WWTP Discharges as Environmental Flows?

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Abstract

Macroinvertebrate communities were used to assess the potential impacts of Gippsland Water's Waste Water Treatment Plant's (WWTP) discharge into Shillinglaw Creek in Drouin (Victoria) during the Spring seasons between 2004 and 2007. In June 2006 a Dissolved Air Flotation Filtration (DAFF) system was installed at the WWTP. Analysis of the data revealed a significant difference in the macroinvertebrate communities prior to and after the commissioning of the DAFF treatment. Consistently higher SIGNAL indices were found downstream of the discharge point and indicate that tertiary treated wastewater does improve the quality of the effluent discharge and as a consequence the condition of the waterway downstream by providing additional flows. The Victorian Environment Protection Authority has historically discouraged water authorities from discharging wastewater into waterways, instead promoting land based reuse. There is an increasing demand for water in the current and future climate. Environmental flows have in the past been generally regarded as a minor priority. It is therefore suggested that tertiary treated wastewater can be used for environmental flows, provided a risk assessment of each individual discharge is conducted.

Introduction

Recent history indicates that in temperate, south-eastern Australia drought, or below average rainfall years have been prevalent. The study site, located near Drouin, located approximately 90 km southeast of Melbourne, Victoria, has had below average rainfall in each of the past years since 1999 (BOM 2008). Drought, combined with human impacts have led to one third of major streams and two thirds of wetlands in Victoria to be considered in a poor to very poor condition (DSE 2004). Such degraded aquatic environments have led to one species of fish to be extinct and 18 native species are now listed as threatened out of the 40 native species of within Victorian freshwaters (DSE 2004).

The Victorian Government in 2004 released the White Paper, in which formularised the Environmental Water Reserve (EWR) as part of a principle to achieve sustainable healthy river systems (DSE 2004). The EWR is designed to set aside water to; (1) maintain the environmental values of the water system and other water services that depend on environmental condition; (2) sustain biodiversity, ecological functioning and water quality; and (3) have legal status and is held by the crown. This is an important landmark for the legal status of the aquatic environment, of which the water provided for the environment has the equivalent legal status to water allocated for consumption (DSE 2004).

The Victorian Government has also previously announced a target of 20% reuse of water by 2010 (DSE 2004), of which most to date has been in the form of land-based reuse, namely irrigation. However, water has, in several cases in suburban Melbourne, been reused in houses, mainly for toilet and garden usage (DSE 2004). This target of wastewater reuse was based upon Environmental Protection Authority (EPA) Victoria's Guidelines for the Use of Reclaimed Water (EPA Victoria 2003*a*). Whose main objective is to encourage the safe and sustainable use of reclaimed water through land-based, urban reuse, industrial reuse and groundwater recharge. Under current legislation the discharge of reclaimed water into the environment is not identified as beneficial reuse.

A resource condition target of the Victorian River Health Strategy (VRHS) aims that by 2011 significant improvements are to be achieved in environmental flow regimes of 20 high value river reaches (DNRE 2002). In the ever increasing search for environmental water one pathway to successfully meet the requirements of the White Paper, the VRHS and the EWR objectives, as well as improving the ecological state of many rivers and streams, is to formally recognise the contribution of WWTP discharges to environmental flows.

Historically, the Victorian Government have encouraged water authorities towards land based reuse of wastewater in a variety of methods, mainly through irrigation and industrial practices, rather than releasing the water back into the waterways has come at a cost to the environment. The treatment of wastewater has vastly improved over recent history and tertiary treatment is common at many treatment plants. The discharge from these tertiary treated WWTPs are usually of good quality, and can often meet Class 'A' standards.

One such discharge is the Drouin WWTP, from which Gippsland Water is currently licensed to discharge tertiary treated water to Shillinglaw Creek. This allows a case study to be conducted regarding the appropriateness of wastewater discharges to be used as environmental flows, in this case base environmental flows.

Rapid Biological Assessment (RBA) programs have generally been designed to compare sites that have been impacted, to reference sites. Reynoldson *et al.* (1997) define the reference condition as the condition that is representative of a group of minimally disturbed sites based on selected physical, chemical, and biological characteristics. The reference condition is the best available reference, as its original condition is usually one that will probably never occur again (Reynoldson *et al.* 1997) as there is the contention that streams are in a constant state of recovery after the last disturbance (Olsen *et al.* 2007) and that patch-specific disturbances are likely to result in heterogeneity of benthic communities (Matthaei and Townsend 2000).

