Preliminary Ecological Water Requirements of Collie River East Branch: Risk Assessment of Salinity Mitigation Diversion Scenarios

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Department of Water
Government of Western Australia

by

Wetland Research & Management

October 2007

Preliminary Ecological Water Requirements of Collie River East Branch: Risk Assessment of Salinity Mitigation Diversion Scenarios

Report Prepared for:

THE DEPARTMENT OF WATER

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Frontispiece (clockwise from top left): East Branch of the Collie at Coolangatta Gauging Station (photo: Katherine Bennett, December 2005), long-necked tortoise *Chelodina oblonga* (photo by Andrew Storey, January 2006), and *Crinia insignifera* taken form Bingham River (photo by Andrew Storey).

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1 INTRODUCTION

1.1 Background

The Collie River flows through the township of Collie in the southwest of Western Australia, approximately 200 km southeast of Perth (see Figure 1). The river contributes flow to one of Western Australia's largest water storage investments, the Wellington Reservoir. However, owing to high salinities of inflow water from the East Branch of the Collie, the water within the reservoir is brackish to saline which is marginal for irrigation use and currently not potable. Between 1945 and 1996, the salinity of inflow water to the Wellington Reservoir increased by over 250% (WRC 2001a). In an attempt to address the regions' water allocation requirements, the Collie River catchment above the Wellington Reservoir has been designated a Recovery Catchment under the State Salinity Strategy 2000. The primary aim is to reduce salinity within the Wellington Dam to 530 mg/L by 2015 and return it to potable quality by 2030. In order to achieve this, the Department of Water propose to reduce saline water inflow to the dam by diverting a proportion of flows from the East Branch of the Collie River at particular times of the year, i.e. at the beginning of the flow season (autumn) when the first flush moves salt from the catchment into the river and, at the end of the season when low flows concentrate salt (spring). It is envisaged this is an effective means to reduce saline inflow to the Wellington Reservoir, since the reach only contributes 25% of flow but greater than 50% of the salt load to the dam. The first stage of the strategy involves diverting saline flows to Griffin's coal mine voids near the Collie township. As part of this proposal, Ecological Water Requirements are necessary for the East Branch of the Collie River, because whilst the plan aims to remove salt, it will also reduce stream flow. It is therefore necessary to examine the likely impacts of diversions and thereby reduced flows on the aquatic biota of the downstream environment of the East Collie.

Salinisation within the Collie River has resulted from extensive clearing of native forest in the upper catchment for pasture prior to the 1970s, with over 35% of the catchment being cleared (WRC 2001a). Although the river flows through mostly State Forest and Nature Reserves of marri-jarrah across the Darling Plateau, once it reaches the coastal plain most of the catchment has been cleared to accommodate a number of land uses. Major land uses in this region include agriculture, coal mining, timber production, and conservation/recreation (Storey *et al.* 1990). Agricultural activities within the lower parts of the catchment have resulted in a network of flood irrigation channels which have considerably changed the natural drainage channels of the river (WEC 2002). The loss of riparian vegetation has led to reduced biodiversity, gullying and bank erosion along drainage lines and has consequently resulted in widespread ramifications in the lower parts of the catchment; *e.g.* increased nutrient, salt and sediment loads and increased surface runoff leading to flooding and channel erosion. Historically, riparian zones would have been wide and densely vegetated, with winter-wet depressions and swamps on floodplains during winter.

The approach adopted by the Department of Water, the state agency responsible for licensing resources, is to consider the environmental requirements through the determination of Ecological Water Requirements (EWRs), upon which the Environmental Water Provisions (EWPs) are based, with the residual water then being available for consumptive uses. In line with protecting ecologically important downstream ecosystems, the Department of Water have commissioned ***Wetland Research & Management*** to determine the Ecological Water Requirements (EWRs) of the East Branch of the Collie River, and assess the risk of different diversion scenarios to the ecological values of the East Branch.

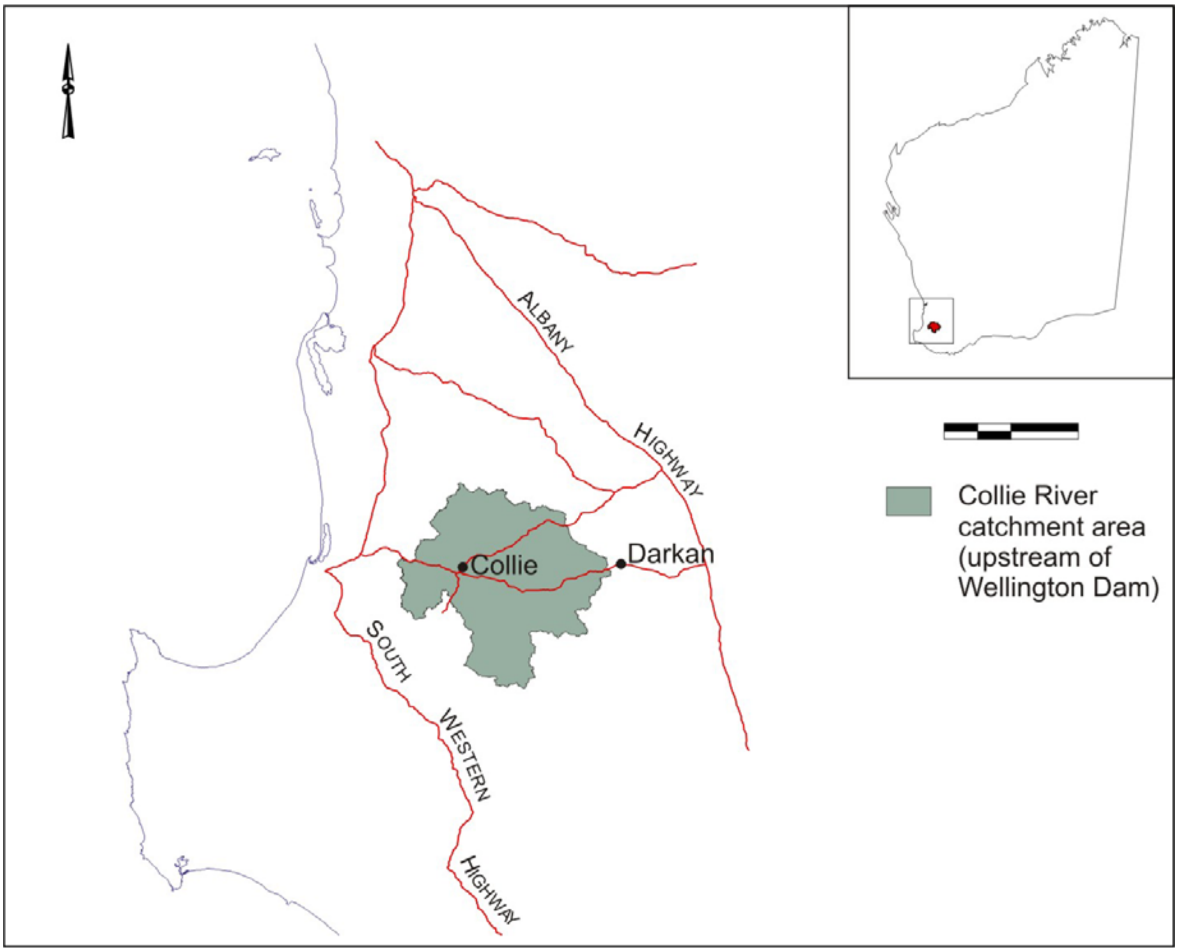


Figure 1. Collie River East Branch location

1.2 Diversion Proposal

Under the current proposal, three scenarios are being considered with respect to the volume of water to be diverted from the East Collie River between Wargyl Pool (upstream) and Buckingham Gauging Station (downstream). The three scenarios are;

- 5 GL diverted over a nominal 90 day period May/June (autumn) and Aug/September (spring)
- 10 GL diverted as above
- 5 GL diverted as above

1.3 Review of Existing EWRs on the Collie River

1.3.1 Existing EWRs

There are a number of existing Ecological Water Requirements in place within the Collie River catchment, including an EWR for the lower Collie River and Henty Brook (Streamtec 1999), one for the Harris River and lower East Branch of the Collie (WEC & Streamtec 2000), and an EWP for the Collie River South Branch (WEC & Streamtec 2001).

As part of this study, it was necessary to review these EWRs, to determine the current hydrological state of the catchment and place the proposed diversion scenarios in context. These existing EWRs were all determined using the "Building-Block Approach" for in-stream flow assessments. They were undertaken before the advent of the Flow Events Method (FEM) which uses Hec-Ras and RAP. As such, they relied on a limited number of channel cross-sections and limited survey of channel morphology. Furthermore, morphological data were not used in a hydraulic model *per se*, but rather hydraulic parameters were calculated by hand and spreadsheet, with no calibration of these parameters against actual flows and depths (apart from the depth and discharge observed on the day of survey). Finally, these previous studies did not put the modelled flows into a context of historical flow regimes (i.e. past frequency and duration of calculated flows). Both methods, however, tend to use comparable expert knowledge on water requirements of specific ecological values (the 'rules of thumb' referred to by Streamtec (1999)), which are based on agreed biological and ecological objectives.

EWRs and/or EWPs calculated in these previous studies have been summarised and expressed as a percent of median annual flow (MAF) remaining in the respective part of the river system (Figure 2). Percentages calculated in Figure 2 were based on median annual flows reported in EWR tables within the various studies (NB except for Central Collie upstream of the confluence of the Southern branch). However, these values were often different to median annual flow data reported on the Department of Water website, i.e. 30.5 GL for the Harris and lower East Branch of the Collie reported by WEC & Streamtec (2000) compared with 41.1 GL on the DoW website, and 17.4 GL for Collie River South (WEC & Streamtec 2001), compared with 20 GL on the DoW website.

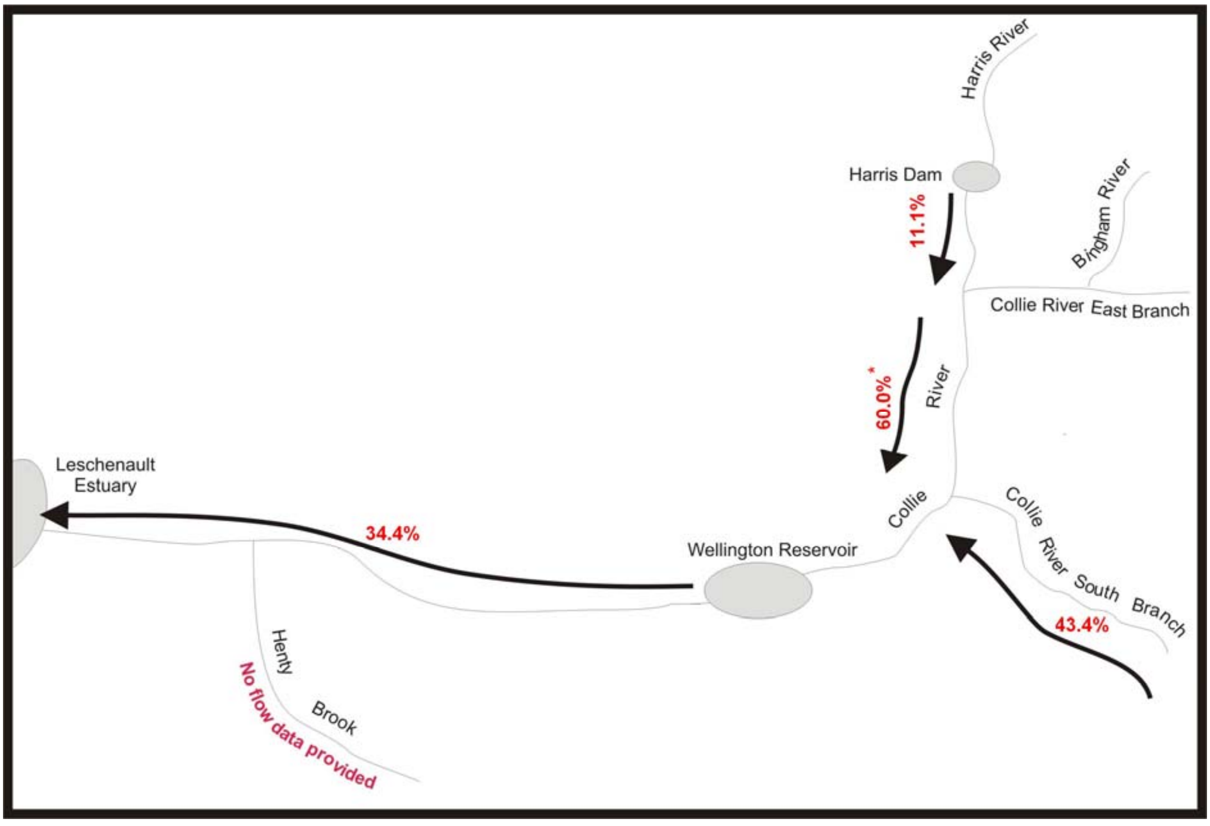


Figure 2. Schematic of the Collie River system, showing past EWRs as a percentage of Median Annual Flow. The value for the Collie River South Branch is an EWP. * - the percent remaining in the Collie River downstream of the Harris is approximated using pre-and post-dam flows for the Harris and the available flow record for Central Collie gauging station (1955 – 1968).

1.3.2 Hydraulic modelling

Generally, previous EWR reports for the Collie catchment do not provide adequate detail of the methods used, particularly in describing measurements of channel morphology and how these data was utilised to model flows. In Streamtec (1999) and WEC & Streamtec (2000), it was reported that "extensive measurements were made of river channels", with cross-sectional profiles being made at 25, 50 and 75 metres within a 100 metre reach using a surveyor's dumpy and staff. It would seem then, that only three channel profiles were made over a distance of only 50 metres, but perhaps slope measurements were made over the entire 100 metre reach. But, given the lack of detail provided in the reports, it is not possible to state this with certainty. Furthermore, locations of reaches were not provided in either report, nor were the location of each individual cross-section with respect to river form and processes, i.e. pool, riffle-zone, backwater etc. Active channel width and depth did not appear to be measured in the field as part of the channel surveys, but were calculated using the methodology outlined by Newbury and Gaboury (1994). This calculation appears to be based on measurements of mean depth, mean width, and slope.

Undertaking measurements of only three cross-sections would not constitute "extensive" in terms of modelling channel hydrology or in characterising in-stream variability in physical or biological attributes; it would be considered the absolute minimum for deriving any hydraulic understanding of the system. Yet, this type of information is important because it provides a measure of the level of rigour and robustness used to achieve the output data, but also provides an indication of the types of physical and biological attributes provided for in the modelling process. Again, three cross-sections will not encompass the variability in channel form along the reach. In addition, using only a 50 metre reach as a means of characterising the channel morphology of an entire river system is probably not sufficient. The small reach chosen may not adequately take into account natural variability of the river system and may influence measurements of channel morphology and ultimately the accuracy of estimates of hydrology used to calculate EWRs. The Department of Water (formerly Department of Environment) currently undertake 15 comprehensive cross-sections over 500 m to 1 km river reach, with all cross-sections surveyed to the same datum. This level of rigour allows for more accurate modelling of flow requirements under the FEM, but also provides for a more accurate hydraulic model using Hec-Ras. However, as mentioned previously, these existing EWRs were undertaken prior to the advent of the FEM using RAP. Nonetheless, the rigour used in previous studies leave the accuracy of any hydraulic calculations open to conjecture, and the Hec-Ras hydraulic model has been available as freeware for accurate hydraulic modelling for many years. The accuracy of estimated threshold flows to achieve specific flow objectives (i.e. depth of water over habitats), therefore must be treated with caution!

1.3.3 Identification of values and processes

A critical step in determining EWRs for any river system is documenting the ecological values and processes to be protected, since flows are provided to protect and/or enhance specific attributes and processes. Values may be derived from the review of prior recent studies or from specific field surveys. In some instances, a decision may be made to restore or enhance degraded values, using historical accounts of past values and/or processes. Ultimately, accurate recording of values is important.

The reporting of ecological values and sampling conducted to document ecological values in the previous EWR reports, however, was unclear and confusing. For example, WEC & Streamtec (2000), for the Harris and lower East Collie, suggested that field sampling for macroinvertebrates was not conducted since "long-term quantitative data" had been collected by Streamtec. The most recent information available for use was from 1997 (WEC & Streamtec 2000). However, no information was given as to sampling protocols used or locations of sampling sites, with relevance to the calculation of EWRs. Later in the report, the authors discuss the use of stable carbon isotope analysis in determining carbon flow and food web structures. It is in this section that WEC & Streamtec (2000) report that "samples of aquatic macroinvertebrates were collected from the Harris River and East Branch of the Collie". However, it is not made obvious whether these macroinvertebrates were included as values in the EWR. In Streamtec (1999) aquatic macroinvertebrates of the Collie River were sampled using the AusRivAS protocols implemented under the Monitoring River Health Initiative (MRHI) of the National River Health Program (NRHP). However, once again no details of habitats sampled or species list of invertebrates collected was offered. AusRivAS scores or model output data were similarly not reported.

Although WEC & Streamtec (2000) report that fish were sampled as part of the EWR, no detail was provided concerning sampling methodology or sites. Similarly, no species list of fish collected during the surveys was provided; however, a list of all known species for the whole of the Collie River was included and was based on published literature and museum records. Streamtec (1999) apparently did not sample fish fauna as part of the EWR for the lower Collie River. EWRs for both reports accounted for ten species of fish, which is very likely an overestimation for the Harris River and lower East Collie (WEC & Streamtec 2000) since some fish included are estuarine species (i.e. big headed goby *Afurcagobius suppositus* and Swan River hardyhead *Leptatherina wallacei*) which are unlikely to be above the Wellington Dam, and one species has not been recorded from the Collie system since 1916 (pouched lamprey *Geotria australis*; Morgan *et al.* 1998) and is no longer likely to occur due to habitat alteration, and another species (i.e. Cobbler) may or may not occur in the Harris. It is somewhat perilous to set flows for values not confirmed to be on the system. In summary, based on the level of information provided, it is not possible to conclude whether ecological values were adequately documented or if what was detailed was actually relevant to the reach/river.

In Streamtec (1999) and WEC & Streamtec (2000), riparian vegetation was described using the foreshore condition assessment methods of Pen and Scott (1995). These assessments were undertaken in order to "estimate the extent of ecologically remnant riparian vegetation and the likely impact of flow regulation" (Streamtec 1999, WEC & Streamtec 2000).

1.3.4 Flow objectives

Flow objectives included in previous EWRs of the Collie River catchment were generally the identified ecological values. They were generic and similar for all previous studies and are reproduced here in Table 1. More recent EWRs using the FEM approach include more specific flow objectives such as winter base flows (for aquatic invertebrates, native fish, riparian vegetation and energy flows), winter bench inundation flows, winter active channel flows (channel form), fish passage flows, floodplain inundation (aquatic fauna, wetlands, riparian vegetation), and flows to inundate riparian vegetation. Under the current FEM approach, the required flows are reported in terms of magnitude, frequency and duration of specific flow events. EWRs in all previous studies however were simply reported as a total volume per month (see Table 2). This approach is uninformative, making it difficult for managers to apply EWRs, since no information regarding magnitude (m^3/s), frequency (number of events) or duration (number of days for each event) of events was provided in these previous EWRs.

Past EWRs of the Collie included a 'seasonal adjustment' (WEC & Streamtec 2001). There is little justification given for this 'objective', and it simply appears to be a value to fill gaps in the hydrograph where there was no specific flow event, but the authors felt there should be a flow. The reports provide little information on what the value applies to or how it was derived. The current method for calculating EWRs uses historical flow data, and therefore the frequency and duration of flow objectives are able to be determined for every month, with no 'gaps' to subjectively fill.

Table 1. Flow objectives used in previous EWR studies of the Collie River catchment.

	Streamtec (1999)	WEC & Streamtec (2000)	WEC & Streamtec (2001)
Channel form			✓
Macroinvertebrates	✓	✓	✓
Fish passage	✓	✓	✓
Pool maintenance	✓	✓	
Bankfull (flood)	✓	✓	
Riparian vegetation	✓	✓	✓
Energy flows	✓	✓	✓
Sub-catchment flows	✓	✓	
Seasonal adjustment			✓

Table 2. Example of EWRs reported by WEC & Streamtec (2000). NB – EWRs are reported as volumes (ML) per month.

