**Monitoring for a specific management objective:
protection of shorebird foraging habitat adjacent to a
wastewater treatment plant.**

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19

20 **Abstract**

21 Intertidal invertebrates are often used in environmental monitoring programs as they are good
22 indicators of water quality and an important food source for many species of fish and birds.
23 We present data from a monitoring program where the primary aim is to report on the
24 condition of the potential invertebrate prey abundance, biomass and diversity for migrating
25 shorebirds on mudflats adjacent to a waste water treatment plant in a Ramsar listed wetland
26 in Victoria, Australia. A key threat to the foraging habitat at this site has been assessed as a
27 reduction in potential prey items as a result of the changes to the waste water treatment
28 processes. We use control charts, which summarise data from intertidal mudflats across the
29 whole shoreline of the adjacent wastewater treatment plant, to elicit a management response
30 when trigger levels are reached. We then examine data from replicate discharge and control
31 sites to determine the most appropriate management response. The monitoring program sits
32 within an adaptive management framework where management decisions are reviewed and
33 the data is examined at different scales to evaluate and modify our models of the likely
34 outcomes of management actions. This study provides a demonstration of the process
35 undertaken in a year when trigger levels were reached and a management decision was
36 required. This highlights the importance of monitoring data from a range of scales in
37 reducing uncertainty and improving decision making in complex systems.

38

39 Keywords: “infauna”, “macroinvertebrate”, Port Phillip Bay”, “Western Treatment Plant”,
40 “East Asian-Australasian flyway”, sewage effluent”, “best practice”.

1.0 Introduction

Monitoring of natural resources is undertaken for a variety of reasons although improving our understanding of complex systems is often a primary aim (Fancy et al., 2009; Vaughan et al., 2001). In addition, monitoring can provide input into decision-making by determining the condition of selected ecosystem elements and so help in prioritising and evaluating responses to management interventions. A particular challenge of managing complex systems is one of scale, where focusing data collection on one spatial, temporal or taxonomic scale might mean important changes at other scales are missed (e.g. Thrush et al., 2005; Witman et al., 2015).

Management of migratory species in particular poses a number of issues that are at a range of scales (Bamford et al., 2008; Webster et al., 2002). Migratory shorebirds (Charadriidae) provide a good example of highly mobile species whose conservation requires international collaboration. Management interventions are usually based at a regional or more local scale and focused on ensuring that adequate habitat is available on the migratory route (Hua et al., 2015; Placyk and Harrington, 2004). For example, many species of migratory shorebirds feed upon intertidal invertebrates (Colwell, 1993; Martins et al., 2013; Placyk and Harrington, 2004) and, where shorebirds are a conservation priority, monitoring of their invertebrate prey as well as the shorebirds themselves is recommended to ensure that there is sufficient food available on the migratory route (Goss-Custard et al., 2002; Smith et al., 2012).

We present data from an ongoing intertidal infauna monitoring program on mudflats adjacent to a wastewater treatment plant in a Ramsar-listed wetland of international importance in Victoria, Australia. The site is an important part of the East Asian-Australasian flyway, and the decline in population size of numerous species using this flyway is of global concern (Bamford et al., 2008; Clemens et al., 2016). While these declines are considered most likely

to relate to factors outside of Australia there is recognition that management of Australian shorebird habitat remains important (Clemens et al., 2016).

A key threat to the foraging habitat at this site has been identified as a reduction in infauna prey for shorebird populations as a result of changes to the wastewater treatment processes and the resultant effluent discharges to, and adjacent to, the foraging habitat (Morris and Keough, 2003a and see also Murray and Hamilton, 2010). There is a large body of evidence to support existing conceptual models showing nutrients from organic inputs (including sewage effluents) can be critical in supporting dense infauna assemblages internationally (Pearson and Rosenberg, 1978; Smith and Shackley, 2006; Weston, 1990) and locally (Morris and Keough, 2002; Poore and Kudenov, 1978). Areas of increased infauna biomass around sewage discharges are considered to represent a zone of moderate enrichment akin to the peak of opportunists of Pearson and Rosenberg (1978). As in other parts of the world, there has been concern that, as water quality discharging from the Waste Water Treatment Plant improves, the reduction in nutrients will negatively impact these shorebird populations through reducing the diversity and biomass of their invertebrate prey assemblages (Alves et al., 2011; Burton et al., 2001). In other words, the high conservation status of some coastal areas may rely on continuing low level enrichment increasing the productivity and biomass of prey items (Morris and Keough, 2003a).

