COMPOUNDING EFFECTS OF AGRICULTURAL LAND USE AND WATER USE IN FREE-FLOWING RIVERS: CONFOUNDING ISSUES FOR WATER MANAGEMENT

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Defining the ecological impacts of water extraction from free-flowing river systems in altered landscapes is challenging as multiple stressors (e.g. flow regime alteration, increased sedimentation) may have simultaneous effects and attributing causality is problematic. Furthermore, patterns of water extraction (e.g. pumping timing and rates) may govern the characteristics of hydrologic impacts and ecological responses. We examined the impacts of land and water use on rivers in the upper Ringarooma River catchment in Tasmania (south-east Australia), which contains intensively-irrigated agriculture, to support implementation of a Water Management Plan (WMP). Temporal trends in river condition were assessed using a 19-year dataset of macroinvertebrate community monitoring. Macroinvertebrates were also sampled at 19 sites before and after a dry irrigation season in 2012/13, while spatial land and water use datasets were used to characterize study sites as having upstream sub-catchments with: (1) low agricultural land use and low water use, (2) high agricultural land use and low water use, or (3) high agricultural land use and high water use. In addition, the response of macroinvertebrate communities in edge water and thalweg microhabitats to diel flow variability was investigated at sites with stable and variable baseflows. Water extraction-related stressors were found to exacerbate impairment associated with agricultural land use (e.g. reduced macroinvertebrate density, more flow avoiding taxa). Increased diel flow variability relating to water extraction appeared to contribute to the degradation by making edge water habitat uninhabitable for many taxa. These findings support the need to implement the Ringarooma WMP in accordance with its environmental flow thresholds, allocation limits and adaptive management provisions. To inform an adaptive management process, further ecological surveillance monitoring and targeted research to test the effectiveness of flow management strategies is being be conducted.

1 INTRODUCTION

It has long been recognised that anthropogenic modification of landscapes impacts the ecological condition of rivers [1, 2]. While water abstraction, via subsequent flow regime alteration, can adversely impact river condition, in an agricultural landscape multiple stressors associated with land use practices may simultaneously affect rivers [3, 4] and disentangling causality can be difficult [5]. Furthermore, spatial and temporal patterns of water extraction (e.g. pumping timing and rates, distribution of exaction points, etc.) may govern the characteristics of hydrologic impacts and ecological responses. In addition, natural variation in geographic and environmental factors influences the ecological state of river systems [6, 7] and this needs to be accounted for when assessing impacts of land and water use.

Free-flowing river systems (i.e. drainages without substantial instream dams) typically retain components of their natural flow regimes; however, direct extraction of water from river channels can suppress low and moderate flow events, especially during dry periods when flows are naturally low and extraction rates are greatest [8]. Such impacts may be more subtle than those caused by large instream dams and their associated flow regulation, but understanding the environmental consequences of this type of flow regime alteration is critical to effective water planning in catchments with high levels of direct extraction, including setting, implementing and evaluating environmental flows [9, 10].

The Ringarooma River catchment in north-eastern Tasmania, south-east Australia has been influenced by water use practices associated with agriculture and forestry operations for over 100 years, while historical alluvial tin mining has also affected substantial stretches of some rivers in the catchment. Since *c.* 1990, consumptive water use in the catchment has increased markedly due to demands from irrigated agriculture, especially during summer-autumn (December-April). A Water Management Plan (WMP) was developed for the catchment by the Tasmanian State Government [11] to formalise water resource management strategies in the catchment and balance the needs of water users and the environment in line with state legislation. The Ringarooma WMP will allow allocations of up to 26% of the long-term (1970-2009) median annual discharge of the Ringarooma River, and up to 44% of the long-term median yield of the river for December-April periods (proportions of which are for non-consumptive use, i.e. for hydro-electric power generation).

Previous coarse assessments on river health in the Ringarooma catchment indicated that recent levels of water extraction were not significantly affecting the environmental integrity of the river system, thus the volumes of allocated water and minor degradation of the river reflected a balanced water planning outcome. However, temporal and spatial trends in river condition have not been thoroughly examined in the catchment, nor have the impacts of water use on its riverine ecosystems.