Parameter	Jan	Feb	Mar	Apr	May	June	Jul	Aug	Sep	Oct	Nov	Dec
Macroinvertebrates	12.6	11.3	12.6	12.2	12.6	12.2	12.6	12.6	12.6	12.6	12.2	12.6
Fish passage									325	405		
Pool maintenance	26	26	26	26	26	26	26	26	26	26	26	26

Since these existing EWRs were undertaken by the same practitioner, similar rules were applied in all studies. For example, all studies used a depth of 5 cm for inundation of riffles for macroinvertebrates. This value is considered adequate to maintain species diversity in riffles, and is still used in current EWRs. However, they do not define the 5 cm rule – whether it is a maximum depth on the riffle, or a mean depth and whether it is for the whole width of the riffle or just the thalweg. Both Streamtec (1999) and WEC & Streamtec (2000) suggested a water depth requirement of 20 cm over riffles for fish passage. The western minnow (*Galaxias occidentalis*; reported to be the most common species in the Collie River) was always used as a justification for this rule, as it was "only found in reaches where the water depth was greater than 20 cm" (Streamtec 1999, WEC & Streamtec 2000). Since no fish sampling was conducted by Streamtec (1999), it is not clear where this information came from as no reference was provided. In an assessment of EWRs for the lower Collie River, WRM (2003a) considered the 20 cm water depth to be overly conservative for small bodied fish (i.e. western minnow, pygmy perch and nightfish), and especially for the western minnow, which is an exceptionally good swimmer compared with it native counterparts. It must be acknowledged, however, that this value was likely selected originally as a conservative value at a time when determining EWRs for fish in Western Australia was in its infancy. According to WRM (2003a), western minnow have been observed in "shallow waters (<1 cm) whilst traversing obstacles and have even been observed crawling over wet rocks to negotiate rapids". Furthermore, western minnow are known to jump small obstacles (such as water falls) and have been observed jumping through v-notch weirs up to 30 cm in height (ARL 1990a). It seems, therefore, drowning-out obstacles to enable fish passage is not always necessary, but would depend on the fish species present. Larger fish, such as freshwater cobbler, would require a depth of at least 20 cm in order to traverse obstacles. This depth may also be more appropriate in small streams rather than large river systems such as the upper reaches of the lower Collie (WRM 2003a). It is clear, therefore, that care must be taken when applying 'rules of thumb', and that consideration must be given to the types of species present and where they occur in the system. Such considerations were not made in any of the previous EWR and EWP reports.

The total EWR calculated by WEC & Streamtec (2000) for the Harris River from the Harris Dam to the confluence with the East Collie was reported as 11.1% of the Median Annual Flow. Subsequent reanalysis of current flow records in this study places the EWR as only 8.8% of MAF (see Figure 3). Whether it is 8.8% or 11.1% of MAF, this EWR is exceptionally low compared with other EWRs for the catchment (see Figure 2) and other south-west rivers in general (ARL 2005, WRM 2005a), which tend to be circa 50% of MAF. Analysis of the current (post-dam) Harris River flow record indicates that the current hydrograph is highly modified compared with the pre-dam natural hydrograph, with the total loss of all high flows (Rob Donohue, Dept of Water, pers. com.). This is of particular concern, as high winter flows serve many purposes (see section 8.1, 1st para), including maintenance of channel form, prevention of encroachment of riparian vegetation into the channel (viz. terrestrialisation), supporting recruitment of riparian vegetation, inundation of off-channel wetlands and sumps, and mobilisation of carbon off upper and mid-level benches to support foodwebs. For the above reasons, it is highly likely that the EWR for the Harris River is not ecologically sustainable.

The adequacy of EWPs for three systems in the south-west, including the Harris River was assessed by Beatty & Morgan (2005) through a study of recruitment, abundances and biology of species present. Although Beatty & Morgan (2005) suggest EWPs for the Harris River "appear adequate" for maintaining populations of native fish, they reported that the low level of recruitment of western minnow downstream of the dam was likely a result of inappropriate timing of releases for fish passage (Beatty & Morgan 2005). Fish passage releases were set for September and October (as advised by WEC & Streamtec 2000), but Beatty & Morgan (2005) indicated that such releases should be extended to include July and August in order to coincide with peak spawning periods of native fish fauna. It was suggested that this would "aid reproductive success, juvenile recruitment and sustainability of populations of native species" (Beatty & Morgan 2005). There are also inconsistencies between the previous studies concerning the timing of fish passage flows. Streamtec (1999) and WEC & Streamtec (2001) recommended fish passage flows should be made during August and September, yet WEC & Streamtec (2000) recommend passage flows for September and October. WRM (2003a) also noted this flaw in logic with timing of fish passage flows. They noted that literature on fish spawning in southwest W.A. reported upstream migrations occurring from late May and early June onwards, as gravid females attempt to access headwaters to spawn. Therefore, fish migration passage flows should occur from early winter onwards (WRM, 2003a). These fish will then move back downstream in September/October, and would not require the same flows to move downstream over obstacles as they do to move upstream.

There are also implications for water quality in the East Branch of the Collie as a result of such a low EWR in the Harris. Since the East Branch of the Collie is primarily saline, reduced flows from the Harris would result in higher salinities (i.e. less dilution) in the lower East Collie, and where the Collie River meets the South Branch. The implications for changes in salinity have not been considered, especially regarding critical salinity thresholds for aquatic fauna.

1.3.5 Comparison of recommended flow regimes

The percent of mean and median annual flow remaining after implementation of EWRs (Harris/East Collie and South Collie) as well as diversion of flows under the three scenarios (5, 10 and 15 GL), is presented in Table 3 and Figure 3, as a cumulative effect downstream at Mungalup Gauging station. For example, after implementation of both the Harris River and South Collie EWRs, and with diversion of 15 GL in the East Collie, only 32.1% of 'pre-European' median annual flow remains at Mungalup Station, downstream of the Collie townsite (Table 3 and Figure 3).

Table 3. Flows remaining after implementation of EWRs (Harris/East Collie and South Collie) and diversion scenarios, expressed as GL and percent of flow. E D = EWR or Diversion scenario; n^1 = percent of flow remaining after implementation of Harris River EWR; n^2 = percent of flow remaining after East Collie Diversion scenarios (5, 10 and 15 GL); and, n^3 = percent of flow remaining after implementation of South Collie EWR. NA = information not available

			Mean annual flow				Median annual flow			
River		Period of record*	pre -dam	current	E D	%	pre-dam	current	E D	%
Harris	Stubbs Farm	1952-1977	39.5	NA	3.4	8.6^1	41.1	NA	3.4	8.3^1
East Collie u/s Harris	Coolangatta	1968-2006	28.3	42.8	5	88.0^2	20.5	39.6	5	87.4^2
					10	76.6^2			10	74.7^2
					15	64.9^2			15	62.1^2
East Collie River d/s Harris	Central	1955-1968	117.3	NA	81.2	69.2^1	94.2	NA	56.5	60.0^1
					5	65.0^2			5	54.7^2
					10	60.7^2			10	49.4^2
					15	56.4^2			15	44.1^2
South Collie	South Collie	1952-2006	26.2	26.5	7.5	28.0^3	20	20	7.5	37.5^3
Collie River d/s Collie townsite	Mungalup	1969-2006	115.1	104.3	79	68.6^1	96.1	88.6	58.4	60.8^1
					60.3	52.4^3			45.9	47.8^3
					5	48.0^2			5	42.6^2
					10	43.7^2			10	37.3^2
					15	39.3^2			15	32.1^2

* = length of pre-dam flow records vary between gauging stations.

It should be noted that there were differences in the length of record between gauging stations. This has resulted in values suggesting there are lower flows at Mungalup than at Central (Table 3). Given there was no overlap in flow records between these stations, with all data for Central being prior to 1968, this is likely an impact of lower rainfall in the southwest since 1975 (see section 4.2). Generally, current flows were used in calculations except where the record did not extend further than Harris Dam construction.

Allowing for these shortcomings in the availability of flow data, the analysis shows a progressive reduction in flows entering Wellington Dam through a progressive history of abstraction and diversions. Reduction in flows at Mungalup to ~ 35% of 'pre-European' flows is likely to have a significant effect on the ecology of the system, and the ecological sustainability of such a reduction in flow is questioned.

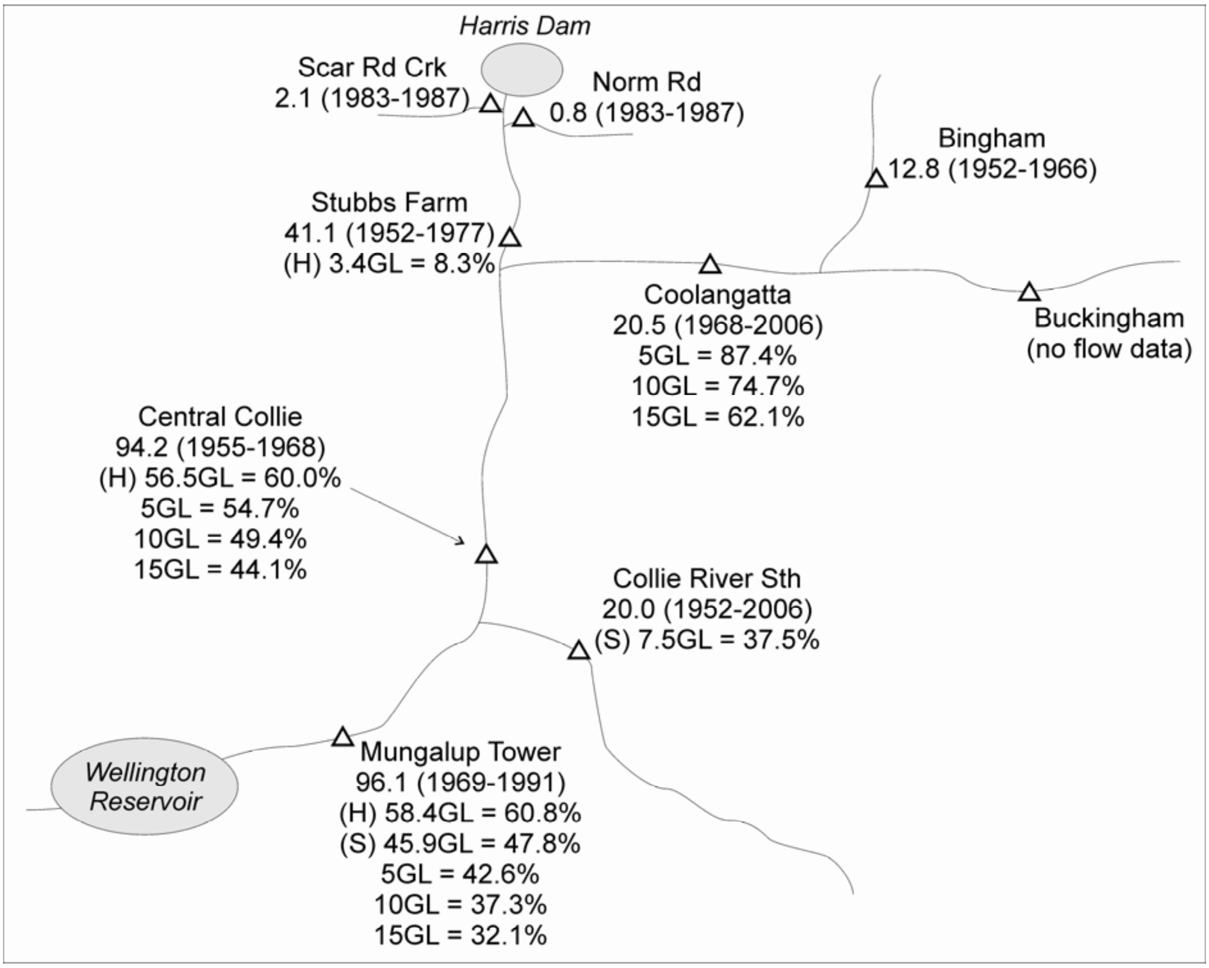


Figure 3. Schematic showing the percentage of median annual flow remaining after implementation of EWRs (Harris/East Collie and South Collie) and diversion scenarios. Median Annual Flow is expressed as GL, and is followed by the period of record used in calculations in parentheses. Remaining flows are first expressed as GL, then percent of flow remaining. Percent of median annual flows is also provided under each diversion scenario; 5 GL, 10 GL, and 15GL. (H) = Harris Dam EWR, (S) = South Collie EWR.

1.4 Study Objectives

The aims of the current project were to:

1. Define the water dependent ecological values of two reaches along the East Branch of the Collie River (adjacent to Buckingham Gauging Station downstream of the proposed diversion, and Coolangatta Gauging Station downstream of the diversion and below inflows from the unregulated Bingham River), and compare this with the values present from an unregulated tributary, Bingham River (near Bingham Gauging Station)
2. Determine the EWRs of water dependent ecological values of the East Branch of the Collie River at Buckingham and Coolangatta;

3. Provide a preliminary assessment of the risk to flow-dependant values associated with change in flow regimes as a result of the diversion scenarios;
4. Make recommendations for future monitoring to assess the effectiveness of the calculated EWRs and to detect effects of changes to the flow regime associated with the diversion on downstream water dependent ecological values.

The study addressed the water requirements of:

- Channel morphology;
- Distribution and extent of key habitat types;
- Riparian and aquatic vegetation ;
- Aquatic faunas (invertebrates and vertebrates);
- Water quality;
- Ecological processes supporting aquatic food webs

A map of the study area, showing location of survey sites is given in Figure 4.

1.5 Statutory/Legislative Framework

The Western Australian approach to ensure that provision is made for the environment in the water allocation decision-making process is by using the concepts of Ecological Water Requirements and Environmental Water Provisions (WRC 2000). These concepts are consistent with the principles in the National Principles for the Provision of Water for Ecosystems (ANZECC/ARMCANZ 2000).

The need to recognise the ecological impacts of flow regulation and diversion has also occurred through a number of Commonwealth Government policies:

- Principles of Ecologically Sustainable Development (1992);
- Intergovernmental Agreement on the Environment (1992);
- COAG recommended fundamental water reforms, including the need to provide water for the environment as part of the introduction of comprehensive systems of water allocations;
- Draft National Water Quality Management Strategy (1994).

The Commonwealth and State agreements on water allocation issues reflect the emerging importance of EWRs in the overall management of river systems. Allocation of water to meet EWRs is based on the premise that the environment has a right to water, that is, the environment has to be regarded as a legitimate user.

In general terms, a water requirement is determined through scientific investigation and community consultation and can be ascribed to a defined value. A water provision is the amount of water allocated from a resource to meet (wholly or in part) the requirement.

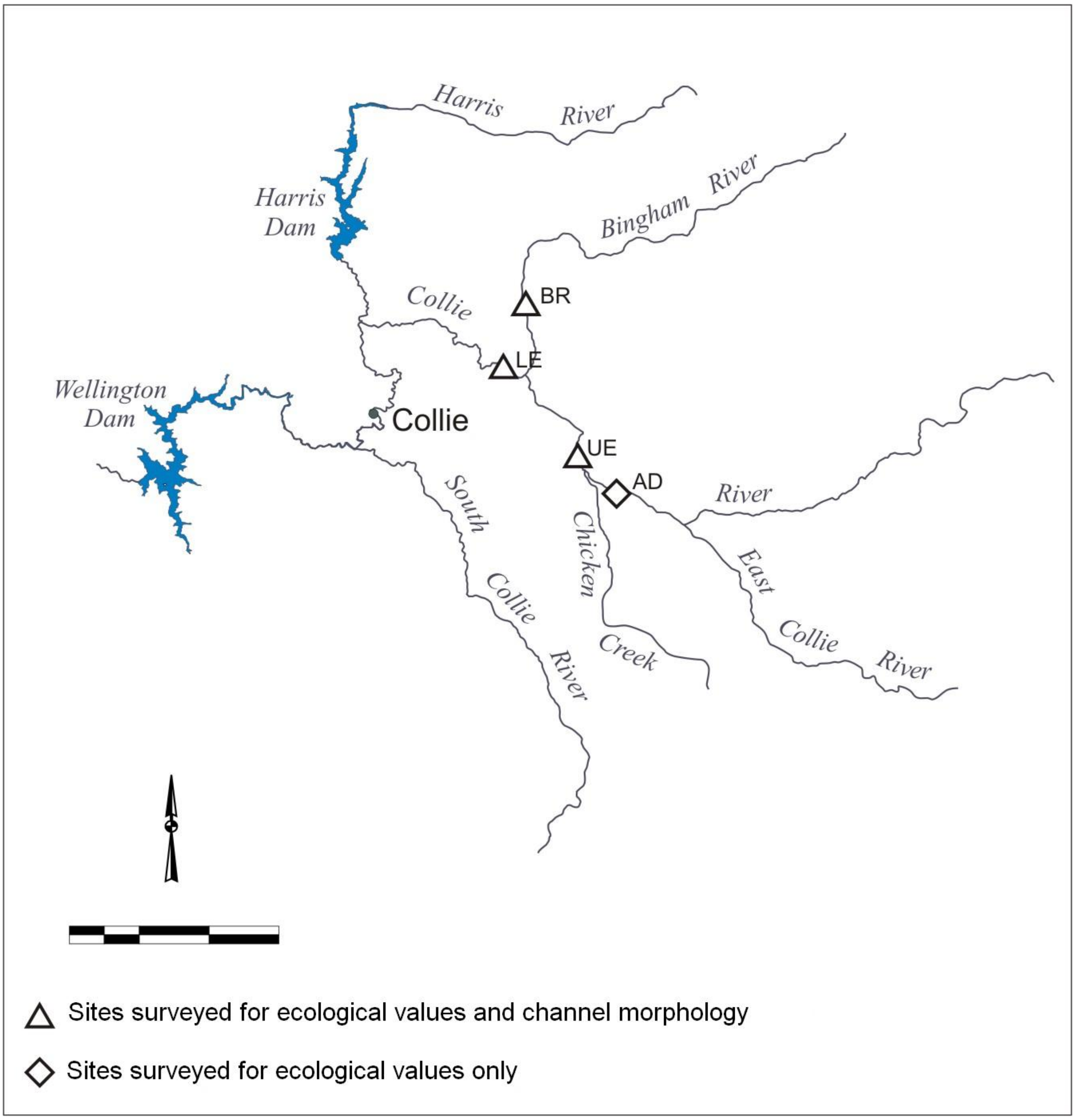


Figure 4. Map of study area showing location of the aquatic fauna sampling sites and survey reaches. BR = Bingham River, LE = Lower East Collie (Coolangatta), UE = Upper East Collie (Buckingham) and AD = Above diversion (above Wargyl Pool).

1.6 Ecological Water Requirements (EWRs)

EWRs are the water regimes required to sustain key ecological values (existing, historical or proposed for restoration) of water dependent ecosystems at a low level of risk. These requirements are a primary consideration in the establishment of EWPs during the water allocation decision-making process.

EWRs are determined on the basis of the best scientific information available. The determination of EWRs initially requires identification of key ecological values including ecological condition and health, biodiversity and rare/endangered species. In undisturbed environments, the usual aim in establishing EWRs is to protect the existing 'natural' ecological values at a low level of risk. The situation is less well defined for ecosystems that have been modified through regulation and/or as a result of land use changes. Where the environment has been disturbed, EWRs can be established to:

- maintain the current key ecological values at a low level of risk,
- maintain and/or enhance current key ecological values,
- identify the likely pre-existing natural ecological values and determine the key values which the EWRs should aim to re-establish, or
- provide for a combination of current key ecological values and key pre-existing natural ecological values.

Some 'value judgements' need to be made when reaching a decision on which key ecological values are to be sustained. WRC (2000) guiding principles in the Environmental Water Provisions Policy for Western Australia, particularly Principles 1-4, provide important information about how these judgements should be made.

1.7 Environmental Water Provisions (EWPs)

Environmental Water Provisions (EWPs) are the water regimes that are determined before any allocation is made to consumptive use. They are defined as "the water regimes that are provided as a result of the water allocation decision-making process taking into account ecological, social and economic demands. They may meet in part or in full the ecological water requirements" (WRC 2000).

They define water regimes that protect both ecological and socio-cultural values of water resources and are set through the water allocation planning process. The degree to which ecological, social and economic goals are met will vary from case to case and may involve compromises between ecological, social and economic goals. DoE's guiding principles provide key information about the setting of EWPs.

The ***Desirable Future State*** of the river system (determined in consultation with the community) provides the context for the development of EWPs.

An integral part of developing EWPs is determining ***Social Water Requirements*** (SWRs). These are defined by WRC (2000) and by Kite *et al* (1998), and are elements of the existing or historic water regime that sustain socio-cultural values.

SWRs are not the primary consideration in the allocation decision making process but they may be established as part of the EWP depending on their impact on ecosystems and the significance of the social value sustained by the water regime. Social requirements may include water for domestic and stock use, recreational pursuits, landscape and aesthetic values and educational or scientific aspects. The desirable future state of the river system also provides a context for SWRs. The determination of SWRs does not however, include uses of water for commercial or economic return.

2 FLOW EVENTS METHOD FOR DETERMINATION OF EWRS

In past years the Building Block Approach has been the preferred method in Western Australia for determining EWRs for rivers and streams (Davies *et al.* 1998, WEC & Streamtec 2000, Storey *et al.* 2001, Storey & Davies 2002, Streamtec 2002). However, following the development of the Flow Events Method (FEM) (Stewardson 2001, Stewardson & Cottingham 2002) and, with its successful application to eastern states rivers (Cottingham *et al.* 2003), the Department of Environment has adopted the FEM as an alternative method for determining EWRs in Western Australia..