The primary aim of this monitoring study is to monitor the condition of potential invertebrate prey abundance, biomass and diversity for migrating shorebirds. The emphasis is on ensuring that there is adequate harvestable prey for shorebirds (West et al., 2005) arriving at the site for the non-breeding season which runs from approximately September to April. The study has been undertaken in an adaptive management framework (e.g. Lindenmayer and Likens,

2009). We present recent data from the study where a management trigger was reached and a decision on intervention was required. We used control charts, a simple graphical tool originally derived for industry that plots when a system is ‘out of control’ to determine when a management intervention is required. While control charts are not frequently used in environmental monitoring (Addison et al., 2015; Morrison, 2008), they are considered a useful tool in aquatic water quality monitoring (ANZECC/ARMCANZ, 2000). In the current study, the control charts were used to provide triggers for management action at the scale of the whole shore management area. Data collected at finer scales provided further information about the drivers of change in intertidal infauna and were used to underpin management decisions by reviewing the outcomes of the possible management interventions (sensu Williams, 2011a).

2.0. Methods

2.1. Site details and background to the monitoring study

The Western Treatment Plant (WTP), owned and operated by Melbourne Water, is a large, 10,550 ha, wastewater treatment plant located on the western shores of Port Phillip Bay, Victoria, Australia. At the time of this study the WTP discharged secondary treated sewage effluent into Port Phillip Bay through three main outlets while a fourth outlet discharged mostly agricultural run-off into the bay. Further freshwater input to extensive intertidal mudflats off the WTP is provided by the Little River which flows through the WTP and into Port Phillip Bay. The intertidal mudflats adjacent to the WTP have historically supported an abundant and diverse intertidal fauna dominated by large polychaete taxa (Dorsey, 1982a; Poore and Kudenov, 1978). While experimental manipulations at this site indicated that moderate changes in prey densities may have little effect on foraging shorebirds (Morris and Keough 2003a) relatively recent (2005) reductions in discharges of nitrogen from the plant

have been linked to reductions in abundance, diversity and size distributions of infauna assemblages (GHD, 2013). The mudflats also support large numbers of resident and migratory shorebirds and can be considered an independent shorebird area within the larger Bellarine Peninsula and Western shoreline Ramsar site (Clemens et al., 2014). In Australia all migratory shorebirds are listed under the Environment Protection and Biodiversity Conservation Act 1999 and the site's inclusion as a Ramsar wetland in 1982 requires sympathetic management of the area to ensure it maintains its ecological character defined at a given point in time (the time of its designation as a Ramsar site with a 6 year reporting framework) (Davis and Brock, 2008; Ramsar Convention Secretariat, 2010). The broad adaptive management framework utilised by Melbourne Water is summarised in Figure 1, and additional details specific to this study are also shown (e.g. use of control charts and model weightings). Further details about each component are summarised below.

2.2. Monitoring

2.2.1. Infauna assessment

Seven sites were chosen for infauna monitoring within the main area of mudflats utilised by foraging shorebirds. While all sites will be influenced by discharges at certain times depending on tide and wind conditions, two sites were situated immediately adjacent to effluent discharges to the bay (145 West and Borrie outlet), two were situated immediately adjacent to freshwater discharges to the bay (Little River and Murtcaim) and the remaining three 'control' sites were at least 1 km from the nearest discharge (Beacon Point, Boat Ramp and The Spit). This provided an opportunity to sample at a number of spatial scales: 1) across the whole WTP shoreline management unit to estimate overall abundance biomass and diversity of macroinvertebrate fauna available to foraging shorebirds (control chart data), 2) by 'treatment', where sites are influenced by effluent, freshwater or act as control sites, and

3) on a site by site basis where changes in macrofauna can be compared across time at individual sites.

