EMRC

Helena River

Fish and Macroinvertebrate Surveys 2010 and 2011

WRM

Wetland Research & Management

November 2011

Helena River

Fish and Macroinvertebrate Surveys 2010 and 2011

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Final Report
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Frontispiece: (left to right) Site 5 in November 2010, site 18 in March 2011 and site 24 in March 2011.

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1 BACKGROUND

This report documents baseline surveys for fish and aquatic macroinvertebrates in the Helena River conducted in late spring 2010 and late summer/autumn 2011. The surveys were undertaken by *Wetland Research & Management* on behalf of the Eastern Metropolitan Regional Council (EMRC), with additional financial support from the Swan River Trust, to aid development of strategies to increase aquatic fauna habitat through restoration and protection of the river and the greater Helena catchment. The current report also makes recommendation on the control of introduced species and provides recommendations for on-going monitoring of fish and aquatic macroinvertebrates as indicators of river health.

1.1 Project Aims

The overall aims of the surveys were as follows:

i) To develop an inventory of fish and macroinvertebrate species in the Helena River, including species present, relative abundances and population structure.
ii) Statistically examine and report on relationships between habitat characteristics and aquatic fauna diversity/species distributions;
iii) To recommend strategies to increase fish populations *e.g*. targeted restocking, habitat restoration activities and feral fish control;
iv) To share information with local community and relevant government agencies.

The current report includes a literature review of the distribution, biology and conservation status of each fish and crayfish species recorded from the system.

2 METHODS

2.1 Sampling Sites

In late spring (November) 2010, fish were sampled at 20 remnant pool sites (Figure 1 & Table 1) on the Helena River. In total, the sites encompassed some 55 km of the system between the confluence with the Swan River and headwater reaches above the Darling Scarp. Sampled sites included pools above and below Mundaring Weir and below the Lower Helena Pipehead Dam (PHD). Of these 20 sites, 17 had previously been surveyed by Wansbrough and Stewart (2006) in May 2006 to assess habitat conditions. Four additional sites included 18, 19, 20 and 22. These were previously sampled for fish and macroinvertbrates by WRM in May 2010 for the Department of Water (WRM 2010), to assess the influence of environmental releases from the PHD on the ecological health of the reach immediately downstream. These additional sites corresponded to Swan River Trust (SRT) monitoring sites. Two sites were also sampled on Warrin Creek (sites 8 & 9), an upland tributary of the Helena River above Mundaring Weir (Figure 1).

In late summer/autumn (February/March) 2011, all 20 sites were to be re-sampled for fish, however many were found to be dry, so substitute sites were chosen as close to the original locations as possible (Table 1). A total of 25 sites were sampled in 2011. At the request of the SRT, 10 of these 19 sites were also sampled for macroinvertebrates, in order to provide baseline data additional to that collected by WRM (2010). The ten sites sampled for macroinvertebrates were located above and below the PHD, again to assess the influence of environmental releases on ecological health.

Photographs of all sites are provided at the end of the Appendix.

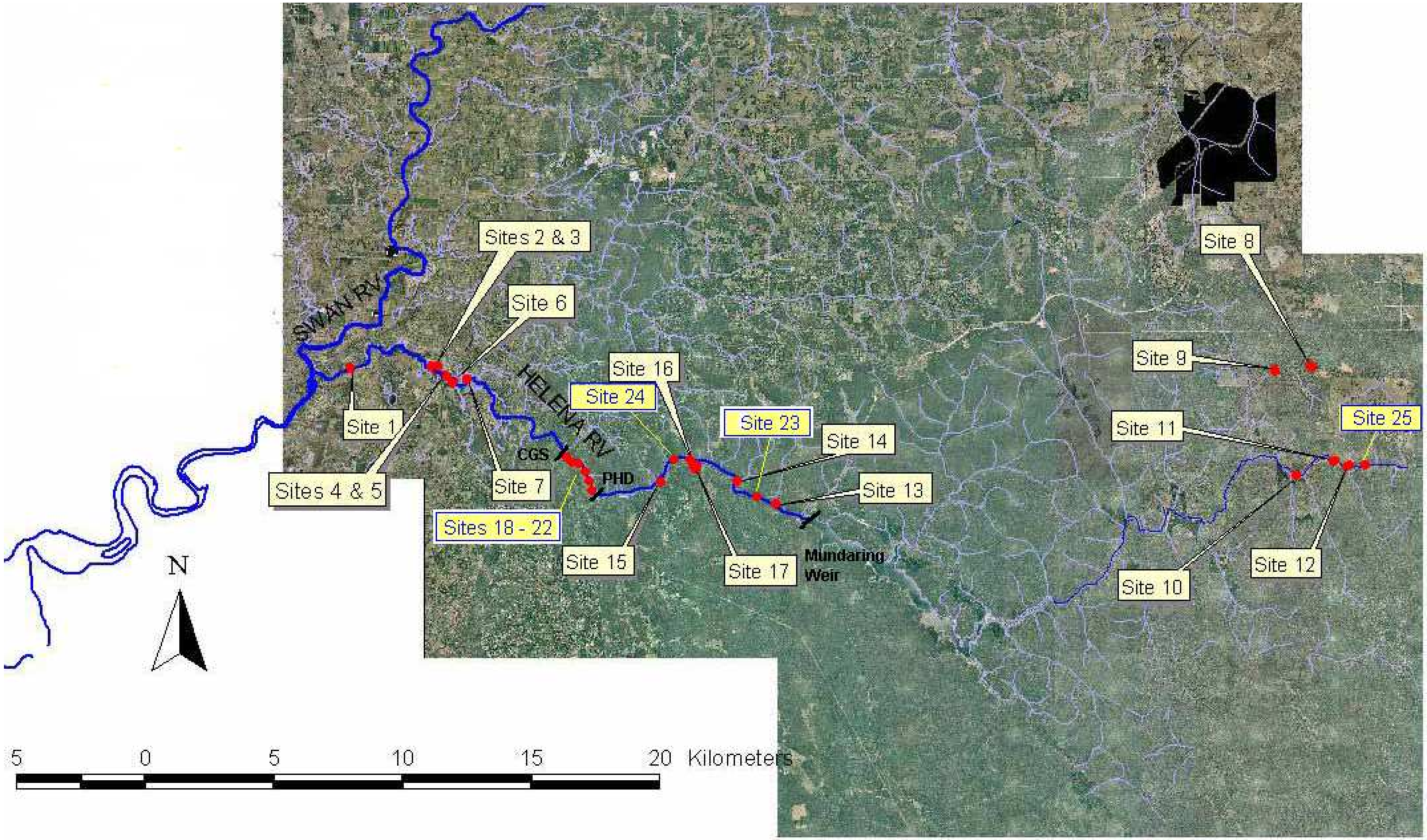

Figure 1. Map showing location of sample sites along the Helena River, Craignish Gauging Station (CGS), the Lower Helena Piphead Dam (PHD) and Mundaring Weir (original map taken from Wansbrough & Stewart 2006).

Table 1. GPS co-ordinates of sites sampled along the Helena River in mid November 2010 and early March 2011. An asterix denotes that the site was sampled for both fish and macroinvertebrates in 2011. Sites at which fish and macroinvertebrates have previously been sampled by WRM (2010) are indicated as well as Swan River Trust (SRT) monitoring sites.

[illegible]	UTM (WGS84, Zone 50)		Sampling dates		Location	Previous surveys
	Easting	Northing	2010	2011		
1	403440	6469830	Nov. 2010	Mar. 2011	Helena Valley floodplain. Within tidal influence of the Swan River. 1.6 km upstream of Swan River confluence.	
2	406680	6469860	Nov. 2010	dry	Helena Valley floodplain. 6.3 km upstream of Swan River confluence.	
3	406740	6469800	Nov. 2010	dry	Helena Valley floodplain. 6.4 km upstream of Swan River confluence.	
4	407030	6469612	Nov. 2010	Mar. 2011	Helena Valley floodplain. 6.7 km upstream of Swan River confluence.	
5	407070	6469623	Nov. 2010	dry	Helena Valley floodplain. 6.8 km upstream of Swan River confluence.	
6	407460	6469390	Nov. 2010	dry	Helena Valley floodplain. 7.3 km upstream of Swan River confluence.	
7	407950	6469590	Nov. 2010	Mar. 2011	Helena Valley floodplain. 8.1 km upstream of Swan River confluence.	
8	440621	6470009	Nov. 2010	Mar. 2011	Warrin Creek, Helena River headwaters above Mundaring Weir. 55 km upstream of Swan River confluence.	
9	440586	6470083	Nov. 2010	dry	Warrin Creek, Helena River headwaters above Mundaring Weir. 53.6 km upstream of Swan River confluence.	
10	440093	6465730	Nov. 2010	dry	State Forest, Helena River headwaters above Mundaring Weir. 52.6 km upstream of Swan River confluence.	
11	441603	6466283	Nov. 2010	Mar. 2011	State Forest, Helena River headwaters above Mundaring Weir. 54.4 km upstream of Swan River confluence.	
12	442070	6466055	Nov. 2010	Mar. 2011	State Forest, Helena River headwaters above Mundaring Weir. 55 km upstream of Swan River confluence.	
*13	419951	6464551	Nov. 2010	Mar. 2011	Bourkes Gully above PHD. 24.9 km upstream of Swan River confluence.	
*14	418387	6465445	Nov. 2010	Mar. 2011	Rifle Range Gully above PHD. 22.9 km upstream of Swan River confluence.	
*15	415455	6465508	Nov. 2010	Mar. 2011	Hardey Road area above PHD. 18.8 km upstream of Swan River confluence.	
*16	416664	6466148	Nov. 2010	Mar. 2011	Hardey Road area above PHD. 20.7 km upstream of Swan River confluence.	
17	416821	6465932	Nov. 2010	Mar. 2011	Nelson Road area above PHD. 21 km upstream of Swan River confluence.	
*18	412554	6466029	Nov. 2010	Mar. 2011	State Forest, below PHD. 12.7 km upstream of Swan River confluence. WRM (2010) Site 5.	SRT monitoring site; WRM (2010)
*19	412383	6466202	Nov. 2010	Mar. 2011	State Forest, below PHD. 12.5 km upstream of Swan River confluence. WRM (2010) Site 7.	SRT monitoring site; WRM (2010)
*20	411964	6466323	Nov. 2010	Mar. 2011	State Forest, below PHD at Craignish Gauging Station. 12.1 km upstream of Swan River confluence. WRM (2010) Site 12.	SRT monitoring site; WRM (2010)
*21	412626	6465833	ns	Mar. 2011	State Forest, immediately below PHD. WRM (2010) Site 4.	SRT monitoring site; WRM (2010)
*22	412468	6466143	ns	Mar. 2011	Downstream of PHD, between sites 18 & 19. WRM (2010) Site 8.	SRT monitoring site; WRM (2010)
*23	419298	6464794	ns	Mar. 2011	Upstream of PHD, between sites 13 & 14.	
24	415722	6466111	ns	Mar. 2011	Upstream of PHD, between sites 15 & 16	SRT monitoring site
25	442351	6466036	ns	Mar. 2011	Above Mundaring Weir, approximately 300 m upstream of site 12.	

2.2 Water Quality and Habitat Variables

In-situ spot measurements of dissolved oxygen (mg/L & % saturation), temperature (^{o}C), electrical conductivity (µS/cm) and pH were made using WTW and TPS field meters at each site on each sampling occasion. Water quality was assessed against ANZECC/ARMCANZ (2000) guidelines for slightly - moderately disturbed freshwater ecosystems in south-west Western Australia. These guidelines specify water quality and sediment guidelines for protecting a range of aquatic ecosystems, from freshwater to marine (ANZECC/ ARMCANZ 2000). The primary objective of the guidelines is to "maintain and enhance the 'ecological integrity' of ecosystems (ANZECC/ARMCANZ 2000). These trigger values may be adopted in the absence of adequate baseline data to develop site or system-specific guidelines.

A range of other habitat characteristics were visually assessed at each site in November 2010 (Table 2). This information was used to assess habitat associations for each species to aid planning for future rehabilitation projects.

Table 2. Water quality and physical habitat characteristics assessed at each site.

Habitat Variable
Temperature (°C)
pH (H^{+})
Dissolved oxygen (% saturation)
Dissolved oxygen (mg L^{-1})
Conductivity (µS cm^{-1})
Mean depth (m)
Mean width (m)
Pool area (m^{-2})
Bedrock (% cover)
Boulders >256 mm (% cover)
Cobbles 64-256 mm (% cover)
Pebbles 16-64 mm (% cover)
Gravel 4-16 mm (% cover)
Sand 1-4 mm (% cover)
Silt <1 mm (% cover)
Clay (% cover)
Detritus (%cover within habitat)
Emergent macrophyte (% cover within habitat)
Submerged macrophyte (% cover within habitat)
Floating macrophyte (% cover within habitat)
Algae (% cover within habitat)
Riparian vegetation (% cover within habitat)
Large woody debris (>10 cm diameter) (% cover within habitat)
Root mats (% cover within habitat)
Snag (<10 cm diameter) density (few, common, many)
Snag complexity (single branch, medium, complex))
Riparian canopy cover (% cover)
Bank angle (degree)
Undercuts (% cover within habitat)
Backwaters present (Y/N & distance to backwaters)

2.3 Macroinvertebrate Sampling

Aquatic macroinvertebrates were sampled with a 250 µm mesh pond net. To provide the most comprehensive species list, all in-stream habitats were sampled at each site – open water column, riffles, littoral edges, macrophytes and any draping or inundated riparian vegetation.


Macroinvertebrate sampling

Macroinvertebrate samples were preserved in 70% ethanol and returned to the laboratory for sorting under microscope to remove animals, which were then identified to species level (where possible) and enumerated to $\log_{10}$ scale abundance classes (*i.e.* 0, 1-10, 11-100, 101-1000, >1000). In-house expertise was used to identify macroinvertebrate taxa using available published keys and through reference to the established WRM voucher collection. Dr Don Edward (The University of Western Australia) was sub-contracted for specialist taxonomic expertise to identify Chironomidae (non-biting midge larvae).

The existence of rare, restricted or endemic species was determined by cross-referencing taxa lists for each site with the WRM database, the Department of Environment and Conservation Wildlife Conservation (Specially Protected Fauna) Notice and the IUCN Red List of Threatened Species.

2.4 Fish and Crayfish Sampling

Fish and crayfish were surveyed using a combination of methods including electrofishing, fyke nets, gill nets and box traps. Sampling methods were standardised as much as practical across habitat types to reduce the influence of sampling method on data collected, although this was not always possible because of the broad range of habitats sampled. Therefore not all methods were used at all sites but were deployed with the intention of maximising the number of species taken. For comparative analysis across years/seasons, the same methods and sampling effort will need to be employed for all future surveys.


Electrofishing

Electrofishing (Smith-Root model 12-B backpack electrofisher) was conducted at all but sites 1 and 9, where the salinity was too high (>10,000 µS cm^{-1}). Electrofishing is an extremely useful and efficient sampling tool in systems with clear, low salinity, slow flow water. Shocking was not continuous, but targeted areas of optimum habitat, whereby the operator would shock, move to a new habitat before shocking again, and so prevent fish being driven along in front of the electrical field.

At most sites, a single large fyke net was set in deeper pools for a 24 hour period. The exceptions were sites 8, 9, 15 and 20, where waters were too shallow and/or remanant pools too small. Fykes comprised a single 10 m leader/wing (7 mm mesh, 1.5 m drop) and a 5 m hooped net (75 cm diam. semi-circular opening, 10 mm mesh). Fykes were set facing downstream with floats placed inside the cod-end to form an air pocket for any long-necked tortoises that may become trapped.


Fyke net with float

Up to four box traps per site were also set overnight. Box traps were baited with a mixture of cat biscuits and chicken pellets. The number of box traps set at a given sites depended on the size of the pool.

Light-weight fine mesh gill nets (10 m net, 2 m drop) were also used. These were set in deeper pools at sites 1, 4, 5, 12 and 18. Gill nets were left in place for a maximum 40 - 60 minutes and promptly cleared to prevent fish kills. Two different sized mesh gill nets were used to target different sized fish, including 10 mm and 13 mm mesh nets (knot to knot).

All species were identified in the field, measured by standard length[1] (SL mm, for fish) or carapace length (CL mm, for crayfish) and then released alive. Fish nomenclature followed that of Allen *et al*. (2002). A permit (Research Permit No. 1720-2010-54) to catch and release fish was sought from and approved by the Department of Fisheries prior to commencement of sampling.

[1] Standard length (SL) = tip of the snout to the posterior end of the last vertebra (*i.e.* this measurement excludes the length of the caudal fin). Carapace length (CL) = anterior tip of the rostrum to the posterior median edge of the carapace.

2.5 Other Aquatic Fauna

Though not specifically targeted for the current study, records were made of opportunistic sightings of any other aquatic fauna such as water rats and tortoise.

2.6 Data Analysis

The abundance and number (richness) of species was plotted in order to illustrate the spatial differences in distribution. Where sufficient numbers of each fish or crayfish species were measured, length-frequency histograms were plotted.

The total abundance of fish, total abundance of crayfish, the abundance of each species, the number of species, the number of non-native and the number of native species were plotted graphically for each sampling site to represent spatial differences in the distribution of each species, and in the fish fauna as a whole. Relationships between each species and physico-chemical/habitat characteristics were investigated by Spearman Rank correlation analysis and the most significant relationships were plotted on x-y graphs.

Analysis of variance (ANOVA) was used to test for significant differences in macroinvertebrate species richness and abundance i) upstream versus downstream of the PHD, and ii) between seasons (May 2010 data *vs* current data). Relationships between macroinvertebrate species richness and physico-chemical/habitat characteristics were also investigated by Spearman Rank correlation analysis.