Implementing the Ringarooma WMP will follow adaptive management principles and involve finalising water allocation limits and examining the condition of rivers in the catchment during a five-year transition period [11]. This transition period will enable the WMP to be phased into practice, with an imperative to ensure levels of water extraction are not impacting significantly on river condition. To support the implementation of the WMP, spatial and temporal trends in river condition were assessed using benthic macroinvertebrates as indicators in the upper Ringarooma catchment where land and water use pressures on riverine ecosystems are quite intense. The ecological effects of diel flow variability in the river system, which is caused by water extraction patterns, were also examined.

2 METHODS

2.1 Study area

The Ringarooma River catchment (area = 930 km^2) has headwaters in the Ben Lomond Ranges (elevations ~1100 m a.s.l.) and drains north into Bass Strait. Annual rainfall varies from ~1800 mm in the headwaters to ~600 mm on the coastal plains. Agricultural activities, such as intensive cropping, dairy farming, cattle grazing, and plantation and production forestry, mostly occur in the middle and upper areas of the catchment, which was the focal region of this study. The upper catchment has predominantly basalt and granitic soils, and rivers in this area have mostly rocky substrate (i.e. cobble-dominated substrata); although granitic fines are prevalent in some mid-catchment tributaries. Several tributaries enter the Ringarooma River in the mid to upper catchment, including the Cascade River, Dorset River, Federal Creek, Frome River, Legerwood Rivulet and Maurice River; most of which enter the Ringarooma River upstream of the town of Branxholm.

The Ringarooma River has a perennial, predictable and highly seasonal flow regime with high flows over winter-spring and low flows during summer-autumn [9]. Across the upper and mid-catchment, a number of agricultural water storages (i.e. farm dams) impound water which would have naturally flowed into Ringarooma River via its tributaries. Flow regime alteration due to water use is most pronounced in rivers is the upper catchment during summer-autumn, with overall reductions in baseflows and significant diel variation in flow being associated with patterns of direct water extraction from the river system for irrigation (DPIPWE, unpublished data).

2.2 Long-term macroinvertebrate community monitoring

Historical qualitative (Australian River Assessment System; AusRivAS) [12] benthic macroinvertebrate data for 13 sites in the upper Ringarooma River that had been sampled on >2 occasions between autumn 1994 and spring 2009 were collated with similar data collected at 19 sites across the upper catchment during November 2012 and March 2013 in the present study. Qualitative sampling was conducted over a 10-m^2 area of substrate in riffle habitat using a kick net (250 × 365 mm, 250 µm mesh). All qualitative invertebrate samples were live picked *in situ* for 30 min, maximising the number of taxa collected, with additional time allocated for picking if new taxa were detected within the final 10 min of the period. The samples were preserved in 90% ethanol.

All macroinvertebrate samples were identified to family level in the laboratory using relevant taxonomic keys [e.g. 13] except in the following cases: Chironomidae (midges) were identified to sub-family level; Nematoda (nematodes), Oligochaeta (worms), Hirudinea (leeches), Acarina (mites) and Turbellaria (flatworms) were identified to order and class level. For simplicity, the terms 'family' and 'taxa' are hereafter used to describe identifications to these various taxonomic levels.

2.3 Seasonal macroinvertebrate sampling across the upper Ringarooma catchment during 2012/13

Quantitative benthic macroinvertebrate samples were also collected during November 2012 and March 2013 from riffle habitat at the 19 sites where qualitative samples were collected (section 2.2). Quantitative sampling involved the collection of 10 surber samples (300 × 300 mm, 250 µm mesh) by hand disturbance of a 0.09-m^2 area of substrate. The 10 samples were pooled for sites and preserved (10% formalin) prior to sub-sampling (20%, box sub-sampler) in the laboratory. The taxa in the sub-samples were identified following the methods that were used for qualitative samples (section 2.2).