All sites were sampled during spring (for construction of control charts), which coincides with the beginning of the migratory non-breeding shorebird season in Victoria, and again during summer (the middle of the non-breeding season) in each year from 2011 - 2015. The spring data and resultant control charts allowed for management interventions to be taken during the migratory shorebird season and shorebird habitat to be re-assessed through the summer data collection. At each site 4 or 5 cores of mud (9 cm internal diameter) were taken to a depth of 10 cm, sieved through a 500 µm sieve, placed in 70% ethanol and returned to the laboratory for later analysis. Samples were sorted and animals identified to the lowest practicable level and the numbers and wet weight biomass of each taxon recorded (no biomass data available for summer 2011). Larger bivalves were excluded from the biomass measure as they are not considered 'harvestable prey' (Zwarts and Wanink, 1993) for the most numerous migratory shorebird species that use these areas (Sharp-tailed Sandpiper *Calidris acuminata*, Red-necked Stint *Calidris ruficollis* and Curlew Sandpiper *Calidris ferruginea*).

2.2.2 Water quality data

Melbourne Water provided water quality data for 145 West, Lake Borrie, Murtcaim Drain and 15 East, all of which discharge intertidally or shallow subtidally (15 East). Data from 15 East was examined as, although not adjacent to shorebird feeding areas, it may still impact intertidal infauna on nearby mudflats at the WTP at certain times. Data provided included daily flow data and weekly nutrient data, including Total Ammoniacal Nitrogen (TAN) and Particulate Organic Carbon (POC). Loads of TAN discharging from the drains were

calculated, using average flow during the week prior to water quality sampling multiplied by the concentration of total ammonia in the effluent on the day of sampling.

POC data is collected monthly but were not available over the entire study period. In order to get a higher resolution of data (weekly measures) and to estimate POC loads from the beginning of the monitoring program, we have calculated POC concentrations as 20% of total suspended solids as this has been shown to be a reliable estimate at 145 West outlet (Nicholson et al., 2011). Loads were calculated as described above.

2.2.3. Rainfall data

We were interested in how freshwater flows and runoff might influence the infaunal assemblages at the WTP. There are no reliable flow meters at either of the freshwater discharges and so we used rainfall data as a proxy for runoff and freshwater flows. Rainfall data were obtained from the Little River rain gauge station (http://www.bom.gov.au/jsp/ncc/cdio/weatherData/av?p_nccObsCode=139&p_display_type=dataFile&p_stn_num=087033). We calculated the average monthly total rainfall, the average daily rainfall for the 3 months prior to each spring sampling occasion and the total rainfall for the preceding 8 months.

2.3. Construction of control charts and control limits (trigger values)

Harvestable prey depends on a variety of factors such as accessibility, size and energy content of infauna taxa, and will vary with species of shorebird (Zwarts and Wanink 1993, West et al 2005). We have used pooled univariate measures of total abundance, biomass and number of taxa plus a multivariate measure of Bray Curtis similarity as proxies of harvestable prey for construction of control charts in this study. These indices are easily measurable, integrate variability at other levels of the ecosystem (e.g. water quality) and should broadly reflect changes in the structure and abundance of the food supply for common species of

shorebird using the mudflats adjacent to the WTP. We also consider changes in common taxa, that previous studies at the WTP have shown to respond to nutrient inputs, to provide supporting information for management decisions and our understanding of the outcomes of management interventions (Morris and Keough, 2001, 2002; O'Brien et al., 2016).

Trigger points (designed to provide a warning of unacceptable change to a system) need to be clearly defined, transparent and accessible to all users and can be identified through the use of control charts. We present data from spring sampling in the first four years of the monitoring program and, as no other long term monitoring data exists, used the mean from the first 3 years of monitoring as the centreline or baseline population ie the 'normal operating conditions' from which deviations in measured values can be assessed. Each data point on the control chart is the mean value of the 7 sampling sites.

Trigger levels are generally set between a value obtained for normal operating conditions and a value beyond which there is potential for unacceptable impacts to occur. In the current study unacceptable impacts were defined as a reduction in intertidal infauna prey items for migratory shorebirds by Melbourne Water (the primary stewards responsible for the maintenance of certain natural values at the WTP site under the EPBC Act 1999). Because there were not enough data available to determine the 'carrying capacity' of the intertidal region, (i.e. how much infauna prey is required to support the current numbers of foraging shorebirds), the measure of the reduction in infauna prey items was based on our experience at the site (expert judgement). Trigger values in this study have been set as a 50% reduction from the 3 year average for the across shore measures of abundance and biomass and a 25% reduction in species richness (number of taxa). These relatively large shifts in measured variables were chosen as previous studies at the WTP have shown that intertidal infaunal invertebrate assemblages can vary over small distances as well as short time scales (Morris

and Keough, 2001, 2003a, b; O'Brien et al., 2011). We consider changes of this magnitude to reflect ecologically important changes in the infauna assemblages that are likely to be over and above the inherent variability observed at this site.