Differences in macroinvertebrate species assemblages amongst sites and between seasons were analysed using non-parametric multivariate procedures in the PRIMER (v6) software package (Clarke & Gorley 2006). Macroinvertebrate species presence/absence and $\log_{10}$ abundance data were examined using Multi-Dimensional Scaling (MDS) ordination (Clarke & Warwick 2001), with analyses based on Bray-Curtis similarity matrices. Ordinations were depicted as two-dimensional plots. The ANOSIM (analysis of similarity) routine within PRIMER was used to test the significance (p <0.05) of the separation of site groupings in MDS ordination space. SIMPER was used to determine those species contributing most to the similarity/dissimilarity between significant site groupings. Relationships between species data and environmental data (water quality & habitat characteristics) were assessed using BIOENV to calculate the smallest sub-set of environmental variables that explain the greatest percentage of variation in the species ordination patterns, as measured by Spearman rank correlation (ρ) (Clarke & Warwick 2001). Where necessary, environmental data were log transformed prior to analysis to meet assumptions of the test. Unless indicated, default values or procedures otherwise recommended by Clarke and Gorley (2006) were employed for all PRIMER routines.

3 RESULTS AND DISCUSSION

3.1 Water Quality and Habitat Variables

3.1.1 Water Quality

In situ water quality is summarised in Appendix Table A1. Water temperatures in late spring ranged from 15.6^{o}C to 26.6^{o}C, compared to 22.8^{o}C to 33.5^{o}C in late summer/autumn. Water temperatures were dependent on ambient air temperature and water volume in remnant river pools. The maximum temperature recorded (33.5^{o}C) was for an extremely shallow pool at Site 15 in the mid reaches of the river.

Spot measurements of pH were generally within the range expected for south-west river systems, *i.e*. pH 6.5 – 8.0 (ANZECC/ARMCANZ 2000). The exceptions were Site 8, which had an acidic pH of 5.6 and Site 7 with an alkaline pH of 8.2 in spring 2010. In late summer/autumn 2011 pH at Site 8 had risen to 7.5, and at Site 7 had fallen to 7.4. More comprehensive measurement of pH at other times of day and year would be required to determine seasonal pH regimes at these sites. Factors that will influence the pH of a given site include local geology, presence of humic substances and metabolic rates within aquatic communities. Metabolic rates are in turn highly dependent on the abundance of aquatic plants and/or organic detritus present, as well as water temperature and degree of shade cover.

Salinity values (as EC) were mostly well above the recommended range of 120 - 300 µS cm^{-1} for freshwaters (ANZECC/ARMCANZ 2000), although relatively low salinities were recorded at two of the smaller pools following rainfall that occurred during the summer/autumn sampling; 102 µS cm^{-1} at Site 15 and 534 µS cm^{-1} at Site 4. High salinity was to be expected at Site 1 (10,100 - 44,400 µS cm^{-1}) due to the tidal influence from the Swan River. To a certain extent, the high salinities at other sites reflected evapo-concentration effects of low water levels following a particularly dry winter-spring. However, very high values of between 3,100 µS cm^{-1} and 18,690 µS cm^{-1} recorded in the Upper Helena (sites 8 - 12 & 25) were considered indicative of secondary salinistation.

Of most concern were the low daytime DO levels recorded in late spring at many of the sites. Hypoxic[2] conditions prevailed at sites 2, 3, 5 and 6 in the lower reaches below the PHD, and sites 16 and 19 in the mid reaches between the PHD and Mundaring Weir (Appendix Table A1). Daytime DO levels were approaching anoxia at sites 3 (3.8%) and 16 (9.4%). Given such low daytime concentrations, DO levels most likely fall to zero overnight at these six sites in late spring. Other sites that may also experience overnight hypoxia (or even anoxia) were those with daytime DO levels of between 40% and 55%, *i.e.* sites 4, 9, 10, 13 and 18. In late summer/autumn, 13 of the 19 sites sampled had very low daytime DO levels (10 - 50%), while the remainder had only moderate levels (52 - 74%).

[2] Hypoxia = low dissolved oxygen concentrations, *i.e.* 1-30% saturation; anoxia = zero dissolved oxygen concentrations, *i.e.* 0% saturation. DO concentration is strongly dependent on temperature. DO measured in mg L^{-1} is the total amount of oxygen in a litre of water. Percent saturation is the amount of oxygen in a litre of water relative to the total amount of oxygen that the water can hold at that temperature (USEPA 1997). Measuring DO as % saturation effectively factors out the influence of temperature. For example, water that is 100% saturated contains the maximum amount of oxygen at that temperature. Water that is 50% saturated contains only half the amount of oxygen that it could potentially hold at that temperature (USEPA 1997).

Hypoxia and anoxia will result in the death of many aquatic organisms. DO levels of less than 50 - 60% saturation (*i.e.* ~5 mg L^{-1}) are known to cause stress to aquatic fauna. Fish and crayfish are particularly susceptible. A number of dead gilgies were found in box traps set overnight at Site 13, where low daytime DO levels of 44% were observed. It is possible the deaths were caused by hypoxic or anoxic conditions overnight, with specimens caught in the traps and unable to move to areas of higher DO (*i.e*. around the edges of the pool). Anoxia in the Middle Helena and in-filling of river pools by fine organic sediments has previously been documented by WRM (2010) during trial environmental water releases from the Lower Helena PHD in May 2010. Automated data loggers revealed overnight anoxia at Site 20 and at a river pool adjacent to Site 19. DO levels at these sites showed a rapid and pronounced response to trial releases of 190 L sec^{-1} from the PHD. DO increased from 0% to 80 - 87% in 6 hours. The trials (conducted in May) suggested a minimum baseflow of 9 L sec-1 is required to maintain DO levels above the threshold 50-60%, below which aquatic fauna would begin to suffer.

DO concentration is dependent on both physical aeration (*e.g*. flow over riffles, high velocities or wind action) and on metabolic rates (*i.e.* rates of photosynthesis and respiration). Very high daytime DO concentrations (*e.g*. supersaturated; >100%) can occur where there is an abundance of macrophyte growth. Conversely, very low day and nighttime concentrations can occur where microbial respiration (consumption of DO) far exceeds photosynthetic production of DO. This is frequently observed in systems that have high loads of organic materials resultant of land clearing and subsequent runoff from rural and urban areas. DO saturation is also influenced by water temperature, both directly and indirectly as metabolic activity increases with temperature. While naturally low DO levels can occur, these are typically confined to still, shallow wetlands with abundant aquatic and riparian vegetation or to drying pools in seasonal creeks. Stratification in deep waterbodies and in estuaries may also result in naturally low DO in the hypolimnion (*i.e*. bottom waters).

3.1.2 Habitat Characteristics

Habitat variables recorded in late spring 2010 are tabulated in Appendix Table A2. A wide range in habitat variables was recorded both above and below Mundaring Weir and below the PHD. In general, the most notable trends amongst sites were:

i) slightly greater proportion of sand and clay substrates at sites in lower reaches downstream of Craignish gauging station (site 20),
ii) greater proportion of pebble, cobble and boulder substrates at sites between Craignish gauging station and the PHD, and
iii) greater pool volumes (area and depth) between the PHD and Mundaring Weir.

3.2 Macroinvertebrates

3.2.1 Taxonomic Composition and Species Richness

Seventy macroinvertebrate taxa ('species') were collected from the 10 sites sampled during February/March 2011 (Appendix Table A3). Five of these sites were located between Craignish Weir and the PHD and five between the PHD and Mundaring Weir. This was significantly fewer than the 83 taxa collected by WRM (2010) in May 2010 from just 6 sites below the PHD (Appendix Table A4). The lower taxa richness in 2011 was considered a reflection of harsher environmental conditions (low water levels, reduced water quality) and natural seasonal variation in community structure. Variations in life history of individual species, their growth rate and life phase, will determine the community composition present at any one time and at any one site. A combined total of 95 taxa were recorded from the 2010 and 2011 sampling occasions, with only 49 taxa (52%) common to both.

The macroinvertebrate 'species' records include larval and pupal stages for groups such as Diptera and Coleoptera. Current taxonomy is not sufficiently developed to allow species-level identification of larval and pupal stages of all members of these groups and they were therefore treated as separate taxa. Different life stages of macroinvertebrate will often have different functional roles in the ecosystem, and so treating these as different species is justified in terms of functional composition.

In both 2010 and 2011, the macroinvertebrate fauna was dominated by Insecta (80%), of which Diptera (two-winged flies) were particularly well represented with 7 families constituting ~36% of insect species collected. Diptera are known to be the most diverse order of insects in freshwater systems (Hutchinson 1993). Next were the Coleoptera (aquatic beetles) with 6 families, accounting for ~28% of the Insecta. Crustacea, including Decapoda (freshwater crayfish & shrimps), Ostracoda (seedshrimps), Cladocera (waterfleas) and Copepoda comprised approximately 9%[3] of the total macroinvertebrate fauna. Gastropoda (freshwater snails) comprised approximately 5% of the total fauna, with 4 species.

The most commonly encountered taxa were ostracods, copepods, cladocera, the mayfly *Tasmanocoenis tillyardi*, corixid hemipterans (true-bugs), gyrinid (whirly-gig) beetles and the chironomids (non-biting midges) *Cladopelma curtivalva*, *Kiefferulus intertinctus* and *Tanytarsus fuscithorax*. All these taxa are typical of lentic (still or slow-flowing) environments, reflecting the lack of flow in the system. There were also a large number of singletons recorded (*i.e*. species recorded from only one site). Some 30% of all taxa recorded in 2010 and 2011 were singletons.

Between-season and between-site comparison of 'species'-level taxa richness are shown in Figure 2A. Large seasonal variation in species richness was evident at most sites. In May 2010, Site 19, upstream of Craignish Gauging Station, supported the highest taxa richness with 40 species, but only 12 species were recorded here in February/March 2011. WRM Site 9 had the lowest richness in 2010 with 27 'species'. This site was not sampled in 2011. Lowest richness in 2011 was 3 species recorded at Site 15. That these sites had relatively fewer species was not unexpected given the small and shallow nature of the remnant pools, particularly so at Site 15 in 2011. The comparatively high number of taxa (32) recorded in 2010 at Site 20, at Craignish Gauging Station, was surprising given the degraded riparian conditions and low day-time dissolved oxygen concentrations. Site 18 showed the least seasonal variation with 36 species recorded in May 2010 and 27 recorded in February/March 2011. Relatively low seasonal variation at this site may be attributable to the persistance of a relatively large, clear, deep water pool and to the forested riparian zone.

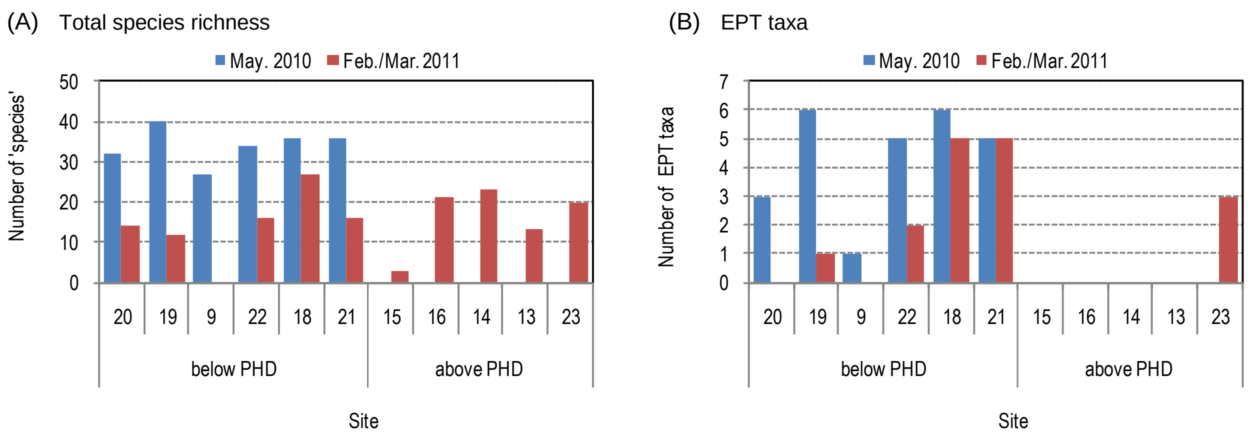

Figure 2. (A) Total macroinvertebrate 'species' richness and (B) EPT taxa richness recorded in May 2010 and February/March 2011 below the Lower Helena Pipehead Dam and between the PHD and Mundaring Weir. Sites are ordered by increasing distance from the confluence with Swan River.

[3] Ostracoda, Copepoda and Cladocera identified to order-level only.

There was no significance difference in average species richness above and below the PHD in February/March 2011 (one-way ANOVA, $p = 0.829$, $F = 5.318$). Average species richness below the PHD was 17, compared to 16 above the PHD. There was however, a significant seasonal difference (one-way ANOVA, $p = 0.0003$, $F = 5.117$). Average species richness in May 2011 was 34.2, compared to 17 in February/March 2011.

3.2.2 Conservation Significant Taxa

Only 13% of all taxa collected in 2010 and 2011 were considered to be south-west regional endemics (refer Appendix Table A3 & A4). These included one listed species, the freshwater mussel *Westralunio carteri*, which was recorded from sites 18 and 21. This species is listed as vulnerable (VU) by the IUCN (2009) and as Priority 4 (P4) by the Department of Environment and Conservation, WA (DEC), *i.e.* "in need of monitoring". Although this species is widespread throughout the south-west, populations are fragmented, increasing the species' vulnerability to disturbance. *W. carteri* occurs naturally in both seasonal and permanent waterbodies. It is a filter feeder and vulnerable to water pollutants and sedimentation. It prefers shallow water habitats with stable, sandy or muddy bottoms and inhabits both permanent and seasonal rivers. *W. carteri* can survive prolonged periods of drought by burrowing into bottom muds and sealing the bivalve. It may thus survive potential drawdown of river pools. It is intolerant of high salinity but levels would likely need to reach 4,000 µS cm^{-1} (~2,500 mg L^{-1}) or greater before causing fatality (Bailey *et al.* 2002).

Of particular concern in regulated rivers is the fact that as part of their life cycle, these mussels have an early larval phase that is parasitic on the gills of native freshwater fish (*e.g*. cobbler). This parasitism is the subject of current research by the Centre for Fish and Fisheries Research at Murdoch University (see *Mussel Watch* http://www.musselwatchwa.com/index.html). Fish appear to be essential for completion of the mussels' life cycle. Mussels may also play an important role in maintaining water quality in the pools which provide refuge for fishes over summer. Mussels also provide a food source for native cobbler and water rats.

Barriers to upstream movement of fish may therefore also restrict gene flow between mussel populations, limit upstream-downstream recruitment of mussels, restrict distributions and prevent recolonisation. As well as weirs and dams, barriers include low flow regimes that make natural barriers (waterfalls, riffle zones) impassable for fish.

3.2.3 EPT Taxa

EPT taxa are those belonging to the genera **E**phemeroptera (mayflies), **P**lecoptera (stoneflies) and **T**richoptera (caddis-flies). These taxa in general are typical of upland streams and are considered more sensitive to disturbance, though sensitivity varies between species. Some, like the caeinid mayfly *Tasmanocoenis tillyardi*, for example, are known to be tolerant of sediment pollution and nutrient enrichment, and are often found in moderately to severely disturbed rural rivers.

In the Helena River, EPT taxa were limited to 4 'species' of Ephemeroptera and 5 'species' of Trichoptera (refer Appendix Tables A3 & A4). This included the more pollution-tolerant *T. tillyardi*, as well as immature mayfly and caddis-fly nymphs which could not be identified to species-level. No Plecoptera were recorded from any of the sampled sites. Between-season and between-site comparison of EPT taxa richness are shown in Figure 2B. Other than Sites 18 and 21, very few sites supported EPT taxa in February/March 2011 sampling, and only one of these (Site 23) was above the PHD. In general, richness and relative abundances tended to be greater at sites 18, 19 and 21. These sites possessed somewhat better water quality, in-stream habitat and riparian vegetation cover, possibly reflecting recent environmental releases or longer term seepage from the PHD. The paucity of EPT taxa at sites between the PHD and Mundaring Weir in February/March 2011, was considered due to the small size and shallowness of remnant pools in these reaches over summer.

3.2.4 Functional Feeding Groups

The functional complexity and 'health' of a river system is influenced by the diversity of functional feeding groups (guilds) (Cummins 1974, Cummins & Klugg 1979), the obligate feeding mode of each species. Current theories of functional organisation of streams in the south-west (see Bunn 1985, 1986, 1988) predict relatively undisturbed, forested upland streams to be dominated by collector (including filterers) and predator feeding guilds, but with a high proportion of shredders. Shredders tend to be less abundant in undisturbed lowland streams as the input of coarse particulate material is less. At the same time, collectors and grazers typically increase. Collectors are also likely to dominate lower reaches of rivers and in particular disturbed reaches where the input of fine particulate material is high.

In the Helena River, predators were the dominant guild at the majority of sites sampled (Figure 3A-B).