2.4 Effects of diel flow variability of macroinvertebrates

To investigate the effect of diel flow variability on benthic biota, quantitative macroinvertebrate samples were collected from edge water (i.e. near wetted perimeter) and thalweg (i.e. near deepest area of river channel) microhabitats at a site in the Ringarooma River upstream of the town of Branxholm ('pulsing' baseflow site, see Results) and at a site in the Dorset River at Dead Horse Hill Road ['stable' baseflow site; 14] during March 2013. Quantitative sampling in these microhabitats followed similar field and laboratory methods that were used for the seasonal quantitative sampling (section 2.3). At both sites, four edge water (depth range = 40-80 mm) surber samples and four thalweg (depth range = 85-330 mm) surbers samples were collected. The taxa in the edge water and thalweg samples were identified following the methods that were used for qualitative samples (section 2.2).

2.5 Data analyses

For all qualitative macroinvertebrate samplings, observed/expected taxa scores ('O/E score') were calculated using the Tasmanian single-season AusRivAS bioassessment models for riffle habitats [12]. Total taxonomic richness ('richness'), percent of taxa belonging to the orders Ephemeroptera, Plecoptera and Trichoptera (%EPT), and percent of taxa considered to be flow obligates ('%Obligates'), facultative users of flow ('%Facultatives') and flow avoiders ('%Avoiders') [15, 16] were also calculated using these qualitative data. Except for O/E scores, the same metrics were also calculated for seasonal and edge/thalweg quantitative macroinvertebrate samples that were collected in 2012/13, along with total density ('density').

According to land use and water allocation data upstream of each seasonal study site in the upper Ringarooma catchment (n = 19), the sites formed three distinct groups in relation to agricultural land use and water use (hereafter referred to as site 'treatments'): (1) low agricultural land use and low water use (LowAg), (2) high agricultural land use and low water use (HighAg), and (3) high agricultural land use and high water use (HighAg+HighWa) [see 14]. These treatments provide a logical and robust means of examining responses of macroinvertebrate communities to the effects of agricultural land use and consumptive water use, and were used in subsequent analyses.

Linear regression was also used to assess temporal trends in seasonal (spring and autumn) metrics that were derived from qualitative macroinvertebrate samples at long-term monitoring sites in the upper Ringarooma catchment. Differences between macroinvertebrate metrics in site treatments were explored using linear mixed-effects models due to unbalanced sample sizes within treatments [17]. Natural log or arcsin transformations of metrics were used where appropriate to satisfy model assumptions. For all models, season and treatment were fixed effects, and sites were treated as random effects. Fixed effects were tested using likelihood ratio (LR) tests derived from comparing models with and without a season × treatment interaction (estimated using maximum likelihood (ML) methods) after their relative support had been assessed using Akaike's Information Criterion (AIC) values from generalised least squares fits (GLS) based on restricted ML (REML). Once the minimum adequate model was found, it was re-fitted with REML to produce unbiased parameter estimates [17]. All models were fitted using the nlme package (v3.1-109) [18] in R 3.0.1 [19]. Two-way fixed-effect ANOVA was conducted on macroinvertebrate metrics (log- or arcsin-transformed where necessary), to identify differences between baseflow types and microhabitats.

3 RESULTS

3.1 Temporal trends in macroinvertebrate community composition

According to long-term qualitative macroinvertebrate sampling the condition of the upper Ringarooma River declined markedly between 1994 and 2013. Significant linear trends (R^2 range = 0.49-0.86, all P <0.01) were evident in several metrics relating to the condition and composition of macroinvertebrate communities at the long-term monitoring site in the river upstream of Branxholm (Figure 1). Typically 16-25 families of macroinvertebrates were recorded during samplings and no temporal trends were evident in richness. However, negative trends were evident in O/E scores for spring, %Facultatives for autumn (not shown), %EPT taxa and %Obligates taxa in spring and autumn, and %Avoiders taxa had positive linear trends in spring and autumn (Figure 1).

Similar temporal trends were also observed at other infrequently monitored sites in the upper catchment that have substantial levels of upstream agricultural land use and water use. Conversely, over the same period the composition of macroinvertebrate communities at reference sites in the Ringarooma catchment was relatively stable.

