Teleosts, agnathans and macroinvertebrates as bioindicators of ecological health in a south-western Australian river

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Abstract

Using standardised techniques to assess bioindicator groups from different food-web levels, the study aimed to provide an ecological baseline upon which monitoring of future changes in the aquatic ecological condition of the Warren River could occur. It also aimed to provide a review of the endemic freshwater fishes of south-western Australia in terms of their appropriateness as indicators of secondary salinisation. Fish and macroinvertebrate communities of the Warren River are described from historical and unpublished data and from sampling during 2006. The Warren River was found to be an important system in terms of freshwater fish conservation housing six of the eight endemic freshwater fishes of south-western Australia and is an important breeding and nursery ground for the Pouched Lamprey (*Geotria australis*). Many of these fishes are potential bioindicators of ecological impacts of salinisation of this and other rivers of the region. The relatively low SIGNAL 2 macroinvertebrate scores at the reference sites suggested that the communities in the main channel of the Warren River were generally those classified as tolerant of pollution. However, a small increase in the pollution sensitive taxa was recorded moving downstream with the most upstream (salinised) site (in the cleared section of the catchment) having the lowest composite and mean SIGNAL 2 score. The use of bioindicator groups from multiple levels of the aquatic food web should be more widely used in assessing and monitoring riverine health in south-western Australia; particularly with regards to the impact of secondary salinisation.

Keywords: Endemic teleosts; Pouched Lamprey; macroinvertebrate communities; aquatic bioindicators; salinisation

Introduction

Aquatic communities are commonly used as indicators of overall catchment condition (Fausch *et al.* 1990) specifically to determine and monitor biological integrity and overall aquatic ecosystem health and overcome the limitations of relying on physico-chemical indicators. These limitations include; inappropriate parameters being measured, synergistic interactions of contaminants being ignored, and spot-measurements not taking into account temporal variability in water quality (Karr & Dudley 1981; Barton & Metzeling 2004). Biological integrity is also more completely assessed by examining multiple levels of ecosystem integration (Simon 1999). Fish community assessment is commonly an important component of studies that aim to assess water quality, biological integrity and ecosystem health and their structure and function can integrate both the direct and indirect stresses on aquatic ecosystems (Simon 1999). The use of fish as indicators of the ecological impacts of secondary salinisation is particularly relevant as salinisation due to anthropogenic driven hydrological changes has been shown to structure fish assemblages (Higgins & Wilde 2005) and has been shown to cause considerable loss of freshwater fish populations (*e.g.*, Morgan *et al.* 2003; Hoagstrom 2009). Utilising both fish and macroinvertebrate communities as bioindicators is an effective way of incorporating multiple levels of food-webs in assessing and monitoring aquatic ecosystem health (*e.g.*, Harris 1995; Chessman 2003; Barton & Metzeling 2004).

Due to secondary salinisation associated with wide-scale clearance of vegetation for semi-intensive agriculture, only ~44% of flow in the largest 30 rivers in the south-west of Western Australia is fresh; the remainder being brackish or saline (Mayer *et al.* 2005). Trends in stream salinity and salt loads of rivers of the region are variable; with those catchments originating in cleared, lower rainfall areas (<1000 mm) usually having fresher flow in their lower reaches (due to freshwater inputs from forested streams and groundwater) compared to their headwaters (Mayer *et al.* 2005).

Secondary salinisation of aquatic ecosystems in south-western Australia due to human activities is known to have altered aquatic ecosystems by, for example altering vegetation and macroinvertebrate communities (Davis *et al.* 2003; Nielson *et al.* 2003; Boulton *et al.* 2007). South-western Australia has the highest proportion of endemic fishes and crayfishes of any Australian Division with 80 and 100% endemicity of these groups, respectively

(Morgan *et al.* 1998; Crandall *et al.* 1999; Allen *et al.* 2002). Within this region, salinisation of previously fresh habitats has also altered the fish assemblages, with salt-tolerant species becoming more prolific and non-salt-tolerant species often becoming restricted to fresher tributaries (Morgan *et al*. 2000, 2003). For example, secondary salinisation within the Blackwood River has resulted in much of the main channel becoming dominated by typically estuarine fishes such as the Western Hardyhead (*Leptatherina wallacei*) and Swan River Goby (*Pseudogobius olorum*), and also by the introduced Eastern Mosquitofish (*Gambusia holbrooki*) (Morgan *et al*. 2003; Beatty *et al.* 2006). Some of the freshwater fishes have been lost from most of the catchment and are now restricted to habitats that remain fresh, particularly those that receive groundwater supplements during dry periods (*e.g.*, Morgan *et al*. 2003; Beatty *et al.* 2006). Despite the known reduction in geographical ranges of many of these fishes due to decline in habitat and water quality, an initial assessment of their individual appropriateness as bioindicators has not previously been carried out.

The Warren River, in south-western Western Australia, has a catchment area of 4350 km^2 and a mean annual flow of ~400000 ML (Pen 1997). It is estimated that between 24 and 35% of the catchment has been cleared which has resulted in this once fresh system becoming increasingly salinised (Hodgkin & Clark 1989; Pen 1997; Smith *et al*. 2006). For example, salinities in the system exceeded 500 mgL^{-1} in the 1960s with average salinities in the 1990s reaching almost 900 mgL^{-1} (Collins & Barrett 1980; Smith *et al*. 2006). Although the Blackwood catchment is considerably more salinised than the Warren River, increasing salinisation of the Warren River and others in the region may result in similar changes to the prevailing aquatic fauna (Morgan *et al*. 2003; Beatty *et al*. 2006).

A desktop review of the significance of the aquatic fauna of the Warren River bioregion, and the adequacy of the reserve system in protecting it, was undertaken by Trayler *et al.* (1996). It highlighted the uniqueness of the aquatic fauna and concluded that the reserve system was important as it housed 86% of the total aquatic fauna of the bioregion. However, the study also stated that there were a number of threatening processes to those fauna that the reserve system alone could not negate. Threats to fauna within reserves (also ubiquitous throughout south-western Australia) included salinisation, land clearing, fire, erosion and deposition of sediment, in-stream barriers (*i.e.*, presence of dams) and the introduction of exotic fish (Trayler *et al.* 1996). Given these identified threats and uniqueness of the fauna in the catchment, there is a need to gather specific baseline information on aquatic bioindicator groups from multiple trophic levels to allow quantification of future changes in aquatic ecosystem health.

The first aim of this study was to gather baseline information on fishes, freshwater crayfishes and macroinvertebrates within the Warren River catchment from a detailed literature review and from sampling additional sites. The subsequent aim was to provide an assessment of these fauna in terms of their suitability as bioindicators of secondarily salinised habitats in this system and other rivers in south-western Australia.

Methods

Fish communities of the Warren River

In order to provide a comprehensive review of the fish fauna of the Warren River, published and unpublished data were examined. The following published and unpublished studies, together with captures during this study, were used and included data for over 75 sites: Christensen (1982), Pen *et al*. (1988, 1991a, b), Jaensch (1992), Morgan *et al*. (1998, 2002), Beatty & Morgan (2006) and Morgan & Beatty (unpublished data collected during various sampling occasions in the Warren River between 2002 and 2006). Distribution of the various study sites was mapped (using *MapInfo*™, Figure 1) and historical distributions of each species within the freshwaters of the Warren River catchment were determined (Figures 2–6).

During December 2006, five main channel sites on the Warren River were examined to determine the fish and crayfish species compositions. These sites included the junction of the Warren River with Wheatley Coast Rd (Site 5 – 116.2845°E, 34.3757°S), Gloucester Rd (Site 4 – 116.0232°E, 34.5052°S), Barker Rd (Site 3 – 115.8973°E, 34.5222°S), Larkin Rd (Site 2 – 115.9403°E, 34.553°S) and near the mouth of the Warren River (Site 1 – 115.8529°E, 34.6021°S) (Figure 1).

In order to ensure all resident species were recorded at each site, fish were captured using a combination of techniques; including a back-pack electro fisher (*Smith Root Model 12-A*), a 5 m seine net (2 mm mesh width) and fyke nets (two 5 x 0.8 m wings, 1.2 x 0.8 m opening and 5 m long pocket with two funnels all comprised of 2 mm mesh) set over a 24 hour period. Presence and relative abundances of freshwater crayfishes were also obtained using eight box-style freshwater crayfish traps (each baited with poultry layer pellets) set over a 24 hour period at each site. All native fish were measured to the nearest 1 mm total length (TL) and were immediately released. Introduced fishes were similarly measured and subsequently euthanased in an ice slurry and preserved in 100% ethanol. Native freshwater crayfish were measured to the nearest 1 mm orbital carapace length (OCL), sexed and released at the site of capture.