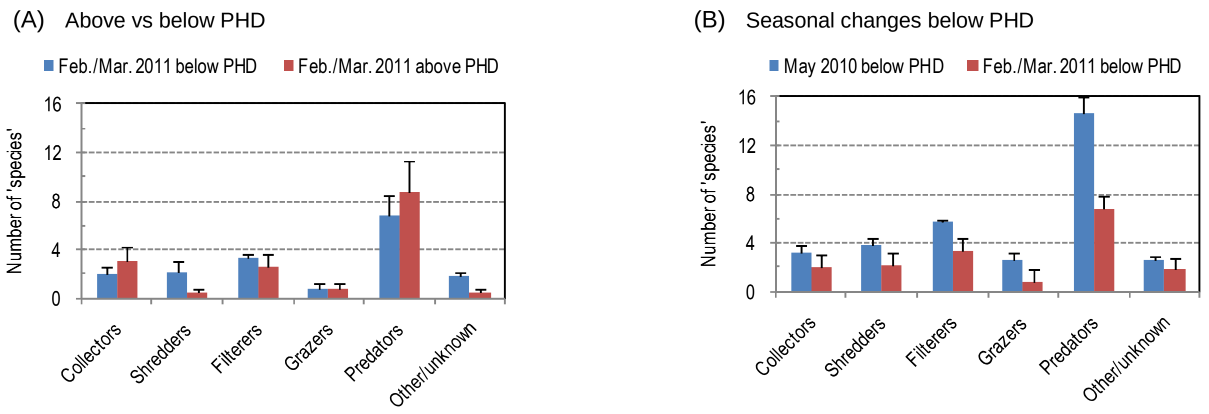

Figure 3. Plots of functional feeding groups illustrating (A) spatial differences above and below the Lower Helena Pipehead Dam in February/March 2011, and (B) seasonal differences between May 2010 and Feb./Mar. 2011 for sites below the PHD. Values plotted are average (±SE) number of 'species' representing each functional feeding group present.

The most commonly occurring predator taxa were Coleoptera (aquatic beetles), followed by Odonata (dragonflies & damselflies) and some Diptera (two-winged flies) (refer Appendix Tables A3 & A4). The next most abundant guild were the filterers. There was little difference in the ratios of feeding groups between above and below the PHD (Figure 3A). The ideal 'healthy' aquatic ecosystem would have greater representation of functional feeding groups other than predators.

There was only a slight seasonal shift in the ratios of feeding groups (Figure 3B), with the most noticeable change being a decrease in the number of predators during February/March 2011. Seasonal and annual variation is to be expected dependent on the availability of food types and life-histories of each species.

3.2.5 Multivariate Analysis of Macroinvertebrate Assemblages

The resultant plot from the MDS ordination analyses on macroinvertebrate $\log_{10}$ abundance data is shown in Figure 4. As results were similar for ordinations on both species abundance and presence/absence, only the results for abundance are shown here. The plot revealed an obvious difference in species assemblages between seasons and, to a lesser extent, between sites above and below the PHD. ANOSIM indicated that the difference between seasons was statistically significant, but that the difference above and below the PHD was not (Table 3). This latter result is likely due to the greater variation (spread) in the data points. Despite not being *statistically* significant, results still suggest the PHD and Mundaring Weir are likely to be of *biological* significance in structuring downstream macroinvertebrate community assemblages.

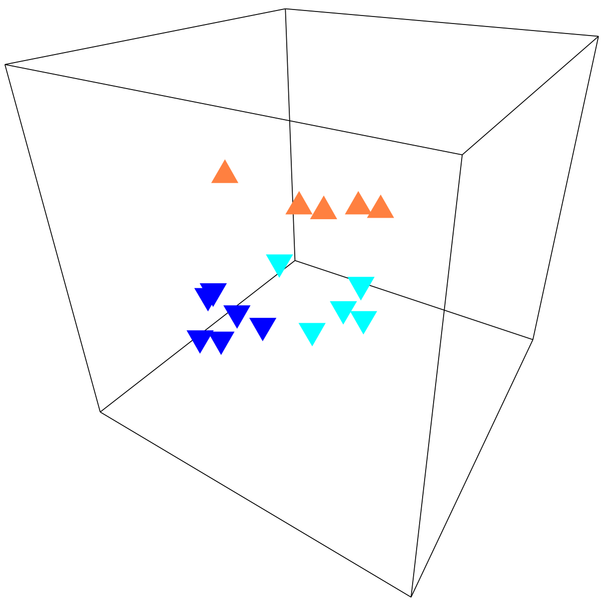

Figure 4. Plot of MDS ordination on macroinvertebrate species $\log_{10}$ abundance data, based on Bray-Curtis similarity matrix. Plot is depicted in 3 dimensions (stress 0.08). Sites are colour coded by sub-catchment and season:

▼ below PHD, May 2011;
▼ below PHD, Feb/Mar 2011;
▲ above PHD, Feb/Mar 2011.

Table 3. Results of ANOSIM (analysis of similarity) on macroinvertebrate species $\log_{10}$ abundance data, showing pair-wise comparisons for season (may 2010 vs Feb./Mar. 2011) and sub-catchment (above vs below PHD). Where R-statistic >0.75 = groups well separated, R-statistic >0.5 = groups overlapping but clearly different, and R-statistic >0.25 = groups barely separable. A *p* value ≤ 0.05 indicates a significant difference.

Pairwise comparison	R-statistic	*p* value
Below PHD May 2010 vs below PHD Feb/Mar 2011	0.684	0.002
Below PHD May 2010 vs above PHD Feb/Mar 2011	0.749	0.002
Below PHD Feb/Mar 2011 vs above PHD Feb/Mar 2011	0.266	0.056

Bray-Curtis percent pairwise similarities were generally low. For example average similarity between sites upstream and downstream of the PHD was only 28% in Feb./Mar. 2011. This was slightly less than the average similarity between seasons of 31%. Analysis with SIMPER showed that the significant differences were subtle, mostly reflecting differences in abundances of species between seasons and sub-catchments, rather than differences in composition. No one species or group of species (*e.g.* EPT taxa, functional feeding group) could be used solely to account for differences above and below the PHD or between seasons. Rather, a large number of taxa, each contributed a small amount to the overall variation. This was a reflection of the large number of singletons encountered and the low number of species common to both 2010 and 2011.

For monitoring purposes, the existing level of similarity amongst sites is not as important as the degree to which that similarity changes over time. Change in pairwise percentage similarity over time can be used to compare how locations alter relative to their baseline condition and relative to each other. This latter comparison allows for natural variation to be incorporated into the monitoring design. The expectation is that abundance of the more sensitive species would increase if habitat conditions at individual sites were to improve and that this may lead to decreased similarity of species assemblage compositions between these and unimproved sites. Conversley, worsening habitat conditions (*e.g*. associated with climate change) may result in increased similarity between sites, as only the most tolerant (and likely cosmopolitan) species will survive.

3.2.6 Relationships between Macroinvertebrate Data and Environmental Data

Univariate analyses revealed no significant relationships between macroinvertebrate species richness and water quality or habitat data. Multivariate analyses however, did reveal significant correlations between environmental data and the patterns in species assemblages observed in the ordinations. A particularly strong correlation (BIOENV, $\rho = 0.930$, $p = 0.01$) was found for a combination habitat variables that included pool size (maximum depth & area), % bedrock substrate and % emergent macrophytes present. In general, a greater diversity of species was found in larger pools with more abundant emergent macrophytes and/or where bed substrates were other than bedrock. Conversely, very small, shallow pools tended to support a reduced diversity and different assemblage of species.

The combination of water quality variables which best explained the variation in species assemblage data were salinity (EC) and temperature (BIOENV, $\rho = 0.601$, $p = 0.001$). This primarily reflected the higher salinity levels and water temperatures recorded during Feb./Mar. 2011 relative to May 2011.

3.3 Crayfish

A total of 417 freshwater crayfish representing 2 native species and 1 exotic species were recorded over the course of the study (Table 4). Raw data for individual sampling occasions are presented in Appendix Table A5. Crayfish species included native gilgies (*Cherax quinquecarinatus*) and native smooth marron (*C. cainii*), and the introduced yabby (*Cherax destructor*). A discussion of each is provided below.

Table 4. Total fish and crayfish species richness (number) and abundance recorded in November 2010 and February/March 2011. Sites are arranged by sub-catchment in order of increasing distance from the confluence with the Swan River.

Species codes: *Ab* = Acanthopagrus butcheri, *Af* = *Aldrichetta forsteri*, *Ac* = *Amniataba caudovittatus*, *Bp* = *Bostockia porosa*, *Gh* = *Gambusia holbrooki* (exotic), *Go* = *Galaxias occidentalis*, *Nv* = *Nannoperca vittata*, *Nvl* = *Nematalosa vlaminghi*, *Pf* = *Perca fluviatalis* (exotic), *Po* = *Pseudogobius olorum*, *Ss* = *Sillago schomburgkii*, *Tb* = *Tandanus bostocki*, *Cc* = *Cherax cainii*, *Cd* = *Cherax destructor* (exotic), *Cq* = *Cherax quinquecarinatus*.

Sub-catchment	Site	Fishes												Crayfish			No. of Species
		Ab	*Af*	*Ac*	*Bp*	*Go*	*Gh*	*Nv*	*Nvl*	*Pf*	*Po*	*Ss*	*Tb*	*Cc*	*Cd*	*Cq*	
Below PHD	1	2	3	1	1	-	160	1	1	-	61	5	-	-	-	1	**10**
	2	-	-	-	-	-	4	-	-	-	-	-	-	-	-	1	**2**
	3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	5	**1**
	4	-	-	-	-	-	31	-	-	-	-	-	-	-	-	1	**2**
	5	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2	**1**
	6	-	-	-	-	-	-	4	-	-	-	-	-	-	-	3	**2**
	7	-	-	-	1	4	29	3	-	-	1	-	-	-	-	7	**6**
	20	-	-	-	-	-	15	34	-	-	3	-	-	-	5	3	**5**
	22	-	-	-	-	-	85	-	-	-	-	-	-	1	11	-	**3**
	19	-	-	-	1	-	185	5	-	-	3	-	-	-	17	1	**6**
	18	-	-	-	4	-	185	8	-	-	3	-	3	23	-	4	**7**
	21	-	-	-	-	1	200	-	-	-	-	-	-	7	10	-	**4**
Between PHD & Mundaring Weir	15	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	**0**
	24	-	-	-	-	-	30	-	-	-	-	-	-	1	-	1	**3**
	16	-	-	-	-	-	119	4	-	-	1	-	-	-	-	5	**4**
	17	-	-	-	-	-	79	3	-	2	-	-	-	1	-	3	**5**
	14	-	-	-	-	-	333	-	-	-	2	-	-	15	-	1	**4**
	23	-	-	-	-	-	99	1	-	1	-	-	-	19	-	4	**5**
	13	-	-	-	18	-	176	1	-	-	-	-	-	-	-	46	**4**
Above Mundaring Weir	10	-	-	-	35	7	-	12	-	-	-	-	-	-	-	40	**4**
	11	-	-	-	9	3	-	20	-	-	-	-	-	-	-	77	**4**
	12	-	-	-	11	32	178	106	-	-	-	-	-	1	-	48	**6**
	25	-	-	-	22	23	129	81	-	-	-	-	-	-	-	17	**5**
	9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	11	**1**
	8	-	-	-	-	-	-	-	-	-	-	-	-	-	-	25	**1**
Total Abundance		**2**	**3**	**1**	**102**	**70**	**2,037**	**283**	**1**	**3**	**74**	**5**	**3**	**68**	**43**	**306**	

3.3.1 Gilgies

Gilgies were the most common and abundant crayfish captured. They were present at all but three of the sites sampled and consituted 73% of all crayfish captured. The male to female ratio was estimated to be approximatley 1 to 1.2. Juveniles of indeterminate sex constituted approximately 17% of all gilgies caught.

Individuals from a range of cohorts, including juveniles, sub-adults and adults, were collected above and below both the PHD and Mundaring Weir (Figure 5). However abundances were much greater above Mundaring Weir, suggesting better recruitment and survival, than below the dams. Poor water quality rather than physical habitat conditions was considered the likely cause of lower abundances recorded downstream of the dams.

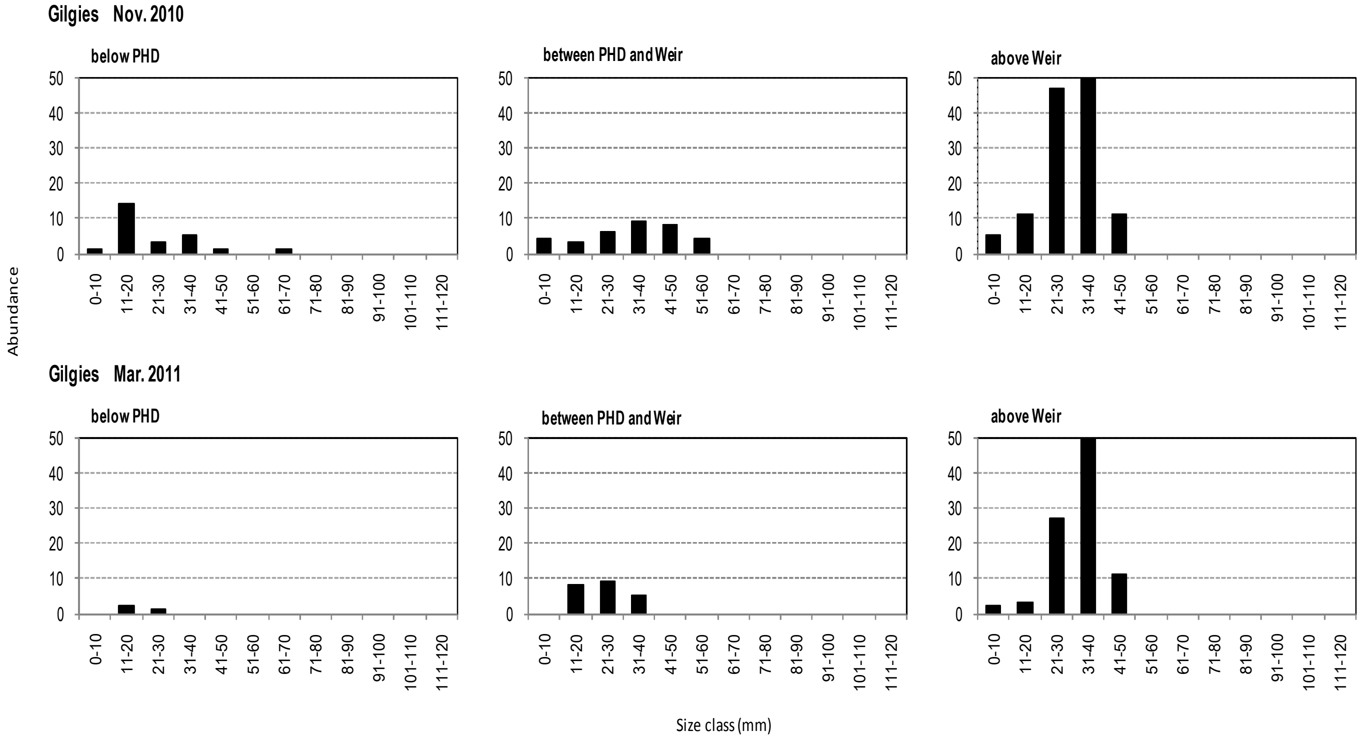

Figure 5. Length-frequency (CL mm) plots for gilgies (*Cherax quinquecarinatus*) collected from the Helena River sub-catchments in November 2010 and February/March 2011.

The physiological tolerances of gilgies are not well understood. During the November 2010 sampling, a number of dead gilgies were found in box traps set overnight at Site 13. At the time, daytime DO levels were low at this site (44%) and it is possible their deaths were caused by hypoxic or anoxic conditions overnight. Gilgies (and yabbies) are known to be tolerant of lower DO levels than fish, however continually reduced levels of 20 - 30% saturation (2 - 3 mg/L) are likely to render them more susceptible to disease (Johnston & Jungalwalla 2005). Sampling by WRM (2010) below the PHD in May 2010, found most gilgies collected were sluggish or moribund and several yabbies had claw and rostrum deformities. It is not known if the poor health of these crayfish was related to low DO or other possible causal agents such as heavy metals, pesticides, herbicides and/or hydrocarbons (WRM 2010). Surrounding lands have a relatively high degree of disturbance and vegetation clearing, with orchards, stock grazing, and suburban development all present.

Gilgies have a range that extends from the Moore River in the north to Bunbury in the south (Shipway 1951). They are known to exploit almost the full range of freshwater environments, and can be found in habitats that range from semi-permanent swamps to deep rivers (Austin & Knott 1996). Gilgies have a well developed burrowing ability that allows them to withstand periods of low water level by retreating into burrows until flows return. Gilgies would appear tolerant of salinities up to at least 18,690 µS cm^{-1} as evidenced by their presence in Warrrin Creek.

3.3.2 Marron

Smooth marron (*Cherax cainii*) were only collected from 8 sites, mostly downstream of Mundaring Weir (Table 4 & Figure 6). There was limited suitable habitat for marron above the Weir. Preferred habitat is deep, broad permanent river pools with low nutrient levels. The generally low abundances recorded during the current study may reflect poor water quality.