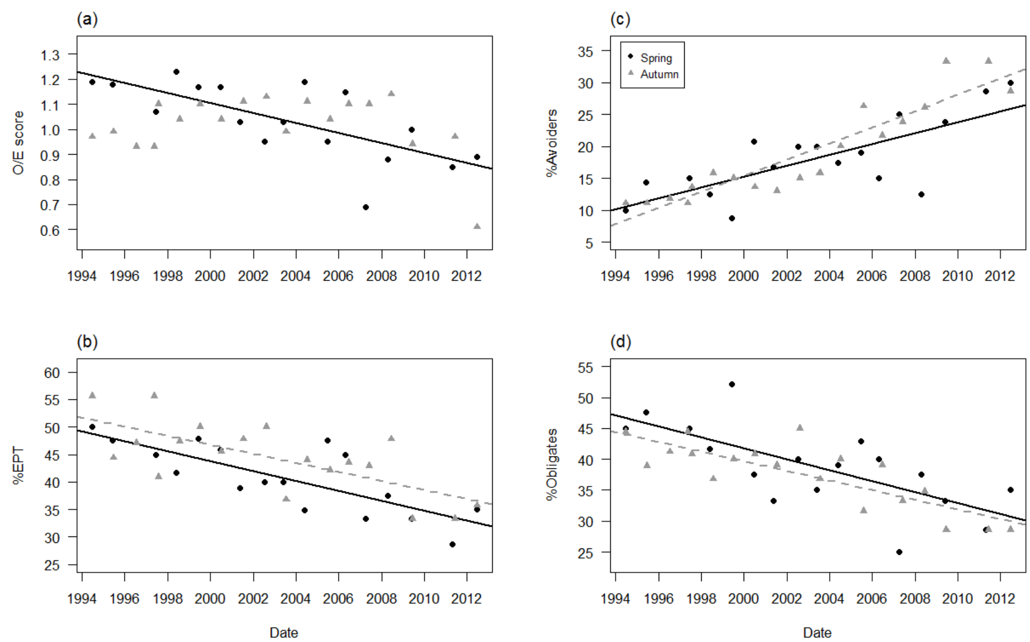

Figure 1. Temporal trends in (a) O/E score, and (b) percent EPT, (c) flow avoiding and (d) obligate macroinvertebrate taxa in riffle habitat in Ringarooma River upstream of Branxholm between 1994 and 2013. Significant linear models (R^2 range = 0.49-0.86, P <0.01) for temporal trends in spring (solid line) and autumn (dashed line) data are shown.

3.2 Spatial patterns in macroinvertebrate communities, and the influence of land and water use

Richness and density in a given site treatment were significantly affected by season (treatment × season interactions: LR = 8.082, P = 0.018; LR = 7.057, P = 0.029; respectively) (see Figures 2a, b,). For both metrics, this appeared to be associated with greater values in spring at the HighAg+HighWa sites compared to autumn (richness: t_{16} = 2.901, P = 0.010; density: t_{16} = 2.676, P = 0.017). Furthermore, across seasons richness and density were significantly lower at HighAg+HighWa sites compared with LowAg sites (t_{16} = 4.267, P <0.001; t_{16} = 2.420, P = 0.028; respectively).

There was a significant difference between seasons in %Obligates (t_{18} = 2.838, P <0.05; Figure 2f), and fewer taxa were flow obligates at HighAg (t_{16} = 2.208, P = 0.042) and HighAg+HighWa (t_{16} = 2.085, P = 0.054) sites compared with LowAg sites. %EPT, % Facultatives and %Avoiders did not differ significantly between seasons (Figures 2c, d, e). However, compared to LowAg sites %Facultatives was lower at HighAg+HighWa (t_{16} = 2.762, P = 0.014) sites, and %EPT was lower at HighAg (t_{16} = 2.179, P = 0.05) and HighAg+HighWa (t_{16} =

3.293, P <0.01) sites. Furthermore, avoiders accounted for higher proportions of the taxa in HighAg (t_{16} = 2.660, P <0.05) and HighAg+HighWa (t_{16} = 3.454, P <0.01) sites compared with LowAg sites (Figures 2c, e).