We have also used a multivariate control chart in this study. This was constructed using Bray-Curtis similarities (square root transform) and the centre line or baseline population was the centroid or average of the first 3 years of monitoring (Anderson and Thompson, 2004; Morrison, 2008). Again, we have used expert opinion to set trigger values. In this case we consider similarities below 70% between the centroid of the first three years of monitoring and the new data to comprise a substantial ecological change in infauna assemblages.

2.4. Model choice

In the event of hitting a trigger value, the adaptive management framework (Figure 1) requires that the 'best model' be chosen with the help of the additional data collected during the study. This additional data, which includes analysis at the treatment scale of data collection, provides support for understanding whether the drivers of any changes in infauna measures are due to the sewage discharges on to the intertidal mudflats. This is because the management options available to Melbourne Water to influence the intertidal infauna assemblages are restricted to two possibilities: increase the nutrient load to the intertidal zone or continue standard operating procedures. Melbourne Water is most likely to increase the flow rates of effluent discharging from the main outlets to increase the nutrient load to the intertidal. We compared the likely outcomes from these two management alternatives as four potential models (*sensu* Williams, 2011b) summarised in Table 1.

In Model 1 we envisaged that increasing the quantity of effluent discharging increased the abundance, biomass and diversity of infauna, while standard operating procedures would

maintain the status quo. In Model 2 we considered the scenario where increasing the quantity of effluent discharging resulted in no change, while standard operating procedures allowed infauna abundance and diversity to return to baseline conditions. In this scenario we assume that the initial reduction in the measured variables was not related to the effluent discharges and so the system will ‘re-set’ itself over time. In Model 3 we consider the likelihood that increasing the effluent discharge results in no change to the infauna but standard operating procedures decreased infauna abundance, biomass and diversity. In this case the assumption is that changes to the effluent have reduced the nutrient input to the shore but increasing effluent flow does not increase the measured variables (effluent does not reach intertidal), while standard operating conditions see a further reduction in variables due to reduced nutrient inputs. In Model 4 we consider the likelihood that increasing the effluent discharge results in a reduction in infauna due to toxicity of contaminants while standard operating conditions results in no change to the infauna measures recorded. Where there are lines of evidence to suggest that the sewage discharges are important in the observed changes then Models 1,3 and 4 need to be considered in the decision framework. Where sewage discharges are not thought to be implicated Model 2 becomes important (Figure 1).

2.5. Statistical analyses

Following the control chart construction of the spring data, we analysed the data using Analysis of Variance (ANOVA) to investigate patterns of change in measured values with two factors; Treatment (control, freshwater discharge and effluent discharge) and Time (spring 2014 versus long term average of spring data 2011-2013). This mensurative experimental approach allowed us to investigate links to effluent discharges and was undertaken to assist with the decision around management alternatives. A 3 Factor ANOVA was undertaken when the summer data became available that also includes Season (spring and autumn) as a factor. This analysis was undertaken to evaluate the success of the

management decision. An outlier was excluded from these analyses as transformation did not reduce its influence. The outlying data point was caused by an almost completely denuded fauna at one of the effluent discharge sites (145 West) on the initial monitoring occasion. No clear reasons for this result were evident although there were high loads of ammonia discharging from the 145 West outlet prior to this sampling occasion. But it was also considered possible that a large deposition of wrack at the site was responsible for the depauperate infauna assemblage.

For all univariate analyses residuals were examined and data were transformed where appropriate. Systat V13 was used for all univariate analyses and Primer V5 was used for all multivariate analyses.

3.0. Results

3.1. Infauna

The control charts in Figure 2 (data pooled across the whole shore management unit) show a reduction in all measures in the spring of 2014 in comparison to the 3 year spring mean, and the reduction in biomass was enough to hit the trigger point. Prior to deciding on which of the two management options was appropriate the data were examined at the scale of treatment to determine where the reductions in measured variables were occurring.

There were significant main effects of time in the 2 factor ANOVA on spring data. There were reductions in the square root of the abundance ($F_{1,20}=8.691$, $P=0.007$), $\log_{10+1}$ biomass ($F_{1,20}=10.741$, $P=0.004$) and number of taxa ($F_{1,20}=5.303$, $P=0.004$) in spring 2014 when compared to the long term average of spring data (2011-2013). These reductions were most pronounced in the freshwater treatment (Figure 3, spring panels) although interaction terms (Time X Treatment) were not significant ($P> 0.05$).