EPA Victoria recommends the use of RBA programs based upon macroinvertebrate sampling to monitor the impacts of wastewater discharges into waterways (EPA Victoria 1998). The RBA program recommended using standardised macroinvertebrate sampling techniques and analyses together with biological indices to detect potential impacts of the discharge.

Biological indicators are increasingly being used to supplement physical and chemical parameters to examine heath of waterways (eg. ANZECC 2000; EPA Victoria 2003*b*). The main types of biological indicators are biotic indices, multimetric indices and predictive models. Biotic and multimetric indices are based on data that are derived across gradients of disturbance, where as the predictive models predict the composition of community assemblages of sites in the absence of stress (Walsh 2006).

Macroinvertebrates occupy a central role in the ecology of rivers and provide a stable basis for biological monitoring, since they can live from several weeks up to several years (Chessman 1995). Additionally, they possess relatively long life cycles, in comparison to some other organisms used in bioassessment, hence this allows the possible identification of potential differences inferred by perturbations (Hawkes 1979). Furthermore, their feeding and behavioural characteristics link them to potential contaminants within the stream and therefore make them good indicators for detecting any specific degradation of stream condition (Reynoldson *et al.* 1995).

Biological indicators, in particular macroinvertebrates, are now common indicators within many sampling biological sampling programs, e.g. SIGNAL, AUSRIVAS and Victorian EPA's RBA Methodology (Chessman 1995; ANZECC 2000; EPA Victoria 2003*b*). EPA Victoria (2003*c*) found that AUSRIVAS appeared more sensitive to habitat impairment than of water quality, where as biotic indices (SIGNAL and number of EPT (Ephemeropera, Plectoptera, Trichoptera) families) are more sensitive to changes of water quality. Although, Walsh (2006) points out that evidence for such claims are scant.

The State Environment Protection Policy (SEPP) objectives are primarily aimed to detect impacts, which act as a trigger to warrant further investigations (EPA Victoria 2003*c*). To be an effective trigger for further investigations they must be sensitive enough to detect low to moderate levels of degradation and produce as few false triggers as possible (Walsh 2006).

As part of Gippsland Water's biological monitoring program of the WWTP at Drouin, the macroinvertebrate communities and *in-situ* water quality were analysed to determine if the wastewater discharge into Shillinglaw Creek had any adverse impacts. This biological monitoring program has been conducted during the Autumn season of 2002, and during the Spring seasons between 2004 and 2007.

In June 2006 a Dissolved Air Flotation Filtration (DAFF) system was installed at the WWTP, its aim to remove greater amounts of the formed floc, which in turn improves water quality (GHD 2005).

To accurately determine if the introduction of the DAFF treatment has any positive impacts upon the subsequent waterway, it is important to understand the extent of the local spatial and temporal variation within Shillinglaw Creek. The spatial variability of streams and rivers is well recognised as an important feature (Palmer and Poff 1997) that maintains diversity, as is the understanding of the factors that drive and maintain heterogeneity (Cooper *et al*. 1998). Disturbance, a dominant factor in stream ecology (Resh *et al*. 1988), can have variable spatial effects creating a mosaic of microhabitats with different disturbance histories (Townsend 1989). Therefore, it is imperative when comparing two sites to one another that they both have the same, or very similar, environmental characteristics, in particular the habitat structure. Sites that support different habitats can introduce inter-habitat variation that can potentially mask the differences on results between sites (Parsons and Norris

1996). An understanding of temporal variation within an aquatic ecosystem requires the monitoring of communities over a range of time scales (Gell *et al.* 2002). Sampling sites were therefore based upon these previses and were located at the same point during each sampling occasion.

The monitoring program tested the following hypotheses:

- The WWTP is having a significant impact upon the macroinvertebrate communities downstream of the discharge; and
- The installation of the DAFF system improves the condition of the waterway.

Methods

Study Site

The Drouin WWTP is located south-west of Drouin (38^{o} 08' S, 145^{o} 50' E), south-eastern Victoria, within Gippsland Water's jurisdiction. Annual rainfall is approximately 1000 mm (BOM 2008).