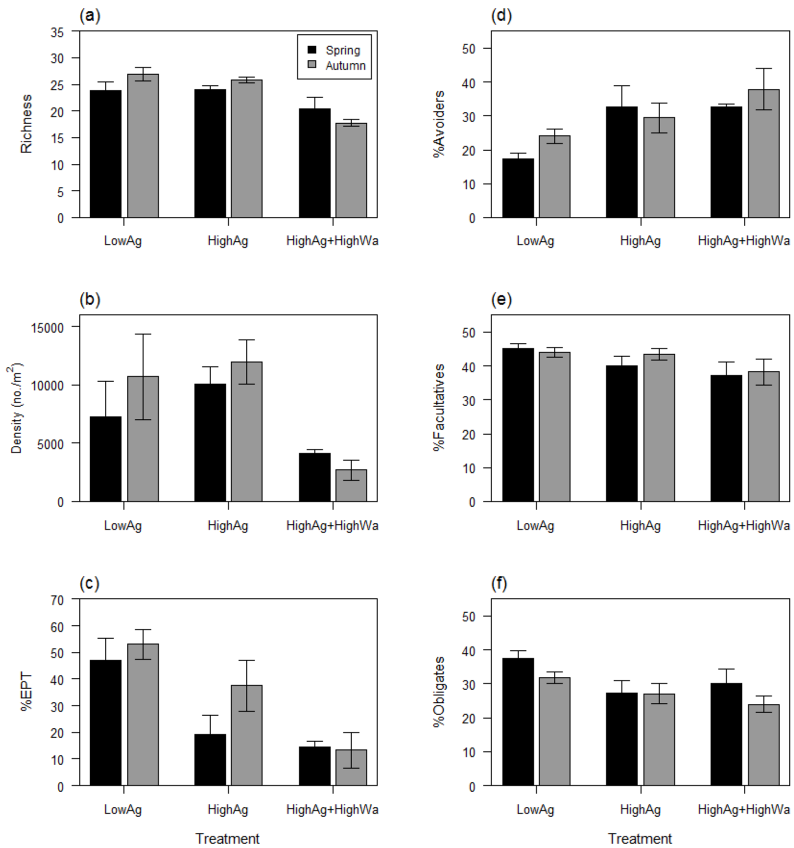

Figure 2. Mean (± s.e.) of macroinvertebrate metrics for site treatments based on sampling during November (spring) 2012 and March (autumn) 2013. The following variables are shown: (a) taxonomic richness (b) density, and percent (c) EPT taxa, (d) flow avoiding taxa, (e) flow facultative taxa, and (f) flow obligate taxa.

3.3 Diel flow variability and its effects on macroinvertebrate communities

Baseflows in the upper Ringarooma River had substantial diel (i.e. within-day) variability during summer-autumn 2012/13, and this was quite pronounced in the river near the town of Branxholm where flow typically varied by 0.35-0.50 m^3 s^{-1} within 24-h periods, which was more than two-fold greater than mean daily flows (Figure 3).

Comparisons between macroinvertebrate communities in edge water and thalweg habitats in upper Ringarooma River (pulsing baseflow) and the Dorset River (stable baseflow) clearly show that pulsing baseflows strongly influence their composition in these habitats (Figure 4). For %Obligates there was a significant baseflow type × habitat interaction ($F_{1,12}$ = 10.960, P <0.01), with on average 15% more flow obligate taxa being found at the stable baseflow site ($F_{1,12}$ = 16.582, P <0.01) and 11% more taxa being associated with thalweg habitats ($F_{1,12}$ = 8.603, P <0.05) (Figure 4f). At the site with the pulsing baseflow there was markedly fewer flow obligate taxa in edge (mean (± SE) = 3.6 ± 3.6%) compared with thalweg habitats (26.3 ± 6.0%) (P <0.01), but there was no difference between these habitats at the site with stable baseflow (P = 0.993).