There was a significant time $\times$ season interaction in the three-factor ANOVA for abundance, biomass, number of taxa and abundance of *Corophium* spp. (Table 2). These variables all increased in summer 2015 to values similar to the long term average (Figures 3 and 4). The large standard error bar around the mean abundance and biomass in the effluent treatment in summer is a result of a large increase in infauna densities at the 145 West site while numbers had not changed at the Borrie Outlet site (see 145 West data below).

There was a significant main effect of treatment for biomass (Table 2) with the control treatments having a lower mean biomass than the river or effluent treatments (Figure 5).

We examined the data from the 145 West effluent site in more detail as the large standard error for abundance and biomass in the effluent treatment appeared to be due to increases in infauna densities at 145 West. There was a significant ($P=0.03$) increase (1 factor ANOVA with 7 levels of the factor 'time'- see Figure 6) in the abundance of infauna in the summer of 2015 when compared to previous sampling occasions. This increase contributed to the overall recovery of the measured variables. The increase in total abundance at the 145 West in the summer of 2015 was mainly due to an increase in polychaete worms including capitellids, spionids and nereids as well as the amphipod *Corophium* spp. (Figure 6).

3.2. Rainfall

While rainfall in the 3 months preceding the spring of 2014 infauna sampling occasion was lower than in 2012 and 2013, it was not lower than 2011. However, total rainfall in 2014 was much lower than 2011 (Table 3). Rainfall in January 2015 was over twice the long term average (64.6 mm cf 31.6 mm 1906-2014) and half of that rainfall fell in a single day (Little River rain gauge).

3.3. Water quality of effluent discharges

Loads of TAN discharging from 145 West were generally lower in 2014-15 than in previous years (with the exception of summer 2013). In contrast TAN loads discharging from 15 East showed a strong seasonal signal, with high loads of TAN discharging in spring and lower loads discharged in summer. In the summer of 2015 however, loads were more than double than for summers of previous years (Figure 7). POC loads had remained fairly consistent over the time period studied (Figure 7).

4.0. Discussion

International best practice for condition monitoring of estuary areas highlight that monitoring should address specific management objectives and form part of an adaptive management cycle. In addition programs should provide a reporting framework at relevant time scales, in a way that is accessible to a wide range of stakeholders (e.g. a simple visual technique) and that will elicit a management response when limits of acceptable change are breached (Hallett et al., 2016a, b).

We present data from a monitoring program which utilised best practice and where we were also able to incorporate data from several spatial scales to facilitate decision making (Veiga et al., 2016; Witman et al., 2015). Initially control charts indicated the need for a management decision and through the analysis at treatment level we were able assess the most appropriate model and management decision. Monitoring data from the smallest spatial scale (in this example, site level information from 145 West) also provided very useful data that were unexpected but that fed directly into the adjustment and our understanding of the models.

4.1. Management framework

4.1.1. Infauna Monitoring data

We have used control charts to determine the need for management interventions at the largest spatial scale sampled (the whole shore management unit). Control charts can be used to provide an explicit link to management action through the setting of management triggers. These triggers are based on a conceptual model of the system and the relationship between human activity and the resulting response of the system (Addison et al., 2015).

We have set the trigger values or control limits as a set reduction in measured values from the initial 3 year mean baseline population. Further monitoring will allow us to adjust the baseline population and control limits as required, (see Addison, 2014 for a detailed example). As the monitoring information increases, our understanding of the system dynamics and the relative risks associated with the management decisions will also increase (Williams and Brown, 2016).

In the current study, while only edible biomass (of the four control chart variables measured) triggered the control limits, all 4 measured variables provided similar information in that there was a reduction in spring 2014 compared to previous spring data.

4.1.2. Model choice and management action

The reduction in the control chart measures of abundance, biomass and number of taxa were primarily observed around the freshwater inputs into the intertidal region (Fig. 3, spring panels) and corresponded with a lower annual rainfall in 2014. For this reason sewage was not considered to be implicated in the cause of the trigger and Model 2 was considered in the decision framework (Figure 1, Table 1). The management options in response to hitting trigger points in the control charts are limited and because the cause of the trigger was not considered to be related to the sewage effluent, but instead to a reduction in freshwater inputs, the management option chosen was to ‘**continue standard operating procedures**’.