The FEM was originally developed by the CRC for Catchment Hydrology (Stewardson 2001) to integrate the various *ad hoc* approaches being used for determining ecological water requirements throughout Australia. It was designed to provide a standardised, "transparent" analytical procedure, applicable to a broad range of river systems. While it advocates a consistent approach, the method still allows for the inclusion of expert opinion (Stewardson & Cottingham 2002). This approach offers an improvement on the less transparent 'Building Block' approach previously used in Western Australia. However, like the 'Building Block' approach, the FEM assumes that the various components of a flow regime, such as summer and winter baseflows, bankfull flows and flood flows (Figure 5), have different ecological functions (Poff *et al.* 1996, Richter *et al.* 1997) and that these need to be assessed independently (Stewardson & Cottingham 2002, Cottingham *et al.* 2003, Stewardson & Gippel 2003). Even periods of no flow may be important; *e.g.* in ephemeral and seasonal systems.

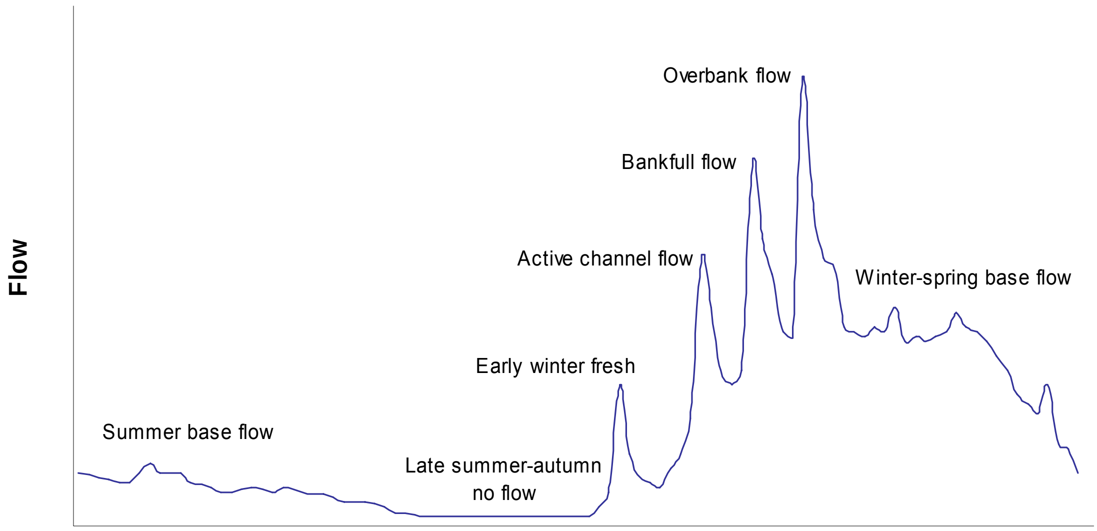


Figure 5. Time series showing typical components (events) of a natural flow regime relevant to river systems in south-west Australia (after Cottingham *et al*. 2003).

One of the main principals of the approach is that the altered river ecosystem (*e.g.* downstream from impoundments, abstraction or land clearing) incorporates integral or 'key' ecological features of the original functional system; *i.e.* some of the natural flow variability should remain. FEM is a modification of the FLOWS method (SKM *et al.* 2002) developed in Victoria, but unlike the FLOWS method it has no *a priori* assumptions about the importance of hydrological events without first considering their significance to key ecological features of the system (Cottingham *et al.* 2003). FEM is an additive method and allows for as many or as few features to be evaluated as required.

All available data on existing and historic ecology, hydrology and channel morphology are used to identify those features likely to be affected by flow modifications. Environmental flow objectives specific to each reach are established, pursuant to determinations of acceptable level of risk posed by flow modifications. Hydraulic models (*e.g.* HEC-RAS) are then used to predict flows required to inundate features such as channel bed and benches to a certain depth in order to maintain key ecological features (*e.g.* water for fish passage; floodplain flows to inundate riparian vegetation and stimulate seed set) and meet environmental flow objectives. The frequency of flows and shape of the hydrograph for each important event is also modelled using the River Analysis Package (RAP) using the modelled water surface profiles from HEC-RAS.

In the instance of a system already regulated by impoundment/abstraction, the method compares the regulated flow regime to modelled unregulated flows for the same period to determine how far the hydrology has changes relative to pre-impoundment.

Figure 6 summarises the standard procedural steps in the application of FEM (and FLOWS) and indicates modifications made for the current study.

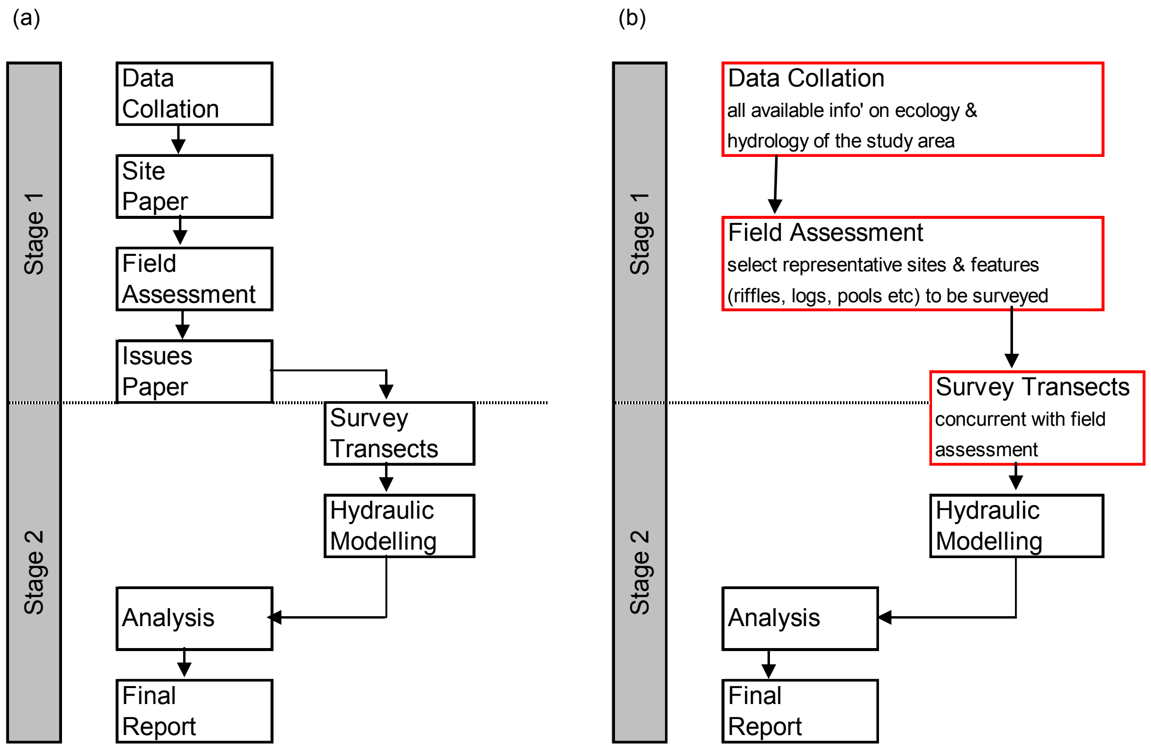


Figure 6. (a) Basic steps in the FEM (& FLOWS) method and (b) steps used in the current study (boxes in red emphasise modifications made for the current study).

3 THE COLLIE RIVER CATCHMENT

The Collie River catchment is part of the broader Collie River Drainage Basin which extends over 3600 km^2. Located within the basin is one of Western Australia's largest water storage investments, the Wellington Reservoir, which lies to the west of the study area. Headwaters of the Collie River East lie on the Darling Plateau and the river forms a confluence with the South Branch to the south of the Collie township. The East Branch is a perennial stream since extensive land clearing in the 1960s and 1970s, and is characterised by fluctuating flows and high salinity levels. It is estimated that 37% of the annual flow of the Collie River arises from the eastern portion of the catchment, including Collie River East Branch, Bingham River and Chicken Creek.

3.1 Climate

The climate of the region is Mediterranean with hot, dry summers and cool, wet winters (Seddon 1972). Temperatures can range from minus zero in winter to greater than 40 °C during summer (Varma 2002). Rainfall is both seasonal and highly predictable, with 75-80% of annual rainfall occurring between May and September (Varma 2002). Long-term (100-year) average annual rainfall at the Collie township is 950 mm. This decreased to 840 mm over the last 20 years. Average annual potential evaporation is around 1650 mm (Varma 2002).

CSIRO (2001) models for global warming predict a general increase in temperature for the south-west of between 0.4 - 1.6°C by the year 2030. A decreasing trend (-20% to +5%) in winter and spring rainfall is also predicted and a ±10% change in summer/autumn rainfall. While the intensity of specific winter rainfall events may increase, their duration is expected to decrease. Correspondingly, the duration of drought events and rates of evaporation are also expected to increase. The ~20% decrease in south-west rainfall experienced over the last 30 - 40 years has resulted in a 30 - 40% decrease in annual streamflow.

3.2 Geomorphology

The Collie River descends the Darling Scarp from the Yilgarn Plateau through the Coastal Plain and discharges into the Indian Ocean from the Leschenault Estuary (Varma 2002). The Darling Scarp is a prominent feature in the landscape, rising to 300 metres above sea level. It separates the Archaean Yilgarn Block from the Phanerozoic sedimentary deposits. Soils are Tertiary laterites over Archaean granites and metamorphic rock (Marchant *et al.* 1987, Wells 1989). In contrast, the Swan Coastal Plain comprises sandy aeolian soils with alluvial clays along the eastern extent (Marchant *et al.* 1987, Wells 1989). As the Collie River flows from the scarp and across the coastal plain, marked changes in physical (i.e. depth, width, river bed material) and riparian vegetation characteristics become evident (Searle and Semeniuk 1985).

The Collie River Catchment is underlain by granites and gneisses of Archaean age, with some dolerite, microdiorite and gabbro dykes (Varma 2002). The crystalline basement is overlain by clay-rich weathered material which, in turn, is overlain by sand and gravely soil (Varma 2002). Landscapes have very low relief of approximately 200 to 250 m above sea level. Although fluvial processes dominate, channels tend to be broad, shallow and flat-floored, with wide floodplain areas (Hill *et al.* 1996).

3.3 Hydrogeology

The Collie River Groundwater Area (CGWA) is a major groundwater resource that occupies most of the Collie Basin (WEC & Streamtec 2000, Varma 2002). Groundwater is recharged directly from rainfall infiltration (approx. 19 x 10^6 m^3/yr) and, to a lesser extent, from streams flowing into the basin (approx. 1 x 10^6 m^3/yr), particularly the South Branch of the Collie (WEC & Streamtec 2000, Varma 2002). Recharge to the basin is about 10% of the annual rainfall (Varma 2002). Groundwater from the basin discharges to the Collie River and its tributaries (near the Collie townsite) and maintains the associated environment (Varma 2002). Discharge to the river system as baseflow occurs down-gradient of gauging stations S612034 on South Branch and S612035 on the East Branch (Varma 2002).

Although mining began in the basin in 1898, groundwater abstraction has only been monitored since 1984 (Varma 2002). In the past, annual groundwater abstraction exceeded recharge, leading to a basin-wide decline in groundwater levels. Since closure of underground mines in 1995, groundwater levels in some parts of the basin have been rising (Varma 2002). Current discharge volumes are approx. 7 x 10^6 m^3/yr (Varma 2002). Groundwater is abstracted from the basin at a rate of 20 GL/annum associated with power generation (44%) and coal mining (56%) (WEC & Streamtec 2000).

Abstractable quantities of groundwater are mainly contained within sandstone of the Muja Coal Measures, Premier Coal Measures, Allanson Sandstone, Ewington Coal Measures and Westralia Sandstone of the Collie Group; within sand and sandstone of the Nakina Formation; and, in surficial sediments (Varma 2002). Sandstone of the Shotts Formation may also contain some groundwater stores. Within the Collie Basin, groundwater flow is through pores of sedimentary units (Varma 2002). Owing to mine dewatering and large-scale abstraction for power generation, the direction of groundwater flow is highly modified from pre-mining patterns. Flow is mainly towards mining areas (Varma 2002).

Generally groundwater in the area is acidic, with pH ranging from 2.6 near the underground and opencut mines to 6.3 close to the southern and southeastern boundaries of the Collie Basin. Outside the basin groundwaters have neutral pH, and acidity within the basin is considered due to contact with sulphide bearing sediments (Varma 2002). The pH of the Collie River may reduce from currently neutral to slightly acidic (around 6.5) under conditions of increased groundwater discharge (Varma 2002). This has the potential to mobilise toxic pollutants, such as arsenic, berilium, copper, nickel, molybdenum and uranium, from Collie Basin coal which could enter the river system (Varma 2002).

Salinities are generally lower than 500 mg/L total dissolved solids (TDS) and thus considered fresh. Throughout the basin, groundwater salinity varies from 40 mg/L in the northeastern part to 4200 mg/L TDS near East Branch in the southeastern part of the basin. Groundwaters outside the Collie Basin, have salinities of 1000 mg/L to 17 300 mg/L TDS (Varma 2002). Varma (2002) considered that under current discharge volumes, discharge of low salinity groundwater to the Collie River would likely result in a 7% decrease in salinity of the Collie River. If groundwater recovered to pre-mining levels, salinity could effectively reduce by as much as 15% (Varma 2002).

3.4 Vegetation

As mentioned above, a large proportion of the catchment has been cleared for agriculture and mining activities. However, of the native vegetation which remains along the East Branch of the Collie River, complexes are dominated by open woodland of *Melaleuca preissiana* (Plate 1)-*Banksia littoralis*, closed scrub of Myrtaceae spp., and sedgelands of *Baumea* and *Leptocarpus* spp (Mattiske and Havel 1998). Common eucalypts include *Eucalyptus marginatus* subsp. *thalassica* and *Eucalyptus wandoo* (Mattiske and Havel 1998).

Plate 1. Paperbark *Melaleuca preissiana* (taken from www.florabase.calm.wa.gov.au).

4 HYDROLOGICAL STATE OF THE CATCHMENT

4.1 Available Data

The first stage in developing EWRs for a river system is to determine the past and current hydrological state of the catchment. This is critical, as typically the further a river system is removed from its historic hydrology, the more the environment, and therefore, its ecological values are impacted. However, there is often minimal data on which to establish current hydrology, and more often than not, even less data for establishing historical state.

In order to assess the current hydrological state of the system, data were sourced from a series of streamflow gauging stations on the Bingham and East Collie Rivers. Stations with their available data series were:

- Buckingham (612038) from 1999 to 2006;
- Bingham River Stenwood (612021);
- Bingham River Palmer (612014) and,
- Coolangatta Farm (612001) from 1968 to 2006.

Analysis of rainfall records also assist in interpreting the past and current hydrological state of the catchment. Long term rainfall records for sites in the vicinity of the East Collie River include Collie (009628, 1907-2004) and Muja (009738, 1963-2004).

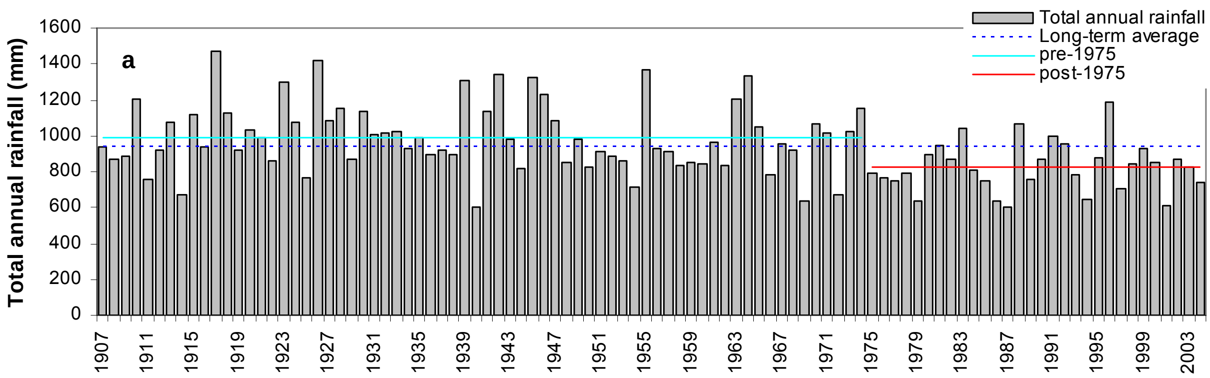
4.2 Rainfall

The average annual rainfall for the area ranged from 722 mm at Muja (Station 009738) to 940 mm at Collie (Station 009628) for the length of record (Figure 7). Considerably higher rainfall events (>1300 mm/year) were recorded from Collie in 1917, 1926, 1942, 1945, 1955, and 1964, and Muja in 1973.

Since 1975 there has been a significant reduction in mean annual rainfall in the southwest of Western Australia. The extent of the reduction between the 'Average Climatic Condition' period (pre-1975) and the 'Dry Period' (post-1975) varies regionally. Storey *et al.* (2001) estimated that mean annual rainfall for the Canning Dam area for the period 1975 - 1997 was 18% lower than the long term mean annual rainfall for 1912 - 1997. Similarly, WEC (2004) reported a 20% reduction in rainfall for the Bickley Brook system and WEC (2002) calculated an 11% reduction in annual rainfall at Collie (Collie townsite, 1911 – 1997). Using more recent data, the influence of this climate change on regional rainfall for the East Branch of the Collie River was assessed as a precursor to modelling catchment hydrology. There has been a significant reduction in mean annual rainfall in the Collie area post-1975 (mean = 826) compared with pre-1975 (mean = 992) (two-tailed t-test, df = 96, t-value = 4.27, $p < 0.001$) (Figure 7a). It is estimated that mean annual rainfall for the period 1975 to 2004 is 87% of the long-term mean annual rainfall (1907 to 2004).

In general terms, reduced rainfall will influence catchment hydrology by reducing stream flows, particularly in upper catchments. However, with increasing distance down the catchment these reduced flows are often counter-balanced by the effects of catchment clearing, which increases the proportion of precipitation resulting in run off. Clearing reduces interception and transpiration by vegetation. As a result, a greater proportion of rainfall runs off the land leading to often greater flow in downstream (i.e. Swan Coastal Plain) reaches due to clearing, even following reduced rainfall. In addition, the speed of run-off is increased. Together, these change the shape of the flood hydrograph from a relatively slow, flat response (i.e. taking a day for a flood peak to develop) to a very rapid response (i.e. a flood peaks in several hours following rainfall). Increased and more rapid runoff often results in channel widening and incision, whereby the bed of the river is degraded. The hydrology of the East Branch of the Collie River is likely affected by clearing, with increasing distance downstream.

The historic change in rainfall and associated change in runoff has implications for calculating EWRs in rivers of southwestern Western Australia. The shape of the river channel is influenced by the action of bankfull discharge which is referred to as 'channel forming flows', and have been calculated to occur approximately one in every two to three years in south-western Australia (WRC 2001b). Reduced rainfall means that bankfull flows for the existing channel now occur at a lower frequency. Frequency of other flow scenarios also will be changed. When developing EWRs for a system, these changes in catchment hydrology need to be assessed and considered.