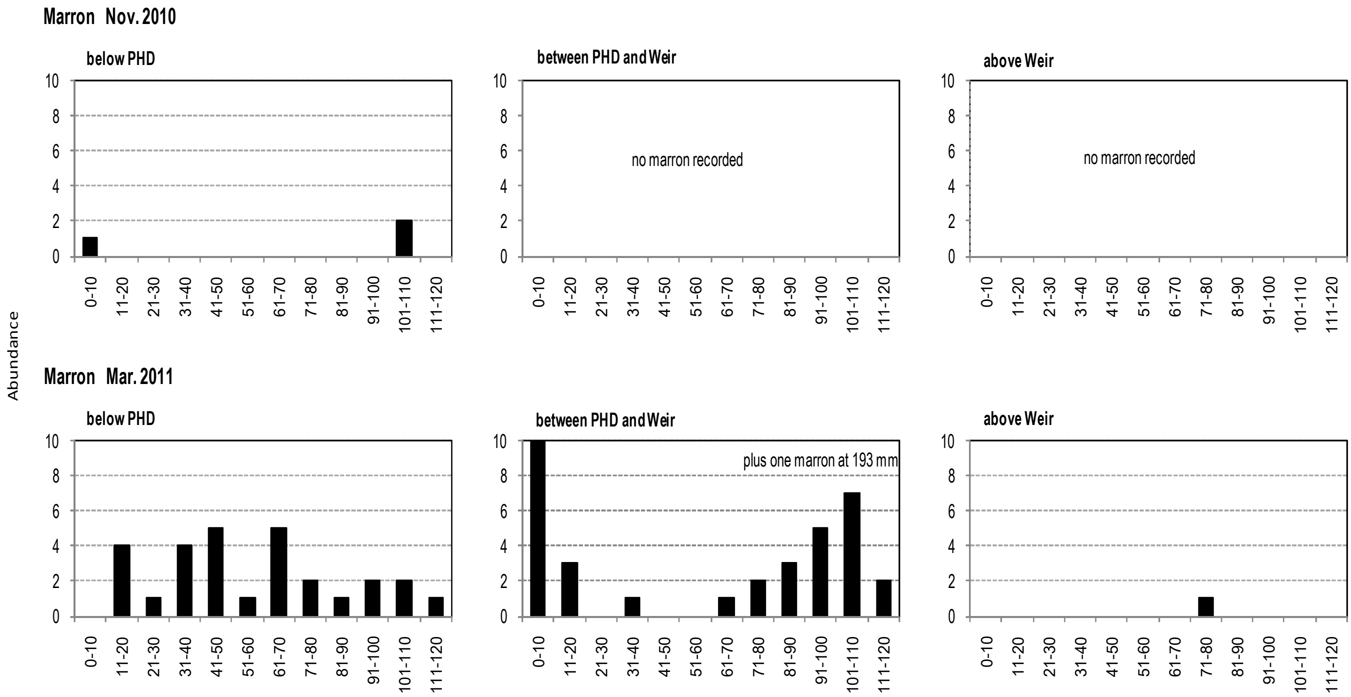

Figure 6. Length-frequency (CL mm) plots for marron (*Cherax cainii*) collected from the Helena River sub-catchments in November 2010 and February/March 2011.

Marron are believed to be far more sensitive to environmental fluctuations than gilgies or the exotic yabby. They have the least tolerance to low DO concentrations, with their optimum DO level being greater than 65%. Aquaculture studies indicate health and growth is best at water temperatures of 24°C and that mortality can occur when temperatures exceed 26°C (Morrisy 1990). If conditions are unfavourable, marron are known to physically remove themselves from the water and migrate overland in search of another permanent waterbody (Morrissy 1978). Low DO levels at many sites may account for the apparent absence of marron at many sites.

During the current study, a range of cohorts were found below Mundaring Weir and below the PHD (Figure 6). Far more individuals were recorded in February/March 2011 when water levels in remnant pools were low, than in November 2010 when water levels were relatively higher. The male to female ratio was estimated to be approximatley 1 to 1.6, with juveniles of indeterminate sex constituting approximately 27% of all marron caught.

Smooth marron have a wide distribution across the southwest from the Hutt River near Geraldton to Esperance (Lawrence & Morrissy 2000, Beatty *et al.* 2003, Lynas *et al.* 2006, 2007).

3.3.3 Yabbies

The introduced yabby (*Cherax destructor*) was present at 4 sites below the PHD. A range of size classes was recorded (Figure 7) suggesting recruitment at these sites. No individuals were caught upstream of the PHD or upstream of Mundaring Weir. The yabby, is a native to south-eastern Australia and has proved to be a highly successful invasive species. Originaly introduced to farm dams in WA in 1932, yabbies have now spread into natural river systems throughout much of the southwest of the state (Lynas *et al*. 2007). The presence of yabbies in natural aquatic systems is of concern owing to their highly aggressive nature and superior competitive ability in comparison with native gilgies and marron (Lynas *et al.* 2004). Yabbies are also tolerant of a wide range of environmental conditions, have the ability to exploit a wide variety of different aquatic habitats, including semi-permanent swamps, billabongs, irrigation channels, and deeper, permanent streams and rivers (Austin 1985), and produces a large number of offspring. They can survive water temperatures from 1°C to 34°C, though in its native Victoria, it will undergo partial hibernation once temperatures fall below 15°C. They tolerate salinities up to ~14,000S µS cm^{-1} and minimum DO concentrations of 45-50% saturation. Yabbies are burrowing crayfish adapted to long-term population survival in the fluctuating environments of impermanent wetlands.

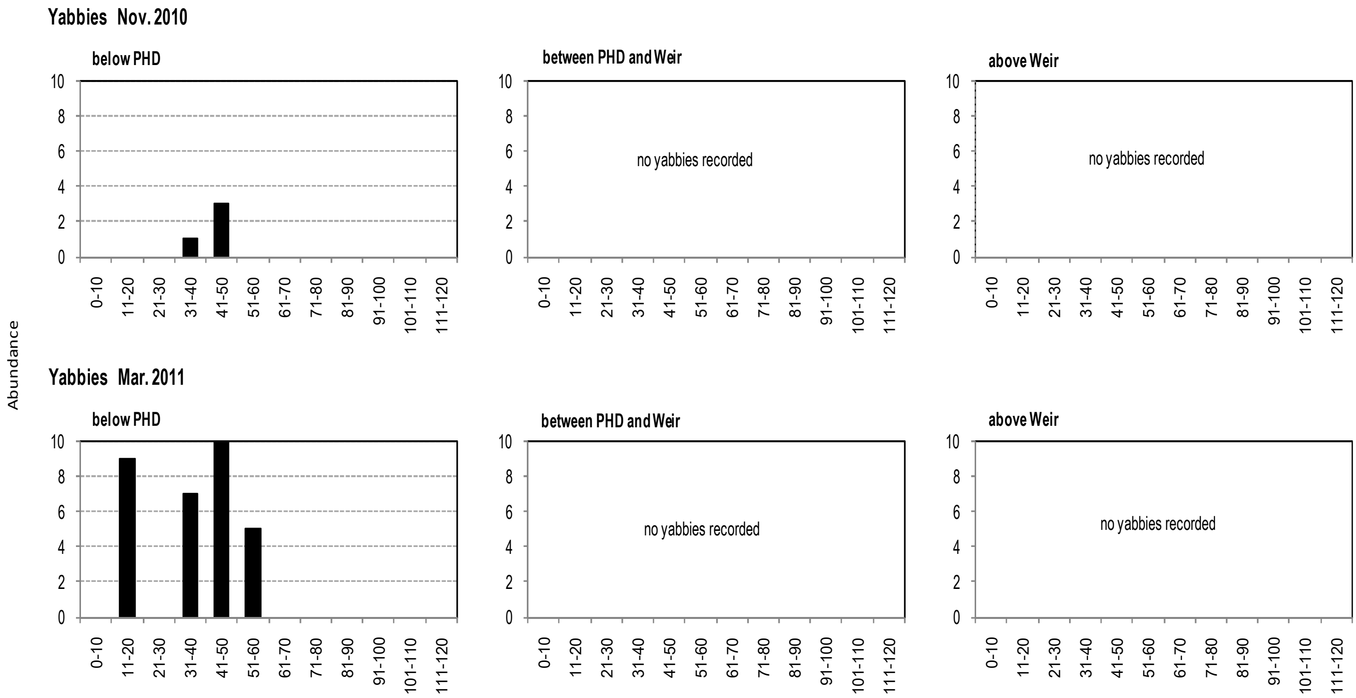

Figure 7. Length-frequency (CL mm) plots for yabbies (*Cherax destructor*) collected from the Helena River sub-catchments in November 2010 and February/March 2011.

3.4 Fishes

A total of 2,584 fish representing 10 native species and 2 exotic species were recorded (Table 4, above, and Appendix Table A5). No fish were collected or observed in Warrin Creek (sites 8 & 9), however fish may frequent this creek in other seasons or other years, when water levels are higher and/or salinities lower.

Native freshwater species recorded from the main channel of Helena River included: western minnow (*Galaxias occidentalis*), western pygmy perch (*Nannoperca vittata*), nightfish (*Bostockia porosa*) and freshwater cobbler (*Tandanus bostocki*). A number of more typically estuarine species were also recorded, mostly from Site 1 near the confluence with the Swan River: yellow-eye mullet (*Aldrichetta forsteri*), black bream (*Acanthopagrus butcheri*), Perth herring (*Nematalosa vlaminghi*), yellowtail trumpeter (*Amniataba caudovittatus*), yellow-fin whiting or western sand whiting (*Sillago schomburgkii*) and Swan River goby (*Pseudogobius olorum*). Although an estuarine species, Swan River gobies are often found far inland in freshwater creeks and tributaries.

The two introduced species recorded were the mosquitofish or gambusia (*Gambusia holbrooki*) and redfin perch (*Perca fluviatalis*). The mosquitofish was by far the most common and abundant species being present at 17 of the 25 sites sampled and constituting 78% (2,037) of all fish caught. Redfin perch were only recorded at two sites (sites 17 & 23) between the PHD and Mundaring Weir, and only 3 individuals were recorded in total.

The western pygmy perch was the most common and abundant of the native fishes. It was present at 14 sites and constituted 11% (283) of all fish caught. The next most common species was the nightfish, which was recorded from 8 sites and constituted approximately 4% (102) of all fish caught.

An unexpectedly low number (3) of native cobbler was recorded. Three individuals were caught in a fyke net set at Site 18 ("Tiger Snake Pool"); a deep relatively clear water pool in State Forest downstream of the PHD.

Though not recorded during the current study, the estuarine western hardyhead *Leptatherina wallacei* may also be present at Site 1, as it has previously been recorded from the Swan River near the confluence with the Helena River (Gill & Potter 1993, Kanandjembo 2001). It is unlikely that western hardyhead would penetrate far inland beyond the more estuarine lower sections of the Helena River. Western hardyhead are however, widespread throughout other reaches of the Swan-Avon catchment that are more strongly affected by secondary salinisation; *i.e.* the Brockman, Mortlock and Dale rivers, as well as the main channels of the Swan, Avon and Canning (Morgan *et al*. 2009).

None of the species recorded during the current surveys are considered rare or restricted in distribution. The western minnow, western pygmy perch, and nightfish are probably the most common native fish in the southwest, occuring across a wide range of habitats, from disturbed, nutrient enriched environments to those that are more or less pristine. The freshwater cobbler however, is locally threatened, with numbers much reduced in many coastal plain rivers due to loss of habitat.

A brief discussion of the known biology and life histories of the species recorded is provided below.

3.4.1 Western Minnow

Relatively few western minnow were recorded downstream of Mundaring Weir and those that were captured, were all considered to be in the 0+ age class (*i.e.* in their first year) (Figure 8). None were captured between the PHD and the Weir during the current study, but were collected by WRM in May 2010 from sites 18 and 19 (WRM 2010), and in November 2007 upstream of Site 13 (WRM unpubl. data). During the current sampling, western minnows were most abundant at sites above Mundaring Weir, in particular sites 12 and 25, where a range of size cohorts were recorded, representing age classes from 0^+ to 3 years. This suggests a stable population with good recruitment.

In general, western minnows occur in rivers, streams, lakes, pools, and readily invade seasonal creeks and swamps connected to permanent water. The species likes both open water and enclosed areas amongst riparian vegetation. It does not appear to have a preference for any particular substrate type (Thorburn 1999). They are often found at the base of waterfalls (and V-notch gauging weirs) where the water is fast flowing and well oxygenated. This may indicate a preference for these conditions, but more likely indicates that fish are being prevented from continued upstream movement by the physical barrier. Fish will jump through V-notch weirs and 'crawl' up wet rock faces in an attempt to traverse barriers (A.W. Storey, UWA, pers. obs.). The western minnow maybe able to pass over barriers of up to ~20 cm height and are able to swim in water as shallow as 1 cm (WRM 2003). Of the fish found in the northern jarrah forest streams, western minnows are probably the strongest swimmer and therefore the least likely to be affected by barriers. This species is also the most likely to cover large distances, other native species being predominantly territorial (EPA 1987).

Although minnows are a freshwater species, recent acute salinity tolerance trials by Beatty *et al*. (2008) indicate they can tolerate salinities up to ~14,000 mg L^{-1} (equivalent to ~25,000 µS cm^{-1}).

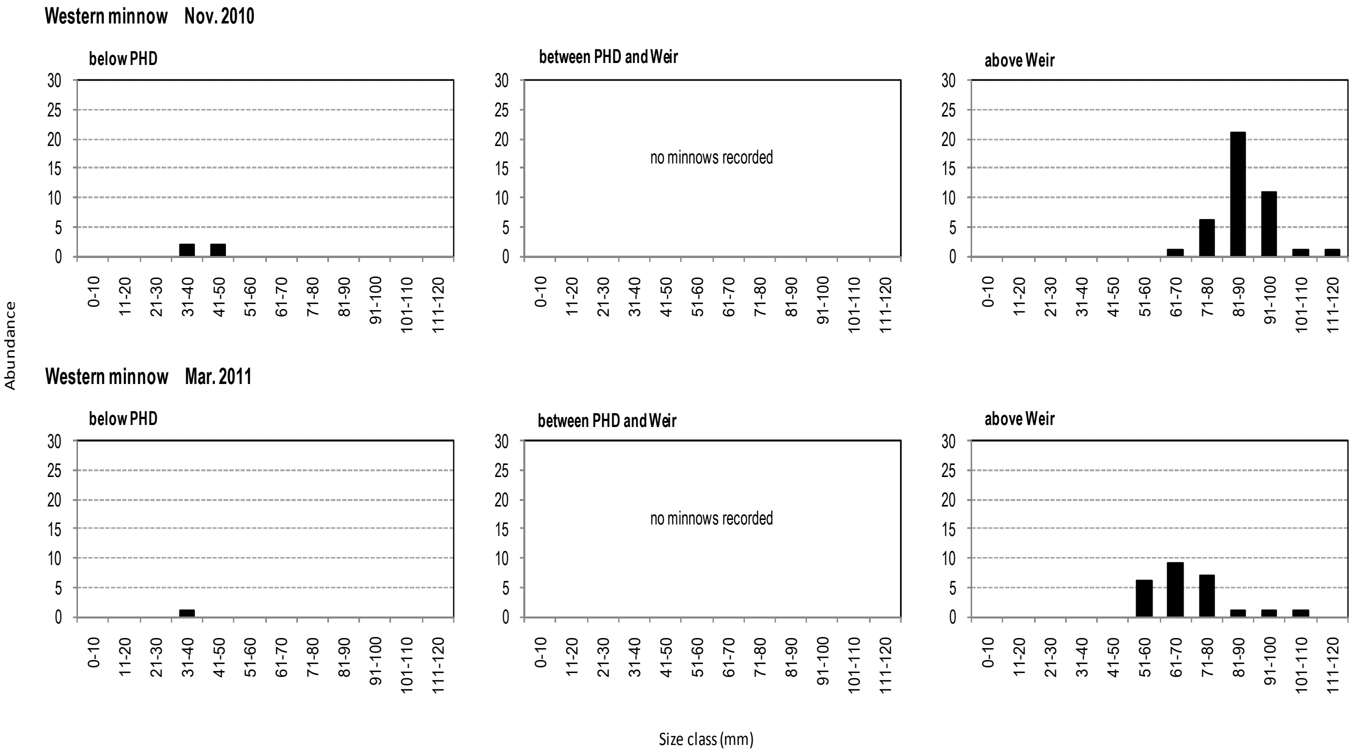

Figure 8. Length-frequency (SL mm) plots for western minnow (*Galaxias occidentalis*) collected from the Helena River sub-catchments in November 2010 and February/March 2011.

3.4.2 Western Pygmy Perch

Western pygmy perch were far more abundant at sites above Mundaring Weir, than at sites below the Weir (Figure 9). This distribution pattern was similar to that observed for western minnows and for nightfish. A range of size cohorts, representing age classes from 0^+ to 5 years were recorded above the Weir, while the majority caught in lower reaches were 0^+ to 1 year individuals. Similar to western minnows, the range in age classes present above the Weir was considered indicative that the population in the upper catchment was stable with good recruitment. Below the Weir, greater numbers of pygmy perch were captured at Site 20 (Craignish Gauging Station).

Western pygmy perch, like western minnows, is a freshwater species common in rivers, streams, and lakes throughout the south-west, and readily re-invade seasonal wetlands *via* flood-ways and up seasonal creeks/drains. Western pygmy perch are often associated with riparian/emergent vegetation and rarely occur in clear, open water, though are frequently captured in turbid open water where there is some woody debris (WRM unpubl. data). Thorburn (1999) found that although western pygmy perch occur in comparatively high densities in several habitat types, especially, snags, macrophyte and grass (and to a lesser extent algae) the major association was with areas containing structure, with the highest number of fish being collected from snags. Data collected by Thorburn (1999) from studies of the Blackwood River, suggest that although western pygmy perch occur across a range of habitats, from those dominated by clay, mud, sand to those dominated by rock, they show greater preference for finer particulate substrates.

Morgan *et al.* (2003) report that while the acute salinity tolerance of western pygmy perch is similar to that of western minnows, pygmy perch are typically only found in habitats with salinities of <~5,000 mg L^{-1} (equivalent to ~10,000 µS cm^{-1}). This may explain their absence from Warrin Creek where salinities reached 7,740 - 18,690 µS cm^{-1} during the current sampling.