Environmental variables

At each main channel site (sites 1–5, see Figure 1) sampled for fish and macroinvertebrates (see below), three replicate measurements of water temperature, conductivity, pH and dissolved oxygen were taken 0.5 m below the water surface. Discharge and flow velocity profiles were also determined for sites 3–5.

Macroinvertebrate communities

In order to determine the macroinvertebrate communities at each reference site in the Warren River, three invertebrate sweeps each of two minutes duration were undertaken in three conspicuous habitats at each site. These habitats included open water, large woody debris, small woody debris, macroalgae (*Chara* sp.), submerged macrophytes, and emergent riparian vegetation. Each sweep was undertaken using a long-handled net (250 μm) deployed in a zigzag motion from

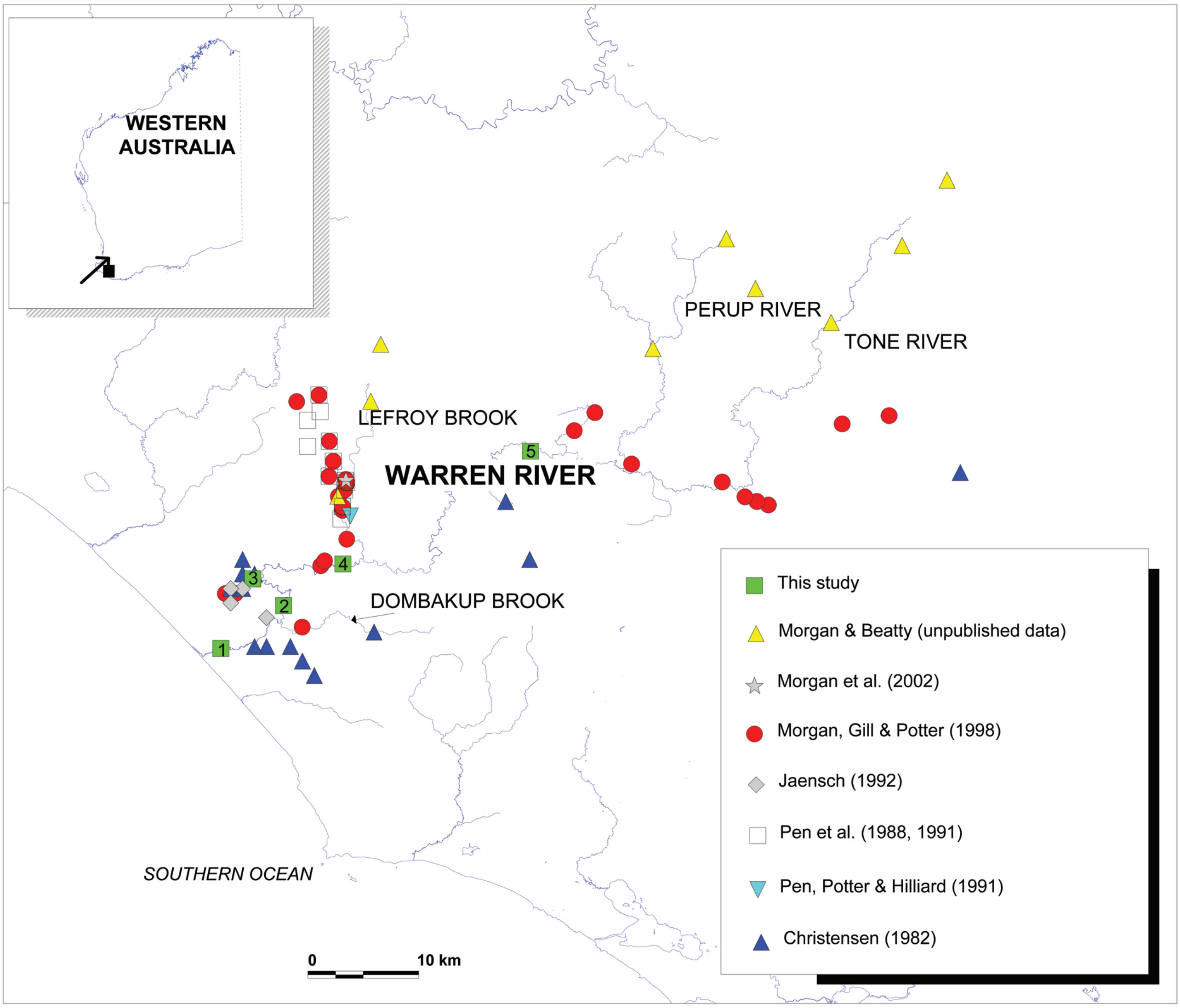


Figure 1. Sample sites from relevant studies utilised to determine the historical distribution of fishes in the Warren River catchment. Sites examined in the study are indicated (1–5).

the benthos to the water surface. The content of each sweep was immediately placed in 100% ethanol.

Invertebrate samples were examined under a dissecting microscope with the majority of taxa identified to family level. Mean relative taxonomic richness (*i.e.*, mean richness per habitat unit) at each site and the total relative taxonomic richness (*i.e.*, total richness across all habitats) of each site were determined. Relative (log-scaled) abundances of each group were recorded.

The macroinvertebrate SIGNAL 2 biotic index, used in the current study, grades invertebrate taxa (1–10) based upon their sensitivity to pollution with the higher the grade the more sensitive the taxon is to pollution (Chessman 2003). By averaging the SIGNAL grades across the taxa present at each site, an overall value is obtained that provides an indication of water quality at the site. Furthermore, SIGNAL 2 scores can also be weighted for abundance of each taxon; not just presence or absence (Chessman 2003). Therefore, it is a commonly used index for assessing aquatic ecosystem health.

The total unweighted SIGNAL 2 score was determined for each site (presence/absence, pooled habitats) as were both the mean unweighted and weighted (for relative abundance) SIGNAL 2 habitat scores at each site. The total unweighted score at each site was the average of the SIGNAL 2 grades of the taxa present at each site (pooled habitats). The mean unweighted score at each site was the mean SIGNAL 2 scores of the three habitats. To determine the mean weighted score of habitats at each site, the sensitivity grade of each taxon collected at each habitat was multiplied by the (log) abundance of that taxon collected, the products then summed for all taxa present in the habitat, and the total divided by the sum of the (log) abundances of all individuals present in the habitat (see Chessman *et al*. 2006). These mean weighted scores of the three habitats were then averaged for each site to give the mean weighted SIGNAL 2 score.

A number of taxa recorded in the Warren River did not have SIGNAL grades assigned (see results), these included; Calanoida and Cyclopoida (copepods), Podocopida (ostracods), Cladocera (water fleas), or Aranae (water spiders). Therefore, these were included in determining the relative taxonomic richness but excluded in determining the above SIGNAL scores at each site.

To further determine the level of differences and similarities in the macroinvertebrate assemblages between habitats and sites, multivariate analysis of invertebrate relative abundance was undertaken using multi-dimensional scaling (MDS) and Analysis of Similarity (ANOSIM) in the PRIMER v5.0 package (Clarke & Gorley 2001). ANOSIM involves the calculation of a test statistic, R, which is a measure of the average rank similarities of replicates within *a priori* designated groups (in this case habitats or sites) compared with the average rank similarity of all replicates among these groups. Thus, an R value of 0 indicates that there are no differences between these groups, whereas a value of 1 indicates that all replicates (invertebrate sweeps) within each habitat or site are more similar to those within those designated groups than they are to any of the replicates from the other groups.

Results

Environmental variables

Conductivity was low (769 – 1694 μS/cm) at all downstream sites when compared to the uppermost site (site 5) where mean conductivity was ~8000 μS/cm. Similarly, discharge was lowest at the most upstream site (~38 l/sec at Wheatley Coast Rd – site 5) and increased substantially at Gloucester Rd and Barker Rd (~1000 l/sec). There were no clear spatial trends in temperature (18.8 – 22.0 °C), pH (7.1 – 7.6) or dissolved oxygen (6.4 – 8.4 ppm).