The Drouin WWTP receives an inflow of 1-1.5 ML per day and has primary, secondary and recently (June 2006) tertiary treatment processes. The method of treatment is as follows; the wastewater gravitates directly into a flume, where it is regulated through a series of three lagoons. The wastewater is then pumped from lagoon 3 through bio-filters for ammonia removal and then into parallel flocculation tanks housed within the DAFF plant. Within each tank PFS is added as a coagulant as well as to remove phosphorous. Once flocculation has occurred, wastewater passes into the two DAFF units, where dissolved oxygen is injected into the tank to attach to the formed flocs, causing the solid particles to rise to the surface. The formed solids are skimmed off the surface and placed into lagoon 2. The treated wastewater is subsequently passed through a filter, which then the discharge enters Shillinglaw Creek via a drain.

For several years the treated wastewater from the WWTP has been released into Shillinglaw Creek during the wetter months (Jun-Nov), while it has been reused for irrigation purposes during the drier months (Dec-May). The usage of the discharge between 2004 and 2007 is as follows:

- 2004-2005: 209 ML was reused, while 328 ML was discharged into Shillinglaw Creek;
- 2005-2006: 204 ML was reused, while 390 ML was discharged into Shillinglaw Creek; and
- 2006-2007: 255 ML was reused, while 143 ML was discharged into Shillinglaw Creek.

Shillinglaw Creek enters King Parrot Creek approximately three kilometres further downstream.

Site Description

Upstream of the discharge point, Shillinglaw Creek had an average stream width of 2.7m and a depth of 0.5m. Sites downstream of the discharge point had an average stream width of 2.4m and a depth of 0.4m.

In King Parrot Creek, upstream of the junction with Shillinglaw Creek, there was an average stream width of 2.8m and a depth of 0.4m. Meanwhile, downstream of the junction with Shillinglaw Creek, King Parrot Creek had an average stream width of 2.5m and a depth of 0.5m.

The substrate within Shillinglaw Creek upstream of the discharge is predominantly silt. Silt is a major feature downstream, with Sites 5 and 6 also containing a small amount of gravel.

The substrate within King Parrot Creek upstream of the confluence with Shillinglaw Creek is predominantly silt, with Site 1 having a combination of boulders, pebbles, gravels and sand. While downstream silt is the main substrate material, however Site 6 is comprised of boulders and sand.

Data Collection

Six sites were sampled along Shillinglaw Creek i.e. 2 upstream and 4 downstream of the point of direct effluent discharge. The upstream and downstream sites from the discharge point were selected so that they were as similar to each other as possible with regards to biotic and abiotic parameters.

Identification of each sampling site was by number in ascending order in the downstream direction, from Site 1 to Site 6 for the sites sampled on Shillinglaw Creek.
Sites 1 and 2 were located upstream of the discharge point, while Site 3, 4, 5 and 6 were downstream. Collection was conducted in a reverse order, travelling upstream. This was necessary so as not to contaminate subsequent samples with drifting or dislodged organisms from prior sampling.

For each sampling site, several physical, biological and environmental variables were measured in parallel with the macroinvertebrate sampling in order to ensure that no other variables influenced the outcomes of the study. *In-situ* water quality data was collected at each site prior to sampling. All recorded parameters measured were as per the standard Victorian RBA method (EPA Victoria 2003*b*).

To obtain a standard assessment of the macroinvertebrate communities at each site, samples were taken from two distinct aquatic habitats, the benthos and littoral fringe (also deemed the riffle and edge), using standard Victorian RBA methods (EPA Victoria 2003*b*). However, to address the variations and overlap between these habitat types, this method was modified to combine these two habitats and obtain accurate descriptions throughout the water column. Therefore, a 5m transect of both the benthic and littoral habitats were combined to comprise a 10m representative sample of the site.

The collected material from each site was live-picked for a minimum of 30 minutes, as per EPA Victoria (2003*b*), and preserved in 70% ethanol. Upon return to the laboratory, the specimens were sorted, identified to family level, using keys as per Hawking (2000) and MDFRC (2008), and enumerated.

Also at each sampling site, a quick assessment of resident fish and zooplankton populations was undertaken. The presence/absence of piscivorous, but particularly zooplanktivorous fish that may reflect predation/prey relationships within the system, can sometimes help to explain depressed or elevated levels of zooplankton activity at various sites.