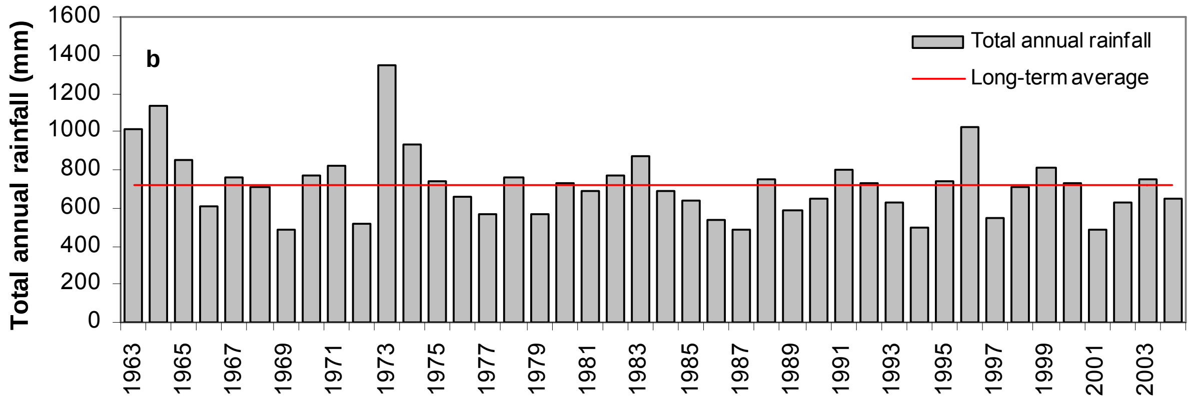


Figure 7. Total annual rainfall and long-term average for a) Collie Station 009628, and b) Muja Station 009738

5 SALINITY

As mentioned above, the East Branch of the Collie River is highly saline. Elevated salinity in freshwater systems can directly impact fauna through effects on osmoregulatory physiology as the maintenance of constant solute body concentration is affected (Bayly 1972, Kefford *et al.* 2003), and indirectly through increased water volumes, longer periods of inundation (Halse *et al.* 2003), and alterations in habitat; the loss of aquatic and fringing vegetation around wetlands (Froend 1987). Increases in salinity particularly impact biota which do not have adaptations for life in saline environments (Beresford *et al.* 2001). Aquatic macroinvertebrates, for example, are known to be particularly sensitive to changes in salinity. However, there is little published information on the specific sensitivity of Western Australian freshwater organisms to increases in salinity and few specific studies on sub-lethal or long-term effects on more sensitive life stages. The majority of literature suggests fauna may be adversely affected even at relatively low salinities of 500 – 1,000 mg/L TDS (Bunn & Davies 1992; Hart *et al.* 2003; Bailey *et al.* 2002). ANZECC/ARMCANZ (2000) give a maximum trigger value of 300 μS/cm (~204 mg/L)[1] for upland rivers and Bailey *et al.* (2002) reported that reduced abundance of 80 - 90% has been recorded in many invertebrate species following rapid increases in salinity from around 300 to 2000 μS/cm (~204 – 1360 mg/L) over a period of days. In inland waters, maximum aquatic biodiversity is typically recorded when salinities are less than 3000 μS/cm (2000 mg/L TDS). It is generally believed that biota of freshwater ecosystems will be adversely affected as the salinity increases to ~1500 μS/cm (1000 mg/L) (Hart *et al.* 1991). In a recent study by Rutherford & Kefford (2005), it was suggested that for high protection level (*i.e.* 95% species), salinities for upland rivers should not exceed ~590 μS/cm (400 mg/L). Horrigan *et al.* (2005) similarly found significant shifts in community composition as salinity reached 800 - 1000 μS/cm and that changes were more prominent in riffle habitats as these supported a greater number of salt-sensitive taxa. They also noted that changes in composition occurred at lower salinity but were more subtle, resulting in "the steady substitute of salt sensitive taxa by opportunistic and salt tolerant taxa" (Horrigan *et al.* 2005). Thus, while community composition changed, species richness remained more or less the same. Increasing salinities likely result in the proliferation of nuisance groups such as mosquitoes (Culicidae) and midges (Chironomidae and Ceratopogonidae), and increases in other tolerant genera, *i.e.* water fleas (Cladocera), seed shrimps (Ostracoda) and ceinid amphipods (Bailey 2002).

The affect of each diversion scenario on the salinity at Coolangatta was examined by modelling daily salt loads over a 25 year period (1st of January 1980 to 31st of December 2005). Data were modelled based on the observed daily flows over this period, with data corresponding to a dry river system (i.e. no flow) not included. This meant that all zero and negative modelled salt loads were not included. Salt load data were then transformed in monthly concentrations (mg/L) and examined graphically (Figure 8). The change concentration under each scenario was calculated as the difference between the observed current mean concentration for each month and modelled monthly mean concentration for each scenario (Figure 9). Concentrations would be expected to decrease between June and October under each diversion scenario, whilst an increase would likely occur during the drier months, i.e. December to March (Figure 9).

Analyses indicate that up to a 600 mg/L reduction in salt concentration will occur in winter under the highest diversion (Figure 9), with a 200 – 300 mg/L reduction in salt concentration in most winter months under the lowest diversion. As noted above, increases in salinity of as little as 200 mg/L has been shown to have detrimental effects on aquatic fauna. This effect may

[1]Salinity as total dissolved solids (TDS) in mg L^{-1} = 1000 x ppt = 0.68 x EC μS cm^{-1} (ANZECC 1992).

depend on ambient salinities and relevant thresholds for target species (i.e. a small increase in salinity may exceed a threshold and result in chronic or acute effects). However, aquatic fauna usually exhibit a range of thresholds, with species demonstrating a progressive increase in tolerance to salinity. It is intuitive that the corollary of species declining with increasing salinity should also apply, whereby species 're-establish' in a system as salinity decreases. The modelled improvements in salinity in the East Branch of the Collie under high and low diversions would likely reduce salt stress on aquatic fauna, and would potentially lead to re-establishment of some sensitive fauna.

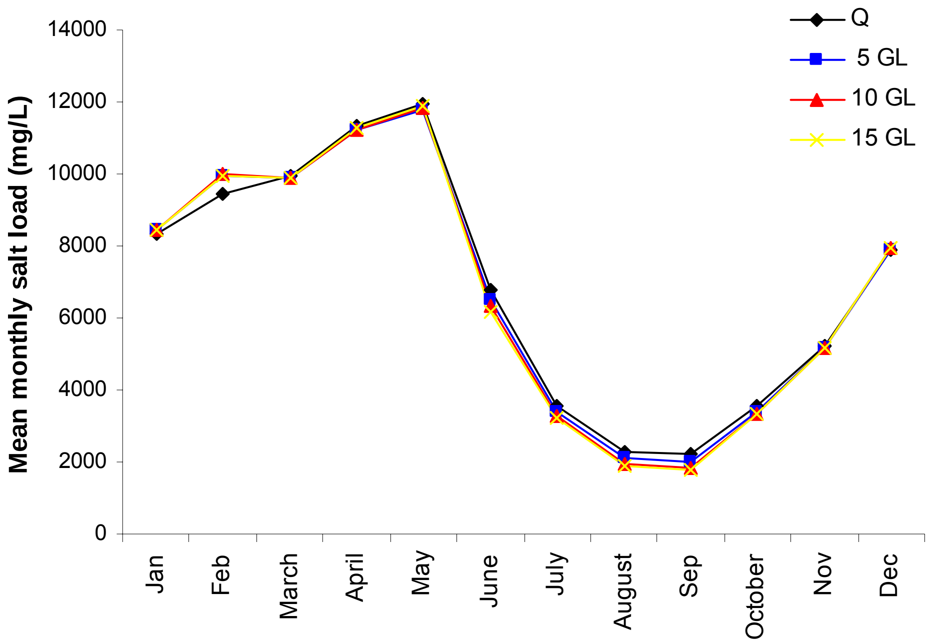


Figure 8. Observed mean monthly salinity (Q) and expected salinities under the three proposed diversion scenarios (i.e. 5 GL, 10 GL, and 15GL diverted).

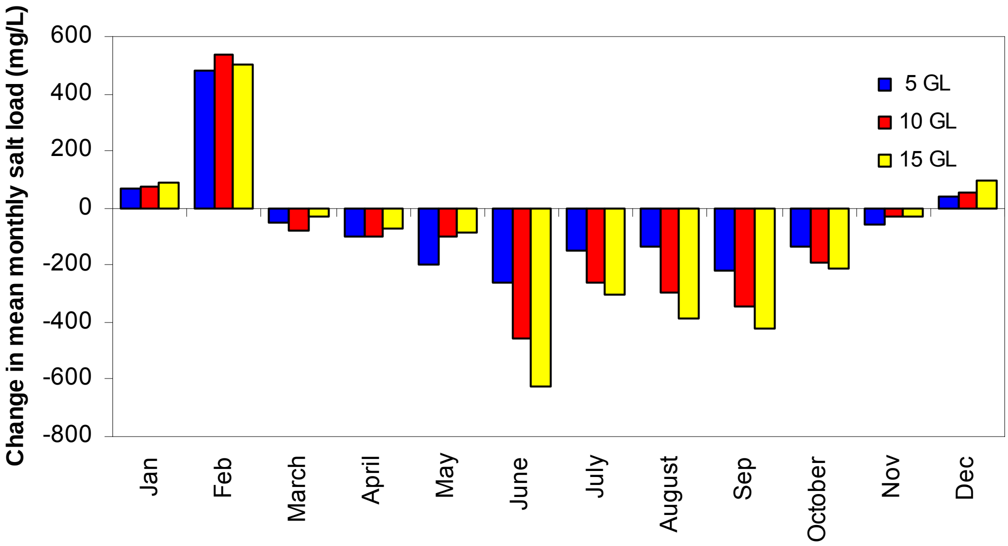


Figure 9. Difference in salt concentration under the three proposed diversion scenarios compared to observed salt concentrations.

6 HYDRAULIC MODELLING

6.1 Introduction

A critical aspect of determining EWRs to protect key water-dependent environmental values and processes is the accurate surveying of the hydraulic geometry of the river channel. This is necessary to build hydraulic models which are then used to calculate the flows required to achieve specific water levels/discharges that drive important hydraulic and ecological processes. The first step in developing a hydraulic model is the accurate surveying of channel morphology using staff and dumpy to characterise the shape and variability of the channel. Replicate surveyed cross-sections (width and depth measurements) are then entered into the selected hydraulic model, which is then used to determine discharges required to achieve desired stage heights. Relationships of observed discharge to stage height are then used to calibrate the model.

6.2 Methods

6.2.1 Survey reaches

Based on the scope of the diversion proposal, system hydrology, location of major tributaries, catchment morphology and extent of clearing, the study area was partitioned into three main sections:

- Upper Section of the East Branch of the Collie River (UE) – near the Buckingham Gauging Station (downstream of the proposed diversion but upstream of the confluence with the Bingham River);
- Lower Section of the East Branch of the Collie River (LE) – near the Coolangatta Gauging Station (6120015), downstream of both the proposed diversion and the Bingham River; and,
- Bingham River (BR) – near the Palmer Gauging Station (612014).

6.2.2 Cross-sectional surveys

Channel surveys were conducted by subcontractors, surveying under the direction and guidance of Department of Environment staff and Dr Andrew Storey of *WRM*. Methods followed those previously used by WRM in EWR studies of river systems in Western Australia (i.e. ARL 2005, WRM 2005a & b).

Replicate channel cross-sections were surveyed to characterise channel morphology of the reach. Cross-sections were located to characterise the shape and variability of the channel over the reach and were positioned to include key hydraulic and ecological features such as points controlling water levels in upstream pools (riffles/rock bars etc), backwaters, pools, riffles, large woody debris and channel constrictions. These features, together with elevations of bankfull[2]

[2] Bankfull level is the bank height to which waters rise to top of channel bank without flowing out onto the floodplain. Active water level is also referred to as the 'Channel Forming Flow' and is a flow that occurs frequently enough to shape the channel. Active level may be at bankfull level (i.e. floodplain level) or at some point below the top, particularly if the channel is incised (deepened) or has been physically modified (channelised and straightened). Both these levels may be determined by looking for features such as upper edge of exposed soil, lowest extent of annual grasses, grooves in the bank, changes in vegetation type and upper edge of water stains on the bank or on vegetation (WRC 2001).

and active channel (depth & width) were measured using a surveyors' dumpy and staff (Plate 2 & Figure 10). Longitudinal distances between each cross-section were measured to determine change in elevation along each reach (i.e. slope).

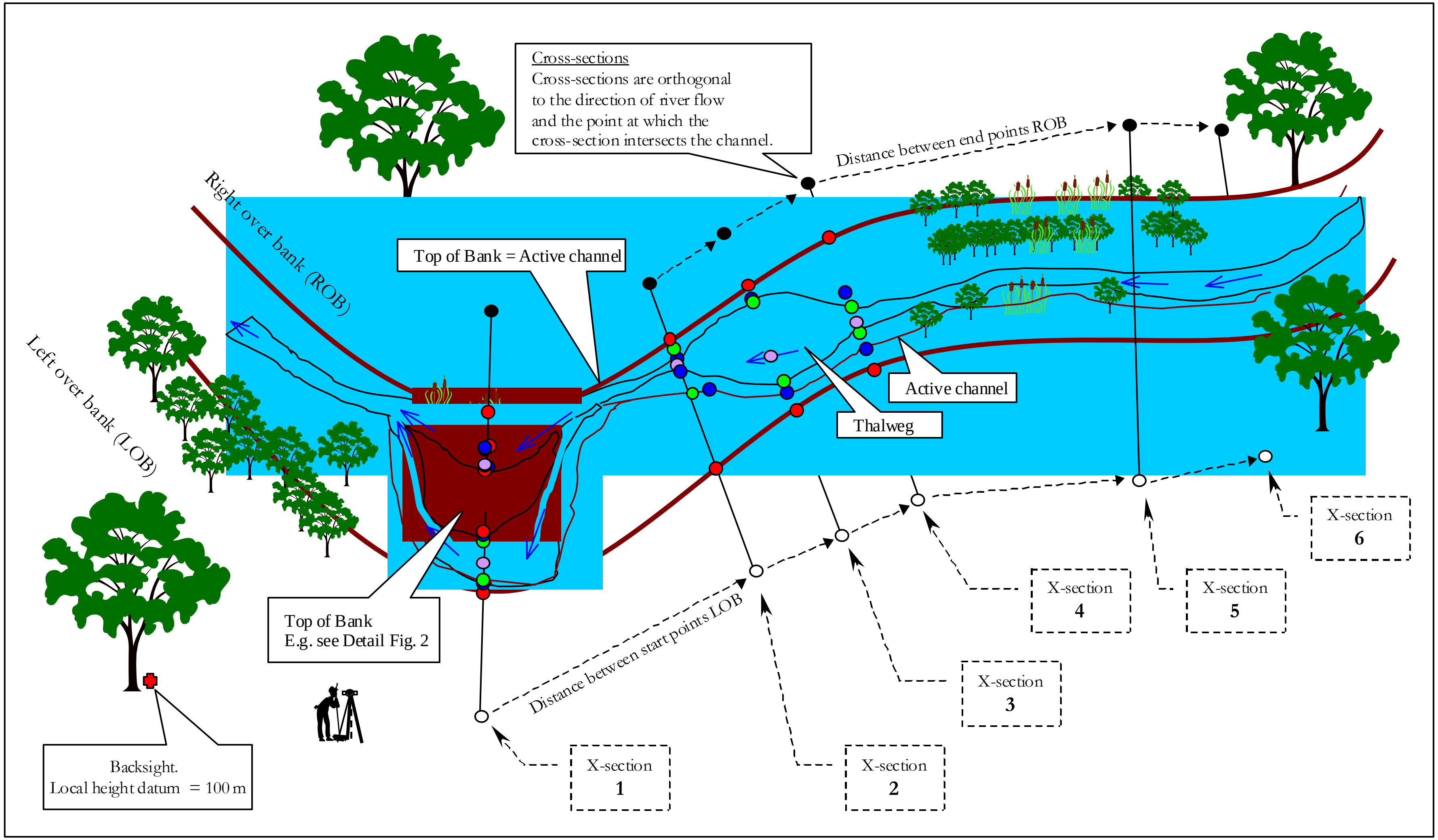
Plate 2. Example of undertaking channel cross-section surveys using dumpy, staff and tape measure (Hill River, August 2004).

Discharge at the time of survey was measured to assist in calibrating the hydraulic model to be developed for the reach. A confined segment of stream of uniform shape was selected and replicate measurements of velocity were taken across the channel using a Universal Current Meter (Hydrological Services Model OSS-B1). Velocity readings were taken at 0.2, 0.6 and 0.8 of total depth at 10 cm increments across the channel width. Measurements were then used to calculate discharge (Q) as:

$$Q = \text{cross-sectional area} \times \text{average velocity}$$

6.2.3 HEC-RAS modelling

A hydraulic model was constructed for each reach using HEC-RAS (**H**ydrological **E**ngineering **C**entre, United States Army Corps of Engineers, **R**iver **A**nalysis **S**ystem), a one-dimensional, steady-state flow backwater analysis model.


Cross-sections
Cross-sections are orthogonal to the direction of river flow and the point at which the cross-section intersects the channel.
Distance between end points ROB
Right over bank (ROB)
Left over bank (LOB)
Top of Bank = Active channel
Active channel
Thalweg
Top of Bank
E.g. see Detail Fig. 2
Distance between start points LOB
Backsight.
Local height datum = 100 m
X-section 1
X-section 2
X-section 3
X-section 4
X-section 5
X-section 6

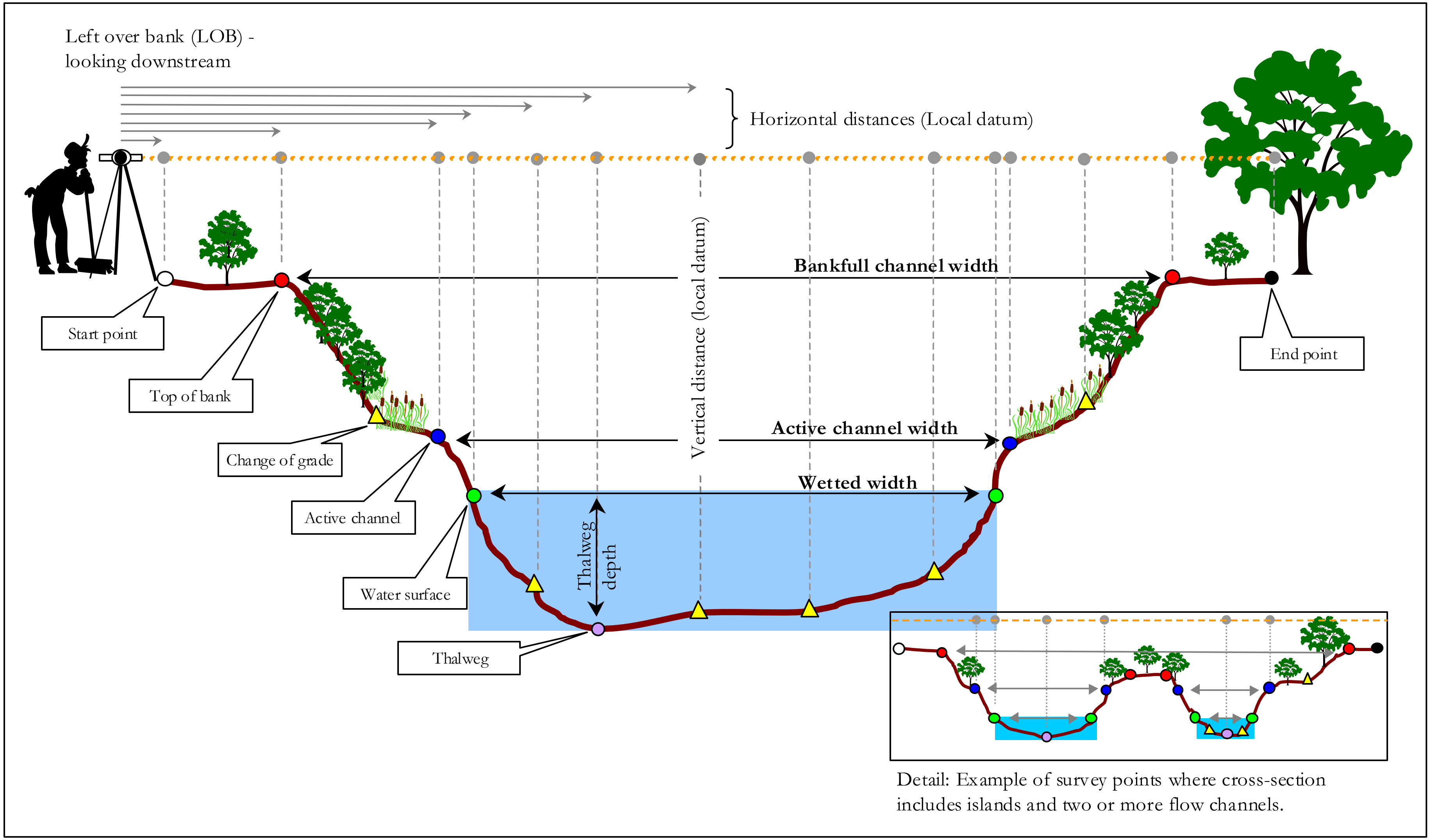


Figure 10. Examples of channel cross-section measurements (illustration provided by Rob Donohue, Dept of Water)

To predict discharge, the hydraulic model relies heavily on subjective estimates of channel bed 'roughness' known as Manning's roughness coefficient or Manning's *n*. Bed roughness is determined by any factor that impedes water flow and includes bed substrates, organic debris (fallen logs etc), in-channel vegetation and channel meander/sinuosity. When discharge is known (Q), Manning's *n* may be calculated empirically using Manning's equation:

$$Q = \frac{AR^{2/3}S^{1/2}}{n}$$

Where, Q is discharge, A is cross-sectional area, S is slope, R is hydraulic radius (area divided by wetted perimeter) and *n* is Manning's *n*. If one or more variables are unknown, *n* may be estimated from Cowan's equation (Cowan 1956), which uses field observations of existing stream condition, and from published tables of roughness coefficients for similar systems; e.g. the national Land and Water database of roughness values for Australian streams (http://www.rivers.gov.au/roughness/index.htm).

The individual HEC-RAS models then need to be calibrated against flows to develop the depth to discharge relationship. Ideally the observed discharge on the day of survey is used as one point in time, and then discharge measurements for a range of discharge events of varying magnitude, related to water levels on selected cross-sections on the reach would be used to construct ratings curves (i.e. stage height vs. flow rate). Recent peak flood flows from the flow record may also be matched with recent flood debris on the reach to gain maximum depth to discharge measurement for the model. These data all assist in showing how stage height (depth) varies with discharge over time. Historic flow data could then be interrogated using the ratings curves from HEC-RAS in the RAP package in order to estimate recurrence intervals of each ecologically important flow event. Ideally, discharge at each site should be measured on a number of occasions over several seasons (& years) to cover a range of stage heights in order to produce the ratings curves and calibrate the HEC-RAS model to each reach. However, this was not possible given the time and budgetary constraints of the current project. Therefore, the models for the East Collie had varying levels of calibration.