Fishes of the Warren River

An examination of the published and unpublished literature, together with unpublished data of the authors and sites sampled during this study (see Figure 1) revealed that 13 species of fish and one agnathan were present in the freshwaters of the Warren River. This includes six of the South-west Coast Drainage Division's eight endemic freshwater fishes. The Freshwater Cobbler (*Tandanus bostocki*) was recorded at four of the five sites sampled during the current study (Figure 2). The Western Minnow (*Galaxias occidentalis*) was the most widespread, in terms of catchment area, of any native fishes captured being found in 35 sites (Figure 2). The Nightfish (*Bostockia porosa*) was also widespread throughout the catchment and is currently known from 39 sites and therefore ranked second in terms of site prevalence (Figure 3). The Western Pygmy Perch (*Edelia vittata*) was the most commonly encountered species in the system, being recorded at 45 sites, and is encountered throughout both the main channel and tributaries (Figure 3). The Black-stripe Minnow (*Galaxiella nigrostriata*) was extremely rare within the catchment and has only been reported from Four Mile Brook on Seven Day Rd (Morgan *et al*. 1998) (Figure 4). Importantly, the Western Mud Minnow (*Galaxiella munda*) may now have disappeared from sections of the Warren River (see Morgan *et al*. 2002) (Figure 4).

The Warren River also provides important breeding and nursery grounds for the anadromous Pouched Lamprey (*Geotria australis*) with larvae being recorded in Big Brook, Lefroy Brook and Dombakup Brook (Figure 5). Three estuarine fishes penetrate large distances up the main channel of the river, namely, the Western Hardyhead (*Leptatherina wallacei*), Swan River Goby (*Pseudogobius olorum*) and the South-western Goby (*Afurcagobius suppositus*) (Figure 5). Four species of fish have been introduced into the river, including Brown Trout (*Salmo trutta*), Rainbow Trout (*Oncorhynchus mykiss*), Redfin Perch (*Perca fluviatilis*, common in the system) and Eastern Mosquitofish (*Gambusia holbrooki*, the most widespread introduced fish in the system) (Figure 6).

Macroinvertebrates and decapods of main channel sites

Freshwater Shrimp (*Palaemonetes australis*) were prevalent at all five reference sites sampled in the current study (Figure 1). Smooth Marron (*Cherax cainii*) was recorded at four of the five sites (not being recorded at the most downstream site) and the Gilgie (*Cherax quinquecarinatus*) was recorded at all sites aside from Larkin Rd (Figure 1).

Taxonomic richness across sites ranged from 13 (Barker Rd) to 19 (Wheatley Coast Road crossing) and is consistent with previous studies of macroinvertebrate communities in this region (*e.g.*, Growns & Davis 1994) (Table 1). The most common taxon was the Chironomidae (non-biting midges), found in all habitats at all sites (Table 1). Other common taxa found at all five sites (and at 80% of habitats) were Caenidae (Ephemeroptera, *i.e.*, mayflies), Palaemonidae (Decapoda, freshwater shrimp), Cyclopoida (copepods in 60% of habitats), Leptoceridae, Hydroptilidae and Ecnomidae (tricopterans in 53, 47 and 53% of habitats, respectively) and Culicidae larvae (*i.e.*, mosquito larvae in 33% of habitats) (Table 1).

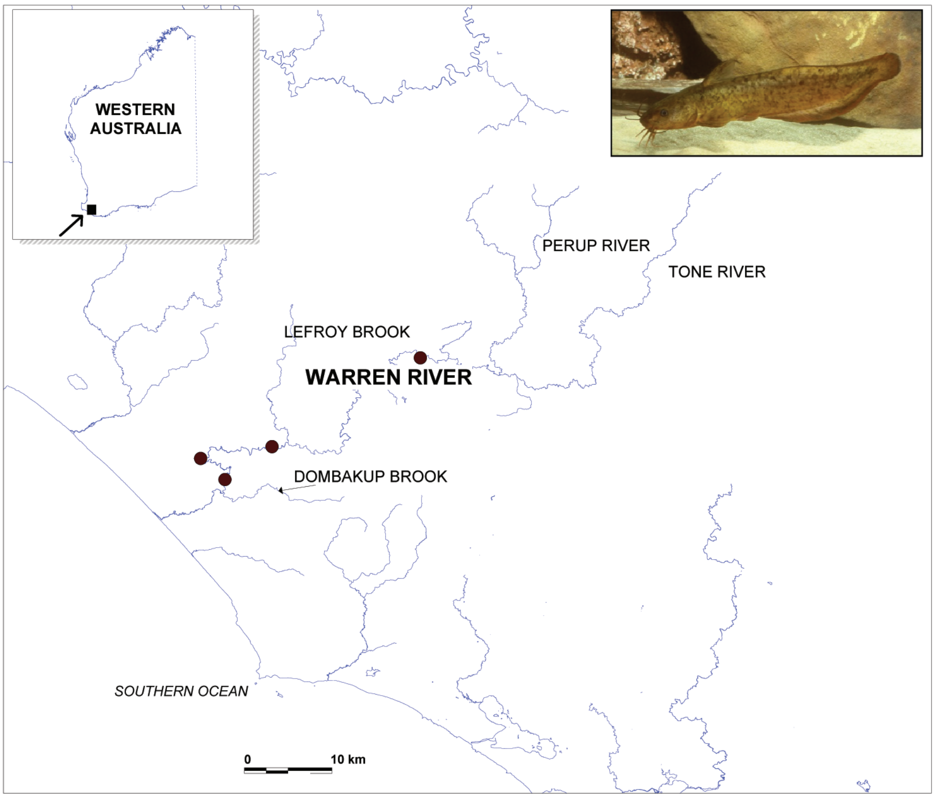
ANOSIM revealed that macroinvertebrate community composition was not significantly influenced by habitat type (Global R = 0.022, p = 0.447). However, from the MDS plot (Figure 7), the macroinvertebrate communities were relatively tightly grouped within habitats at three of the sites (*i.e.*, the mouth, Wheatley Coast Rd and Barker Rd) and ANOSIM revealed that macroinvertebrate community composition was significantly influenced by site (Global R = 0.258, p = 0.031). Pair-wise ANOSIM between sites found that the community composition at the most upstream (Wheatley Coast Rd) and the most downstream (mouth) sites were most distinct from the other sites, although only at a significance level of 0.1. For example the mouth site was different from the Larkin Rd (R = 0.82, p = 0.1), Barker Rd (R = 0.63, p = 0.1), and Wheatley Coast Rd (R = 0.53, p = 0.1), with the Wheatley Coast Rd site also being different from Barker Rd (R = 0.63, p = 0.1).

The relatively low composite macroinvertebrate SIGNAL 2 scores (ranging between 2.9 at the Wheatley Coast Rd site and 3.8 at the Barker Rd site), the mean weighted SIGNAL 2 score (ranging between 3.1 (±0.2) at the Wheatley Coast Rd site and 3.8 (±0.2) at the Barker Rd site) and unweighted SIGNAL 2 score (ranging between 3.1 (±0.2) at the Wheatley Coast Rd site and 4.1 at the Barker Rd (±0.2) and Larkin Bridge (±0.3) sites)

Table 1

Occurrence (Occ) and relative abundance (Log10) of macroinvertebrate taxa present in each habitat at each site in the Warren River in December 2006. Shown are: mean and total taxonomic richness, total unweighted SIGNAL 2 scores, and mean (±1 S.E.) weighted and unweighted SIGNAL 2 scores at each site. N.B. Habitat codes are: Open = open water, LWD = large woody debris, SWD = small woody debris, Algae = submerged *Chara*, Mac = submerged macrophyte, and Riprn = emergent riparian vegetation. No SIGNAL 2 scores are available for Calanoida and Cyclopoida, Podocopida, Cladocera, or Aranae.*=SIGNAL 1 score.