350 The assumption was, that if there was an increase in rain over the spring and summer period,
351 the abundance would increase in the freshwater treatment to reflect the long term average.
352 The infauna abundance did return to ‘normal’ in freshwater and control treatments in summer
353 2015 and rainfall was higher than the average in January 2015 with half of the rainfall falling
354 on one day. These changes observed in the freshwater treatment were consistent with our
355 conceptual understanding that nutrients and POC discharging from the outlets and rivers
356 stimulate the productivity of the near shore environment. In addition they provide support for
357 the conceptual model whereby a reduction in nutrient inputs (the process we have identified
358 as a threat to the conservation objectives) can be managed through changes to human activity
359 i.e. increasing effluent discharge (Addison, 2014). Similarly, the fact that effluent and
360 freshwater discharge treatments had a higher biomass than control treatments further supports
361 this model (Table 2 main effect of treatment and Figure 5).

362 River discharges have been shown to be important drivers of infauna community structure
363 through changes to the sedimentary environment, changes in salinity and inputs of nutrients
364 and organic matter (Lautenschlager et al., 2014; Nishijim et al., 2013). A pulse rainfall event
365 in January 2015 was likely to have affected invertebrate composition at WTP freshwater
366 discharge sites with increases in polychaetes (Little River site) and amphipods (Murtcaim
367 Drain site). While the rainfall event was not unusually large it followed a long period of low
368 flows and so may have been enough to flush out sediment as well as the nutrients and POC
369 associated with this sediment.

370 There was also a large increase in faunal abundance at the effluent site 145 West, which was
371 not explained by increases in rainfall. The increase in capitellid, spionid and nereid
372 polychaetes as well as the amphipod *Corophium* spp.—all species associated with nutrient
373 inputs at the WTP (Dorsey, 1982b; Morris and Keough, 2003a, b; Parry et al., 2013; Petch,

1989) and other geographic regions (Nicastro and Bishop, 2013; Pearson and Rosenberg, 1978; Smith and Shackley, 2006) strongly suggest that there was an increase in TAN or POC at the 145 West site prior to the summer 2015 sampling occasion. There was no evidence that nutrient inputs from the 145 West outlet were any greater than usual for that time of year, (Figure 7). However TAN loads discharging from an outfall to north east of the study area (15 East) were considerably greater in the summer of the 2014/15 season than they had been in the summer of previous years (Figure 7).

Model simulations at the WTP (Greer, 2012) indicate that discharges from 15 East are likely to have impacted the 145 West site, albeit with a 20–50 fold dilution. These findings are broadly consistent with model simulations run for different wind conditions (Longmore et al., 2000). Wind conditions in the summer of 2014/15 were similar to the long term average in that they were dominated by southerlies and westerlies (<http://www.bom.gov.au/climate/dwo/IDCJDW3003.latest.shtml>). Model simulations under these conditions did indicate that the plume from 15 East would influence 145 West although effluent again would be strongly diluted (Longmore *et al.* 2000). There was no influence of POC on 145 West in the scenarios undertaken (Greer 2012).

It is possible that a small increase in TAN loads reaching the 145 West site from 15 East combined with the existing loads of TAN from 145 West over the summer period could have been enough to cause the increase in the species strongly associated with nutrient inputs at the WTP. While loads from 15 East had been high in the preceding winter / spring periods, the summer of 2015 may have provided a sustained nutrient input that allowed juvenile recruitment not seen in previous years. The smaller-scale, site level information (data from 145 West in this example), again supports our conceptual framework and feeds directly into our understanding of the likely outcomes of changes to nutrient inputs reaching the intertidal.

4.1.3. Review

The minimum threshold of invertebrate prey abundance to maintain historical levels of foraging shorebirds is currently unknown. The provision of longer term monitoring may provide clarity in the future, especially when combined with current bird population data. For example, summer counts of shorebirds have been higher in the past 2 years than the previous 12 (Menkhorst et al., 2015) suggesting that the current levels of infauna abundance, biomass and diversity across the whole shore can support high levels of shorebird foraging. This also suggests that the control limits we are using are precautionary and consistent with a strong conservation focus.