Statistical Analyses

Biological Indicators

To characterise the condition of the stream, the biological indices were performed as required by EPA Victoria (2003*b*). These are the number of families, number of EPT families, number of key families and the EPA Victoria SIGNAL Index. These indicators were then compared to the SEPP objectives (EPA Victoria 2003*c*). Also the Chessman (1995) SIGNAL index was calculated to provide a more robust assessment of the condition of the stream. The use of multiple indices gives the assessment additional reliability, as each of these indices have their individual limitations (Tiller and Metzeling 2002). However, the AUSRIVAS Observed/Expected score was not calculated as the AUSRIVAS predictive model involves the observed taxa at a selected site compared to the taxa expected to occur at a site with similar characteristics in the absence of human impact (Reynoldson *et al.* 1997; Walsh 2006). Victoria EPA (2003*c*) found that AUSRIVAS appeared more sensitive to habitat impairment than of water quality, where as biotic indices (SIGNAL and number of EPT families) are more sensitive to changes of water quality. It was therefore decided that the AUSRIVAS score would not be calculated.

Multivariate Statistical Analyses

The sampling technique of the macroinvertebrate communities is qualitative, therefore Bray-Curtis similarity matrices were constructed with presence/absence transformations using PRIMER v5 (Clarke and Warwick 2001).

Comparisons between sites and sampling occasions were conducted using one-way Analysis Of Similarity (ANOSIM) within PRIMER v5 (Clarke and Warwick 2001). ANOSIM was used to test the statistical significance of the hypotheses, of which tested the following macroinvertebrate groups (Clarke and Warwick 2001):

- Upstream and downstream of the discharge point in Shillinglaw Creek; and
- Before and after the installation of the DAFF treatment facility.

Results

All Samples

A total of 4,863 individuals from 71 taxa were found in Shillinglaw Creek over the 5 years of data collected for this study.

Spring Samples Only

A total of 4,228 individuals from 60 taxa were obtained throughout all of the samples collected between 2004 and 2007.

The order Trichoptera was the most diverse, closely followed by the Odonata, with 10 and 9 families, respectively. The Diptera was the most abundant order, with 1,254 individuals from 5 families, of which the most abundant family was the Chironomidae (1,143 individuals), which comprised 27% of the total number of macroinvertebrates collected. The molluscs were also abundant, of which the introduced snail *Physa acuta* comprised 12%, and the Hydrobiidae 6.5% of the total number. There are no immediately obvious trends within the descriptive analyses, although Autumn 2002 had the highest average diversity, found amongst the samples upstream of the discharge (Table 1).

Descriptive Analyses

Table 1 Abundance, diversity and SIGNAL scores for each year surveyed, averaged upstream and downstream of the discharge point

Year	Indices	Ave. Upstream of Discharge	Ave. Downstream of Discharge
2002	Number of Organisms	113.5	103.5
	Number of Families	21.5	16.75
	EPA SIGNAL Index	5.43	4.98
2004	Number of Organisms	255	220
	Number of Families	21.5	15.25
	Chessman SIGNAL Index	5.50	5.01
	EPA SIGNAL Index	5.97	5.60
2005	Number of Organisms	128	143.75
	Number of Families	14.5	15.75
	Chessman SIGNAL Index	5.23	5.08
	EPA SIGNAL Index	5.81	5.67
2006	Number of Organisms	118	112
	Number of Families	12.5	13.5
	Chessman SIGNAL Index	4.82	5.36
	EPA SIGNAL Index	5.42	6.03
2007	Number of Organisms	272.3	195
	Number of Families	18	19
	Chessman SIGNAL Index	4.39	5.27
	EPA SIGNAL Index	5.17	5.73

Table 2 Average Bioindicators before and after the installation of the DAFF treatment (excluding samples collected during Autumn 2002)

	Before DAFF Installation		After DAFF Installation	
	Average Control Samples	**Average Impact Samples**	**Average Control Samples**	**Average Impact Samples**
EPA SIGNAL Index	5.89	5.63	5.30	5.88
Chessman SIGNAL Index	5.36	5.04	4.61	5.32

Before the installation of the DAFF facility the control samples had higher scores of both SIGNAL indices than the impact samples (Table 2). This trend may be a result from the discharge. The opposite trend occurred after the installation with the impact samples having higher scores than the control sites (Table 2).

The control samples before the installation of the DAFF treatment had higher average scores from both SIGNAL indices than after its installation (Table 2). This trend was not significant (Table 6). The opposite trend was observed with the impact sites having higher average scores after the installation than those before it (Table 2). The latter trend however was significant (Table 6).

Multivariate Analyses

All Years

The results from one-way ANOSIM analysis found there to be a significant difference between all samples collected in Autumn 2002 to all other samples collected in Spring (2004-2007) (Global R = 0.414, p = 0.02, 999 permutations).