				Wheatley Coast Rd			Gloucester Rd			Barker Rd			Larkin Bridge			Mouth		
ORDER/SUB ORDER	**FAMILY**	**Occ %**	**SIGNAL grade**	**Open**	**Algae**	**SWD**	**Open**	**Riprn**	**Mac**	**Open**	**Algae**	**LWD**	**Open**	**LWD**	**SWD**	**Open**	**LWD**	**Riprn**
Calanoida		**13**		3			2											
Cyclopoida		**60**			4	3	3			3			2		2	3	3	3
Ephemeroptera	Caenidae	**80**	**4**	3	2	3	3	1	2		2	2	1			2	3	3
	Leptophlebiidae	**47**	**8**						1	3	2	1	1	1	3			
Diptera	Stratiomyidae	**7**	**2**			1												
	Culicidae	**33**	**1**			1		2			1			1				1
	Chironomidae	**100**	**1**	3	3	3	3	2	2	3	3	3	3	3	3	4	4	4
	Ceratopogonidae	**33**	**4**	1			1				1	1				3		
Coleoptera	Dytiscidae	**20**	**2**		1			1			1							
	Gyrinidae	**13**	**4**			1									1			
	Hydrophilidae	**7**	**2**													1		
Trichoptera	Leptoceridae	**53**	**6**	1	2	3	3				2		2			2		3
	Hydroptilidae	**47**	**4**					2		3	2	2			2	2		2
	Ecnomidae	**53**	**4**	1		2			2		2		1	1		2	2	
Decapoda	Palaemonidae	**80**	**4**	1	3	3		2	3	1	2	2		1	2		2	1
Isopoda	Amphisopidae	**13**	**1**	1	1													
	Cirolanidae	**13**	**2**											1	3			
Podocopida	Cyprididae	**47**		1	4	3							2	1	1			1
Cladocera	Daphniidae	**27**					1			1						3	2	
	Macrothricidae	**7**											1					
Araneae		**33**						1			1		1			1		1
Acarina (Hydracarina)	Pionidae	**7**	**6**														1	
Mollusca	Hydrobiidae	**13**	**4**		1	1												
	Physidae	**7**	**1**		1													
	Planorbidae	**20**	**2**					1						1				1
Amphipoda	Ceinidae	**13**	**2**		2	3												
	Perthidae	**7**	**4**			2												
Anisoptera		**27**	**3**	1	1								1		1			
Oligochaeta	Megadrile worms	**13**	**2**													3	1	
Mean (and total) richness per site					**11.7 (19)**			**6.3 (15)**			**7.7 (13)**			**9.0 (16)**			**9.7 (16)**	
Total SIGNAL 2 score					**2.9**			**3.6**			**3.8**			**3.6**			**3.3**	
Mean weighted SIGNAL 2 score					**3.1 (0.2)**			**3.3 (0.5)**			**3.8 (0.2)**			**3.4 (0.4)**			**3.2 (0.1)**	
Mean unweighted SIGNAL 2 score					**3.1 (0.2)**			**3.5 (0.6)**			**4.1 (0.2)**			**4.1 (0.3)**			**3.9 (0.3)**	



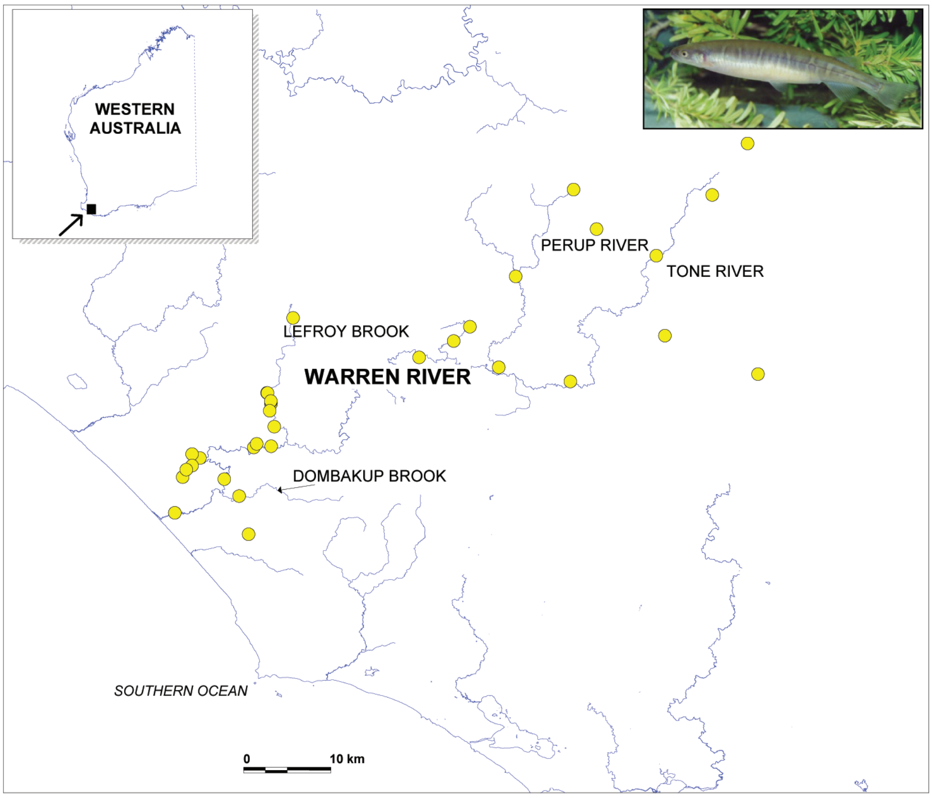
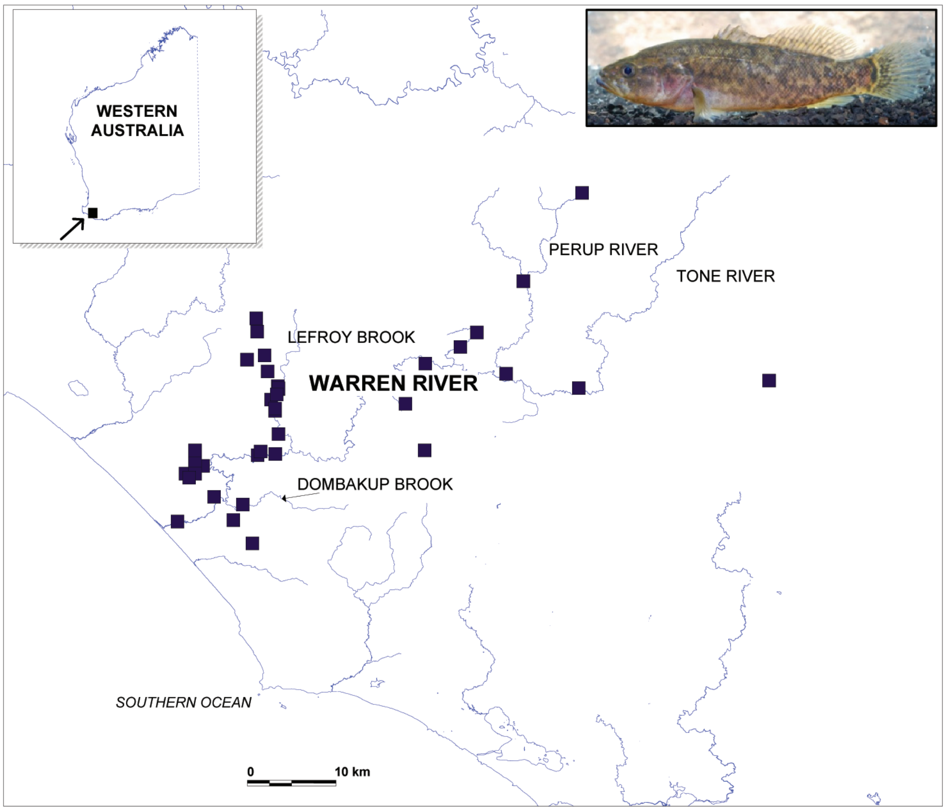


Figure 2. Distributions of the Freshwater Cobbler (top, photo M Allen) and Western Minnow (bottom, photo D Morgan) in the Warren River catchment.