The results from the fourth year of monitoring can be used to adjust our initial understanding of the likely outcomes of management actions. This is an ongoing process as the adjustments we suggest here are only based on a single sampling occasion after hitting trigger levels and so the all available evidence needs to be reviewed within each management cycle. While we were only able to choose one management action to test after triggering a control limit, there was support for the conceptual model that increases in nutrient loads will increase the abundance, biomass and diversity of intertidal infauna. The data also presented the possibility that in the current environment these increases may not need to be very large. For example, in the scenario presented in Model 1 of Table 1, increases in effluent flow that result in only a small amount of nutrients reaching the intertidal are likely to have the desired effect in terms of increasing measured values. Similarly support is also provided for the ‘continue standard operating procedures’ management option resulting in a return of measured variables to the baseline when the reduction in measured values cannot be linked to effluent discharges (Model 2 in Table 1). As a result of this we consider the outcomes from Model 1 to be much more likely than the other models. We have also reduced the uncertainty around Model 2 when a reduction in measured variables is not linked to effluent discharges.

The outcomes from Model 3 are now considered less likely than previously thought and we have no new information that relates to Model 4.

The review phase allows our understanding of the possible outcomes of the models to be adjusted, incorporating any new findings and the results of the management actions. It also allows for adjustments to our understanding of the models from unexpected findings that result from the monitoring program (in this case the changes observed at the 145 West site allowed us to speculate that small amounts of nutrients reaching the intertidal zone will have a measurable effects on infauna populations). Provisional control chart parameters are also adjusted at this stage as required. This phase may also affect the management framework in that changes to our understanding of the models may directly affect the choice of management actions towards the beginning of the cycle. For example in the case of a further year of low freshwater inputs, the choice of Model 2 and the management action of ‘continue standard operating procedures’ might not be considered appropriate. In this instance the system would be unlikely to re-set itself in the short term and an increase in effluent flows might be the preferred option.

5.0. Conclusions

Collecting data at a range of scales has allowed us to use trigger values or control limits relevant to our primary conservation objective (protection of shorebird habitat) as well as increase our understanding of the drivers influencing intertidal invertebrate diversity, abundance and biomass at this site. The combination of this information resulted in increased confidence in our models of potential outcomes of management interventions and the management decision framework.

446 Figure captions

447 **Figure 1.** Generalised Adaptive Management framework used by Melbourne Water with
448 additional details relating to the intertidal monitoring program at the Western Treatment
449 Plant.

450

451 **Figure 2.** Control charts of four proxies of harvestable prey a) total infauna abundance, b)
452 'edible' biomass, c) number of taxa per core and d) multivariate control chart constructed
453 using Bray-Curtis similarities. Solid lines indicate average value for the first 3 years of
454 monitoring (Spring data) and the dotted lines indicate trigger values (see text for further
455 details relating to control chart construction). Boxes show mean values for each variable
456 pooled across the whole shore management unit.

457

458 **Figure 3.** Mean abundance, biomass and number of taxa (+/- standard error) of invertebrate
459 infauna sampled in spring and summer periods. The long term average is the mean of the
460 first 3 years of monitoring data calculated for each season separately. 2014 - 2015 is
461 monitoring data for spring 2014 (spring panels) and summer 2015 (summer panels).

462

463 **Figure 4.** Mean number of *Corophium* spp. (+/- standard error) sampled in spring and
464 summer periods. The long term average is the mean of the first 3 years of monitoring data
465 calculated for each season separately. 2014 - 2015 is monitoring data for spring 2014 (spring
466 panels) and summer 2015 (summer panels).

467 **Figure 5.** Edible biomass (mean and standard error) at the three different treatment types
468 over the entire monitoring period.

469

470 **Figure 6.** Patterns in mean density and mean abundance of capitellids, spionids, nereids and
471 *Corophium* spp. at a single site only (145 West) during the monitoring period. Graphs show
472 means +/- standard errors.

473

474 **Figure 7.** Total loads (kg day^{-1}) of Total Ammoniacal Nitrogen (TAN) and Particulate
475 Organic Carbon (POC) discharging in the 3 months prior to each sampling occasion from 145
476 West and 15 East outlets. POC loads are estimated from total suspended solids data (see text
477 for details).

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480

481 **Tables**

482 **Table 1.** Possible Outcomes of management options on intertidal infauna measured variables
 483 (– indicates no change in infauna variables and arrows indicate increase or decrease in
 484 measured variables). The possible outcomes are expressed as four different models.