Buckingham Reach was not flowing at the time of survey, and there was no suitable flood debris on the reach to allow a peak flow calibration, therefore this model was totally un-calibrated. Bingham Reach was not flowing at the time of survey, but there was flood debris, so this model was calibrated against flood debris and recent peak flows only, and Coolangatta Reach had a minimal baseflow at the time of survey, and had flood debris, so was calibrated against a baseflow and flood debris left by recent peak flows.

Preliminary rating curves were then developed for each model using a range of hypothetical discharges covering the range of known discharges, using log-scale discharge categories. Normal depth was used as the downstream boundary condition, with slope derived from changes in water surface level over the reach.

6.3 Results

6.3.1 Reach characteristics

The survey reaches for the East Collie project were selected to be representative of each area, but also to capture flow changes related to tributaries (i.e. Coolangatta Reach captures inflows from Bingham River), and the proposed flow diversions (off-take above Buckingham Reach).

Buckingham Reach was characterised by long, wide, deep pools at the bottom end of the reach (Plate 3), and shorter, narrower, shallower pools along the upper half of the reach, interspersed with broad, low-profile, uniform gravel/cobble riffles/runs. The reach had little intact riparian vegetation, reflecting disturbance from clearing and agriculture. Similarly there was little emergent or trailing vegetation or woody debris along the reach. The channel was in poor condition as a result of erosion and siltation. Banks along much of the reach were unstable, bare and eroding. Sedimentation of Duderling Pool would appear to be a major issue with the top end of the pool infilled by sand/silt eroded from further upstream. This material is now being stabilised by grasses and saplings (Plate 4).

Coolangatta Reach was also characterised by long, wide, deep pools along the length of the reach (Plates 5 & 6), interspersed with sand/gravel runs between the pools. These runs tended to be held together by woody debris and root mats from adjacent Melaleuca trees. The reach tended to have relatively good riparian vegetation close to the channel, principally mature Melaleuca trees, and these helped stabilise the banks and maintain channel form. The shallower reaches tended to have good loads of woody debris, with Melaleuca trees growing into the channel. This vegetation and woody debris combined to stabilise the channel and moderate flows by increasing roughness.

The Bingham River reach was characterised by short, shallow, narrow pools (Plates 7 & 8), interspersed with sand/root mat shallows/runs between the pools. These runs were held together by woody debris and root mats from tea-trees and Melaleuca trees growing in the channel. The reach tended to have relatively dense riparian vegetation adjacent to and growing into the channel. This helped stabilise the banks and bed, and maintain channel form. In many places the channel was indistinct, dispersing through the vegetation in a poorly formed channel. The vegetation and woody debris combined to stabilise the channel and moderate flows by increasing roughness. The system was characterised by several shallow basins adjacent to the channel, which may be connected for varying lengths of time when the system flows.

Plate 3. The upper East Branch of the Collie River: Buckingham site

Plate 4. Top end of Duderling Pool showing infilling with sand

Plate 5. Lower East Branch below the Coolangatta Gauging Station

Plate 6. Lower East Branch at the Coolangatta Gauging Station looking upstream

Plate 7. The Bingham River at the gauging station

Plate 8. Bingham River downstream of the gauging station

6.3.2 HEC-RAS models

Hydraulic models were constructed for each reach using HEC-RAS. The key outputs from each model are:

- Graphical presentation of each cross-section along the reach;
- Longitudinal elevation of the whole reach;
- Summary table of hydraulic properties for the reach;
- A ratings curve for the reach.

Longitudinal profiles for each reach (Figure 11) show the overall slope and longitudinal shape of the reach. On the plots, the black line, 'Ground' in the legend represents the thalweg, being the deepest, central point of the channel. The small black squares on the ground line show the exact points where each cross-section was located. The blue lines indicate water levels for the three flows, and the left overbank (LOB) top of bank level is indicated. The water level for the lower flow profile is in-filled with blue shading.

HEC-RAS models output a range of hydraulic parameters which describe the hydraulic properties of each cross-section on each reach for each flow profile. The model has a choice of

254 different parameters, a selection of the more commonly used parameters are presented in Table 4 for the reach. The final main output of interest from HEC-RAS is the ratings curve for the reach, which shows the relationship between discharge and stage height (water depth). Under ideal conditions, this would be derived by repeated gauging of discharge at the selected reach for a range of flow scenarios over time, with water depth recorded on each occasion. These observed readings would then used to calibrate the HEC-RAS model and produce a rating curve.

Table 4. Hydraulic parameters output by HEC-RAS.

HEC-RAS Plan: Plan 01 Buckingham Reach: East Collie Profile: PF 7 – discharge of 0.02 m^3/sec

Reach	River Station	Profile	Q Total	Min Ch El	WS Elev	Crit WS	EG Elev	EG Slope	Vel Chnl	Flow Area	Top Width	Froude # Chnl
			(m3/s)	(m)	(m)	(m)	(m)	(m/m)	(m/s)	(m2)	(m)	
Buckingham	15	PF 7	0.02	17.67	18.04	0.00	18.04	0.000002	0.01	1.67	7.92	0.01
Buckingham	14	PF 7	0.02	16.88	18.04	0.00	18.04	0.000000	0.00	7.18	12.20	0.00
Buckingham	13	PF 7	0.02	16.39	18.04	0.00	18.04	0.000000	0.00	13.12	13.89	0.00
Buckingham	12	PF 7	0.02	17.68	18.04	0.00	18.04	0.000001	0.01	2.92	9.44	0.00
Buckingham	11	PF 7	0.02	17.58	18.04	0.00	18.04	0.000001	0.01	2.13	7.96	0.01
Buckingham	10	PF 7	0.02	16.48	18.04	0.00	18.04	0.000000	0.00	10.07	9.31	0.00
Buckingham	9	PF 7	0.02	18.00	18.03	18.03	18.04	0.045933	0.41	0.05	2.83	1.00
Buckingham	8	PF 7	0.02	17.05	17.70	0.00	17.70	0.000000	0.01	2.88	6.56	0.00
Buckingham	7	PF 7	0.02	16.29	17.70	0.00	17.70	0.000000	0.00	7.04	6.46	0.00
Buckingham	6	PF 7	0.02	17.64	17.69	17.69	17.70	0.036712	0.49	0.04	1.27	0.86
Buckingham	5	PF 7	0.02	16.73	17.33	0.00	17.33	0.000001	0.02	1.33	2.99	0.01
Buckingham	4	PF 7	0.02	13.60	17.33	0.00	17.33	0.000000	0.00	71.39	36.51	0.00
Buckingham	3	PF 7	0.02	14.45	17.33	0.00	17.33	0.000000	0.00	49.43	25.92	0.00
Buckingham	2	PF 7	0.02	16.40	17.33	0.00	17.33	0.000000	0.00	4.96	7.94	0.00
Buckingham	1	PF 7	0.02	17.17	17.33	17.25	17.33	0.000930	0.17	0.12	1.43	0.18

HEC-RAS Plan: Plan 01 Coolangatta Reach: East Collie Profile: PF 7 – discharge of 0.02 m^3/sec

Reach	River Station	Profile	Q Total	Min Ch El	WS Elev	Crit WS	EG Elev	EG Slope	Vel Chnl	Flow Area	Top Width	Froude # Chnl
			(m3/s)	(m)	(m)	(m)	(m)	(m/m)	(m/s)	(m2)	(m)	
Coolangatta	15	PF 7	0.02	15.35	17.06	0.00	17.06	0.000000	0.00	15.08	14.42	0.00
Coolangatta	14	PF 7	0.02	16.93	17.03	17.03	17.06	0.034699	0.69	0.03	0.60	0.99
Coolangatta	13	PF 7	0.02	16.56	16.78	0.00	16.78	0.000016	0.02	0.99	5.45	0.02
Coolangatta	12	PF 7	0.02	13.75	16.78	0.00	16.78	0.000000	0.00	45.91	21.14	0.00
Coolangatta	11	PF 7	0.02	16.58	16.78	16.67	16.78	0.000693	0.11	0.19	2.00	0.11
Coolangatta	10	PF 7	0.02	16.40	16.47	16.47	16.49	0.110528	0.62	0.03	0.89	1.01
Coolangatta	9	PF 7	0.02	15.59	16.41	0.00	16.41	0.000000	0.00	7.72	14.79	0.00
Coolangatta	8	PF 7	0.02	15.52	16.41	0.00	16.41	0.000000	0.01	3.97	6.57	0.00
Coolangatta	7	PF 7	0.02	16.30	16.41	16.35	16.41	0.002460	0.10	0.21	3.40	0.12
Coolangatta	6	PF 7	0.02	16.04	16.20	0.00	16.20	0.001449	0.11	0.19	2.05	0.12
Coolangatta	5	PF 7	0.02	15.24	16.20	0.00	16.20	0.000000	0.00	6.85	11.51	0.00
Coolangatta	4	PF 7	0.02	15.21	16.20	0.00	16.20	0.000000	0.00	8.63	16.29	0.00
Coolangatta	3	PF 7	0.02	16.06	16.20	0.00	16.20	0.000892	0.05	0.40	8.51	0.08
Coolangatta	2	PF 7	0.02	16.04	16.12	0.00	16.12	0.020481	0.22	0.10	2.53	0.35
Coolangatta	1	PF 7	0.02	15.06	15.11	15.11	15.13	0.171716	0.58	0.04	1.03	1.00

HEC-RAS Plan: Plan 01 Bingham Reach: East Collie Profile: PF 7 – discharge of 0.02 m^3/sec												
Reach	River Station	Profile	Q Total	Min Ch El	WS Elev	Crit WS	EG Elev	EG Slope	Vel Chnl	Flow Area	Top Width	Froude # Chnl
			(m3/s)	(m)	(m)	(m)	(m)	(m/m)	(m/s)	(m2)	(m)	
Bingham	17	PF 7	0.02	19.20	19.22	19.22	19.22	0.144576	0.25	0.08	7.59	0.76
Bingham	16	PF 7	0.02	17.36	19.07	0.00	19.07	0.000000	0.00	13.84	12.55	0.00
Bingham	15	PF 7	0.02	18.99	19.05	19.05	19.07	0.233654	0.63	0.03	1.08	1.14
Bingham	14	PF 7	0.02	18.16	19.00	0.00	19.00	0.000000	0.00	6.08	12.44	0.00
Bingham	13	PF 7	0.02	18.95	19.00	18.97	19.00	0.004956	0.10	0.21	5.97	0.17
Bingham	12	PF 7	0.02	18.60	18.70	18.70	18.72	0.179982	0.62	0.03	0.83	0.99
Bingham	11	PF 7	0.02	17.10	18.65	0.00	18.65	0.000000	0.00	8.47	9.33	0.00
Bingham	10	PF 7	0.02	18.54	18.62	18.62	18.64	0.169687	0.69	0.03	0.64	1.02
Bingham	9	PF 7	0.02	17.77	18.55	0.00	18.55	0.000003	0.01	1.66	3.31	0.01
Bingham	8	PF 7	0.02	18.27	18.55	0.00	18.55	0.000103	0.04	0.51	2.97	0.03
Bingham	7	PF 7	0.02	16.65	18.55	0.00	18.55	0.000000	0.00	9.86	10.49	0.00
Bingham	6	PF 7	0.02	18.45	18.53	18.53	18.55	0.165826	0.61	0.03	0.90	0.99
Bingham	5	PF 7	0.02	18.16	18.54	0.00	18.54	0.000002	0.01	1.79	7.89	0.01
Bingham	4	PF 7	0.02	17.79	18.54	0.00	18.54	0.000002	0.01	1.57	3.86	0.01
Bingham	3	PF 7	0.02	18.43	18.51	18.51	18.53	0.185959	0.65	0.03	0.82	1.05
Bingham	2	PF 7	0.02	17.54	18.33	0.00	18.33	0.000000	0.00	4.48	9.92	0.00
Bingham	1	PF 7	0.02	18.30	18.31	18.31	18.33	1.298656	0.47	0.04	5.38	1.64

Codes:

Q Total = total flow in cross-section.
Min Ch El = minimum channel elevation.
WS Elev = calculated water surface from energy equation.
Crit WS = critical water surface elevation; water surface corresponding to the minimum energy on the energy versus depth curve.
EG Elev = energy grade line for given WSEL.
EG Slope = slope of the energy grade line.
Vel Chnl = average velocity of flow in main channel.
Flow Area = total area of cross-section active flow.
Top Width = top width of the wetted cross-section.
Froude # Chnl = Froude number for the main channel.

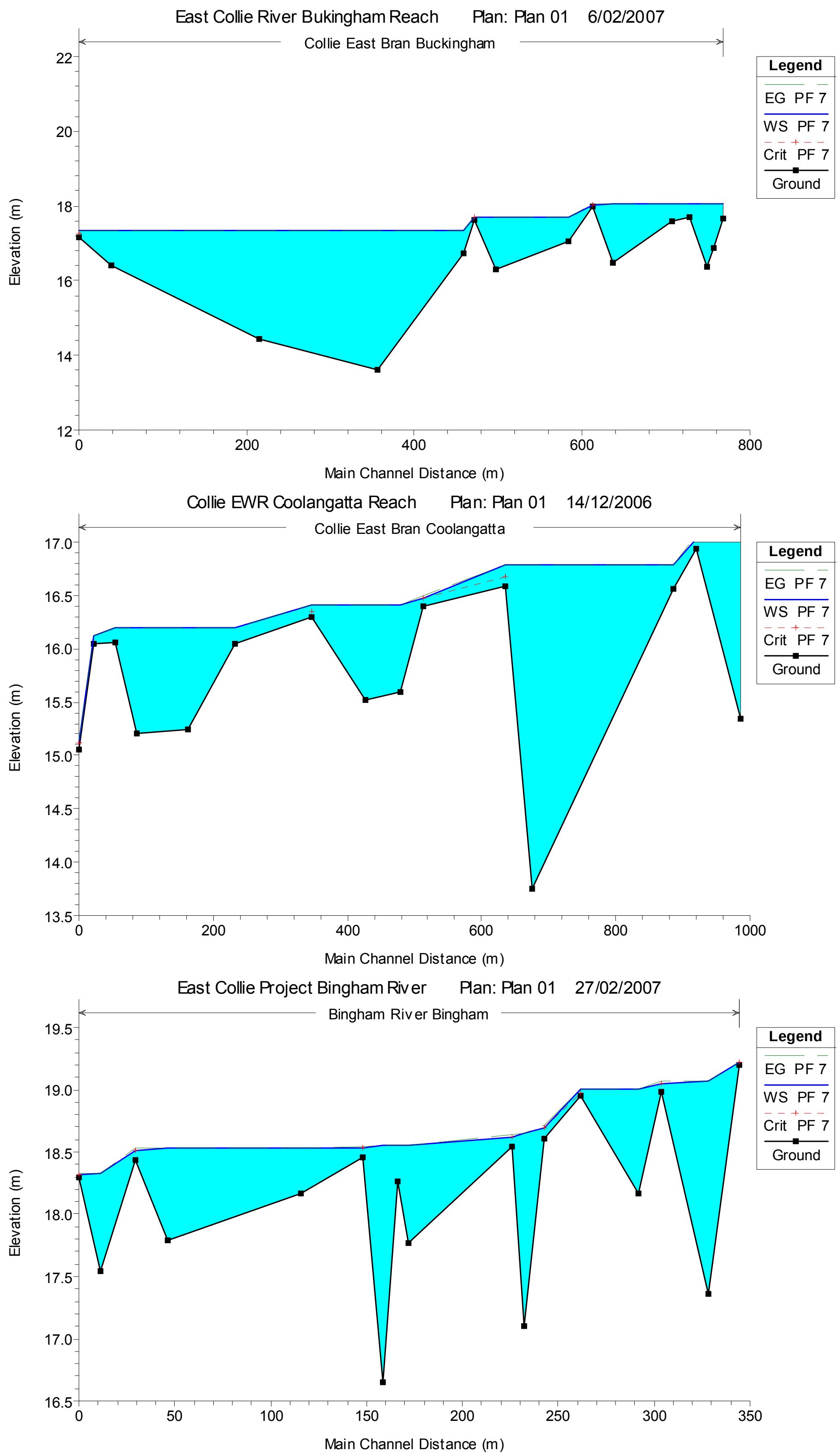


Figure 11. Longitudinal profiles of Buckingham Reach (top), Coolangatta Reach (middle) and Bingham Reach (bottom) showing location of cross-sections, water levels for a flow profile (0.02 m^3/sec), left overbank (LOB) top of bank level, with water level for the lower flow profile in-filled in blue.

7 WATER-DEPENDENT ECOLOGICAL VALUES

A critical step in the determination of EWRs for an aquatic system is the identification of key water-dependent ecological values. In undisturbed environments, the usual aim in establishing EWRs is to protect the existing 'natural' ecological values at a low level of risk. However, for ecosystems that have been modified (i.e. through regulation and/or land use changes), EWRs can be established to:

- maintain and/or enhance current key ecological values,
- identify the likely pre-existing ecological values and determine the key values which the EWRs should aim to re-establish, or
- provide for a combination of current key ecological values and key pre-existing natural ecological values.

Identification of past and present ecological values, therefore, is an important part of any EWR study. Current ecological values were determined from field surveys of aquatic fauna (macroinvertebrates and fish), riparian habitat condition and selected physico-chemical parameters undertaken between the 27th of February and the 1st of March 2006.

7.1 Methods

7.1.1 Site locations

Four sites were selected to represent the range of values likely to exist in the East Collie and Bingham rivers (Table 5 & Figure 4). Three of these sites corresponded with survey reaches, and the fourth was located in the Upper East Branch of the Collie, above the proposed diversion and in the vicinity of Wargyl Pool. This site was sampled for macroinvertebrates and fish because of its considerable salinity levels when compared with the reach immediately downstream of the Chicken Creek confluence where the system is supplemented with fresh water to maintain pools.

Table 5. Names and locations of each site sampled

Site Code	Location
UE	Upper East Collie; near the **Buckingham** Gauging Station, downstream of the proposed diversion but upstream of the Bingham River
LE	Lower East Collie; near the **Coolangatta** Gauging Station, downstream of both the proposed diversion and the Bingham River
BR	**Bingham** River; near the upper gauging station
AD	Above Diversion; East Collie upstream from the proposed diversion & above Wargyl Pool

7.1.2 Physico-chemical parameters

In situ measurements of basic water quality parameters were made at each sampling site using hand held field meters. Parameters and units of measurement are presented in Table 6.

Table 6. *In situ* water quality parameters measured at each site.

Parameter	Code	Units
pH	pH	H^+
Water Temperature	Temp	ºC
Conductivity	Cond	µS/cm
Salinity	Sal	ppt
Dissolved Oxygen	DO	% sat.
Dissolved Oxygen	DO	mg/L
Redox	ORP	mV
Turbidity	Turb	NTUs

7.1.3 Macroinvertebrate fauna

At each site, the aim of the sampling programme was to maximise the number of taxa collected to facilitate the characterisation of the fauna with respect to their likely water requirements. To this end, as many habitats as possible were sampled, including trailing riparian vegetation, woody debris, open water column and benthic sediments. Within a site, each pool was sampled thoroughly and combined to form one composite sample for each site. Sampling was conducted with a 250 µm mesh net to selectively collect the macroinvertebrate fauna. This sampling protocol was based upon a standardised approach adopted for previous macroinvertebrate surveys (see Storey *et al.* 1993; Edward *et al.* 1994).

Samples were preserved in 70% alcohol and returned to the laboratory where they were sorted under low power microscope to remove animals. All taxa were identified to the lowest possible level (species, where possible) and enumerated to $\log_{10}$ scale abundance classes (i.e. 1 = 1 individual, 2 = 2 - 10 individuals, 3 = 11 - 100 individuals, 4 = 101 - 1000 individuals and 5 = 1000+). Specialist taxonomic expertise by Dr D.H. Edward, The University of Western Australia, was subcontracted to identify Chironomidae.

The existence of rare, restricted or endemic species was determined by cross-referencing taxa lists for each site/habitat with an established database (School of Animal Biology, The University of Western Australia), the Department of Conservation and Land Management (CALM) Wildlife Conservation (Specially Protected Fauna) Notice 2003 (Government Gazette 11 April 2003, pp. 1158-1167) and with the 2004 IUCN Red List of Threatened Species (IUCN 2004). A recent, comprehensive study by Sutcliff (2003) on the conservation significance of south-west Odonata, Plecoptera and Trichoptera was also consulted.

Due to the general lack of data on distributions of Australian aquatic macroinvertebrates and taxonomic uncertainties, conclusions on the rarity or otherwise of many macroinvertebrate groups/species cannot be made with certainty.

7.1.4 Fish fauna

The fish fauna was sampled at each site by a range of methods with the aim of maximising the number of species recorded to facilitate the characterisation of the fauna with respect to their likely water requirements. Methodology depended upon the characteristics of each site, but generally consisted of electrofishing, sweep netting of trailing riparian vegetation, fyke nets, box traps and direct observation.