Samples collected during different seasons (Autumn and Spring) are different and are hard to compare to each other. When a single sampling occasion of a season (Autumn) is compared to multiple occasions of a different season (Spring) may lead to erroneous conclusions (Reece *et al.* 2001). The main determinant for the difference between the sampling seasons is that Autumn season sampling collects the recruited individuals in the community while the Spring samples collect the emergent stages of some invertebrates. Therefore, the Autumn 2002 data was omitted from the dataset, as it may obscure the results.

All Spring Samples (2004-2007)

Influence of the WWTP discharge

Table 3 Comparison between samples upstream and downstream of the discharge

Year	R Statistic	Significance Level	Actual Permutations
2004	-0.125	0.733	15
2005	-0.393	1.0	15
2006	-0.089	0.667	15
2007	-0.354	0.867	15
All Years Combined	-0.029	0.620	999

All four years of spring sampling showed no significant difference from macroinvertebrate communities sampled upstream of the discharge to those downstream of the discharge (Table 4). Also, when all data sets for all four years were combined, there were no significant differences between the control and impact samples. These results suggest that the discharge from the WWTP was not having any negative effects. It must also be stated that these tests also showed there to be no significant positive effects of this discharge either.

Comparison between years

Using only the sample data from the control sites, all one-way ANOSIM analyses found there to be no significant differences in comparisons of any two years (Table 4). Therefore no observable temporal trends were found amongst all control samples.

Using only the sample data from the impact sites, all one-way ANOSIM analyses also found there to be no significant differences in comparisons of any two years (Table 5). Therefore there were also no observable temporal trends within the impact samples.

The absence of a significant difference between any of the years amongst the samples upstream of the discharge and amongst samples downstream of the discharge infers that there is no or very little temporal variation between these treatment groups. This validates the test to determine if the introduction of the DAFF facility to the treatment plant is having any affect upon the macroinvertebrate communities.

Table 4 All Samples Upstream of the Discharge

Treatment Pairs	R Statistic	Significance Level	Actual Permutations
2005, 2006	0.375	0.333	3
2005, 2007	1.0	0.333	3
2005, 2004	1.0	0.333	3
2006, 2007	0.0	0.667	3
2006, 2004	1.0	0.333	3
2007, 2004	1.0	0.333	3

Table 5 All Samples Downstream of the Discharge

Treatment Pairs	R Statistic	Significance Level	Actual Permutations
2005, 2006	0.339	0.114	35
2005, 2007	0.323	0.114	35
2005, 2004	0.49	0.057	35
2006, 2007	0.188	0.20	35
2006, 2004	0.339	0.114	35
2007, 2004	0.417	0.086	35

Influence of the DAFF treatment

Within the control samples, there was found to be no significant difference between the samples collected before the installation of the DAFF treatment to those collected after its installation (Global R = 0.354, p = 0.086, 35 permutations) (Table 6). This result is expected, as the control samples are not influenced by the WWTP.

The impact samples however, did show a significant difference between before and after the installation of the DAFF treatment (Global R = 0.228, p = 0.017, 999 permutations) (Table 6).

This result combined with the SIGNAL scores indicates that the effect of the DAFF treatment was positive. Further, the overall improvement in condition of the waterway downstream of the discharge, it could be suggested, is due to the discharge of the WWTP and the introduction of the DAFF treatment.

Table 6 Differences before and after the DAFF installation

Treatment Pairs	R Statistic	Significance Level	Actual Permutations
Control Samples	0.354	0.086	35
Impact Samples	0.228	0.017	999

Discussion

It was found through the investigation that the WWTP discharge was having no significant effect or improvement upon the macroinvertebrate communities within Shillinglaw Creek prior to the installation of the DAFF treatment. However, the commissioning of the DAFF treatment shown trends in the improvement of the condition of the waterway with higher SIGNAL scores downstream than upstream of the discharge.

The commencement of the DAFF treatment within the WWTP during June 2006 provides Shillinglaw Creek with improved water quality. This, with the addition of increased flow, positively improved the overall condition of the waterway, as shown by the macroinvertebrate community response and the SIGNAL scores.