Management option	Increase effluent discharge rate	Continue standard operating procedures
Model 1	↑	-
Model 2	-	↑
Model 3	-	↓
Model 4	↓	-

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Table 2. P values from 3 factor ANOVA for a number of measured variables for intertidal infauna. Significant values ($P < 0.05$) are indicated by bold text. Due to a loss of degrees of freedom for biomass due to missing data it was not possible to calculate the 3 factor interaction term.

	Time	Treatment	Season	Time X Treatment	Time X Season	Treatment X Season	Time X Treatment X season
Error df = 43	df=1	df=2	df=1	df=2	df=1	df=2	df=2
Total abundance ($\sqrt{}$)	0.98	0.217	0.047	0.278	0.043	0.734	0.725
Total biomass ($\log_{10} + 1$) NB error df = 38	0.743	0.001	0.258	0.486	0.007	0.794	
Total number of taxa	0.865	0.080	0.180	0.234	0.014	0.959	0.492
Polychaetes ($\log_{10} + 1$)	0.811	0.631	0.439	0.805	0.826	0.871	0.877
Capitellids ($\log_{10} + 1$)	0.595	0.626	0.973	0.624	0.582	0.722	0.848
Spionids ($\log_{10} + 1$)	0.366	0.408	0.360	0.909	0.478	0.281	0.905
Amphipods	0.893	0.435	0.844	0.904	0.052	0.844	0.662
<i>Gammaropsis</i> sp. ($\log_{10} + 1$)	0.050	0.451	0.496	0.960	0.672	0.939	0.754
<i>Corophium</i> spp. ($\log_{10} + 1$)	0.548	0.148	0.492	0.637	0.043	0.976	0.854
<i>Paracorophium</i> sp. ($\log_{10} + 1$)	0.464	0.996	0.764	0.556	0.448	0.944	0.940

Table 3. Rainfall data taken from Little River rain gauge station for time periods prior to the spring sampling occasion in each of the 4 years of the intertidal monitoring program. Values show means (and standard deviations).

Rainfall summary data	2011	2012	2013	2014
Average daily rainfall previous 3 months (mm)	0.91(0.3)	2.14(0.7)	1.6(0.6)	0.9(0.28)
Average total rainfall previous 3 months (mm)	27.7(9.3)	65.5(18.9)	50.5(17.5)	26.9(9.0)
Total rainfall previous 8 months (mm)	397	378	336	254

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Figures

Figure 1. Generalised Adaptive Management framework used by Melbourne Water with additional details relating to the intertidal monitoring program at the Western Treatment Plant.

657 **Figure 2.** Control charts of four proxies of harvestable prey a) total infauna abundance, b)
658 'edible' biomass, c) number of taxa per core and d) multivariate control chart constructed
659 using Bray-Curtis similarities. Solid lines indicate average value for the first 3 years of
660 monitoring (Spring data) and the dotted lines indicate trigger values (see text for further
661 details relating to control chart construction). Boxes show mean values for each variable
662 pooled across the whole shore management unit.
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666 **Figure 3.** Mean abundance, biomass and number of taxa (+/- standard error) of invertebrate
 667 infauna sampled in spring and summer periods. The long term average is the mean of the
 668 first 3 years of monitoring data calculated for each season separately. 2014 - 2015 is
 669 monitoring data for spring 2014 (spring panels) and summer 2015 (summer panels).

Figure 4. Mean number of *Corophium* spp. (+/- standard error) sampled in spring and summer periods. The long term average is the mean of the first 3 years of monitoring data calculated for each season separately. 2014 - 2015 is monitoring data for spring 2014 (spring panels) and summer 2015 (summer panels).

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Figure 5. Edible biomass (mean and standard error) at the three different treatment types over the entire monitoring period.

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685 **Figure 6.** Patterns in mean density and mean abundance of capitellids, spionids, nereids and
 686 *Corophium* spp. at a single site only (145 West) during the monitoring period. Graphs show
 687 means +/- standard errors.
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Figure 7. Total loads (kg day⁻¹) of Total Ammoniacal Nitrogen (TAN) and Particulate Organic Carbon (POC) discharging in the 3 months prior to each sampling occasion from 145 West and 15 East outlets. POC loads are estimated from total suspended solids data (see text for details).

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