Sweep netting consisted of aggressive sweeps of the marginal and emergent vegetation with 250 μm and 1 mm mesh aperture dip nets. Mesh box traps baited with cat biscuits were set over night at each site. Traps were cleared the following day.

Due to the high electrical conductivities of the water, electrofishing was only carried out at one site (pool 5 in the upper Bingham River). Electrofishing was conducted with a Smith-Root Model 12B backpack shocker (set to 200 volts DC output with a pulse frequency of 70 Hz; mode switches at J4). Electrofishing is an extremely useful and efficient sampling tool in small rivers with clear, low salinity water. At pool 5 in the upper Bingham River, the electrofisher was used along the banks, within riparian vegetation, over macrophytes, amongst large woody debris and in open 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.

Principles of electrofishing: a DC voltage is passed from a negative electrode (cathode) to a positive electrode (anode) whilst the electrodes are immersed in the water. If a fish is caught in the electrical field generated, a process referred to as galvanotaxis occurs. This is the involuntary movement of the fish towards the anode, until it reaches an electrical field strong enough to stun it. Stunning is referred to as galvanoarcosis. The Smith-Root electrofisher uses a pulsed DC current, which is more effective than a flat DC signal because the body of the fish flexes with each pulse, accentuating the involuntary swimming action towards the anode. Once the current is switched-off, or the fish removed from the electrical field, the fish quickly recovers. The operator carries the anode (in the form of a modified pond net) whilst trailing the cathode (a stainless steel cable approximately 3.5 m long, referred to as a 'rat tail'). The Smith-Root backpack electrofisher has an effective range of approximately 3 m around the operator. Galvanotaxis can be used to 'pull' fish and crayfish out from under debris, logs, boulders and bank undercuts.

Fyke netting was conducted at the majority of sites. Fyke nets were set at a 45° angle to the bank in pools deeper than 1 metre to create a complete barrier to fish passage (Plate 9). The fyke comprised a single 10 m leader and a 5 m net of approx 8 mm mesh size. A float was placed at the cod-end (closest to the bank) to provide an air space for tortoise. The fyke net was set amongst snaggy habitat at night and retrieved the following day.

All fish and tortoise captured by any method were identified, measured and returned to the water alive.

Plate 9. The Fyke net set up at the Upper East Site (near Buckingham Gauging Station). Photo taken by Andrew Storey, March 2006.

7.1.5 Faunal EWRs

EWRs for aquatic fauna (fish and macroinvertebrates) were developed by considering the biology, life history and habitat requirements of taxa recorded from the field survey. This was achieved using expert knowledge, and by reference to available literature on reproductive biology and ecology of specific taxa, including aspects such as preference for seasonal or perennial flows, habitat requirements, diet, and general biology such as timing of breeding and migration.

7.2 Results & Discussion

7.2.1 Physico-chemical parameters

The physico-chemical characteristics of each site are detailed in Table 7. Weather conditions over the sampling period were generally hot and humid. Water temperatures were generally high and ranged from 21 °C (pool 4 from the Lower East Collie; Buckingham) to 29.9°C (pool 1 from the site above the proposed diversion on the Upper East Collie), likely reflecting differences in water depth (i.e. shallower pools have greater daily ranges in temperature than deeper pools), time of day when sampled (waters sampled in the morning will likely be cooler than those sampled in the afternoon), and water colour (tannin-stained water tends to absorb more heat than clear water).

Water quality was assessed against the water quality guidelines for the protection of aquatic ecosystems (ANZECC/ARMCANZ 2000), using data specific to slightly – moderately disturbed freshwater ecosystems of south-west Western Australia (Table 8). The ANZECC guidelines specify biological, sediment and water quality guidelines for protecting the 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 freshwater and marine ecosystems, including biological diversity, relative abundance, and ecological processes" (ANZECC/ARMCANZ 2000).

Table 7. *In situ* water quality results.

Site Code	Pool	Temp ° C	pH	Sal ppt	Cond (µs/cm)	DO (%)	DO (mg/L)	ORP mV	Turb NTU
UE	1	26.0	4.58	2.03	3671	60.5	4.9	441	60.2
UE	2	24.7	7.01	3.72	6547	39.7	3.2	250	8.5
UE	3	25.2	7.57	4.05	7098	61.9	4.8	202	8.2
LE	1	25.4	7.88	4.63	8700	87.4	7.0	265	43.1
LE	2	25.1	7.56	5.38	9500	71.7	6.1	216	71.8
LE	3	24.1	7.58	5.57	10000	80.6	6.4	186	56.5
LE	4	21.0	7.82	5.18	9200	68.0	6.0	22	135.2
BR	1	26.0	7.48	3.65	6476	106.0	8.5	47	123.7
BR	2	26.0	7.56	3.24	5830	82.7	6.4	-32	136.3
BR	3	27.8	8.13	3.10	5605	110.9	8.4	64	72.8
BR	4	25.5	7.65	3.85	7099	73.0	5.7	118	80.0
BR	5	28.0	8.30	0.52	1085	117.2	9.3	333	91.4
AD	1	29.9	8.05	25.50	40000	136.0	8.9	172	37.6
AD	2	26.2	7.18	15.05	24800	92.1	6.9	174	31.5
AD	3	29.5	8.68	21.84	34900	128.0	8.5	150	22.0

Table 8. ANZECC/ARMCANZ (2000) default physico-chemical trigger values for slightly disturbed Western Australian ecosystems.

Ecosystem Type	DO % saturation[b]	pH
Upland River[a]	90 - na	6.5 – 8.0
Lowland River[a]	80 - 120	6.5 – 8.0
Lakes & Reservoirs	90 – no data	6.5 – 8.0
Wetlands	90 - 120	7.0 – 8.5

Na = not applicable

[a] All values during base river flow not storm events

[b] Derived from daytime measurements; may vary diurnally and with depth.

Dissolved oxygen (DO) levels fell below guidelines within a number of pools in the East Collie and Bingham Rivers (Tables 7 & 8). This is likely a seasonal effect resulting from low flows reducing the rivers to a series of pools over summer.

DO concentration in any water body is the net result of biological processes (respiration and photosynthetic rates) and physical re-aeration (*e.g. via* wind action and water flow). Physical processes include re-aeration, which is the exchange of oxygen between the surface of the water and the atmosphere, at the water-air interface. Biological processes include metabolic rates, *i.e.* photosynthesis and respiration by aquatic biota. Low day-time DO is usually due to a combination of factors: fewer in-stream primary producers (algae and macrophytes) and microbial-rich benthic sediments coupled with high water temperatures that limit oxygen saturation. Bacteria and other benthic invertebrates associated with excessive organic material (e.g. plant detritus) consume DO to the extent where, during summer, anoxic conditions would likely prevail. Anoxia not only causes the localised extinction of fauna, but also results in the mobilisation of nutrients and some heavy metals from sediments causing further water quality problems. Even low (<40% saturation) overnight DO levels may increase the potential for water quality problems through desorption (release) of nutrients (e.g. phosphorus) and heavy metals from sediments. The toxicity of many compounds such as ammonia is thereby increased as they become more bioavailable. Low dissolved oxygen levels can also promote the growth of toxic anaerobic bacteria such as *Clostridium botulinum*.

Only one pool sampled during the current study was fresh (pool 5 on the Bingham River) as defined by the Department of Environment (2003)[3]. Other pools from the Bingham River were saline. The upper East Collie (Buckingham) recorded brackish to saline waters, with salinity increasing considerably above the proposed diversion, in the vicinity of Wargyl Pool. In this area, salinities as high as 25.5 ppt were recorded (see Table 7). This is likely due to high salinity runoff from adjacent land, in conjunction with concentration effects associated with low flows and evaporation. Notably, salinity dropped from 25 ppt to 15 ppt from above to below Wargyl Pool, indicating probably fresher groundwater inputs at this point. The lower reach of the East Branch of the Collie River (Coolangatta), below the confluence with Bingham River, was saline at the time of sampling; salinity ranged from 4.63 to 5.57 ppt. This reach likely recorded lower salinities than upstream sites due to dilution effects resulting from inflow of lower salinity water from the Bingham River in addition to pumping of fresh bore water to maintain pools. High temporal variability in the concentration of dissolved salt is a common phenomenon in the majority of aquatic ecosystems in Australia, even those which are not undergoing salinisation (Hart *et al.* 2003; Nielson *et al.* 2003). Generally, there is a concentration effect with higher salinities at lower flows and lower salinities at higher flows (Hart *et al.* 2003; Nielson *et al.* 2003).

In natural systems, pH is determined by atmospheric and geological factors (*e.g.* the result of inundation of limestone-rich or, conversely, humus-rich substrates) and to some extent, rates of primary productivity. Anthropogenic determinants include acid rock drainage (ARD), acid sulphate soils, acid deposition (*e.g.* acid rain) and agriculture. Most river systems in Western Australia have a natural pH range circum-neutral. According to the ANZECC/ARMCANZ (2000) guidelines, the natural pH range is typically between 6.5 and 8 in lowland rivers within south-west WA (Table 8). All pools sampled in the current study recorded circum-neutral to basic pH within guidelines, with the exception of pool 1 from the Upper East Collie (Buckingham; Table 7). This pool is just below the Chicken Creek confluence and contained acidic water, recording a pH of 4.58. This is likely a result of pumping acidic bore water to maintain pools in this reach. However, the pH effect is rapidly reduced downstream of the pumping site, returning to circum-neutral with approximately 100 m of the confluence. An increase in acidity can have profound impacts on aquatic fauna, particularly molluscs. Low pH typically results in decreased abundance and diversity of aquatic fauna, usually through indirect effects of altered trace metal toxicity (Dallas & Day 1993). Naturally acidic streams and wetlands in south-western Australia (pH ~4) tend to have lower macroinvertebrate species diversity and abundance (Pusey & Edward 1990, Davis *et al.* 1993). Even small changes in natural pH may have sub-lethal effects on aquatic fauna by altering ionic and osmotic balances (Dallas & Day 1993). Sub-lethal effects include reduced fecundity and impaired growth and development. Other impacts to aquatic systems include a decline in some algal species, cessation of nitrification in wetland systems, and sub-lethal and acute responses in some fish. In addition, acidic conditions can lead to the mobilisation and leaching of naturally occurring contaminants from soils, particularly aluminium. This can lead to toxic effects in aquatic fauna.

[3] < 1.5 ppt = fresh, 1.5 – 3 ppt = brackish, 3 - 35 ppt = saline, > 35 ppt = hypersaline

ORP, or Oxidation-Reduction Potential, is a measure of a systems capacity to oxidise material. Positive oxidation values indicate oxidation (e.g. biological conversion of ammonia to nitrites and nitrates), whilst negative values suggest reduction (e.g. removal of nitrates). Redox (Reduction-Oxidation) was highly variable across sites and pools, and ranged from -32 mV in the Bingham River (pool 2) to 440 mV in the Upper East Collie (pool 1; Buckingham).

7.2.1.1 Patterns in physico-chemical data

Principal Components Analysis (PCA) on the 15 samples from March 2006 reduced the physico-chemical dataset to five principal components, with over 74% of the variation being explained by PC1 and PC2 (Table 9).

Table 9. Eigenvalues, percent of variance and cumulative percent of variance explained by the five components from PCA performed on water quality data.

PC	Eigenvalues	% Variation	Cumulative % Variation
1	3.96	49.5	49.5
2	2.03	25.4	74.9
3	1.08	13.4	88.3
4	0.59	7.4	95.7
5	0.27	3.3	99.0

There was some separation of study reaches within ordination space (Figure 12). Two of the pools from the Upper East Branch of the Collie, above the proposed diversion separated from all other pools along PC1, on the basis of higher salinity and dissolved oxygen levels.

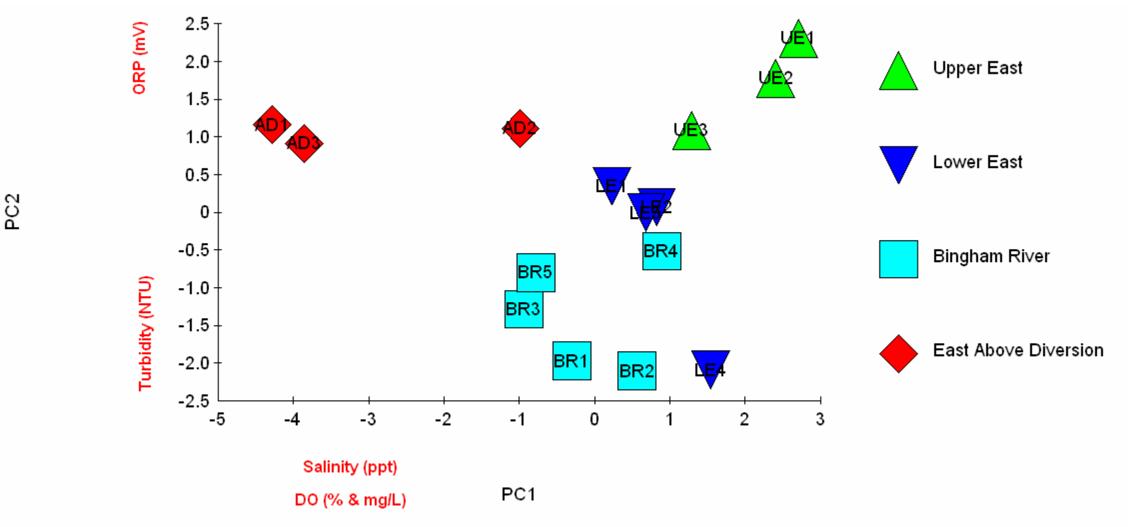


Figure 12. PCA ordination of physico-chemical characteristics for all pools sampled during March 2006.

7.2.2 Macroinvertebrates

7.2.2.1 Taxonomic composition

A total of 109 taxa were recorded from the four reaches in January 2006 (Table 10). The composition of macroinvertebrate taxa was typical of lotic (flowing) freshwater systems throughout the world (Hynes 1970), and was dominated by Insecta (78%), with the majority being Diptera (31% of the insects). Molluscs only comprised 4% and Crustacea 7% of the total fauna. Generally, insects comprise between 70 and 80% of the invertebrate fauna from undisturbed streams in the northern jarrah forest of Western Australia (Bunn *et al.* 1986, Storey *et al.* 1990). Bunn and Davies (1992), suggest that salinisation of rivers is likely to be accompanied by a shift in community structure from an insect dominated to crustacean dominated system.

The invertebrate fauna comprised one Hydrazoa (hydra), Oligochaeta (aquatic worms), four species of Gastropoda (snails), four types of micro-crustacea (water fleas, seed shrimps etc), one Amphipoda (side swimmers), three types of other macro-crustacea (freshwater crayfish and shrimp), nine Arachnida (water mites and spiders), one Collembolla (springtail), 26 Diptera (flies), six Trichoptera (caddis flies), two species of Ephemeroptera (mayflies), 13 Odonata (dragonflies & damselflies), nine Hemiptera (true bugs) and 29 taxa of Coleoptera (aquatic beetles).

The taxonomic listing also includes records of larval and pupal stages for groups such as Diptera and Coleoptera. Current taxonomy in Australia is not sufficiently well developed to allow identification of all members of these groups to species level. In many instances, it is likely that these stages are the same species as the larval/adult stages recorded from the same location. However, because this could not be definitively determined, they were treated as separate taxa.

7.2.2.2 Taxa richness

The East Branch of the Collie River recorded a total taxa richness of 91, the majority of which were collected from the lower reaches, below the proposed diversion and Wargyl Pool. In the Bingham River, 72 types of taxa were collected. However, it must be noted that invertebrates collected from the East Branch of the Collie were collected from three reaches, whilst in the Bingham River all taxa came from just one reach.

Of the 109 taxa, seven occurred at all four sites. These common taxa included oligochaetes, the amphipod *Austrochiltonia subtenuis*, the chironomid *Procladius paludicola*, the diptera larvae Ceratopogoniinae spp., the trichoptera *Oecetis* sp., the odonate Coenogrionidae spp., and the dytiscid *Necterosoma darwini* (Table 10). Of the remaining taxa occurrences, 22 occurred at three sites, 31 at two sites, and 49 taxa (45% of taxa) were singletons, occurring at only one site (Table 10). Collecting a high number of singleton taxa is quite common in studies of freshwater systems in Western Australia. In a study encompassing 223 wetlands across the wheatbelt, Pinder *et al.* (2004) similarly found a high proportion of singletons.

Table 10. Macroinvertebrate taxa occurrences across sites. Abundances are log_{10} scale classes: 1 = 1 individual, 2 = 1- 10, 3 = 11 – 100, 4 = 101 - 1000, 5 = >1000. Cons. Cat. = conservation category where: C = common taxa recorded from other states/territories; I = indeterminate; I* = indeterminate but probably occurs only in SW; S = endemic to SW but commonly occurring; L = endemic to SW but with a restricted distribution; and, E = exotic/introduced.

Class/Order	Family	Species/taxa	Cons cat	Upper East Collie (AD)	Upper East Collie	Bingham River	Lower East Collie
CNIDARIA							
HYDROZOA	**Hydroida**	*Hydra* sp.	I	0	0	1	0
ANNELIDA							
OLIGOCHAETA		Oligochaeta spp.	C	2	2	3	2
MOLLUSCA							
GASTROPODA	**Ancylidae**	*Ferrissia petterdi*	C	0	0	2	0
	Lymnaeidae	*Lymnaea stagnalis*	E	0	0	2	0
	Physidae	*Physa acuta*	E	0	0	3	0
	Planorbidae	*Glyptophysa* sp.	I	0	0	2	1
CRUSTACEA							
CLADOCERA		Cladocera spp.	I	0	2	2	3
COPEPODA	**Cyclopoida**	Cyclopoida spp.	I	4	4	0	4
		Calanoida spp.	I	0	0	4	0
OSTRACODA		Ostracoda spp.	I	0	1	4	4
AMPHIPODA	**Ceinidae**	*Austrochiltonia subtenuis*	S	3	3	4	4
DECAPODA	**Palaeomonidae**	Palaeomonidae spp.	I*	0	0	4	0
	Parastacidae	*Cherax cainii*	S	0		2	2
		Cherax quinquecarinatus	S	0	2	2	2
ARACHNIDA							
ARANAE		Aranae spp.	I	0	1	2	0
	Tetragnathidae	*Tetragnatha* sp.	I	0	2	2	0
ACARINA		*Acarina* spp. (nymphs)	I	0	2	0	0
Prostigmata	**Eylaidae**	*Eylais* sp.	I	0	1	1	0
	Hydrodromidae	*Hydrodroma* sp.	I	0	2	2	2
	Hydryphantidae	*Diplodontus* sp.	I	0	0	0	2

Class/Order	Family	Species/taxa	Cons cat	Upper East Collie (AD)	Upper East Collie	Bingham River	Lower East Collie
	Unionicolidae	*Koenikea* sp.	C	0	2	0	0
		Unionicola sp.	I	0	0	0	2
	Trombidioidea	Trombidioidea spp.	I	0	0	1	0
COLLEMBOLLA	**Entomobryoidea**	Entomobryoidea spp.	I	0	2	0	0
INSECTA							
DIPTERA	**Chironomidae**	Chironomidae spp. (pupa)	I	0	1	2	2
	Chironominae	*Chironomus aff. alternans*	C	0	0	0	3
		Cladopelma curtivalva	C	0	0	1	0
		Cryptochironomus griseidorsum	C	0	1	1	1
		Dicrotendipes ?conjuctus	S	0	0	3	0
		Dicrotendipes conjunctus	S	0	3	0	0
		Dicrotendipes sp.	I	0	0	2	2
		Kiefferulus intertinctus	C	0	3	3	0
		Kiefferulus martini	C	0	2	1	0
		Parachironomus sp. VSCL35	C	0	0	2	0
		Polypedilum (Pentapedilum) leei	C	0	0	0	2
		Polypedilum nubifer	C	0	0	2	2
		Rheotanytarsus sp.	I	0	0	2	0
		Tanytarsus ?fuscithorax	C	0	2	3	3
		Tanytarsus sp.	I	4	2	3	0
	Orthocladiinae	*Larsia ?albiceps*	C	0	0	0	1
		Limnophyes ?pullulus	C	0	1	0	0
		Parakiefferiella sp. VCD2	I	0	0	2	0
	Tanypodinae	*Paramerina levidensis*	S	0	1	2	1
		Procladius paludicola	C	4	3	3	3
	Ceratopogonidae	Ceratopogonidae spp. (pupa)	I	1	0	2	0
		Ceratopogoniinae spp.	I	3	3	3	3
		Forcypomiinae spp.	I	0	2	0	0
	Stratiomyidae	Stratiomyidae spp.	I	0	0	0	1
	Tabanidae	Tabanidae spp.	I	0	2	2	2