There were also significant differences between baseflow types and habitats for richness ($F_{1,12}$ = 176.333, P <0.0001; $F_{1,12}$ = 8.333, P <0.05, respectively), density ($F_{1,12}$ = 39.840, P <0.0001; $F_{1,12}$ = 5.415, P <0.05, respectively) and %EPT ($F_{1,12}$ = 218.176, P <0.0001; $F_{1,12}$ = 19.396, P <0.001, respectively) (Figure 4a, b, c). According to these metrics, compared with the site with the stable baseflow, on average the macroinvertebrate community at the site with the pulsing baseflow comprised 14 fewer taxa, had *c.* 50% less EPT taxa and contained *c.* 5000 fewer individuals per m^2. At the site with the pulsing baseflow, compared to thalweg habitat the community in edge water habitat was less taxonomically rich and contained a smaller proportion of EPT taxa (both P <0.05), and appeared to contain fewer individuals per m^2 (weak difference: P = 0.07). Conversely, richness, density and %EPT were similar in edge and thalweg habitats at the site with the stable baseflow.

There was no significant difference between habitats for %Avoiders, but there were on average 14% more flow avoiding taxa at the site with the pulsing baseflow ($F_{1,12}$ = 12.005, P <0.01) (Figure 4d). As expected, %Facultatives (i.e. taxa for which flow is largely unimportant) were similar across the baseflow regimes and habitats (Figure 4e).

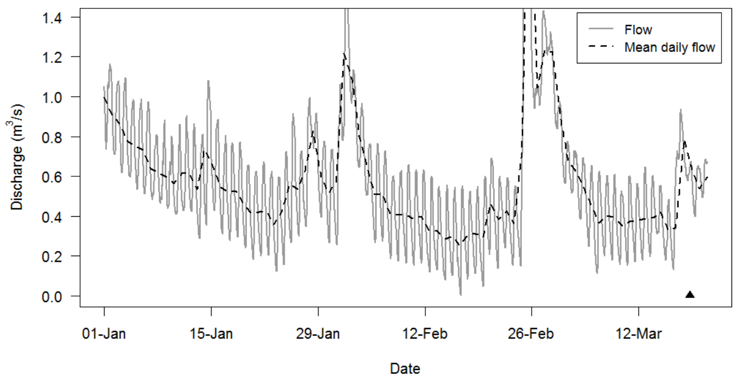

Figure 3. Instantaneous and mean daily flow in the Ringarooma River at Branxholm between 1 January and 20 March 2013. The timing of macroinvertebrate sampling in edge water and thalweg habitat in the river at this site is indicated (solid triangle; 16:30-17:15 h, 18 March 2013). To illustrate diel flow variability only flows ≤1.4 $m^3 s^{-1}$ are shown.

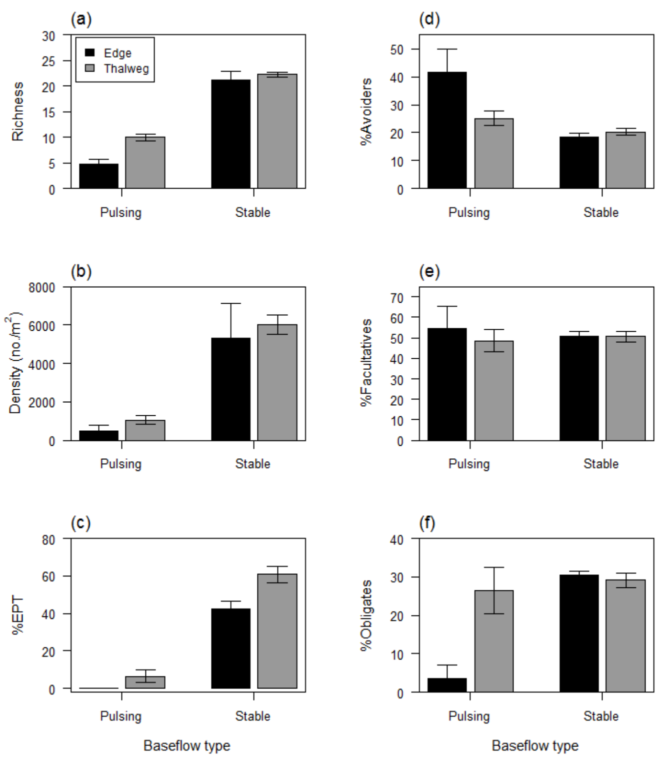

Figure 4. Mean (± s.e.) of macroinvertebrate metrics for samples from edge water and thalweg habitats in reaches with pulsing and stable baseflows, March 2013. A reach in the Ringarooma River upstream of Branxholm and a reach in the Dorset River at Dead Horse Hill Road were used as pulsing and stable baseflow sites, respectively. The following metrics are shown: (a) taxonomic richness (b) density, and percent (c) EPT taxa, (d) flow avoiding taxa, (e) flow facultative taxa, and (f) flow obligate taxa.