The Victorian Government, through the White Paper, recognised the underlying principle that future management of water is based on the understanding that a healthy economy and society is based upon a healthy environment (DSE 2004). We know that rivers are a major part of lives for many people eg. drinking water, Indigenous groups, fisheries, boating and water based activities to name a few. One third of Victorian rivers are considered to be of poor to very poor condition (DSE 2004), as determined by the Index of Stream Condition (ISC). This very principle of a healthy society would be severely jeopardised if the condition of many waterways continues to decline.

Recently, there is much recognition worldwide of the degraded state of many rivers, not just in Victoria. Effort to restore rivers to a state where they are able to continue to provide the services required is increasing. Environmental flows are just a part of this process (Crook and Koster 2006).

Future climate change predictions may, however, change the very way in which water is used for every purpose. As patterns of precipitation change within a catchment, so will the seasonal, annual and interannual storage cycles of water (FAO 2003), which will in turn effect the irrigation demand, due to a combination of decreased precipitation and increased evaporation arising from increased temperatures (Bates *et al.* 2008).

Within Australia, the Murray-Darling Basin Commission estimates that by 2023 there will be 5% less inflows into the Murray-Darling Basin, which equates to 1,100 GL less water (DSE 2004). Although this is vastly different system to the study site for this study, this gives a dramatic insight to the impact of climate change within Australia.

The climatic change predictions for the Port Phillip and Western Port catchments, which include the study site and Melbourne, suggest that by 2030 the annual rainfall will be 4% less than the 1990 average, with the spring season seeing the largest decrease of 7% of precipitation (DSE 2008). This data also indicates that this region will be exposed to higher evaporation rates (3% increase at 2030, compared to 1990 levels), with the winter season experiencing the largest increase (8% higher than in 1990) (DSE 2008). These results are also expected to increase further into the future. DSE (2008) also indicates that this region will see an increase in the intensity of rainfall events, with a decrease in the number of rainy days per year.

All these climate change predictions, in conjunction with the current conditions of Victorian waterways, paint a bleak picture for the future functions and ecology of these systems. The Victorian State Government with the White Paper (DSE 2004) does however, recognise that "…within urban systems, the Environmental Water Reserve can be enhanced by substituting water used for consumptive purposes with recycled or reused water, or by using recycled water, of adequate quality, to boost environmental flows…" as well as through more efficient and conservation approaches to water usage.

Historically, WWTP discharge into waterways has been discouraged due to poor water quality as a result of insufficient treatment. In many cases this discharge has had negative impacts on its receiving waters. However, treatment technology has come a long way in the last ten years where the tertiary treatment of wastewater in now common practice for WWTP discharges into waterways amongst Victorian water authorities. This tertiary treatment of wastewater often results in effluent quality meeting Class 'A' standards. This standard of water is often of higher quality than the receiving environment.

A WWTP discharge into a waterway is just one method that can be used to improve environmental flows. WWTP discharges into creeks and rivers may prove to be a most viable method of reclaiming water for the environment. WWTP's are generally located topographically at a lower elevations of townships, and in close proximity to a waterway, meaning that only small amounts of infrastructure would be needed to allow wastewater to be released directly into a waterway or retention dam. A retention dam could store water to then be discharged into a waterway as a fresh or pulse flow. Pulse flows are small flow events that exceed base flows for several days and are important for improved habitat and water quality throughout the waterway (Cottingham *et al.* 2005). They are also important triggers for spawning and migration of many fish species (Cottingham *et al.* 2001; Bunn *et al.* 2006).

Victorian Catchment Management Authorities and Melbourne Water have undertaken extensive investigations of environmental flows requirements for priority river systems across the state (eg. Barwon, Thompson, Yarra, Goulburn, Glenelg and Wimmera Rivers). These investigations have identified which components of the flow regime are not being met.

In many cases there may be limited opportunities to enhance the environmental water reserve in either the short or longer term by more recognised means, such as market adjustment, water buyback or implementation of other savings. In these cases the discharge of high quality treated wastewater may provide a short or long term solution to enhance the environmental water reserve. These discharges can contribute to an overall net environmental benefit to a flow-stressed river system.

Wastewater that is to be released back into the natural environment would have to be subject to an ecological risk assessment. This includes risks to the environment as well as to the human population.

This investigation has identified a significant improvement in macroinvertebrate community condition through water quality improvements. This coupled with river health activities such as frontage management, instream habitat etc. can provide an overall improvement in ecological condition.

It is recommended river managers and regulators consider that high quality WWTP discharges can have a net environmental benefit to river condition. These discharges can provide a legitimate solution to enhance the EWR in either the short or long term.

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