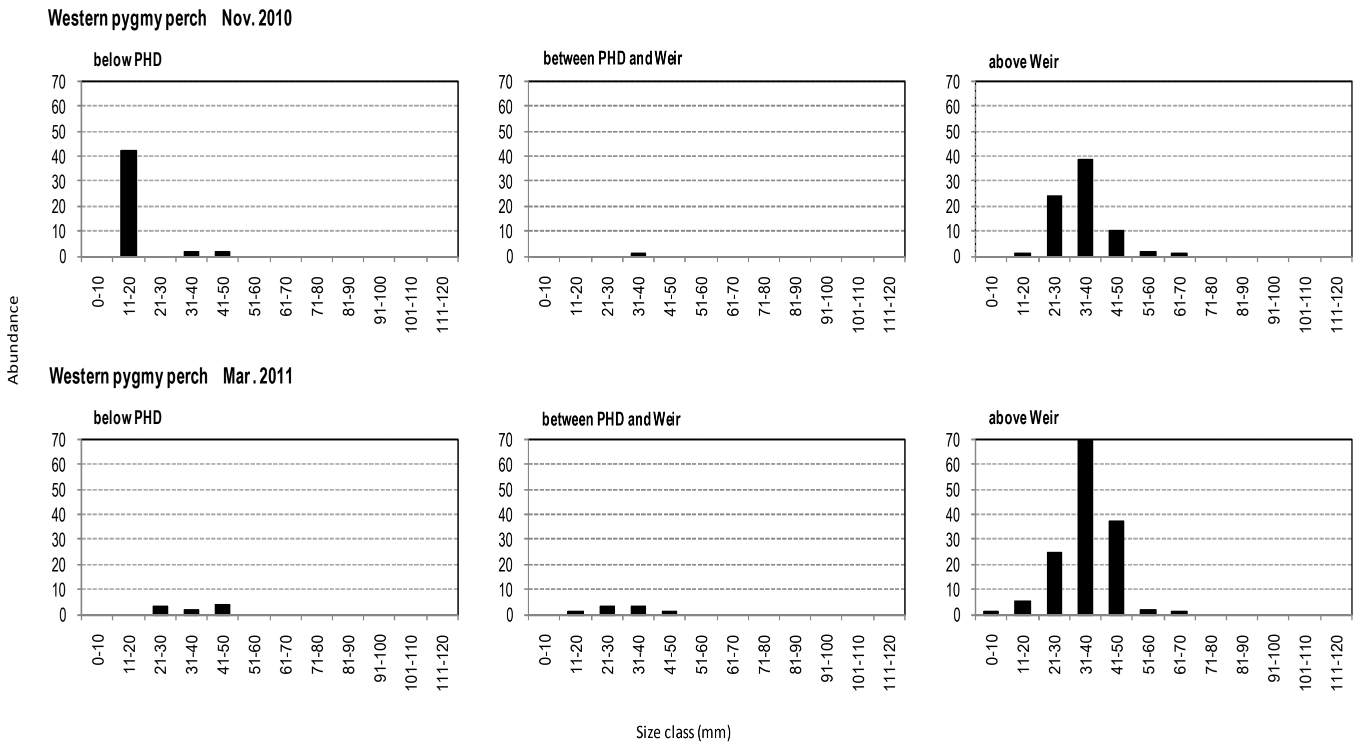

Figure 9. Length-frequency (SL mm) plots for western pygmy perch (*Nannoperca vittata*) collected from the Helena River sub-catchments in November 2010 and February/March 2011.

3.4.3 Nightfish

Nightfish were recorded above and below both the PHD and Mundaring Weir, but abundances above the Weir were much greater than below (Table 4, above). Nightfish were most abundant at site 10, where 35 individuals caught.

Based on standard length (Figure 10), the 3 individuals recorded below the PHD were 0^+ to 2 year olds. In between the PHD and Mundaring Weir, a range of size cohorts representing 0^+ - 3^+ year old fish were collected. A similar range of size classes was collected from above the Weir, but with a far greater number of 1 - 2 year individuals. Similar to western minnows and pygmy perch, the wide range of nightfish size cohorts present was considered indicative that the population above the Weir was stable with good recruitment.

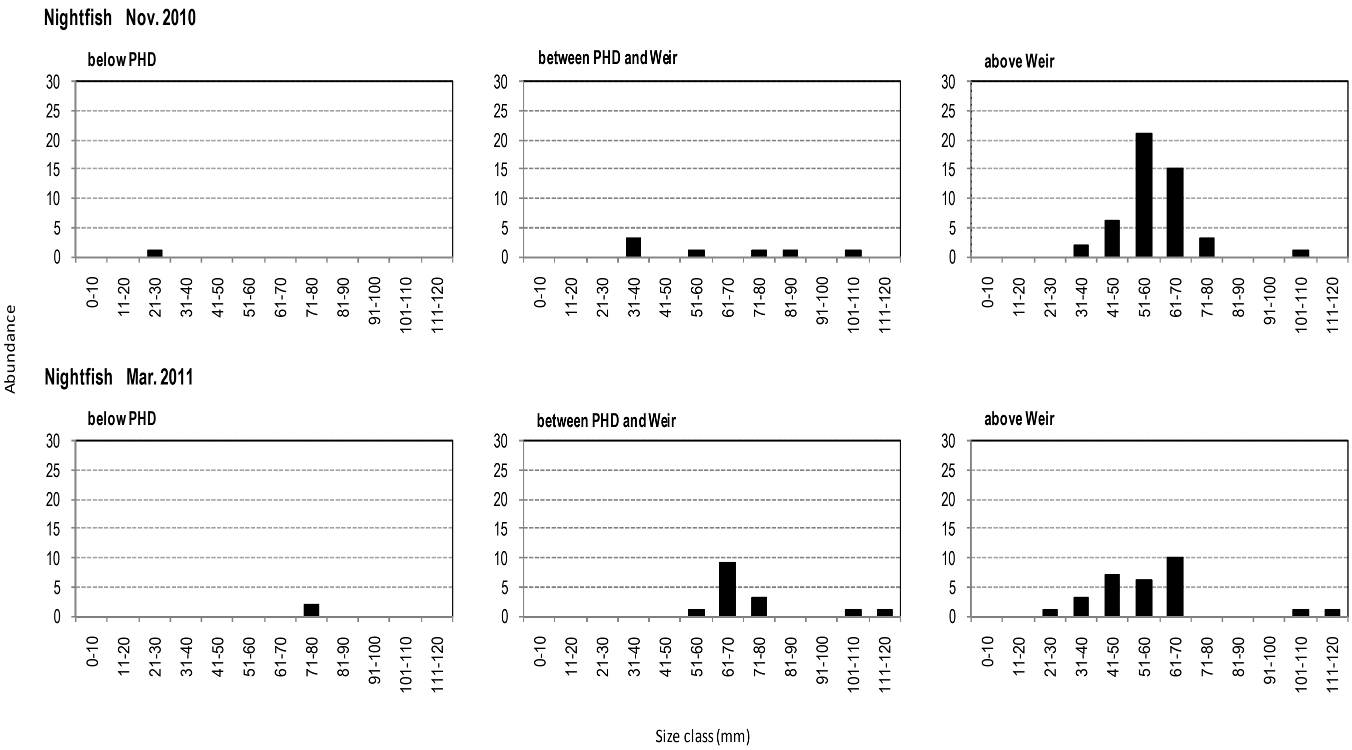

Figure 10. Length-frequency (SL mm) plots for nightfish (*Bostockia porosa*) collected from the Helena River sub-catchments in November 2010 and February/March 2011.

Nightfish are very widely distributed in south-west Western Australia, occuring in rivers, streams, lakes and pools. They are a solitary, bottom dwelling fish, and, as the name suggests, are more active during the night than during the day. They commonly occur under ledges, rocks, logs, amongst root mats and inundated vegetation. Pen and Potter (1990) recorded that the majority of nightfish live for at least 6 years. Thorburn (1999) reported that nightfish appears to have a preference for lower salinity water (<1,000 µS cm^{-1}) and low flows (up to 2,000 s^{-1}). However, recent work by Beatty *et al*. (2010) on the Brockman River suggests it can tolerate, and possibly breed in, salinities of at least ~8,000 µS cm^{-1}.

Thorburn (1999) noted that like the western pygmy perch, nightfish rarely frequent open water. They are most commonly found in association with more complex cover types, in particular, snags along the banks of narrower reaches and tributary sites. Like pygmy perch, they do not appear to have a strong preference for any particular bed substrate type, although Thorburn (1999) recorded highest densities from finer substrate types, especially mud and fine sand.

3.4.4 Freshwater Cobbler

The freshwater cobbler caught at Site 18 ("Tiger Snake Pool") below the PHD were likely 2-3 years old, based on their size (330mm, 335 mm & 350 mm SL). Cobbler of this size (and larger) typically inhabit deeper river pools of the main channel of rivers. Site 18 was one of the deeper and larger pools sampled for the current study. Cobbler reach a maximum size of up to 450 mm SL (550 mm total length), and are by far the largest native freshwater species in the south-west. They are believed to live for at least 9 years. Studies by Morrison (1988) suggest that 1 - 2 year old fish are not sexually mature, but all fish 5 or more years old are sexually mature. This species appears to need to attain a weight of at least 500 g to become sexually mature. They spawn in a 'nest' made of gravel and rocks with a sandy central depression into which the female deposits her eggs (Morrison 1998).

Freshwater cobbler are endemic to the south-west of the state occuring between the Franklin River on the south coast and the Moore River north of Perth (Allen *et al.* 2002). They have a sporadic distribution, are tolerant of brackish waters (at least 10, 000 µS cm^{-1}) and typically occur in slow flowing larger waterbodies such pools in the main channel of rivers (Beatty *et al.* 2010). In the Capel River in the south-west of the state, there is anecdotal evidence from farmers that cobbler numbers have declined over the past 40 years due, at least in part, to a reduction in weed beds and an associated reduced abundance of freshwater shrimp (*Palaemonetes australis*).

3.4.5 Swan River Goby

The Swan River goby was only recorded from sites below Mundaring Weir, with highest abundances (61) occuring at Site 1 near the confluence with the Swan River. Sizes of individuals ranged from 15 mm to 50 mm. The species only lives for about a year and is thought to be sexually mature once they have attained ~25 mm total length, usually between 5 and 7 months of age (Gill *et al.* 1996). Although length may not necessarily reflect age, because water temperature greatly affects growth rate and hence sexual maturity (Gill *et al.* 1996). Water temperatures within the range 20 - 25^{o}C appear most conducive to reproductive success.

The Swan River goby is a typically estuarine species that has a wide distribution from the Murchison River, north of Perth, as far east as Esperance. It occurs in estuaries, rivers, and both freshwater and hypersaline lakes (Gill *et al.* 1996, Morgan *et al.* 1998). It can penetrate long distances inland up secondarily salinised rivers (*e.g.* the Avon River and the Blackwood River), and even occurs in some isolated hypersaline lakes. It is usually found close to the bottom, over mud substrates and sometimes amongst weeds or adjacent to rocky areas.

3.4.6 Other Native Estuarine Species

Five species of estuarine and marine vagrants were recored at Site 1 during the February/March 2010 sampling. These included the estuarine yellowtail trumpeter (*Amniataba caudovittatus*) and black bream (*Acanthopagrus butcheri*), the marine estuarine-opportunists yellow-eye mullet (*Aldrichetta forsteri*) and yellow-fin whiting (*Sillago schomburgkii*), and the semi-anadromous Perth herring (*Nematalosa vlaminghi*). All these species are known to opportunistically enter the lower reaches of tributary rivers of south-west estuaries in search of shelter and food (*e.g. Palaemonetes australis*, planktonic crustaceans, bivalves, insects, small fish and/or detritus). Their presence in rivers like the Helena, typically coincides with high, stable salinities over summer (refer Potter *et al.* 2000).

Marine estuarine-opportunists such as yellow-eye mullet and yellow-fin whiting, regularly use estuaries as nursery areas, but the adults spend most of their life in marine environments and can complete their entire life cycle in near shore marine waters. Semi-anadromous species are marine species that need to return to estuaries or freshwater environments to spawn and recruit. In the case of Perth herring, adults return to estuaries and are typically not found far upstream or in fresh waters. The estuarine species such as black bream and yellowtail trumpeter, spend their entire life within estuarine environments (see Chubb *et al*. 1981, Lenanton *et al.* 1984 Potter & Hyndes 1999, Potter *et al.* 2000).

In south-western estuaries, including the Swan River, black bream and yellow-eye mullet are thought to have a protracted spawning period that extends from early autumn to late winter or spring. Perth Herring recruit over spring and summer, while yellow-fin whiting appear to recruit only over summer (see Chubb & Potter 1984, Lenanton *et al.* 1984, Potter *et al.* 2000).

3.4.7 Introduced Gambusia (Mosquitofish)

Large numbers of gambusia (mosquitofish) representing a range of cohorts were recorded throughout the river (Figure 11). Greatest abundances were recorded from sites immediately below the PHD and between the PHD and Mundaring Weir (Table 4, above). Gambusia are short lived with both males and females reaching sexual maturity with 4-6 weeks. Males live only for 3-6 months, attaining a maximum length of around 35mm, whereas females may live up to 15 months growing to a length of around 60 mm (MacDonald & Tonkin 2008).

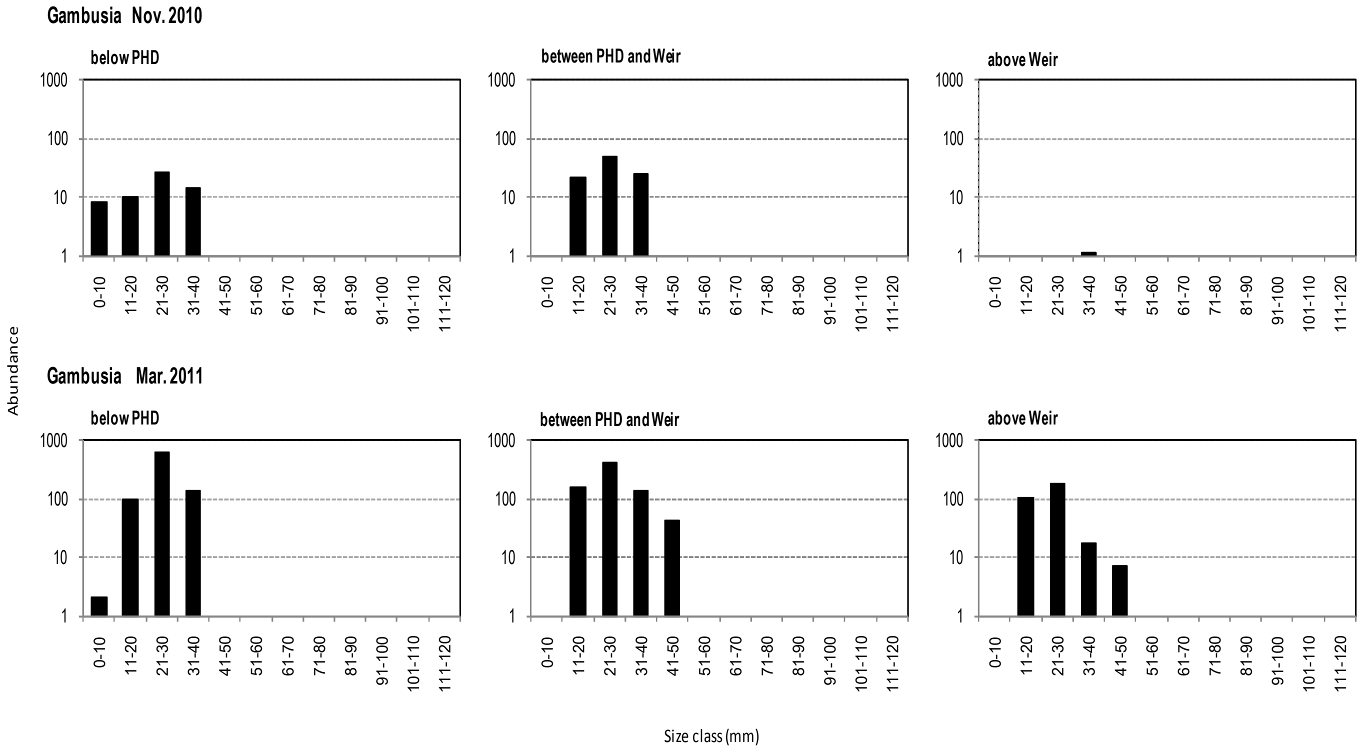

Figure 11. Length-frequency (SL mm) plots for introduced gambusia (*Gambusia holbrooki*) collected from the Helena River sub-catchments in November 2010 and February/March 2011.

Gambusia are native to central America and were introduced to fresh waters around Perth in 1936 (Mees 1977) in an attempt control mosquitoes, but with little success. Through intentional introduction and natural dispersal, they are now widespread and abundant in streams and reservoirs throughout the southwest (Morgan *et al.* 1998), dominating the fish fauna in lowland areas (Pusey *et al.* 1989).

Gambusia have been implicated in the decline of several small native species in Australian waters, primarily through their aggressive behaviour and increased competition for limiting resources (Myers

1965, Arthington & Lloyd 1989). One of the major concerns with gambusia is their ability to produce large populations in still, warm-water habitats over summer. Under favourable conditions, they can have up to 9 broods per year (MacDonald & Tonkin 2008). Gambusia have a high tolerance for environmental fluctuations and anthropogenic disturbance. They are able to tolerate extremely low DO levels (1 mg L^{-1}) (Pyke 2005), salinities 1.5x that of seawater (Morgan *et al.* 2004) and water temperatures up to 35°C (MacDonald & Tonkin 2008). Their tolerance of chemical pollutants and biocides is also relatively high when compared with other fish species (Pyke 2005). Gambusia tend to prefer shallow waters (<15 cm), and they are commonly found to occupy waters that are too shallow for many other fish (1-15 cm). Research by Gill *et al*. (1999) suggests the abundance of mosquitofish and their impact on native species is often greatly reduced in rivers with a complexity of in-stream habitats that offer a range of refugia for native fishes.