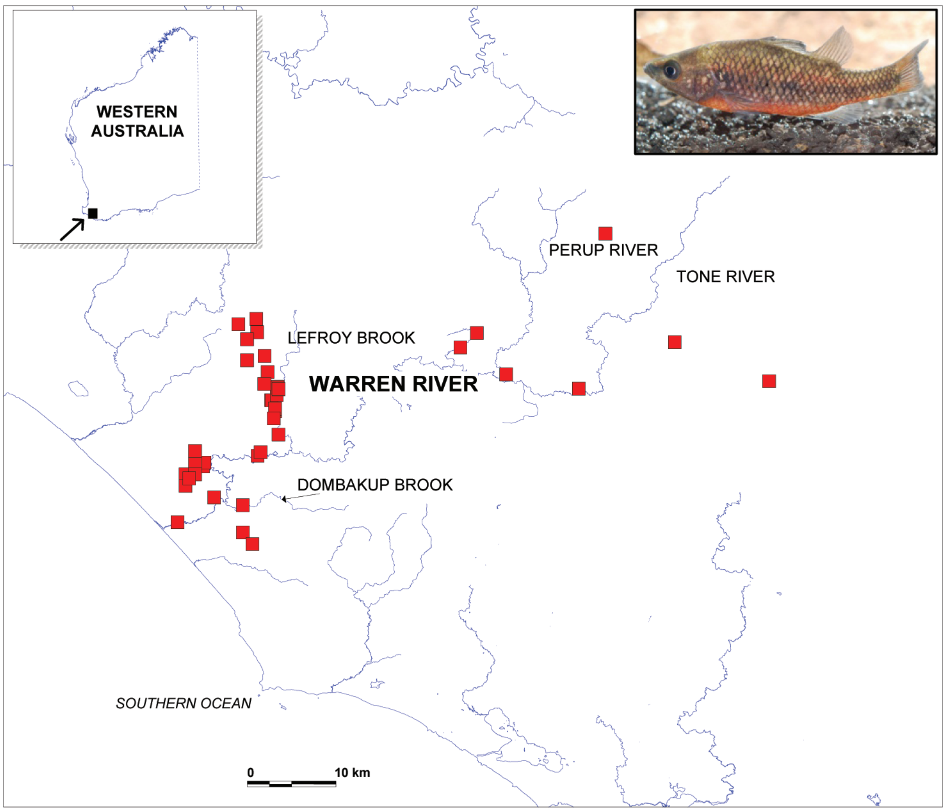


Figure 3. Distributions of the Nightfish (top) and Western Pygmy Perch (bottom) in the Warren River catchment. Photos S Beatty.

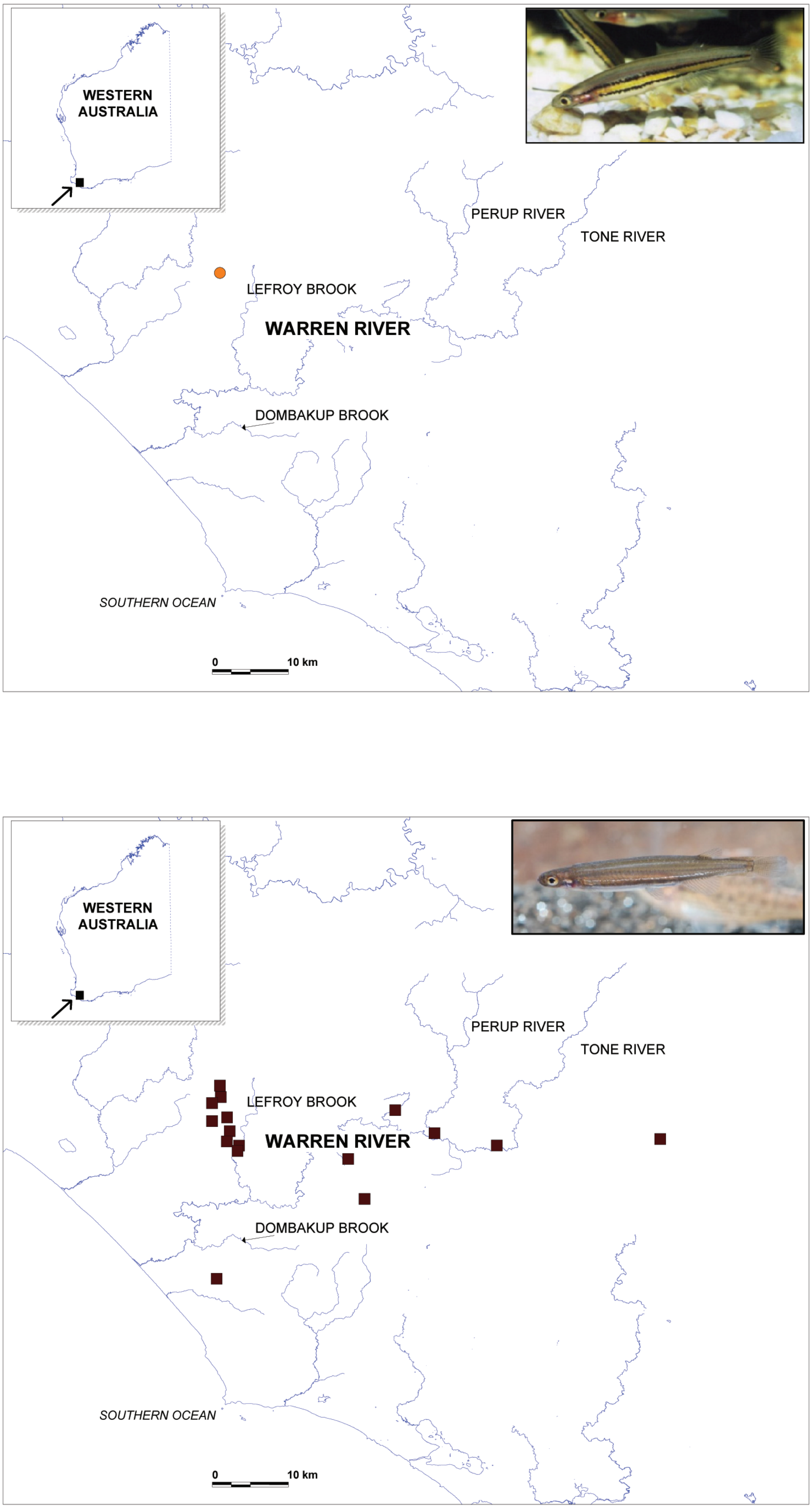
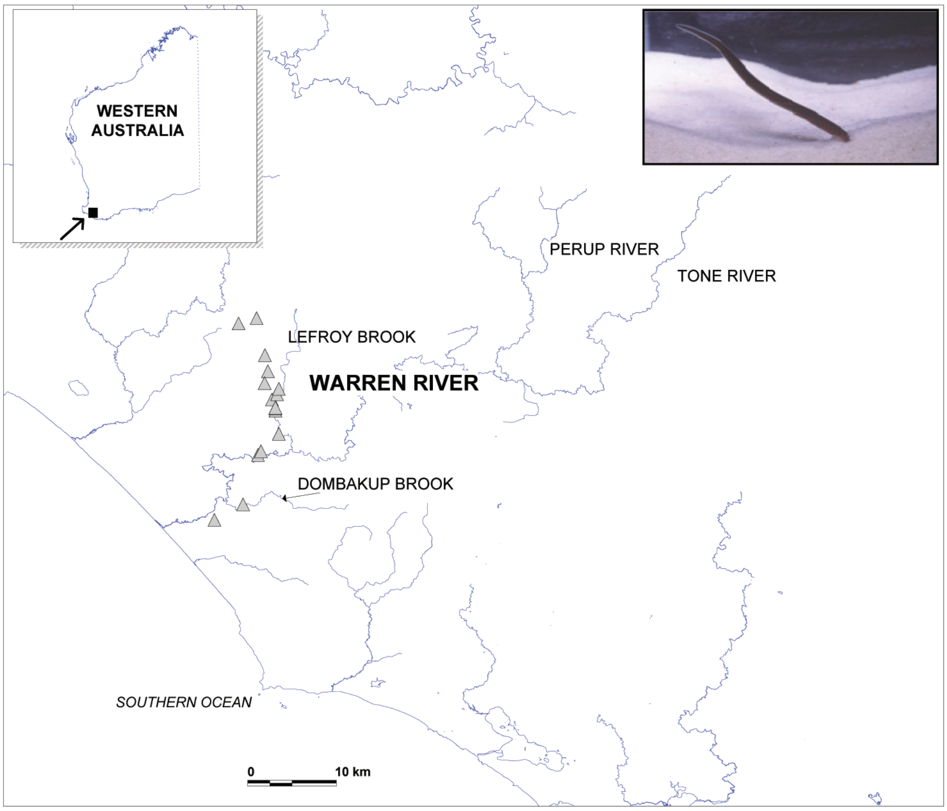


Figure 4. Distributions of the Black-stripe Minnow (top, photo G. Allen) and Mud Minnow (bottom, photo S. Beatty) in the Warren River catchment.