Class/Order	Family	Species/taxa	Cons cat	Upper East Collie (AD)	Upper East Collie	Bingham River	Lower East Collie
	Muscidae	Muscidae spp.	I	0	2	0	0
TRICHOPTERA	**Ecnomidae**	*Ecnomus* sp.	I*	0	0	0	3
	Hydroptilidae	*Acritoptila/Hellyethira* spp.	I*	0	0	1	1
	Leptoceridae	*Notalina spira*	C	0	3	3	3
		Oecetis sp.	I*	3	2	2	2
		Triplectides australis	C	0	3	3	2
		Leptoceridae spp. (pupa)	I	0	0	2	0
EPHEMEROPTERA	**Baetidae**	*Cloeon* sp.	C	0	0	3	2
	Caenidae	*Tasmanocoensis tillyardi*	C	0	0	3	3
ODONATA							
Zygoptera	**Lestidae**	*Austrolestes annulosus*	C	3	0	0	2
	Coenogriondae	Coenogrionidae spp.	I	1	3	2	2
		Austroagrion cyane	C	0	2	2	1
		Ischnura aurora	C	0	2	0	0
		Xanthagrion erythroneurum	C	0	1	0	1
	Gomphidae	*Austrogomphus collaris*	S	0	0	2	0
		Austrogomphus lateralis	S	0	0	2	0
Anisoptera	**Aeshnidae**	*Hemianax papuensis*	C	0	2	2	0
	Hemicorduliidae	Hemicordulia sp. (imm.)	I	0	1	2	0
		Hemicordulia australiae	C	0	1	1	2
		Hemicordulia tau	C	0	2	2	2
	Libellulidae	*Diplacodes haematodes*	C	0	0	1	2
		Orthetrum caledonicum	C	0	0	3	1
HEMIPTERA	**Veliidae**	*Microvelia* sp.	I	0	3	1	0
	Corixidae	Corixidae spp. (imm.)	I	0	2	0	0
		Agraptocorixa eurynome	C	0	2	0	0
		Agraptocorixa sp. (F)	C	0	2	0	0
		Sigara truncatipala	C	0	2	0	0
		Sigara sp. (F)	C	0	3	0	0
		Diaprepocoris sp.	C	2	0	0	0

Class/Order	Family	Species/taxa	Cons cat	Upper East Collie (AD)	Upper East Collie	Bingham River	Lower East Collie
		Micronecta sp.	C	0	4	1	3
	Notonectidae	*Anisops* sp. (imm./F)	I	0	3	0	0
COLEOPTERA	**Dytiscidae**	*Allodessus bistrigatus*	C	0	1	0	0
		Antiporus gilberti	C	0	0	2	0
		Lancetes lanceolatus	C	0	0	0	2
		Limbodessus inornatus	S	0	3	0	2
		Limbodessus shuckhardi	C	0	2	1	0
		Megaporus howitti	C	0	2	2	0
		Megaporus solidus	S	0	1	2	2
		Necterosoma darwini	S	1	2	3	1
		Necterosoma pencillatus	C	0	1	0	0
		Necterosoma regulare	C	0	0	2	2
		Rhantus suturalis	C	0	2	2	0
		Sternorpiscus brownii	S	0	0	3	2
		Sternopriscus multimaculatus	C	0	2	1	0
		Sternopriscus sp. A (F) (cf. *multimaculatus*)	C	0	1	2	1
		Sternopriscus sp. B (F) (cf. *brownii*)	I*	0	1	3	2
		Necterosoma sp. (L)	I	2	1	0	3
		Megaporus sp. (L)	I	0	0	0	3
	Gyrinidae	*Macrogyrus ?angustatus*	C	0	2	2	0
		Aulonogyrus/Macrogyrus sp. (L)	I	0	1	2	2
	Haliplidae	*Haliplus gibbus/fuscatus* (F)	C	0	0	1	0
	Hydraenidae	*Octhebius* sp.	I	0	2	0	1
	Hydrochidae	*Hydrochus* sp.	I	0	3	2	0
	Hydrophilidae	*Berosus* sp. (F)	I	2	0	0	0
		Berosus sp. (L)	I	2	0	0	0
		Enochrus ?esuriens (female)	C	0	1	0	0
		Helochares tenuistriatus	S	0	0	1	0
		Hydrophilus latipalpus	C	0	1	0	0
		Paracymus pygmaeus	C	0	2	3	0
		Paranacaena sp.	I	0	2	0	0
		TOTAL NO. TAXA		**15**	**66**	**72**	**52**

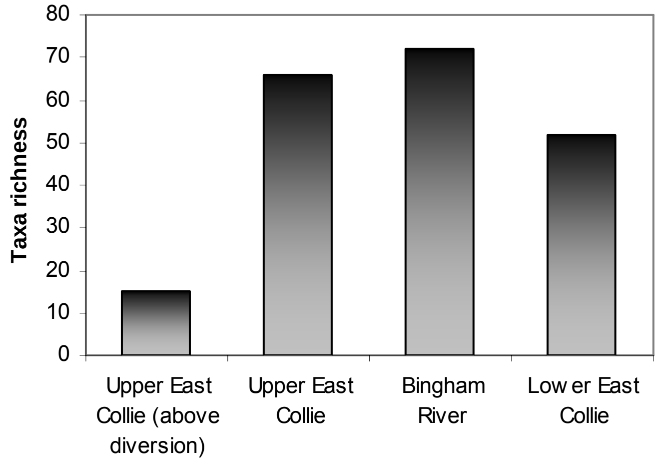


Figure 13. Taxa richness recorded from each reach.

Aquatic invertebrate biodiversity varied between sites (Figure 13). The highly saline reach of the Upper East Branch of the Collie (above the proposed diversion) recorded a notably lower number of taxa (15) than other reaches sampled in the East Branch of the Collie and the Bingham River. All other reaches had comparable taxa richness, with the Bingham River supporting the greatest number of taxa (72) (see Figure 13). Results from this study support the literature which suggests that taxa richness declines as salinity increases (i.e. Bunn & Davies 1992, Mitchell & Richard 1992, Wollheim & Lovvorn 1995).

Pools within the highly saline reach were characterised by no cover, little habitat, open substrates and water with a green hue. Therefore, the low number of invertebrates recorded is likely a result of both physical tolerances associated with the saline conditions, as well as low habitat diversity. The majority of taxa collected from this reach are known to be highly tolerant of adverse water quality, including high salinities. For example, the chironomid *Procladius paludicola* is common throughout Western Australia and is known to occur in a range of environmental conditions from lotic to lentic, polluted to pristine. A number of *Tanytarsus* species are also known to be widespread and pollution tolerant. In addition, the damselfly larvae *Austrolestes annulosus* is known to tolerate saline conditions (Hawking & Theischinger 1999), and is commonly found in secondary saline systems across Australia, including the Muir-Unicup peat wetlands in the southwest (WRM 2005c). In fact, the macroinvertebrate community recorded from this reach was comparable to that of the Hotham River, another saline river system in south-western Australia (Bunn & Davies 1992), in which the macroinvertebrate assemblages were also dominated by the amphipod *Austrochiltonia subtenuis* and *Tanytarsus* chironomids.

There is little published data on aquatic fauna of the study area. Although EWRs have been previously undertaken on a number of areas of the Collie River, none specify the number or types of macroinvertebrates collected (see section 1.3.3 above).

A number of sites within the Collie basin were sampled as part of a program to develop a biomonitoring system for rivers based on macroinvertebrates (First National Assessment of River Health (FNARH)) (Halse *et al.* 2002). However, the study was based on the AusRivas approach, and as such classification of macroinvertebrates was only to Family level. The Collie basin was classified as 'severely impaired' based on its AusRivas score, however, condition was uneven across the catchment, with the final score being influenced by the location of sites (Halse *et al.* 2002).

In addition, a study was conducted as part of water resource planning to ensure the maintenance of environmental and social values of the Collie River (Kay *et al.* 2000). In particular, this study examined the effect of supplementing pools in summer with groundwater that has become acidic through de-watering and acid mine drainage. Pools on both the South and East Branches of the

Collie River were sampled in spring 1998 and autumn 1999. Using family level identification and the AusRivAS program, Kay *et al.* (2000) found aquatic fauna was adversely affected for only a short distance downstream of groundwater pumping (Kay *et al.* 2000).

In the current study, the East Branch of the Collie River recorded an average species richness of 44, while the Bingham had a total species richness of 72 for the one reach sampled. Streamtec (1998) assessed the biodiversity of aquatic invertebrates in the Harvey system and reported forested upland (first-order) streams had the greatest taxa richness (70), when compared with lowland rivers (20) and drains (15 taxa). Lowland rivers, such as the reaches of the East Collie and Bingham rivers sampled in the present study, were found to be characterised by cosmopolitan species with a dominance of collectors (Streamtec 1998). Upland fauna, associated with forested permanent streams, is generally considered to have higher conservation value than lowland reaches (Bunn 1985, 1986, 1988; Storey *et al.* 1990, 1991; Storey and Edward 1989; Streamtec 1998).

With respect to comparisons with other rural river systems in southwest Western Australia, species richness in the East Branch of the Collie River was comparable, whilst in the Bingham River it appeared to be high. Bunn & Davies (1992) recorded an average species richness of 35 in the intermittent stream Thirty-four Mile Brook during October, and an average richness of 37 in October and 20 in March for the perennial Hotham River. Unfortunately, no sampling was undertaken for this study during January, so there will undoubtedly be seasonal differences in the invertebrate assemblages when compared with the current study.

Macroinvertebrates of the Ludlow River (to the southwest of the study area) were sampled in November 2005 (WRM 2006b) and January 2006 (WRM 2006c). The average species richness in January was 44 (WRM 2006c), whilst in November it was 40 (WRM 2006b). Henty Brook, a tributary of the Collie River, was sampled by WRM in November 2005 (WRM 2006a). The average taxa richness reported was 42 (WRM 2006a). Invertebrate assemblages were similar to those recorded in the current study, with an abundance and diversity of Insecta, particularly Diptera.

In a study of streams from the Harvey River catchment to the north, Creagh *et al.* (2003) also reported a similar macroinvertebrate fauna to the East Branch of the Collie and Bingham Rivers. Like the current study, aquatic fauna in these systems was dominated by predators and collectors with a high proportion of Coleoptera and Diptera. Comparable taxa richness was reported from these degraded systems; Samson Brook recorded an average of 34 species per site, McKnoe Brook 37, Wokalup 45 and Wellesley 43 (Creagh *et al.* 2003). The number of species recorded from each river during the current study was also comparable to numbers recorded by Storey *et al.* (1991) and Storey (1999) from moderately disturbed middle order and lowland reaches on the North Dandalup and Canning River systems, respectively.

7.2.2.3 Conservation significance

The conservation status of fauna was based on levels of endemicity and rarity. Of the 109 taxa recorded during the current survey, 12% (13 taxa) were endemic to the southwest, including; the freshwater crayfish, *Cherax quinquecarinatus* and *Cherax cainii*; the amphipod *Austrochiltonia subtenuis*; the chironomids *Dicrotendipes ?conjunctus*, and *Paramerina levidensis*; the dragonflies *Austrogomphus collaris* and *Austrogomphus lateralis*; and, the dytiscids *Limbodessus inornatus* (formerly *Liodessus inornatus)*, *Megaporus solidus*, *Necterosoma darwini*, *Sternopriscus browni* and *Helochares tenuistriatus*. A further five taxa (6%) were indeterminate, but considered likely to be southwest endemics. These included the shrimp Palaeomonidae spp., the trichoptera *Ecnomus* sp., *Acritoptila/Hellyethira* spp. and *Oecetis* sp., and *Sternopriscus* sp. B (female specimen but likely to be *Sternopriscus browni* based on gross morphology and colour pattern). While the palaeomonid shrimp would previously have been identified as *Palaeomenetes australis*, Brenton Knott (UWA pers. comm.) suggests the taxonomy needs to be revised. The species common to the southwest of Australia is likely to belong to a new genus (Brenton Knott, UWA, pers. comm.). None of these species were considered rare within the southwest.

Plate 10. The introduced *Lymnaea stagnalis* (photo taken from http://members.tripod.com/arnobrosi/gallery.html#L)

Of the remaining taxa, 45% (49 taxa) were cosmopolitan in distribution, occurring across Australia and overseas, while 37% (40 taxa) were indeterminate due to insufficient information. Two introduced taxa (2%) were collected (Table 10) from the Bingham River. These were both species of freshwater snail, *Lymnaea stagnalis* (Plate 10) and *Physa acuta* (Plate 11). *L. stagnalis* is native to Europe and was likely introduced to Australia for the aquarium trade, however, they are also easily transported on living weed or in containers imported by aquarium suppliers of live fish (Smith 1996). This is not one of the Lymnaeid species responsible for the sheep liver fluke, *Fasciola hepatica*. Native to Europe, the Nearctic Region and the Neotropical region, *P. acuta* is thought to have been introduced by European settlers (Smith 1996). It appears to be actively spreading and is now found across Australia in many slow and non-flowing waterbodies (Smith 1996). This snail is a long-established occupant of wetlands and river systems within various parts of southwestern Australia. Although lacking an operculum, *Physa* are able to survive dry periods by sealing their shells with a mucus plug, known as an epiphragm.

Plate 11. The introduced *Physa acuta* (photo taken from www.applesnail.net/content/photos/photo_nonapple_1.htm)

All of the four reaches sampled in January 2006 supported taxa considered endemic to the southwest of the state, with the Bingham River supporting the greatest number of these taxa and the highly saline pools of the East Collie (above the proposed diversion) supporting the least (Table 10 and Figure 14). Introduced species were species of gastropod and were only collected from the Bingham River survey reach.

Figure 14. Proportions of species from each conservation category collected from each reach (E = restricted endemic, S = common south-west endemics, I* = introduced exotics, I = indeterminants, C = cosmopolitan).

Generally, taxa recorded were considered tolerant of a wide range of environmental conditions and are common, ubiquitous and frequently encountered in freshwater systems across Western Australia.

7.2.2.4 Functional feeding groups

The functional complexity and 'health' of a river system is manifested in the diversity of functional feeding groups[4]. Functional feeding groups were assigned to each species. Literature sourced included; Williams 1980; Barnes 1987; Bunn 1988; Boulton 1989; Cartwright 1997; Davis and Christidis 1997; St Clair 2000; Gooderham and Tsyrlin 2002; and the ARL database). The proportions of each functional feeding group from invertebrates collected from the East Branch of the Collie and the Bingham Rivers are presented in Figure 15. The region was found to have high proportions of predators and collectors, with very few shredders, grazers, or filterers. The assemblage structure with respect to functional feeding groups was quite similar between the Bingham River and the East Branch of the Collie; the Bingham recorded slightly less predators and a greater proportion of grazers than the Collie (Figure 15).

Upland rivers of the northern jarrah forest in Western Australia generally have high proportions of shredders owing to the greater abundance of riparian, overhanging and aquatic vegetation (Bunn 1986; Davies 1993; Streamtec 1998). However, the East Collie and to a lesser extent the Bingham Rivers have little coarse particulate matter for shredders to feed on since they are located within agricultural areas, and as such, have little overhanging riparian vegetation due to historical clearing. Perhaps the low proportion of shredders can also be attributed to the seasonality of the water bodies studied, restricting the presence of some shredder taxa which require perennial flows. The high proportion of collectors is perhaps to be expected since they are known to dominate disturbed river reaches where the input of fine particulate material is high (Nessimian 1997; Velasquez and Miserendino 2003). An increase in grazers would also be expected in streams where algal production is high due to nutrient enrichment and/or a more open vegetation canopy resulting in increased light and higher water temperatures.

[4] Functional feeding groups: 'shredders' feed on coarse particulate matter (CPOM > 1mm); 'collector's feed on fine particulate matter (FPOM < 1mm); 'filterers' filter suspended particles from the water column and are often viewed as a subset of collectors; 'grazers' are those animals that graze or scrape algae and diatoms attached to the substrate; 'predators' capture live prey.

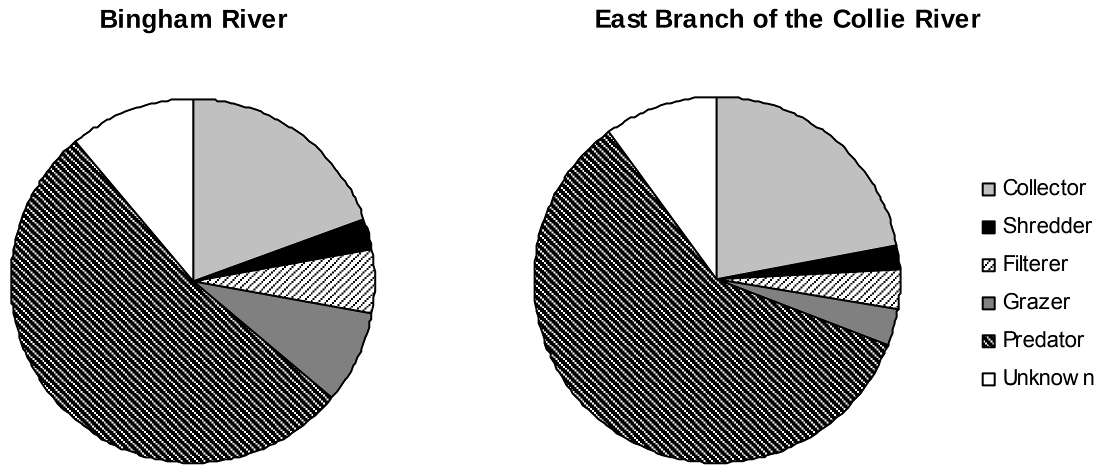


Figure 15. Proportion of functional feeding groups from the Bingham River and all reaches of the East Branch of the Collie.

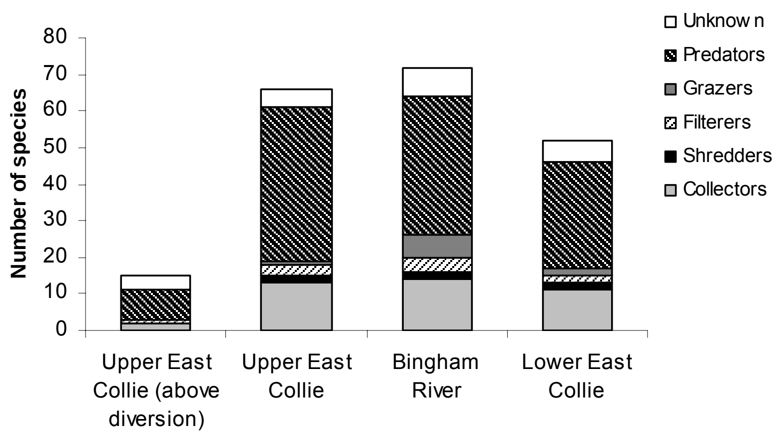


Figure 16. Proportion of functional feeding groups from individual reaches.

The relative proportion of each functional feeding group from individual reaches is presented in Figure 16. The highly saline pools of the Upper East Collie (above the proposed diversion) recorded a high abundance of predators, with very few collectors or filterers, and no shredders or grazers. This is a reflection of the low habitat diversity and lack of vegetation characteristic of pools from this reach. The functional organisation of the remaining reaches of the East Collie was comparable (Figure 16).

7.2.3 Freshwater crustaceans

Two Western Australian endemic freshwater crayfish were recorded during the current study, *Cherax quinquecarinatus* (gilgie; Plate 12) and *Cherax cainii* (marron; Plate 13). Gilgies have a range from the Moore River in the north to Bunbury in the south (Shipway 1951). They were collected from the Bingham River, the lower East Collie, and the upper East Collie (in the fresher waters below Chicken Creek). Gilgies 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). These crayfish have a well developed burrowing ability, digging short burrows under stones on the stream bed or in the banks along the margins (Shipway 1951). In this way, gilgies are able to withstand periods of low water level by retreating into burrows until flows return. A study on the microhabitat characteristics of marron, gilgies and the introduced yabby within the Canning River system near Perth, determined that gilgies are more commonly found in areas with higher flow velocity and dissolved oxygen concentrations than marron, *Cherax cainii* (Lynas *et al.* 2006).

Plate 12. Gilgie, *Cherax quinquecarinatus*. Photo taken by Jess Lynas (specimen collected from Elgin Drain, near Capel, November 2005).

Plate 13. Marron, *Cherax cainii*. Photo by Jess Lynas, January 2006 (collected from Yoganup, near Capel).

Little is known of their physiological tolerances. However, Shipway (1951) suggests that they are able to tolerate more extreme environmental conditions than marron, and may survive longer periods out of water. Beatty *et al.* (2005) suggested the reproductive biology of *C. quinquecarinatus* may help explain their apparent success throughout the range of aquatic habitats found in southwest Western Australia. That is, they show traits of both r- and K-strategists (Beatty *et al.* 2005). Beatty *et al.* (2005) studied the population biology of gilgies from a permanent urban stream, Bull Creek, in the southwest of the State. Populations within this waterway have been under considerable pressure from overfishing. Traits found to be associated with r-strategists were early maturation (they breed at the end of their second year), an extended late winter-summer spawning period (with multiple spawning events), and high mortality rates (Beatty *et al.* 2005). Qualities typical of K-strategists included relatively slow growth rates, low fecundity, and moderately sized eggs (Beatty *et al.* 2005). They are small crayfish, reaching a total body length of 80 mm (Riek 1967).