3.4.7 Introduced Redfin Perch

Three 0+ year redfin perch were recorded upstream of the PHD; two (53mm & 55mm SL) at Site 17 and one (45 mm SL) at Site 23. Redfin perch are an exotic species native to Europe. Male redfin perch reach sexual maturity in their first year, and females in their second. Peak spawning typically occurs in August-September.

They were first introduced into the state in the 1890s and have since been released as a sport fish into many south-west rivers (Morgan *et al.* 2002). Redfin perch are aggressive piscovores that are tolerant of a wide range of environmental conditions. They grow and breed more rapidly in the temperate waters of south-western Australia than in their native habitats, though they appear intolerant of water temperatures >31°C and salinities >10,000 mg L^{-1} (~15,000 µS cm^{-1}) (Morgan *et al*. 2004).

Redfin perch are known to predate heavily on both native crayfish and native fish (Morgan *et al*. 2002). They have been implicated in the decline of the native fishes and crayfish (in particular western pygmy perch and marron), through predation and competition (Morgan *et al*. 2004). In the Helena River, stable populations of redfin perch unlikely to establish outside the PHD or Mundaring Reservoir unless there are permanent deep, cool pools with suitable spawning habitat such as aquatic vegetation or sunken logs. Water temperature appears to play a crucial role in spawning which, in Australia, typically occurs when temperatures are between ~8 °C and 15 °C (Morgan *et al*. 2002).

The Department of Fisheries WA list them as a nuisance species and, if caught, it is now illegal to release them live to the water.

3.4 Relationships between Fish and Crayfish Data and Environmental Data

Significant relationships between species richness or abundance and environmental data are plotted in Figure 11. Generally, the majority of relationships were not significant, however, there were associations between the total number of native species (fish and crayfish) present at a site and the type of snag habitat (rho = 0.641, p <0.01) and the amount of large woody debris present (rho = 0.601, p <0.01). This supports the findings of Storey (1998) who reported significant relationships between species richness and both extent of riparian cover and the width of the vegetated buffer in the Canning River when sampled in February 1998. The inference being that native fish and crayfish rely on overhanging and fringing riparian vegetation which provides shade and shelter and, in forested areas, is also a source of snags and woody debris for in-stream habitat.

Storey (1998) recorded relatively strong positive correlations between percentage cover of cobble and boulder substrate and abundance of individual species such as native western minnows, nightfish and pygmy perch. The abundance of minnows was also found to be significantly related to the width of the vegetated buffer at a site. However, the current study found no such relationships for individual species. It may be that fish and crayfish of the Helena are concentrated into sub-optimal habitat due to lower water levels over summer-autumn. Indeed, the relationships found by Storey (1998) were not evident during repeat sampling of the Canning in 2006 (WRM 2007). It was postulated that this may have been because sites in the Canning did not represent a continuum of water quality or habitat conditions ranging from 'poor' to 'good', and therefore strong species-habitat correlations were unlikely to be found (WRM 2007). This is also likely the case in the Helena River. Observed fish-habitat correlations do not always indicate direct cause and effect and care must be taken with interpretations.

In contrast to Storey (1998), targeted studies by Thorburn (1999) on habitat preferences of fish in the Blackwood River, found greater occurrence and density of western pygmy perch and nightfish in sandy habitat, rather than cobble habitat. However, similar to both Storey (1998) and the current study, Thorburn (1999) found strong associations between abundances of pygmy perch and nightfish and abundance of snags and macrophytes. In any river system, it is likely that a range of inter-dependent physico-chemical factors, rather than any one variable, will influence fish abundance and distribution. There is a wealth of published literature on the local and catchment-scale variables that determine fish communities in general. For example, as noted by WRM (2007), low diversity in forested reaches on the Darling Scarp typically has nothing to do with percentage cover of riparian vegetation (or snag abundance). Nor is it limited by regulation (dams, weirs) alone. Rather, it is the naturally shallow, faster flowing habitat conditions that limit colonisation by deep-bodied fishes that require deeper permanent pools and/or slow-water environs.

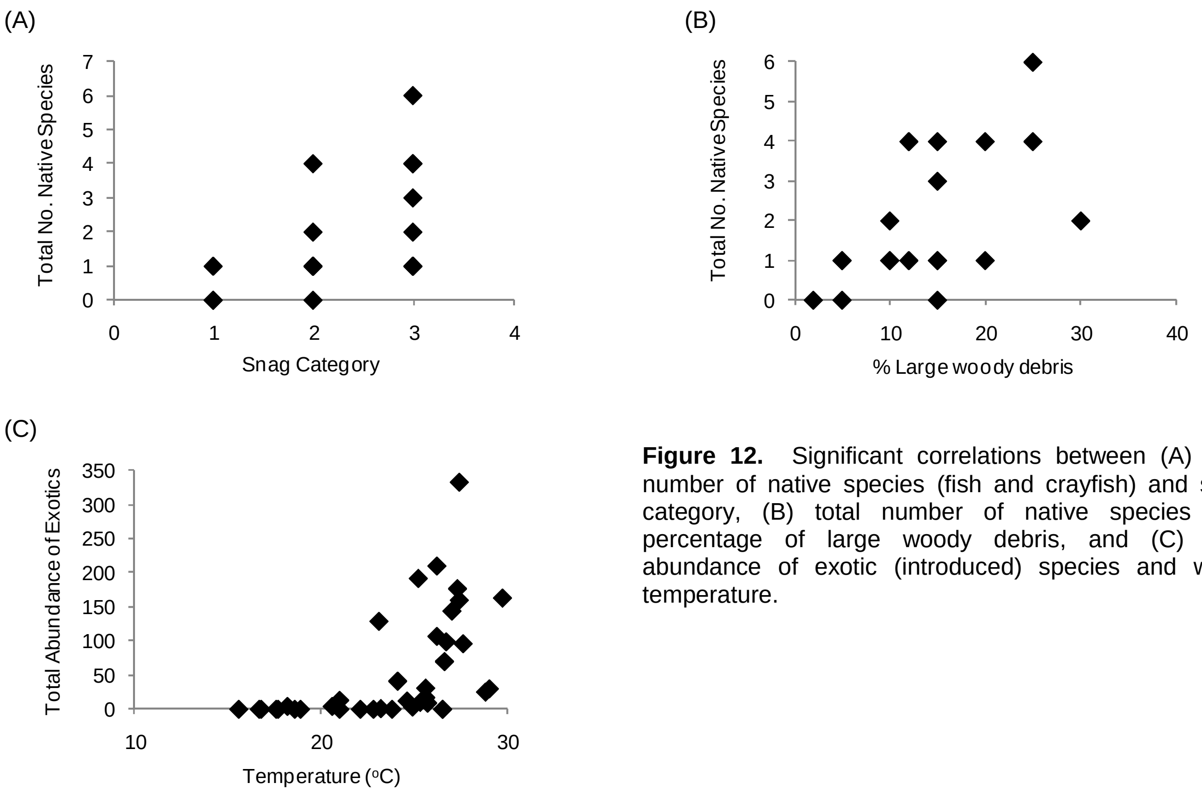

Figure 12. Significant correlations between (A) total number of native species (fish and crayfish) and snag category, (B) total number of native species and percentage of large woody debris, and (C) total abundance of exotic (introduced) species and water temperature.

The current study found a strong positive correlation between water temperature and the abundance of introduced species present at a site (rho = 0.705, p <0.01). This result reflected the dominance of gambusia and their known preference for still, shallow and hence warmer waters. High salinities were also considered to be a contributing factor to the relatively low diversity and abundance of freshwater fish and crayfish in Warrin Creek. Conversely, tidally influenced site 1 on the lower Helena River (near the confluence with the Swan River) supported greater fish species diversity due to the presence of estuarine-marine species.

Abundances of native freshwater fish and crayfish were generally low throughout the Helena River. This was unlikely due to degraded physical habitat conditions alone, as many sites were considered to be only moderately disturbed. It is highly likely that survival of fish and crayfish is limited by the prevalence of very low dissolved oxygen concentrations.

3.5 Tortoise

Opportunistic records of long-necked tortoise (*Chelodina oblonga*) were made for individuals that became trapped in fyke nets or were visually observed whilst electrofishing. Tortoise were recorded in low numbers above and below the PHD, but not above Mundaring Weir (Table 5). Juvenile tortoise as well as adults were collected at Site 1.

This species is endemic to the south-west of Western Australia, inhabiting both permanent and seasonal waterbodies. Tortoises can survive short periods of drying by either burrowing into moist, soft bottom substrates or by migrating overland to more permanent water bodies. Since tadpoles, fish, crayfish and aquatic invertebrates constitute a large part of their diet, tortoises tend to eat only when open water is present.

Table 5. Records of long-necked tortoise. Sites are arranged by sub-catchment in order of increasing distance from the confluence with the Swan River.

Sub-catchment	**Site**	**Incidental records of *Chelodina oblonga***	
		Nov. 2010	Feb./Mar. 2011
Below PHD	1	8	-
	4	-	1
	7	2	-
	18	3	1
Between PHD and Weir	16	1	-
	17	-	1

Where permanent water is available, long-necked tortoise may nest twice during the breeding season; *i.e.* in September-October and again in December-January (Cogger 2000). In seasonal systems, nesting typically only occurs during spring. Males are sexually mature at a carapace length of about 140 mm, while females mature at around 160-170mm (Kuchling 2006, Browne-Cooper *et al*. 2007). Nesting sites are typically located some distance away from water bodies in more open sandy areas. If water quality or physical habitat conditions are poor, females can indefinitely halt breeding.

4 CONCLUSIONS

Based on data collected during the course of the current study, the following conclusions are drawn:

Macroinvertebrates

- The macroinvertebrate fauna of the Helena River is generally typical of other seasonally flowing south-west rivers in rural and urban catchments. That is, dominated by insects, with a high proportion of dipterans (two-winged flies) and aquatic beetles.
- There are few sensitive taxa (EPT taxa) with a predominance of cosmopolitan species (only 13% were considered to be south-west endemics). Sensitive taxa that are present tend to occur with greater frequency and abundance at sites which possess relatively better water quality, in-stream habitat and riparian vegetation cover (*e.g*. sites 18, 19 & 21). The paucity of EPT taxa at sites between the Lower Helena pipehead dam and Mundaring Weir, was considered due to the small size and shallowness of remnant pools in these reaches over summer.
- Predators (such as aquatic beetles and some dipterans) were the dominant functional feeding guild at the majority of sites sampled. The ideal 'healthy' aquatic ecosystem would have greater representation of other functional feeding guilds. A high ratio of predators has been described in overseas studies as correlated to high benthic organic matter which benefits prey species with short-life cycles and consequently high turn-over rates, which in turn supports high numbers of predators (Yamauro & Lamberti 2007).
- Sites above and below the PHD support similar species richness, however community assemblages differ, suggesting the flow regulation to be influential in structuring communities. Community differences were due to differences in richness and abundance of a large number of taxa, each contributing a small amount to the overall variation. No one species or group of species (*e.g.* EPT taxa, functional feeding group) could be used solely to account for differences above and below the PHD.
- One listed species, the freshwater mussel *Westralunio carteri*, recorded from sites 18 and 21. This species is listed as vulnerable (VU) by the IUCN (2009) and as Priority 4 (P4) by the Department of Environment and Conservation, WA (DEC). Mussels have an early larval phase that is parasitic on the gills of native freshwater fish (*e.g*. cobbler). As such, barriers to fish migration (*e.g.* PHD & Mundaring Weir) may restrict restrict gene flow between mussel populations, limit upstream-downstream recruitment of mussels, restrict distributions and prevent recolonisation.

Fish and Crayfish

- The Lower Helena, downstream of the PHD, and in particular downstream of Craignish Weir, supports poor diversity (richness) and abundance of native freshwater fish and crayfish (Figure 13). A greater diversity of fish occurs near the confluence with the Swan River due to the presence of five species of estuarine and marine vagrants;
- Sites between the PHD and Mundaring Weir, consistently support high diversity but low abundances of native species, and an increasing abundance of exotics (gambusia, redfin perch and yabbies);
- Sites in the main channel of the Upper Helena, above Mundaring Weir, support high diversity and high abundance of native fish and crayfish. Only two sites were found to have high abundance of exotic gambusia;
- Warrin Creek supports only low diversity and abundance of fish and crayfish, more typical of lowland sites in the main channel of the Helena River. Very high salinities are at least one factor likely influencing the low diversity.

- Numbers of native species (fish and crayfish) appear to be positively correlated with the type of snag habitat and the amount of large woody debris present. This is in accord with other studies of south-west rivers that have found similar relationships between species richness and abundance of snags, macrophytes, riparian cover and/or vegetated buffer widths (see Storey 1998, Thorburn 1999).
- Hypoxia (and likely overnight anoxia) in the lower and middle reaches is a major concern. It is caused by high biological oxygen demand (BOD) coupled with in-filling of river pools by fine organic sediments. Adequate concentrations of dissolved oxygen (DO) are fundamental for the survival of aquatic species and for the maintenance of ecological processes. Hypoxia and anoxia not only causes the localised extinction of fauna, but also results in desorption (release) of nutrients (*e.g.* phosphorus & ammonium) and heavy metals (*e.g.* from fertilizers or industrial chemicals) bound in sediments causing further water quality problems.

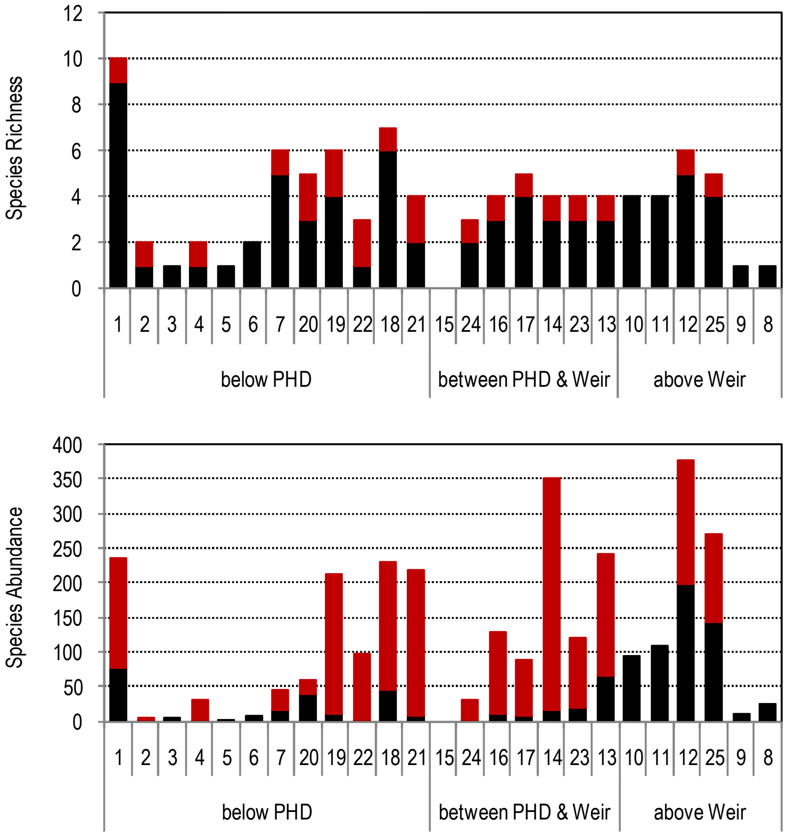

Figure 13. Summary plots for comparison of total species (fish and crayfish) richness (top) and abundance (bottom) amongst sites and indicating proportional contribution of native (black) and exotic (red) species. Sites are arranged by distance from the confluence with the Swan River. Note: comparisons use data from <u>all</u> capture methods employed in November 2010 and February/March 2011.

5 RECOMMENDATIONS FOR MANAGEMENT

Major Issues

Based on results from the current study, the major issues with respect to management appear to be:

1. Maintenance of DO levels above thresholds (*i.e.* 50-60% saturation) known to cause stress in aquatic biota, particularly fish. This pertains mainly to the reaches below the PHD and between the PHD and Mundaring Weir;
2. Control of exotic species (gambusia, redfin perch and yabbies);
3. Restoration of riparian vegetation buffer zones in reaches below the PHD and reintroduction of rush/sedge communities to improve habitat availability for aquatic fauna.

It is therefore recommended that management actions target reaches below Mundaring Weir.

Recommended Management Actions

<u>Maintenance of DO Levels – Flushing Flows</u>

There is a need for flushing flows to be provided to the middle and lower reaches of the Helena River. Such flows would help remove accumulated organics and inorganics from the river pools, thereby reducing biological oxygen demand and increasing pool volume. This in turn will limit hypoxic and anoxic events and lessen the risk of asphyxia to aquatic fauna.

It is recommended that the feasibility of ecological releases from Mundaring Weir should be investigated in collaboration with the Water Corporation. Such releases have already been recommended to the Department of Water (DoW) and the Water Corporation in a recent study by WRM (2010) which found a minimum baseflow of ~9 L sec^{-1} would be required to maintain DO levels in the Helena River between Mundaring Weir and Scott Street road bridge. It may be possible to ameliorate poor quality hypolimnetic water in Mundaring Reservoir if it were first treated by passing through a biological filter, such as an artificial wetland, constructed to resemble a natural wetland (WRM 2010). This type of 'passive' remediation of poor quality water is achieved largely through microbial activity, sustained by organic detritus generated from a large biomass of wetland plants. The wetland would need to be of sufficiently large size to provide adequate residence time for remediation. Waters from the wetland could then be discharged *via* gravity-flow to the downstream river channel. It may be possible to establish such a wetland alongside the river channel between the base of the Weir and Mundaring Weir Road, close to the pump stations, where semi-wetland conditions already exist.