4 DISCUSSION

Temporal and spatial declines in metrics relating to ecological condition (e.g. %EPT taxa) were evident in rivers in the upper Ringarooma River catchment. These findings are supported by previous studies of the impacts of agricultural land use [e.g. 20, 21] and water use [e.g. 3, 8] on riverine ecosystems. In Tasmania, Magierowski *et al.*, [22] found strong negative relationships between land use (especially stock grazing) and macroinvertebrate community structure in rivers across state. The present study applied a similar approach to Magierowski *et al.* [22] in the Ringarooma catchment and found that while agricultural land use has adverse effects on macroinvertebrate communities, stronger negative impacts were related to the combined effects of agricultural land use and water use. These impacts were most evident in autumn following a prolonged period of low flows in the catchment during which flows in the Ringarooma River at Branxholm basically ceased (minimum of 0.003 $m^3 s^{-1}$ in mid-February 2013); which is highly unnatural for this strongly perennial river. These findings reflect the anthropogenic modification of the landscape in the upper Ringarooma catchment and the substantial proportions of summer-autumn flows that are allocated in some stretches of the river system.

The long-term monitoring site at in the Ringarooma River at Branxholm is strategically located downstream of some of the more intensively-farmed areas of the upper catchment [14], where the effects of land and water use on the Ringarooma River are likely to manifest. The magnitude of the changes to the structure of the macroinvertebrate communities at this site over the 19-year monitoring period are quite substantial, with *c*. 15% reductions in the proportions of EPT and flow obligate taxa, and a 20% increase in taxa which are flow avoiders.

During December-April periods, when baseflows are <*c*. 1.4 $m^3 s^{-1}$ in the Ringarooma River a strong, regular, diel pulsing flow pattern is evident in the upper Ringarooma River, and this pulsing appears to be strongly associated with water extraction from the river system (DPIPWE, unpublished data). This pulsing baseflow regime is hydrologically similar to ‘hydropeaking’ below hydro-electric dams [23], and appears to have comparable flow-related impacts. The diel flow pulsing was found to significantly impact macroinvertebrate communities in the upper Ringarooma River, with littoral substrata that are dewatered on a daily basis being uninhabitable for many taxa. The diel flow pulsing appears to reduce the quantity and quality (e.g. via dewatering and siltation of substrate) of instream habitat, and we contend that it has contributed to the overall decline in the condition of the upper Ringarooma River.

These findings support need to implement the Ringarooma WMP in accordance with its environmental flow thresholds, allocation limits and adaptive management provisions [11], and raise several issues which water managers are addressing during the plan’s implementation. Firstly, it is imperative that an acceptable level of impact from water use in this river system is defined while the WMP is being phased into practice, as this will provide a basis from which to make decisions regarding water management. Secondly, the implementation of the WMP will recognise the temporal decline in the condition of the upper Ringarooma River system, and seek to lessen impacts from water use, especially during summer-autumn periods. Thirdly, flow management strategies are focusing on dampening the diel flow pulsing that occurs during low flow periods to mitigate its ecological impacts by using instantaneous flow measurements (instead of mean daily flows) to manage water extraction from the river system and water managers and water users are working together to trial alternative water extraction patterns. Finally, to inform an adaptive management process, further ecological surveillance monitoring and targeted research to test the effectiveness of management strategies is being conducted during the implementation of the Ringarooma WMP.

5 ACKNOWLEDGMENTS

We thank Shivaraj Gurung (DPIPWE) for assistance with spatial data analyses, David Nicholls (DPIPWE) for contextual information about water planning in the Ringarooma catchment, and Regina Magierowski (University of Tasmania) and Leon Barmuta (University of Tasmania) for advice and assistance with data analyses. Tom Krasnicki (DPIPWE) and Marty Jack (DPIPWE) provided support with field and laboratory work.

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