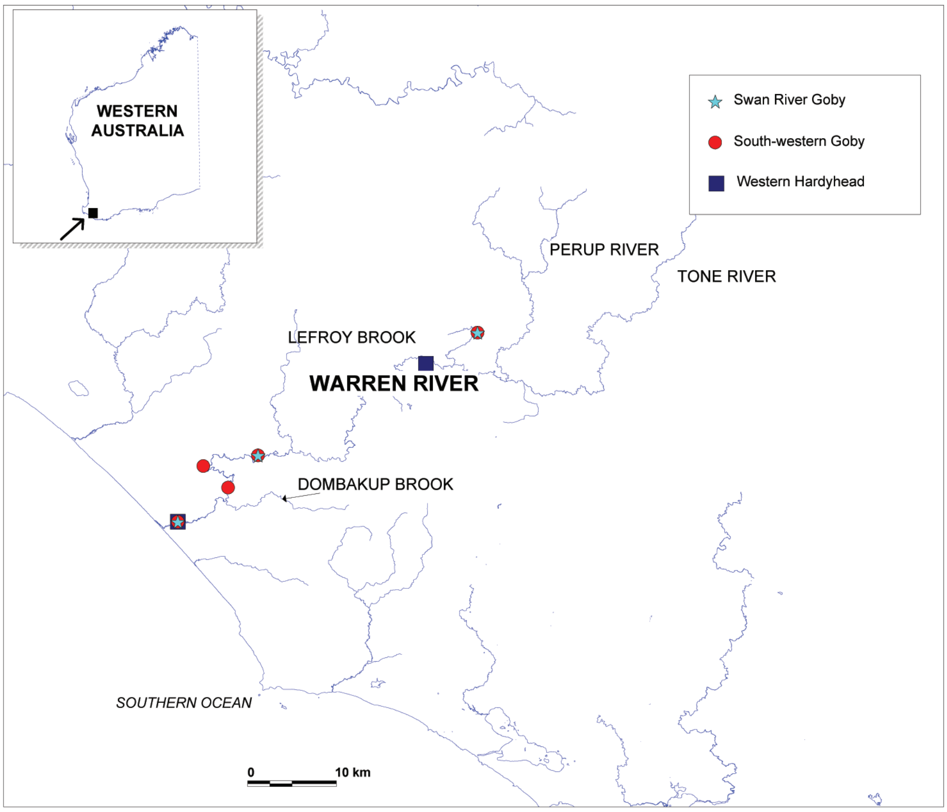


Figure 5. Distributions of the Pouched Lamprey (top, photo D Morgan) and estuarine fishes (bottom) in the Warren River catchment.

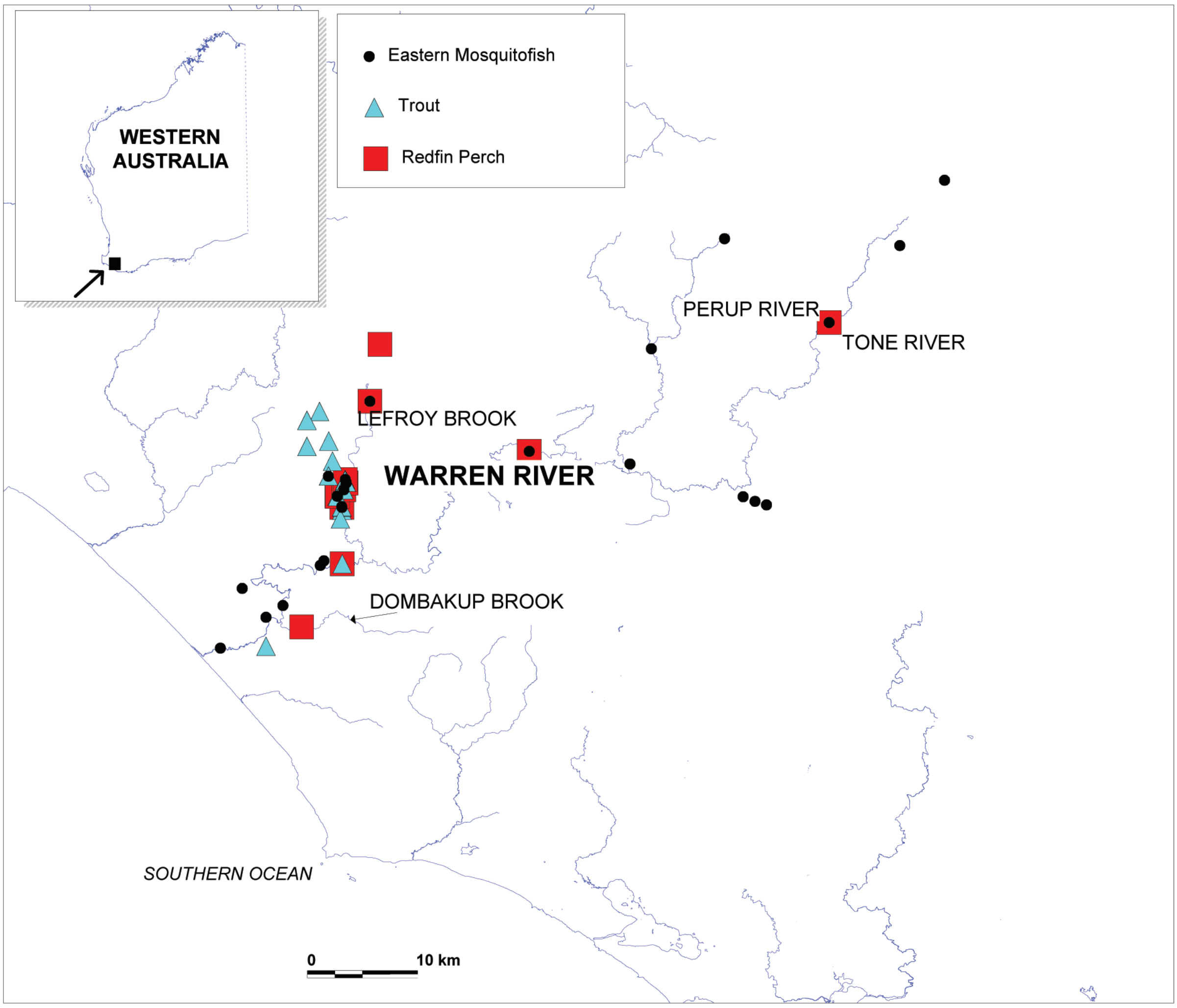


Figure 6. Historical distributions of the introduced fishes in the Warren River catchment.

suggests the taxa present in the main channel of the Warren River were generally tolerant of pollution (Table 1). However, a small increase in pollution sensitive taxa was detected moving downstream with the most upstream site (within cleared farmland) having the lowest composite and mean SIGNAL 2 scores (Table 1).

Discussion

Macroinvertebrates are commonly the key faunal group used as bioindicators to assess and monitor riverine health in south-western Australia (*e.g.*, Bunn & Davies 1992; Growns & Davis 1994; Smith *et al.* 1999; Armstrong *et al.* 2005). This is understandable given this group forms the basis of the predictive models in the widely used Australian River Assessment System (AusRivAS) that was derived from the British River Invertebrate Prediction and Classification System (RIVPACS, Wright *et al.* 1993), and also forms the basis for the SIGNAL 2 biotic index (Chessman 2003); employed in the current study. In south-western Australia, the AusRIVAS models were found to distinguish between undisturbed and severely disturbed sites but did not detect more subtle impacts (Smith *et al.* 1999). However, by examining multiple levels of ecosystems, biological integrity can be more comprehensively assessed (Simon 1999) with fish communities known to indicate direct and indirect stresses on aquatic ecosystems (Simon 1999). Therefore, incorporating both fish and macroinvertebrate community composition is often used in assessing and monitoring aquatic ecosystem health (*e.g.*, Harris 1995; Chessman 2003; Barton & Metzeling 2004; Hering *et al.* 2006; Solomon *et al.* 2006; Wang *et al.* 2007).

However, freshwater fish communities can be highly endemic and, compared to macroinvertebrates, relatively depauperate and therefore require an understanding of environmental tolerance and habitat requirements at the species level for them to be used as bioindicators. The freshwater fishes of south-western Australia are a prime example, as eight of the ten species are endemic; the highest rate of freshwater fish endemism of any drainage division in Australia (Allen *et al.* 2002).