As well as preventing fish and crayfish deaths, It is expected that generally higher dissolved oxygen levels (*i.e.* >50-60%) will provide better habitat for native fish and crayfish and lead to increased biodiversity. Flushing should also limit the rate of pool aggradation and provide greater capacity to support increased species richness, abundance and biomass.

The provision of flushing flows is considered vital for improved ecosystem health and biodiversity in the middle and lower reaches. Restoration of other habitat features such as natural vegetation, large woody debris *etc*, will be of little aid if hypoxia and anoxia persist due to an inadequate flow regime in regulated reaches.

<u>Other Water Quality Issues</u>

It is recommended that seasonal water quality data be collected and analysed to determine if pollutants (*e.g*. pesticides, herbicides, heavy metals, hydrocarbons) may also be responsible for the paucity of fish and crayfish in the lower and middle reaches. If budgets permit, it is recommened that water quality is sampled once a month for a minimum two years to provide a baseline data set. To limit costs, not all of the 20 sites sampled for the current study would need be to sampled for water quality. A sub-set taken from each sub-catchment could be used, the exact number dependent on budget. It is however recommended that a minimum of at least 3 sites from each sub-catchment be included. Preference should be given to sites known to be in the receiving environment for point and/or diffuse source pollutants.

Control of Gambusia

In Australia, there is little published research on the successful control, let alone eradication, of gambusia from river systems (MacDonald & Tonkin 2008). Manual removal (*e.g.* by fishing out an entire site, or poisoning with rotenone) has successfully eradicated gambusia from some relatively enclosed systems such as small creeks, wetlands, lakes and dams. However, it is unlikely to be successful in larger rivers if only a few sites are targeted; gambusia will readily re-invade from either upstream or downstream. More practical options are to:

i) Control numbers of gambusia by limiting the availability of still, shallow, warm, open habitat favoured for breeding (refer Pusey *et al*. 1989, Fairfax *et al*. 2007, MacDonald & Tonkin 2008. Release of flushing flows over summer will help achieve this.

ii) Enhance habitat and cover for native fishes by re-planting streamside vegetation, encouraging growth of native aquatic plants and sedges (Gill *et al.* 1999, Pyke 2008), and re-introducing structure such as rocks, logs (and/or railway sleepers)and branches. This will also improve habitat for macroinvertebrates which would increase food availability for native fish and crayfish. It will also help reduce high summer water temperatures (*i.e*. >30^{o}C) which can be lethal to native fish, but which are readily survived by gambusia.

Control of Yabbies

There has been no research into the control of introduced yabbies in natural waterways. It would be extremely difficult to totally eradicate them from the Helena River if populations are more widely established than recorded in the current study. If only isolated populations exist, it may be possible to eradicate them by concerted trapping efforts coupled with public education campaigns. Yabbies are also capable of migrating long distances overland and will readily escape from farm dams into the river. Recommended management actions are:

i) Control numbers of yabbies by targeted trapping when water levels are low, thereby maximising the number of individuals caught.

ii) Public education campaign on the differences between yabbies and native crayfish, stressing that yabbies should not be released into the river system.

iii) Provision of flushing flows to improve water quality and increase availability of habitat for native marron and gilgies.

Control of Redfin Perch

It is likely that redfin are being illegally stocked in the PHD (and probably Mundaring Reservoir). Numbers in the PHD can be controlled by targeted trapping when water levels are low. Public education campaigns should also be used to alert people that it is now illegal to release them live into dams and waterways.

Restoration of Riparian and In-stream Vegetation

Native fishes are generally associated with instream habitat, *i.e.* reaches with deeper pools with rocks or snags and some degree of shade from either riparian vegetation and/or aquatic macrophytes. Increased biodiversity and abundance of native fish species is typically associated with increased heterogeneity of habitat. The presence of crayfish is more typically influenced by seasonality of flows and water quality, than by habitat complexity. Marron require permanent water whereas gilgies and yabbies occur in either seasonal or permanant waters.

As well as providing streamside shade and shelter, large (≥30 m) vegetation buffer zones can improve water quality by reducing the flow of overland runoff and the input of nutrients, chemical pollutants and organic and inorganic sediments. The can also reduce air-borne drift of pesticides and herbicides and wind-borne weed seeds.

Vegetation restoration should focus on those sites where there is little or no riparian vegetation and/or few native rush/sedge communities. This may require preparatory earthworks at sites where the channel is down-cut and river banks are steep and require stabilisation. Practicalities of access for earth moving equipment will determine which sites are most suitable for this type of restoration. It is also extremely cost effective to protect those sites that still retain remnant vegetation. Work on downstream sites will also be far more successful if upstream sites are in good condition.

The re-introduction of riparian vegetation and aquatic macrophytes should only be done after flushing flows have been re-established. These flows will help prevent excessive encroachment of the channel (such as already present at sites 16 & 17), which can exacerbate in-stream habitat loss and night-time anoxia.

It is recommended that EMRC contact the River Restoration Action Team (River RATs) at the DoW for specific information and support for restoration of in-stream and riparian vegetation zones.

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APPENDIX

Table A1. Water quality at each of the sites sampled in 2010 and 2011. Sites are arranged by sub-catchment and in order of increasing distance from the confluence with the Swan River.

Sub-catchment	Site	Date	Time	pH	EC µS cm^{-1}	DO mg L^{-1}	DO %	Temp °C
Below PHD, below Craignish weir	1	15/11/2010	0930	7.6	10,100	6.1	63.4	17.6
Below PHD, below Craignish weir	1	28/02/2011	1100	7.2	44,400	1.7	27.4	27.4
Below PHD, below Craignish weir	2	16/11/2010	1000	6.7	1,620	0.6	14.6	18.2
Below PHD, below Craignish weir	3	16/11/2010	0900	6.4	1,456	0.3	3.8	16.7
Below PHD, below Craignish weir	4	15/11/2010	1400	6.9	1,126	3.7	40.2	15.6
Below PHD, below Craignish weir	4	28/02/2011	1230	7.2	534	0.8	10.4	25.6
Below PHD, below Craignish weir	5	15/11/2010	1300	6.8	1,103	1.8	19.1	17.7
Below PHD, below Craignish weir	6	16/11/2010	1300	7.4	1,338	2.7	34.1	18.6
Below PHD, below Craignish weir	7	16/11/2010	1145	8.2	1,958	10.6	95.4	20.6
Below PHD, below Craignish weir	7	28/02/2011	1530	7.4	1,258	1.3	22.0	28.8
Below PHD, above Craignish weir	20	19/11/2010	1430	7.1	683	5.0	63.1	24.9
Below PHD, above Craignish weir	20	1/03/2011	1000	7.3	927	1.3	12.3	25.6
Below PHD, above Craignish weir	22	2/03/2011	1600	7.2	995	1.9	18.8	27.6
Below PHD, above Craignish weir	19	19/11/2010	1330	7.2	803	3.0	39.6	25.3
Below PHD, above Craignish weir	19	1/03/2011	0800	7.3	939	1.5	16.8	25.2
Below PHD, above Craignish weir	18	19/11/2010	1100	7.5	673	4.6	52.1	24.1
Below PHD, above Craignish weir	18	1/03/2011	1200	7.4	926	1.5	19.4	27.0
Below PHD, above Craignish weir	21	2/03/2011	1500	7.7	882	2.7	43.4	26.2
Between PHD & Mundaring Weir	15	19/11/2010	0930	7.7	1,139	7.4	94.8	26.5
Between PHD & Mundaring Weir	15	1/03/2011	1700	7.0	102	2.1	36.9	33.5
Between PHD & Mundaring Weir	24	3/03/2011	1300	7.1	1,020	2.5	20.4	29.0
Between PHD & Mundaring Weir	16	18/11/2010	1430	7.4	882	9.5	9.4	24.6
Between PHD & Mundaring Weir	16	1/03/2011	1500	7.0	1,304	2.6	41.2	26.2
Between PHD & Mundaring Weir	17	18/11/2010	1330	7.2	938	8.3	101.2	26.6
Between PHD & Mundaring Weir	17	2/03/2011	1030	7.4	1446	1.6	15.4	25.7
Between PHD & Mundaring Weir	14	18/11/2010	1230	7.2	885	7.8	104.1	26.5
Between PHD & Mundaring Weir	14	2/03/2011	1200	7.9	1851	5.8	73.9	27.4
Between PHD & Mundaring Weir	23	3/03/2011	1100	7.9	1,742	3.7	50.4	26.7
Between PHD & Mundaring Weir	13	18/11/2010	1120	7.2	918	4.0	44.4	21.0
Between PHD & Mundaring Weir	13	2/03/2011	1030	7.7	1,207	3.6	51.8	29.7
Above Mundaring Weir	10	17/11/2010	1200	6.6	3,110	5.3	55.3	21.0
Above Mundaring Weir	11	17/11/2010	1315	7.6	3,110	8.2	91.3	22.1
Above Mundaring Weir	11	4/03/2011	1430	7.5	3,410	4.3	64.5	23.8
Above Mundaring Weir	12	17/11/2010	1430	7.6	3,160	7.8	94.1	23.2
Above Mundaring Weir	12	3/03/2011	1500	7.8	3,750	5.1	66.6	27.3
Above Mundaring Weir	25	4/03/2011	1100	7.4	3,340	5.2	58.5	23.1
Above Mundaring Weir, Warrin Ck	9	17/11/2010	1115	6.9	18,690	5.0	54.2	16.8
Above Mundaring Weir, Warrin Ck	8	17/11/2010	1030	5.6	7,740	7.3	69.6	18.9
Above Mundaring Weir, Warrin Ck	8	4/03/2011	0930	7.5	2,340	3.2	34.5	22.8

Table A2. Habitat data collected from each of the sites in November 2010. Sites are arranged by sub-catchment and in order of increasing distance from the confluence with the Swan River.

Site	% Channel	% Riffle	% Pool	Ave depth (m)	Max depth (m)	Ave Width (m)	Max width (m)	Pool area (m2)	Riparian Cover (%)	Bedrock (%)	Boulders (%)	Cobbles (%)	Pebbles (%)	Gravel (%)	Sand (%)	Silt (%)	Clay (%)	Leaf litter/mats (%)	Emergent Macrophyte (%)	Submergent Macrophyte (%)	Floating macrophyte (%)	Algal cover (%)	Draped riparian vege (%)	Root mats (%)	LWD (%)	Snags abundance class	Snag complexity code	Organics (%)	Buffer (m)	Bank Angle (°)	Undercut (%)	Backwaters (p/a)
1	50	0	50	0.5	0.6	8.5	11.5	160	40	0	0	0	0	0	95	5	0	1	5	0	0	0	0	0	15	2	2	79	1	38	0	0
2	85	0	15	0.3	0.5	11.0	11.5	2	5	0	0	0	0	0	95	5	0	25	15	0	0	0	10	0	10	3	3	40	0	30	0	0
3	20	0	80	0.5	0.7	12.0	14.1	28	30	0	0	0	0	0	80	20	0	15	15	0	0	0	10	0	20	3	3	40	0	70	25	0
4	10	0	90	0.6	1.5	6.5	7.5	90	35	0	0	0	0	0	65	5	30	20	2	0	0	0	2	0	5	2	2	71	1	50	0	0
5	30	0	70	0.7	2.0	7.0	11.1	90	70	0	0	0	0	0	65	5	30	20	3	0	0	0	2	0	15	2	2	60	1	55	0	0
6	50	0	50	0.4	0.7	5.5	9.0	84	45	0	0	0	0	0	20	50	30	25	15	0	0	0	2	0	30	3	3	28	1	60	2	0
7	70	0	30	0.5	1.5	8.0	9.3	90	30	0	1	1	1	0	40	37	20	45	5	0	0	0	0	0	25	3	3	25	1	45	0	0
20	0	0	100	0.5	0.8	10.6	11.0	560	15	5	10	15	5	0	20	35	10	25	11	4	0	0	2	0	10	2	2	48	30	75	0	0
19	0	0	100	0.4	0.6	7.5	8.0	490	15	0	40	15	5	0	10	25	5	14	0	0	0	0	1	0	20	3	3	65	30	55	0	0
18	0	0	100	1.2	2.2	15.0	17.0	1200	10	45	20	5	0	0	15	15	0	10	0	5	0	0	3	0	25	3	3	57	30	60	5	0
15	0	0	100	0.3	0.5	14.0	15.0	105	0	74	1	2	3	0	0	20	0	7	0	3	0	0	0	0	2	1	2	88	30	20	0	0
16	0	0	100	0.6	1.0	12.0	14.0	360	10	8	17	0	2	3	35	30	5	10	20	40	0	5	3	0	12	3	3	10	30	50	5	0
17	0	0	100	0.5	1.9	15.0	16.0	1500	10	6	4	0	0	0	80	10	0	10	15	55	0	0	2	0	10	2	3	8	2	45	0	0
14	0	0	100	1.5	2.0	28.0	35.5	875	2	5	5	2	0	0	78	10	0	10	10	35	0	0	5	0	5	1	2	35	30	40	0	0
13	0	0	100	0.7	1.8	9.0	12.3	180	10	0	0	0	0	0	55	40	5	30	8	0	0	0	12	0	15	3	3	35	30	50	5	0
10	60	0	40	0.4	0.7	7.5	9.0	260	60	0	0	0	0	0	30	50	20	15	2	0	0	0	1	0	70	3	2	12	2	8	0	0
11	70	0	30	0.2	0.5	2.3	7.2	63	25	0	0	0	0	0	85	15	0	10	30	0	0	0	1	0	15	3	3	44	30	65	5	0
12	0	0	100	0.8	1.2	9.3	12.0	340	15	100	0	0	0	0	0	0	0	20	3	0	0	0	5	0	12	2	2	60	30	25	5	1
9	0	0	100	0.3	0.5	5.0	6.0	30	30	0	0	0	0	0	50	20	30	40	20	0	0	0	14	0	15	2	2	10	30	80	0	0
8	0	0	100	0.5	0.6	2.0	2.6	15	0	0	0	0	0	0	60	20	20	10	15	0	0	0	5	0	5	1	2	65	30	80	40	0

Table A3. Aquatic macroinvertebrates recorded from the Helena River during February/March 2011.

Values are $\log_{10}$ abundance classes, where 1 = 1 individual, 2 = 2-10 individuals, 3 = 11-100, 4 = 101-1000 *etc.* Codes: Cons. Cat. = conservation category, where A = Australia, C = cosmopolitan (Australia and beyond, however not necessarily worldwide), I* = indeterminate but probably occurs only in south-west of WA, I = indeterminate, S = endemic to southern half of WA, SW = endemic to the south-west of WA. FFG = functional feeding group, where 1 = Collectors, 2 = Shredders, 3 = Filterers, 4 = Grazers, 5 = Predators, 6 = Other/unknown. (L) = larva; (F) = female; (imm.) = immature life stage.

Phyla/Class/Order	Family/Sub-family	Species	Cons. Cat.	FFG	Site 13	Site 14	Site 15	Site 16	Site 18	Site 19	Site 20	Site 21	Site 22	Site 23
NEMATODA		Nematoda sp.	I	6	–	–	–	–	–	–	–	1	–	–
ANNELIDA														
OLIGOCHAETA		Oligochaeta spp.	I	1	3	2	–	2	–	–	2	–	1	2
MOLLUSCA														
GASTROPODA	**Ancylidae**	*Ferrissia petterdi*	A	4	–	–	–	–	–	–	–	–	–	1
	Physidae	*Physa acuta*	A	4	–	2	–	1	3	–	2	2	–	–
CRUSTACEA														
CLADOCERA		Cladocera sp.	I	3	–	–	–	–	–	–	–	–	–	–
COPEPODA		Calanoida sp.	I	3	1	–	–	–	3	–	–	–	–	–
	Cyclopoida	Cyclopoida sp.	I	3	–	–	–	–	4	–	4	2	–	–
		Harpacticoida sp.	I	3	1	–	–	–	–	–	–	–	–	–
OSTRACODA		Ostracoda spp.	I	3	2	2	–	2	3	3	3	2	2	1
DECAPODA	**Palaemonidae**	*Palaemonetes* sp.	SW	1	–	3	–	–	3	–	–	2	2	2
	Parastacidae	Parastacidae sp. (imm.)	I	1	–	–	–	–	–	–	–	–	–	2
		Cherax quinquecarinatus	SW	1	–	–	–	–	–	–	–	–	–	2
ARACHNIDA														
ACARINA		Hydracarina spp.	I	6	–	–	–	1	–	–	2	–	2	–
	Oribatidae	Oribatidae spp.	I	6	–	–	–	1	–	1	2	–	1	–
INSECTA														
EPHEMEROPTERA	**Baetidae**	Baetidae sp.	I	2	–	–	–	–	–	–	–	–	–	2
		Cloeon sp.	I	2	–	–	–	–	4	–	–	2	–	–
	Caenidae	*Tasmanocoensis tillyardi*	A	2	–	–	–	–	2	1	–	3	1	2
	Leptophlebiidae	*Atalophlebia* sp. AV17	S	2	–	–	–	–	–	–	–	2	–	–
ODONATA														
Zygoptera	**Zygoptera**	Zygoptera (imm.)	I	5	–	1	–	–	1	–	–	–	1	–
	Coenogriondae	*Ischnura* sp.	A	5	–	–	–	–	1	–	–	–	–	–
Anisoptera		*Anisoptera* spp. (imm.)	I	5	–	1	–	–	2	1	–	2	–	–
	Aeshnidae	*Aeshna (Adversaeschna) brevistyla*	A	5	–	1	–	–	–	–	–	–	1	–
	Hemicorduliidae	*Hemicordulia australiae*	A	5	1	–	–	2	–	–	–	–	–	–
		Hemicordulia tau	A	5	–	2	–	2	–	–	–	–	–	–
		Procordulia affinis	SW	5	–	–	–	1	–	–	–	–	–	–
	Libellulidae	*Orthetrum caledonicum*	A	5	–	1	–	–	–	–	–	–	–	1
HEMIPTERA	**Corixidae**	*Sigara* (*Tropocorixa*) *mullaka*	SW	5	–	–	–	–	1	–	–	–	–	–
		Micronecta robusta	A	5	–	–	1	–	1	–	–	–	–	–
	Nepidae	*Laccotrephes* (*Laccotrephes*) *tristis*	A	5	–	–	–	1	–	–	–	–	–	–
COLEOPTERA	**Dytiscidae**	*Hyphydrus* sp.	I	5	–	3	–	–	–	–	–	–	–	–
		Hyphydrus elegans	A	5	–	–	–	1	–	1	–	–	–	–
		Lancetes lanceolatus	A	5	–	–	–	–	–	–	–	–	–	2
		Megaporus sp.	I	5	–	2	–	–	–	–	–	1	–	1
		Megaporus solidus	SW	5	–	–	–	–	–	1	–	–	–	–
		Necterosoma darwini	S	5	–	3	–	–	–	1	–	–	–	–
		Necterosoma penicillatum	A	5	–	3	–	–	1	2	–	2	1	–
		Onychohydrus scuttelaris	A	5	–	1	–	–	–	–	1	–	–	–
		Platynectes (Platynectes) sp.	A	5	–	–	–	1	–	–	–	–	–	–
		Rhantus suturalis	A	5	–	–	–	–	–	1	–	–	–	–
	Gyrinidae	*Macrogyrus* sp. WRM01	I	5	–	–	–	–	2	–	–	–	–	–
		Macrogyrus (*Tribologyrus*) *australis*	A	5	–	2	–	2	2	–	–	–	–	–
	Hydrophilidae	*Helochares* sp. (L)	I	5	1	1	–	2	–	–	–	1	–	1
		Helochares (*Hydrobaticus*) *tenuistratus*	SW	5	–	1	–	–	–	–	–	–	–	–
		Hydrophilus (*Hydrophilus*) *albipes*	A	5	2	1	–	–	–	–	–	–	–	–

Table A3 continued.