Fishes of the Warren River and south-western Australia: potential as bioindicators

The Warren River was found to be an important system in terms of freshwater fish conservation; housing

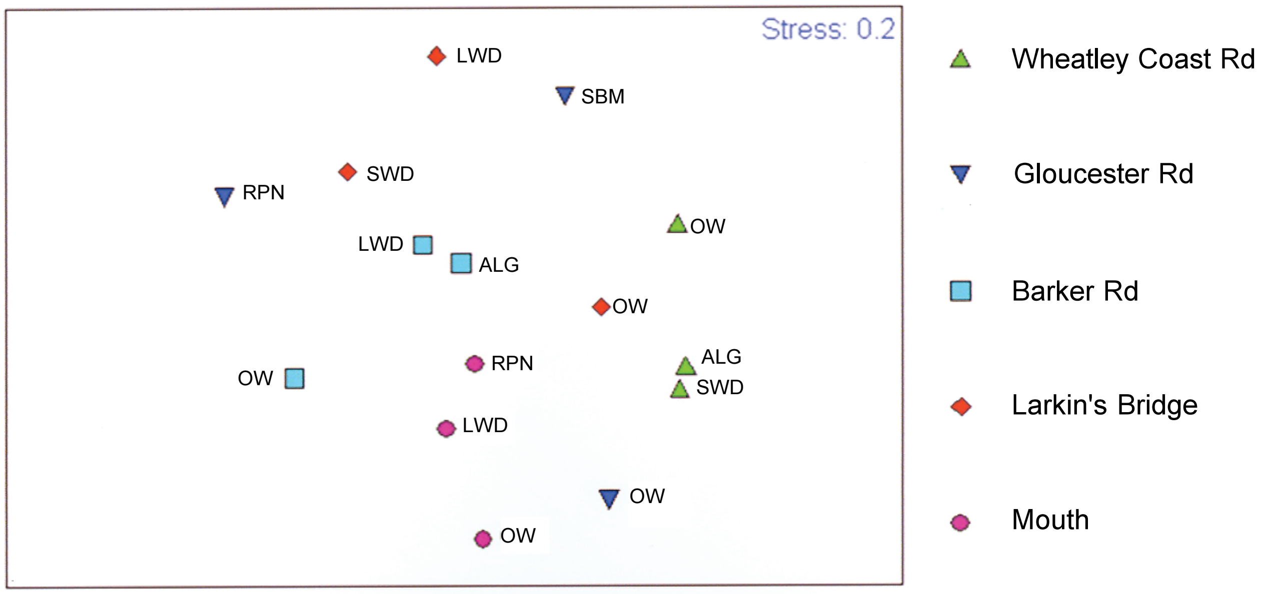


Figure 7. Ordination of the macroinvertebrate communities (using log abundances) at the sites sampled in the Warren River in December 2006. N.B. Habitat codes are: OW = open water, RPN = emergent riparian vegetation, SBM = submerged macrophytes, ALG = submerged macroalgae, SWD = small woody debris, LWD = large woody debris.

six of the eight endemic freshwater fishes of south-western Australia and is one of the important breeding and nursery grounds for the Pouched Lamprey. Sampling in the main channel of the river in the current study found that the reference sites housed four of these species *i.e.*, Western Minnow, Western Pygmy Perch, Nightfish and Freshwater Cobbler. This result is to be expected given that the other two species (*i.e.*, Black-stripe Minnow and Western Mud Minnow) are generally found in smaller tributaries or wetlands (Morgan *et al.* 1998). It is known that a number of the freshwater fishes of the South West Coast Drainage Division of Australia have been lost from inland reaches of major river systems and are now restricted to fresh habitats; particularly those that receive groundwater supplements during dry periods (Morgan *et al.* 1998, 2003; Beatty *et al.* 2006). However, specific gradual and acute salinity tolerances of these endemic species are currently unknown aside from the acute tolerances of the Western Minnow, Western Pygmy Perch and Balston's Pygmy Perch (Beatty *et al.* 2008). Nonetheless, as discussed below, a number of these species has the potential to be complementary to macroinvertebrate taxa in indicating ecological stress; particularly with regard to secondary salinisation.

Prior to this study, there were no documented reports of Freshwater Cobbler from the Warren River (Christensen 1982; Pen *et al.* 1988, 1991a, b; Jaensch 1992; Morgan *et al.* 1998, 2002). The species is usually only encountered within the main channel of river systems (Morgan *et al.* 1998; Beatty *et al.* 2006) and thus its absence from many of the previous studies that concentrated their sampling effort in the tributaries, is not unexpected. This species is thought to be tolerant of at least brackish or slightly saline conditions as it is restricted to the seasonally salinised lower main channel of the Blackwood River (Beatty *et al.* 2006), but is no longer found in the upper reaches of this or other major (many of which are salinised) rivers of the south-west such as the Blackwood River (Morgan *et al.* 2003; Beatty *et al.* 2006). This species, therefore, has the potential to be a useful bioindicator of salinisation in main channels of rivers of this region.

The Western Minnow is known to be relatively salt-tolerant with its acute salinity tolerance recently being recorded at ~14 ppt (Beatty *et al.* 2008); however, it has been recorded in salinities up to 24 ppt (Morgan & Beatty, unpublished data). This tolerance allows it to occupy secondarily salinised habitats where other non-salt tolerant natives have been excluded, as in the Blackwood River (Morgan *et al.* 2003). Therefore, its tolerance of moderate salinisation results in it potentially being a useful bioindicator of more severely salinised river reaches in this region. Its absence from the fresh habitat of Big Brook above Big Brook Dam in the Warren River is probably due to that structure (constructed in 1986) preventing its upstream spawning migration coupled with predation by introduced fishes (see below and Pen *et al.* 1988, 1991b; Morgan *et al.* 2002).

The Western Pygmy Perch has a similar acute salinity tolerance to the Western Minnow, *i.e.*, ~14 ppt (Beatty *et al.* 2008). However, along with the Nightfish, its inland range in south-western Australia has been greatly reduced and elevated salinity probably has other effects on both their life-cycles (*e.g.*, sperm viability, dietary requirements or low salinity tolerance of larvae) that prevent these two species occupying salinised systems. Therefore, both the Western Pygmy Perch and the Nightfish could be appropriate bioindicators of ecological impacts of salinisation in this region.

The less common Black-stripe Minnow is typically restricted to floodplain habitats that are better represented in adjacent catchments along the south-coast of south-western Australia (Morgan *et al.* 1998; Morgan & Beatty 2008). Based on its natural distribution (Morgan *et al.* 1998), it is unlikely that this species is tolerant of elevated salinities. Its specialised habitat requirement (*i.e.*, peat wetlands in the south-west corner of the South West Coast Drainage Division (Morgan *et al.* 1998)), however, renders it not being suitable as a bio-indicator of secondary salinisation within rivers of the south-west.

The Western Mud Minnow was once extremely common and widespread within the Big Brook catchment (see Pen *et al*. 1988, 1991a, b), however, a combination of the construction of the dam and the introduction of non-native fishes into this tributary have been implicated with the loss of this and other endemic fishes in this system (Morgan *et al*. 2002). As with the Black-stripe Minnow, it is unlikely that this species is tolerant of elevated salinities and its loss from the main channel of the Blackwood River has been attributed to secondary salinisation (Beatty *et al.* 2006; Phillips *et al.* 2007). Therefore, although other changes in habitats may impact the species, its absence from saline but otherwise suitable habitat within rivers in the South West Coast Drainage Division results in it potentially being a suitable bioindicator of secondarily salinised habitats.

The Warren River is arguably the most important breeding and nursery ground of the Pouched Lamprey in Western Australia and it contains those habitat types required to support larval (ammocoete) and metamorphosed (juvenile) lampreys, *i.e.*, well oxygenated sandy (for metamorphosed lampreys) and silty (for ammocoetes) substrates (Morgan *et al.* 1998). Therefore, as with the Western Mud Minnow, the Pouched Lamprey has certain microhabitat requirements, however, it is invariably found only in fresh habitats (Morgan & Beatty unpublished data). This species should, therefore, be considered an appropriate bioindicator of secondarily salinised habitats.