Phyla/Class/Order	Family/Sub-family	Species	Cons. Cat.	FFG	Site 13	Site 14	Site 15	Site 16	Site 18	Site 19	Site 20	Site 21	Site 22	Site 23
		Paracymus pygmaeus	A	5	–	–	–	1	–	–	–	–	–	–
DIPTERA	**Chironomidae**	Chironomidae sp. (pupa)	I	6	–	–	–	–	–	–	–	–	1	–
	Chironominae	*Chironomus* aff. *altermans*	C	1	3	1	–	1	–	–	3	–	–	–
		Cladopelma curtivalva	A	5	2	–	–	–	–	–	–	–	1	–
		Cladotanytarsus sp. (VSC12)	I	1	–	1	–	–	4	–	–	3	3	2
		Dicrotendipes sp. (V47)	S	1	–	–	–	–	2	–	–	–	–	2
		Kiefferulus intertinctus	C	3	3	–	–	1	–	2	2	–	1	–
		Kiefferulus martini	C	3	2	–	–	–	–	–	–	–	–	–
		Parachironomus sp. (VSCL35)	I	6	–	–	–	–	3	–	–	–	–	–
		Polypedilum sp. (VSCL8)	I	1	–	–	–	–	–	–	–	–	–	2
		Tanytarsus fuscithorax	I	3	4	3	–	2	3	3	2	3	3	3
	Orthocladinae	*Nanocladius* sp. (sp. 1 of Cranston)	A	6	–	–	–	–	2	–	–	–	–	–
		Parakiefferiella sp. (near *variegatus*)	I	5	–	–	–	–	2	–	–	–	–	–
	Tanypodinae	*Paramerina levidensis*	C	5	–	–	–	–	1	–	–	–	–	–
		Procladius paludicola	A	5	3	2	–	2	–	–	–	–	–	2
	Ceratopogonidae	Ceratopogonidae sp. (pupa)	I	5	–	–	–	–	–	–	1	–	–	–
		Ceratopogoniinae spp.	I	5	–	2	2	2	2	1	2	–	2	2
		Dasyheleinae sp.	I	5	–	–	1	1	–	–	1	–	–	–
	Tabanidae	Tabanidae sp.	I	5	–	–	–	1	–	–	–	–	–	1
	Tipulidae	Tipulidae sp.	I	5	–	–	–	–	1	–	1	–	–	–
TRICHOPTERA	**Hydroptilidae**	*Acritoptila*/*Hellyethira* sp.	I*	4	–	–	–	–	3	–	–	–	–	2
	Leptoceridae	*Oecetis* sp.	I*	5	–	–	–	–	3	–	–	–	–	–
		Triplectides australis	A	2	–	–	–	–	2	–	–	1	3	–
		Triplectides sp. AV1	SW	2	–	–	–	–	–	–	–	1	–	–
		Total number of 'species'			***13***	***23***	***3***	***21***	***27***	***12***	***14***	***16***	***16***	***20***

Table A4. Aquatic macroinvertebrates recorded by WRM (2010) from the Helena River in May 2010. Other than WRM Site 9, sites correspond to those sampled during the current study.

Values are $\log_{10}$ abundance classes, where 1 = 1 individual, 2 = 2-10 individuals, 3 = 11-100, 4 = 101-1000 *etc.* Codes: Cons. Cat. = conservation category, where A = Australia, C = cosmopolitan (Australia and beyond, however not necessarily worldwide), I* = indeterminate but probably occurs only in south-west of WA, I = indeterminate, S = endemic to southern half of WA, SW = endemic to the south-west of WA. FFG = functional feeding group, where 1 = Collectors, 2 = Shredders, 3 = Filterers, 4 = Grazers, 5 = Predators, 6 = Other/unknown. (L) = larva; (F) = female; (imm.) = immature life stage.

Phyla/Class/Order	Family/Sub-family	Species	Cons. Cat.	FFG	Site 18	Site 19	Site 20	Site 21	Site 22	WRM Site 9
CNIDARIA										
HYDROZOA	**Hydroida**	*Hydra* sp.	I	5	–	–	–	–	4	–
ANNELIDA										
OLIGOCHAETA		Oligochaeta spp.	I	1	–	1	–	1	2	–
MOLLUSCA										
GASTROPODA	**Ancylidae**	*Ferrissia petterdi*	A	4	3	–	2	3	1	–
	Hyriidae	*Westralunio carteri*	IUCN VUN	3	3	–	–	2	–	–
	Lymnaeidae	*Austropeplea lessoni*	A	4	1	–	–		–	–
	Physidae	*Physa acuta*	A	4	3	–	3	2	3	3
CRUSTACEA										
CLADOCERA		Cladocera spp.	I	3	4	5	5	4	4	4
COPEPODA		Cyclopoida spp.	I	3	4	5	4	4	3	3
		Calanoida spp.	I	3	5	5	4	3	5	5
OSTRACODA		Ostracoda spp.	I	3	3	3	4	3	3	2
DECAPODA	**Palaemonidae**	*Palaemonetes* sp.	SW	1	2	–	2	–	–	2
	Parastacidae	*Cherax quinquecarinatus*	SW	1	2	–	–	2	–	–
		Cherax cainii	SW	1	–	–	–	1	–	–
		Cherax destructor	Exotic	1	–	–	–	2	–	1
ARACHNIDA										
ACARINA		Hydracarina sp.	I	6	–	–	1	–	–	1
COLLEMBOLA		Collembola	I	2	–	–	–	1	–	–
INSECTA										
EPHEMEROPTERA	**Baetidae**	Baetidae sp. (imm.)	I	2	–	1	–	–	–	–
		?*Cloeon* sp.	I	2	3	2		2	1	–
	Caenidae	*Tasmanocoenis tillyardi*	A	2	4	2	1	4	2	–
	Leptophlebiidae	*Atalophlebia* sp. AV17	S	2	2	2	–	3	2	–
ODONATA										
Zygoptera		Zygoptera sp. (imm.)	I	5	–	3	–	–	–	–
	Coenogriondae	*Austroagrion sp.*	A	5	–	–	2	–	–	–
		Ischnura sp.	A	5	–	–	–	–	1	–
		Coenogrionidae spp. (imm.)	I	5	–	–	–	1	–	–
Anisoptera		Anisoptera spp. (imm.)	I	5	2	3	–	–	–	–
	Gomphidae	*Austrogomphus (Austrogomphus) collaris*	S	5	2	–	–	–	–	–
	Aeshnidae	*Aeshna (Adversaeschna) brevistyla*	A	5	–	1	2	1	2	–
	Libelluloidea	*Hesperocordulia berthoudi*	SW	5	–	–	2	–	–	–
		Orthetrum caledonicum	A	5	1	–	–	–	–	–
		Libelluloidea spp. (imm.)	I	5	2	2	2	2	4	2
	Telephlebiidae	*Austroaeschna anacantha*	SW	5	–	–	–	–	–	1
HEMIPTERA	**Corixidae**	Corixidae spp. (imm.)	I	5	–	3	3	3	3	2
		Micronecta robusta	A	5	–	2	1	–	1	
	Notonectidae	Notonectidae sp. (imm.)	I	5	–	–	2	–	–	–
COLEOPTERA	**Dytiscidae**	*Antiporus* sp. (F)	I	5	–	2	–	–	–	–
		Hyphydrus sp. (L)	I	5	–	2	–	–	3	3
		Hyphydrus elegans	A	5	–	1	–	–	2	2
		Limbodessus shuckhardi	A	5	–	2	–	–	–	3
		Megaporus solidus	SW	5	1	–	–	1	–	–
		Necterosoma sp. (L)	I	5	–	3	–	–	3	3
		Necterosoma darwini	S	5	–	2	–	1	–	–
		Rhantus suturalis	A	5	2	2	–	–	–	2
		Sternopriscus browni	SW	5	1	–	–	1	2	–

Table A4 continued.

Phyla/Class/Order	Family/Sub-family	Species	Cons. Cat.	FFG	Site 18	Site 19	Site 20	Site 21	Site 22	WRM Site 9
	Gyrinidae	*Macrogyrus* (*Tribologyrus*) *australis*	A	5	1	–	–	–	–	–
		Aulonogyrus/Macrogyrus sp. (L)	I	5	2	2	2	3	1	2
	Haliplidae	*Haliplus gibbus/fuscatus* (F)	I	5	–	1	1	–	–	1
	Hydrophilidae	*Berosus* sp. (L)	I	5	–	–	1	–	–	–
		Enochrus (*Methydrus*) *elongatus*	A	5	–	–	1	–	–	–
		Helochares (*Hydrobaticus*) *tatei*	A	5	–	–	–	–	1	–
		Hybogralius sp. (L)	SW	5	–	–	–	–	2	–
		Paracymus sp. (L)	A	5	–	–	1	–	–	–
	Hygrobiidae	*Hygrobia* ?*wattsi*	SW	5	–	–	4	–	–	–
	Scirtidae	Scirtidae sp. (L)	I	3	–	–	1	–	–	–
DIPTERA	**Chironomidae**	Chironomidae spp. (pupa)	I	6	2	2	–	2	1	2
		Chironomidae sp. V53	I	6	–	–	–	–	–	2
		Chironomidae sp. V83	I	6	2	–	–	–	–	–
		Chironomidae sp. VCD2	I	6	–	–	–	1	–	–
	Chironominae	*Chironomus* aff. *altermans*	C	1	–	3	4	–	2	3
		Cladopelma curtivalva	A	5	2	3	2	3	3	2
		Cladotanytarsus sp. (VSC12)	I	1	3	–	–	3	–	–
		Cryptochironomus griseidorsum	C	5	–	–	–	1	1	–
		Dicrotendipes sp. (V47)	S	1	2	2	–	–	–	–
		Kiefferulus intertinctus	C	3	2	1	2	–	2	3
		Parachironomus sp. (VSCL35)	I	6	1	–	–	–	–	–
		Polypedilum sp. (VSCL8)	I	1	–	–	–	2	–	–
		Tanytarsus fuscithorax	I	3	3	2	–	2	3	3
	Tanypodinae	*Ablabesmyia notabilis*	C	5	–	–	–	–	1	–
		Paramerina levidensis	C	5	–	1	–	–	–	–
		Procladius paludicola	A	5	2	–	–	2	2	2
	Ceratopogonidae	Ceratopogoniinae spp.	I	5	2	3	1	2	1	–
		Dasyheleinae sp.	I	5	–	2	–	–	–	–
	Culicidae	Culicidae sp. (P)	I	6	–	1	–	–	–	–
		Anopheles sp.	I	5	–	2	3	1	–	2
		Culex sp.	I	5	–	–	4	–	–	–
	Dolichopodidae	Dolichopodidae sp.	I	5	–	1	–	–	–	–
	Tabanidae	Tabanidae sp.	I	5	–	1	–	–	–	–
	Tanyderidae	Tanyderidae sp.	I	2	2		–	2	–	–
	Tipulidae	Tipulidae sp.	I	1	–	1	2	–	–	–
TRICHOPTERA	**Hydroptilidae**	*Acritoptila/Hellyethira* sp.	I*	4	2	–	–	2	1	1
	Leptoceridae	*Notoperata* sp. AV1	SW	2	–	–	–	2	2	–
		Oecetis sp.	I*	5	2		–	–	–	–
		Triplectides australis	A	2	3	2	2	–	–	–
		Leptoceridae spp. (imm)	I	2	–	1	1	–	–	–
		Total number of 'species'			***36***	***40***	***32***	***36***	***34***	***27***

Table A5. Fish and crayfish species richness (number) and abundance recorded in November 2010 and February/March 2011. Sites are arranged by sub-catchment in order of increasing distance from the confluence with the Swan River. Sites denoted by * were dry in February/March 2011 and, where possible, were substituted by sites denoted by **.

Species codes: Ab = Acanthopagrus butcheri, Ac = *Amniataba caudovittatus*, Af = *Aldrichetta forsteri* , Bp = *Bostockia porosa*, Gh = *Gambusia holbrooki,* Go = *Galaxias occidentalis*, Nv = *Nannoperca vittata*, Nvl = *Nematalosa vlaminghi*, Pf = *perca fluviatalis*, Po = *Pseudogobius olorum*, Ss = *Sillago schomburgkii*, Tb= *Tandanus bostocki*, Cc = *Cherax cainii*, Cd = *Cherax destructor*, Cq = *Cherax quinquecarinatus*.

November 2010

Site	Ab	Af	Ac	Bp	Gh	Go	Nv	Nvl	Pf	Po	Ss	Tb	Cc	Cd	Cq
1	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
*2	-	-	-	-	4	-	-	-	-	-	-	-	-	-	1
*3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	5
4	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1
*5	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2
*6	-	-	-	-	-	-	4	-	-	-	-	-	-	-	3
7	-	-	-	1	4	4	-	-	-	1	-	-	-	-	7
20	-	-	-	-	-	-	34	-	-	-	-	-	-	3	3
19	-	-	-	1	9	-	4	-	-	3	-	-	-	1	1
18	-	-	-	3	41	-	4	-	-	3	-	1	3	-	2
*15	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
16	-	-	-	-	12	-	-	-	-	-	-	-	-	-	4
17	-	-	-	-	70	-	-	-	-	-	-	-	-	-	3
14	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
13	-	-	-	3	13	-	1	-	-	-	-	-	-	-	27
*10	-	-	-	35	-	7	12	-	-	-	-	-	-	-	40
11	-	-	-	2	-	2	11	-	-	-	-	-	-	-	35
12	-	-	-	11	1	31	54	-	-	-	-	-	-	-	27
*9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	12
8	-	-	-	-	-	-	-	-	-	-	-	-	-	-	11

February/March 2011

Site	Ab	Af	Ac	Bp	Gh	Go	Nv	Nvl	Pf	Po	SS	Tb	Cc	Cd	Cq
1	2	3	1	1	160	-	1	1	-	61	5	-	-	-	1
4	-	-	-	-	31	-	-	-	-	-	-	-	-	-	-
7	-	-	-	-	25	-	3	-	-	-	-	-	-	-	-
20	-	-	-	-	15	-	-	-	-	3	-	-	-	2	-
**22	-	-	-	-	85	-	-	-	-	-	-	-	1	11	-
19	-	-	-	-	176	-	1	-	-	-	-	-	-	16	-
18	-	-	-	1	144	-	4	-	-	-	-	2	20		2
**21	-	-	-	-	200	1	-	-	-	-	-	-	7	10	-
**24	-	-	-	-	30	-	-	-	-	-	-	-	1	-	1
16	-	-	-	-	107	-	4	-	-	1	-	-	-	-	1
17	-	-	-	-	9	-	3	-	2	-	-	-	1	-	-
14	-	-	-	-	333	-	-	-	-	2	-	-	15	-	1
23	-	-	-	-	99	-	1	-	1	-	-	-	19	-	2
13	-	-	-	15	163	-	-	-	-	-	-	-	-	-	19
11	-	-	-	7	-	1	9	-	-	-	-	-	-	-	42
12	-	-	-		177	1	52	-	-	-	-	-	1	-	21
**25	-	-	-	22	129	23	81	-	-	-	-	-	-	-	17
8	-	-	-	-	-	-	-	-	-	-	-	-	-	-	13

Photographs of sampling sites

 Site 1	 Site 2
 Site 3	 Site 4
 Site 5	 Site 6


Site 7


Site 8


Site 9


Site 10


Site 11


Site 12


Site 13


Site 14


Site 15


Site 16


Site 17


Site 18


Site 19


Site 20


Site 21


Site 22


Site 23


Site 24

Site 25