The three estuarine species recorded in the Warren River are essentially confined to the main channel and, as is the case with other salinised systems of south-western Australia, they may be relatively recent colonists of the inland reaches of the river (Morgan *et al*. 2003; Beatty *et al*. 2006). The extent of the upstream penetration of these estuarine species is highly likely to be a key bioindicator of ecological impact of salinisation of the region (Morgan *et al.* 2003; Beatty *et al.* 2006).

Introduced fishes of the Warren River

The Warren River is arguably one of the most popular areas for trout anglers in Western Australia and is annually stocked with hatchery reared fish by the Department of Fisheries, Western Australia (Department of Fisheries 2009). However, the degree of natural recruitment to the majority of these trout populations has not been assessed, nor has a detailed assessment of their ecological impact been undertaken in rivers of the south-west region. Interestingly, a recent study by Tay *et al.* (2007), who analysed both gut content and assimilated diet using stable isotope analysis, revealed the iconic Smooth Marron to be the most important (61% by overall volume) dietary component of Rainbow Trout in a water supply reservoir. Brown Trout have previously been recorded as self-sustaining in Big Brook Dam (Warren River catchment) (Morgan *et al.* 2004). Furthermore, self-sustaining populations of Rainbow Trout have recently been recorded in another reservoir (Tay *et al.* 2007) and stream (Morgan *et al.* 2008) in this region; therefore a cessation of the stocking regime alone may not result in the eradication of this species from the south-west.

Redfin Perch, an invasive species known to naturally reproduce in those systems it occupies (Morgan *et al.* 2002, 2004), was also widespread in the Warren River. Unlike the trout species, Redfin Perch has been the focus of a biological and impact study in Big Brook (including Big Brook Dam) (Morgan *et al*. 2002), the first of its kind for the species in Australia. The study recorded a localised loss of endemic fishes in Big Brook Dam that was attributed to its introduction to that system (Morgan *et al.* 2002). The species has an upper salinity tolerance of 10 ppt (Privolnev 1970) and therefore it can also potentially be used as an indicator of salinisation in those rivers in which it has colonised.

The Eastern Mosquitofish is another aggressive invasive species that is well known for fin-nipping native fishes (Gill *et al*. 1999). It is also very tolerant of saline conditions (up to at least ~58 ppt) being commonly found in the upper, cleared farmland reaches of river systems in the south-west and therefore although an indicator of degraded aquatic habitats in this region (see Morgan *et al.* 2003, 2004), is not a suitable indicator of salinisation.

The most upstream site (5) (Wheatley Coast Rd) had the highest movements (and captures) of the introduced Redfin Perch and Eastern Mosquitofish. This may indicate that this site was the most disturbed (as also suggested by the macroinvertebrate SIGNAL 2 index, see next section); however, it may also reflect the fact that these species are generally more prevalent in slower flowing environs (Morgan *et al.* 2002, 2004). Additional sampling of more upstream sites would determine whether these introduced species are more prevalent in more degraded habitats.

Aquatic invertebrates of the Warren River

The Smooth Marron was recorded in the main channel at four of the five reference sites sampled in the current study (being absent at the downstream-most site). This species is the third largest freshwater crayfish in the world, is fast growing, and is likely to be important in structuring food webs through consuming multiple levels of foodwebs (Beatty *et al.* 2005; Beatty 2006). The Warren River is a well recognised component of the recreational fishery for this species (Department of Fisheries 2008). Smooth Marron is known to be tolerant of slightly elevated salinities (lethal at ~17 ppt, Morrissy 1978). This species is an appropriate bioindicator for relatively large increases of salinity with its inland range in salinised rivers having been substantially reduced (Morrissy 1978).

Most of the invertebrate taxa recorded in the current study were consistent with those recorded by Trayler *et al*. (1996), who documented the different groups of aquatic fauna of the Warren River bioregion, including fish, frogs and invertebrates. Although noting that it was probably an underestimation (due to potential presence of undescribed species), the latter study found the region contained a relatively high proportion of the south-west endemic macroinvertebrate taxa and 17% of those found were locally endemic (Trayler *et al.* 1996).

The relatively low composite SIGNAL 2 scores at all sites suggest the macroinvertebrates present in the main channel of the Warren River are those generally tolerant of pollution. A steady increase in the salinity of the Warren River has already occurred during the past 50 years (Smith *et al*. 2006). As it is thought that the invertebrate fauna of the bioregion is intolerant of even small increases in salinity (Growns & Davis 1994; Trayler *et al*. 1996), it is likely that the macroinvertebrate communities have already been partially altered to consist of taxa that have a degree of salinity tolerance, or may have become adapted to environmental extremes and are now relatively insensitive to additional human disturbances such as land clearing or sediment loads (see Chessman *et al*. 2006). If the latter were the case, this would limit their usefulness as bioindicators in this system. However, a small increase in the pollution sensitive taxa was recorded at more downstream sites. This suggests that these communities may still be useful for monitoring current and future ecological impacts of water quality change (particularly salinisation) within the Warren River.

The slight increase in the SIGNAL 2 scores recorded here appears to contrast with Chessman (2003) who generally found lower scores further from the source of rivers, and higher scores at higher altitudes. This typical scenario may be the case in pristine upland areas in the Warren bioregion as relictual aquatic fauna are often found in elevated areas above previous zones of marine incursions (such as headwaters, granite outcrops, and permanent spring flows); as they provide moist, organic rich environments (Main & Main 1991; Trayler *et al*. 1996). Chessman (2003) also found that SIGNAL 2 scores were positively correlated with dissolved oxygen and negatively correlated with turbidity, conductivity (*i.e.*, salinity) and nutrients. Therefore, the subtle reverse SIGNAL 2 trend in terms of proximity to the source recorded in the current study may be due to the sites further downstream being in a more natural condition. This may be a result of them being located within the forested reserve system (*i.e.*, State Forest and/or National Park) and having lower salinities as a consequence of groundwater intrusion (Smith *et al*. 2006) compared with more upstream sites. A stronger pattern of decreasing SIGNAL 2 index may be revealed at sites even further upstream in the more saline farmland region. These base-line macroinvertebrate data suggest that on-going (and expanded) monitoring of the macroinvertebrate communities of the Warren River will provide a useful tool for assessing its long-term environmental condition.

Trayler *et al*. (1996) highlighted the importance of the reserve system in the Warren Bioregion stating that 25% of the region was held in National Parks or nature reserves (A-class) at that time; with 86% of the aquatic fauna being represented in reserves. The rivers and stream zone system (introduced in the 1970s and since modified), was stated as being of particular importance to these fauna as it was designed to provide buffer zones for water-ways against logging operations (Trayler *et al*. 1996). These buffer zones are important for preventing water quality decline in these systems as logging activities have been shown to reduce the macroinvertebrate richness. For example, clear-felling in Carey Brook (in the adjacent Donnelly River catchment) resulted in increased sediment loads and an associated decrease in the taxonomic richness and abundance of macroinvertebrates (Growns & Davis 1994).

The macroinvertebrate communities recorded in the current study represent an important ecological component that can be repeatedly monitored to assess ecological condition in the Warren River. It is recommended that additional sampling of fishes and macroinvertebrates occur at these sites and at more degraded upstream areas (*e.g.*, saline areas such as within the Tone River) during both spring and summer with both the SIGNAL 2 index and the AusRivAS assessment being applied to provide a more comprehensive base-line for long-term monitoring of the environmental condition of the river.

Conclusions

The current study has provided an assessment of two key bioindicator communities of the Warren River to allow future changes in its condition to be assessed; particularly in light of proposed management actions to address rising salinisation (see Smith *et al*. 2006). The study has identified a number of fishes as potentially appropriate bioindicators of salinisation of this and other rivers in the region. These species include the Freshwater Cobbler, Western Minnow, Western Pygmy Perch, Nightfish and the Western Mud Minnow. However the specific acute and gradual salinity tolerances of these species should be determined prior to their further incorporation in predictive ecological health models. The relatively low SIGNAL 2 scores at all sites suggested the macroinvertebrate taxa present in the main channel of the Warren River were generally tolerant of pollution, however, a small increase in the pollution sensitive taxa was recorded moving downstream with the upstream-most site having the lowest composite and mean SIGNAL 2 score. The use of multiple bioindicator groups should be more widely incorporated in assessing and monitoring riverine health in this region rather than relying on single levels of aquatic foodwebs as has commonly occurred in the past.

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