Marron were collected from the Bingham River and the lower East Collie. They have high conservation significance and are considered a flagship species[5] (Nicholl and Horwitz 2000). Flagship species are important ecologically because not only are they able to respond to restoration activities and/or recover from endangerment but they also hold significant charismatic appeal. Generally, flagship species are used to enhance public understanding of environmental issues. In addition, the decline of marron throughout much of its range has brought about sympathy towards its plight (Nicholl and Horwitz 2000). Therefore, the elimination of marron from any area is considered to have significant conservation implications (Nicholl and Horwitz 2000).

Marron are a riverine crayfish. They are characteristically "K-selected species" and inhabit the deeper and broader water of permanent river systems (Riek 1967). They typically belong to large, stable populations which inhabit low nutrient waters. Marron are highly sensitive to environmental fluctuations (Morrissy 1983; Morrissy *et al.* 1984; Holdich and Lowery 1988). They have a long annual period of ovarian development with a single springtime breeding season. There is a tendency for breeding failure in highly eutrophic waters (Morrissy 1983).

[5] a priority conservation grouping similar to keystone, umbrella and indicator species or groups.

Unlike gilgies, the burrowing habit in marron is not strongly developed. Riek (1967) suggested this is a consequence of the relatively poor development of chelae muscles in marron, thus restricting their ability to construct burrows. Shipway (1951) noted marron were more content to seek shelter under logs or stones in the bed of streams rather than to burrow. It is generally accepted that marron do not burrow to escape drought (Maguire *et al.* 1999; Lawrence and Jones 2002). However, Hutchings (1987) states that while marron do not normally burrow, they will dig shallow excavations when under stress (e.g. from over crowding).

7.2.4 Tortoises

Long-necked tortoises (Plate 14), also known as the narrow-breasted snake-neck tortoise (*Chelodina oblonga*), were recorded during the January 2006 survey, from the Bingham River survey reach and the Upper East Collie (downstream of Chicken Creek and the proposed diversion). These tortoises are restricted to the south-west of Western Australia, with a distribution extending from the Hill River in the north to the Fitzgerald River National Park in the south-west.

Plate 14. Long-necked tortoise (*Chelodina oblonga*) caught from pool 5 (upstream of the bridge) in the Bingham River in January 2006 (photo by Andrew Storey).

7.2.5 Frogs

No tadpoles were caught in macroinvertebrate sweeps but an adult frog, *Crinia insignifera*, was recorded from the Bingham River (Plate 15). Since frogs play an important role in functional ecosystems and have a requirement for water during their life cycle, it was thought relevant to consider their ecological water requirements. Frogs spend much of their lives in moist environments, such as marshes, swamps and along the riparian zone of rivers, due to their permeable skin which makes them susceptible to desiccation. In addition, frogs need water during certain stages of their life cycle in which to lay eggs and for tadpoles to survive and metamorphose.

Plate 15. *Crinia insignifera* collected from the Bingham River survey reach. (Photo by Andrew Storey, January 2006).

The Sign-bearing or squelching froglet, *Crinia insignifera*, is found in coastal plain habitats near temporary swamps and marshes between Gingin and Busselton. This species is not easily distinguishable from the false western froglet *C. pseudinsignifera*, by either call or identification of tadpoles. The western most extent of distribution for *C. pseudinsignifera* is the Darling Scarp, with this frog often being found in temporary swamps and pools in more arid areas in the eastern part of its range. Where their ranges meet along the Darling escarpment, *C. insignifera* and *C. pseudinsignifera* hybridise. Both species are similar in ecology and are common to the south-west of Western Australia. Female squelching froglets lay between 66 and 268 eggs in small clumps in shallow water during winter (Tyler *et al.* 2000). The tadpoles take approximately three to five months to develop.

A number of other frog species are known from the region. A study of the nearby Kemerton region (to the west of the Collie River and ~3 km east of the Leschenault Inlet) identified a number of frog species, including Glauert's froglet *Crinia glauerti*, squelching froglet *Crinia insignifera*, Lea's frog *Geocrinia leai*, Gunther's toadlet *Pseudophryne guentheri*, the slender tree frog *Litoria adelaidensis*, and the western green tree frog *Litoria moorei* (Bamford and Watkins 1983).

7.2.6 Fish fauna

7.2.6.1 Taxa richness

Two introduced species (the mosquitofish *Gambusia holbrooki* and redfin perch *Perca fluviatilis*) and four native species were recorded during the January 2006 survey, including the night fish *Bostockia porosa*, the western minnow *Galaxias occidentalis*, the western pygmy perch *Edelia vittata*, and the freshwater cobbler *Tandanus bostocki* (Table 11). A total of eight native species have been previously recorded from the Collie River system (Allen *et al.* 2002, Morgan *et al.* 1995, Morgan *et al.* 1998, WEC & Streamtec 2001, WRM 2003a), but only seven of these are likely to inhabit the East Branch of the Collie River (Table 12). This conflicts with both WEC & Streamtec (2000) and WEC & Streamtec (2001) which state that only seven species are known from the area (including one species, *Geotria australis*, which was omitted from our list because it is not considered to currently occur in the Collie River).

Table 11. Fish species recorded from each sampling reach during field surveys.

Scientific name	Common name	Reach Lower East Collie	Upper East Collie	Upper East Collie Above Diversion	Bingham River
Bostockia porosa	Nightfish	Present	Common		
Galaxias occidentalis	Western minnow	Common	Common	Common*	Common
Edelia vittata	Western pygmy perch		Common		Present
Tandanus bostocki	Cobbler				Present
Gambusia holbrooki	Mosquitofish	Abundant	Present	Abundant	Abundant
Perca fluviatilis	Redfin perch	Present			Present

* Minnows in this reach were clearly affected by the high saline conditions, they all appeared skinny and unhealthy compared with populations downstream of Chicken Creek.

Table 12. Summary of native fish species known to occur in the Collie River system.

Scientific name	Common name	Previously recorded from:	Previously recorded by:
Bostockia porosa	Nightfish	• Collie River (Collieburn Pool, Cox's Pool, Western Collieries, & Davies' Pool) • Lower Collie River (between Wellington Dam and Burekup Weir) • Lower Collie River (below Burekup Weir, downstream to Australind bypass)	• Morgan *et al*. 1998 • WRM 2003a • WRM 2003a
Galaxias occidentalis	Western minnow	• Collie River (Collieburn Pool, Cox's Pool, Round Pool, Western Collieries, & Davies' Pool) • Lower Collie River (in a pool against the base of Burekup Weir)	• Morgan *et al.* 1998 • WRM 2003a
Edelia vittata	Western pygmy perch	• Collie River (Cox's Pool, Round Pool, Western Collieries, & Davies' Pool) • Lower Collie River (below Burekup Weir, downstream to Australind bypass)	• Morgan *et al.* 1998 • WRM 2003a
Tandanus bostocki	Cobbler	• Lower Collie River (below Burekup Weir, downstream to Australind bypass)	• WRM 2003a
Afurcagobius suppositus	Big headed goby	• Lower riverine/estuarine parts of the Collie River system	• Morgan unpub. data
Pseudogobius olorum	Swan River goby	• Lower Collie River (between Wellington Dam and Burekup Weir) • Lower Collie River (below Burekup Weir, downstream to Australind bypass)	• WRM 2003a • WRM 2003a
Leptatherina wallacei	Western Hardyhead	• Lower riverine/estuarine parts of the Collie River system	• Morgan unpub. data

WEC & Streamtec (2000) recorded five species whilst sampling as part of the Ecological Water Requirements study for the Harris River and East Branch of the Collie River. However, the report does not detail which species were collected during surveys, and which species were simply included because they are known from previous studies of the river system (i.e. Morgan *et al.* 1998, Allen *et al.* 2002).

Morgan *et al.* (1998) sampled a section of the Collie River upstream of Wellington Reservoir. The survey was part of a larger study of south-western fish conducted between 1994 and 1996 and incorporated museum records. No additional species were recorded during these surveys, however, museum records indicate that adult pouched lamprey (*Geotria australis*) were present from the lower Collie River in 1916, but this species has not been recorded since (Morgan *et al.* 1998). Recently, they have been collected from the Brunswick River, and amnocoetes are known from the Leschenault Estuary (Morgan *et al.* 1998).

The pouched lamprey belongs to an ancient lineage of jawless fishes whose morphology has remained largely unchanged for approximately 280 million years. *Geotria australis* is the only surviving species in Australia, though six genera with 41 species are known to occur elsewhere (Allen *et al.* 2002). Although most abundant in river systems south of Margaret River, museum records indicate they have been found as far north as the Swan River (Morgan *et al.* 1998). Habitat alteration (including the construction of dams, extraction of groundwater and agricultural practices) and salinisation are believed to have lead to their loss from many areas. In particular, agriculture in the southwest has reduced the abundance of suitable amnocoete beds due to increased run-off adversely affecting the composition of the substrate. Pouched lamprey amnocoetes require soft substrate beds within which to burrow. Amnocoetes spend 4-5 years in freshwaters, before metamorphosing and migrating to the sea. Adults remain in the open ocean for at least two years before returning to the rivers to spawn. Spawning is believed to take place in November. Given the intensity of agriculture in the area, coupled with the high salinity, it is unlikely that pouched lampreys would use the East Branch of the Collie River as spawning habitat.

Pen & Potter (1990, 1991a, b, c & d) surveyed the southern branch of the Collie, above Wellington Dam, and recorded western minnow, pygmy perch, nightfish, redfin perch and mosquitofish. Furthermore, Morgan (unpublished data) sampled tributaries of the Brunswick River, within the Collie River catchment, and collected native species including the western minnow, pygmy perch, and nightfish, and introduced brown trout, mosquitofish and redfin perch. The western hardyhead (*Leptatherina wallacei*) and big headed goby (*Afurcagobius suppositus*) are also known from the lower riverine/estuarine parts of the Collie River system (Morgan unpub. data)

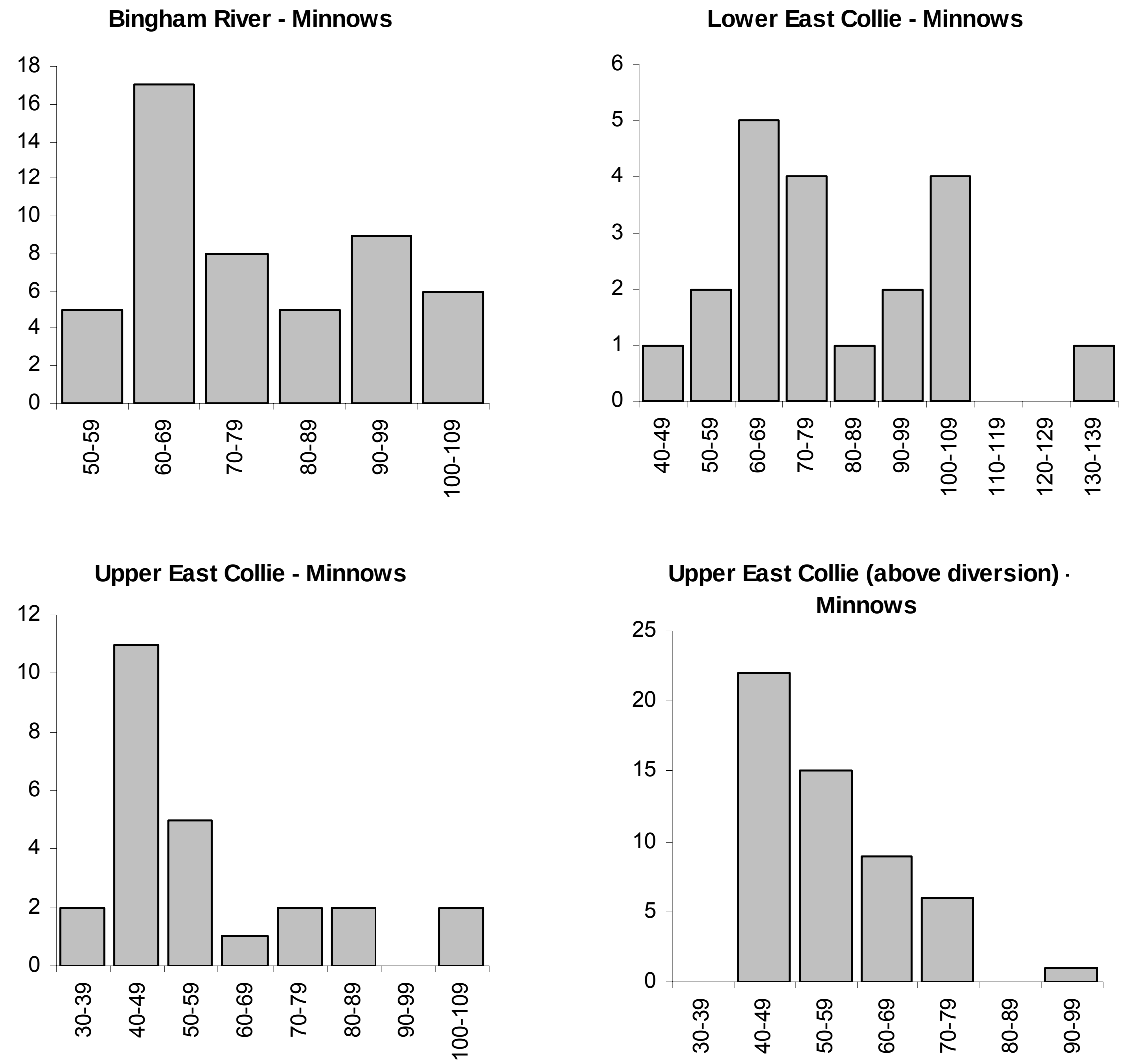
In 2003, WRM sampled the lower Collie Ecological Water Requirement studies already undertaken (see Streamtec 1999 & WEC 2002). A total of five native freshwater species and three introduced species were collected (WRM 2003a). In addition to those native species recorded from the current study, WRM (2003a) also collected the swan river goby (*Pseudogobius olorum*). Introduced species collected were mosquitofish, rainbow trout (*Onycohynchus mykiss*) and redfin perch (*Perca fluviatilis*).

Flow requirements of fish species known to occur in the East Branch of the Collie and Bingham Rivers are discussed in Section 7.

7.2.6.2 Length-frequency analysis

Length frequency data were collected for dominant fish (Figure 17).

Of interest was the fact that minnows collected from the highly saline pools above the proposed diversion site were obviously affected by the poor conditions. In comparison to minnows collected from more downstream reaches, they were skinny and appeared in poor health generally. Stress associated with sampling, handling and measuring was enough to kill the majority of fish collected from the single pool in this reach that contained fish, whereas fish collected and handled in an identical manner at other sites were not adversely affected by handling.



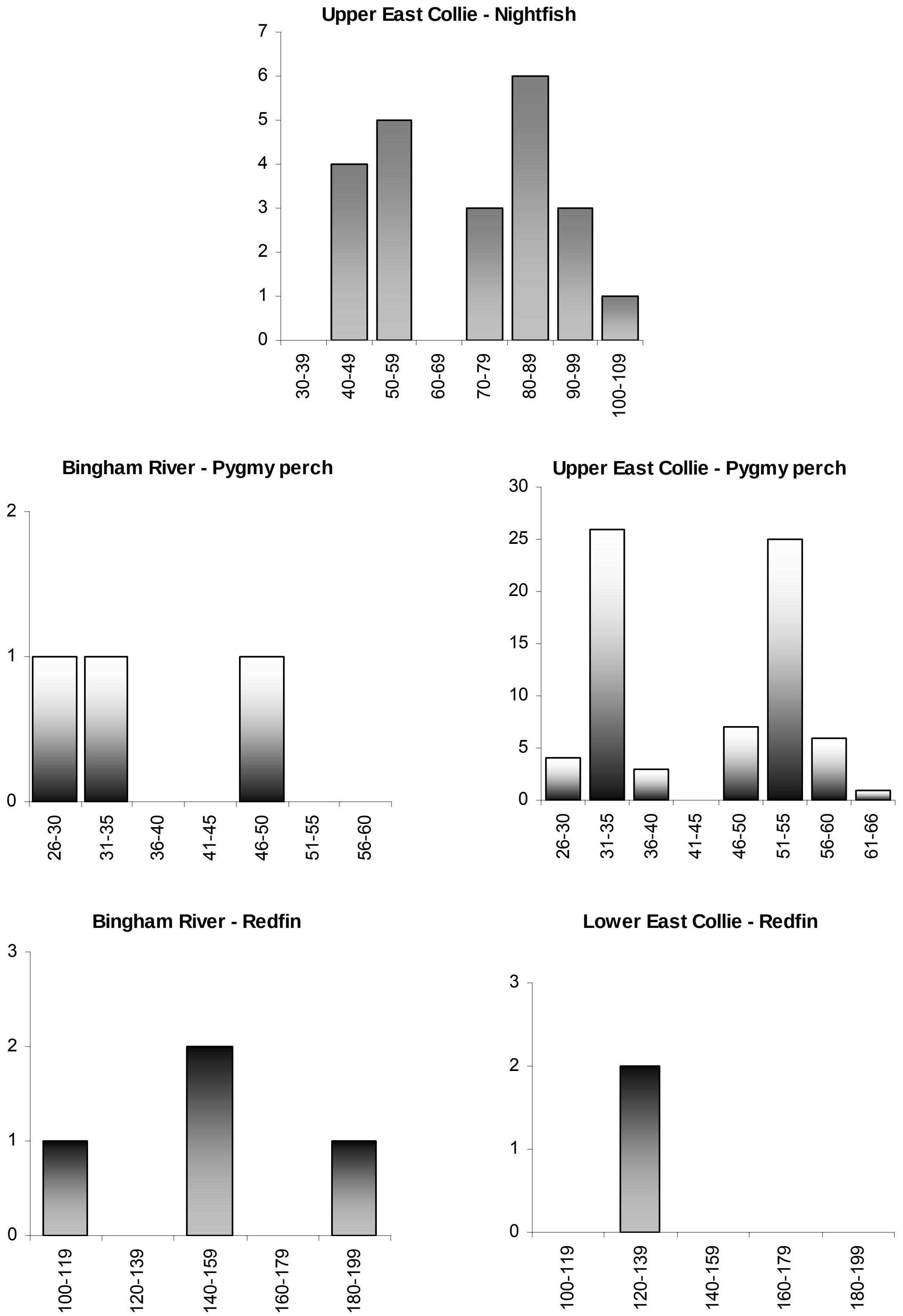


Figure 17. Length-frequency histograms for fish caught during January 2006 surveys.

7.2.6.3 Conservation significance

The southwest of Western Australia has a depauperate indigenous freshwater fish fauna with a high degree of endemism compared with the rest of the continent (Pusey *et al.* 1989). Native species recorded from the East Collie and Bingham Rivers are regional endemics, but are common throughout south-western Australia. Of these, western minnows, pygmy perch and nightfish are the most abundant and widespread. There is anecdotal evidence that the distribution of both lampreys and cobbler is becoming increasingly restricted in the south-west due to habitat loss and flow regulation.

7.2.6.4 Reproduction and life history characteristics of fish species

The field survey of the East Branch of the Collie River and the Bingham River recorded four native and two introduced fish species. A further two native species are confirmed to occur in the Collie River system. For the purposes of determining environmental flows, all six native species are presumed to be present.

In order to understand how changes in flow regime may affect individual species, it is necessary to understand the life history strategies and breeding times of all fish species. Some species have been extensively studied, while there is little information on other species. Life history requirements and ecology were reviewed using a range of sources including; Shipway (1949), Christensen (1982), Merrick & Schmida (1984), EPA (1987), Morrison (1988), Allen (1989), Pusey & Morrison (1989), Pusey *et al.* (1989), ARL (1990a), Pen & Potter (1990), Pusey & Edward (1990), Hutchinson (1991), Pen & Potter (1991a, b, c & d), Hewitt (1992), Hutchinson, (1992), Gill & Potter (1993), Pen *et al.* (1993), Sarti (1994), Gill & Humphries (1995), Watts *et al.* (1995), Pusey & Bradshaw (1996), Morgan *et al.* (1998), Thorburn (1999), Morgan & Gill (2000), Allen *et al.* (2002), Morgan *et al.* (in press), WRC (2002). Although there is likely to be some variation in breeding season and development in different river systems, the overall flow requirements will remain the same.

Western minnow (*Galaxias occidentalis* Ogilby, 1900 (Galaxiidae))

The western minnow (Plate 16) is one of the most widely distributed endemic species in south-western Australia, with a range extending from the Arrowsmith River 300 km north of Perth, to Waychinnicup Creek, 80 km east of Albany (Allen *et al.* 2002). Western minnows are known to inhabit rivers, streams, lakes, and pools within this area. They readily invade seasonal creeks and swamps connected to permanent water. Western minnow have been commonly reported from the base of waterfalls (and V-notch gauging weirs) where the water is fast flowing and well oxygenated. While this may indicate a preference for these conditions it is more likely that the physical barrier prevents continued upstream movement. Western minnows are found in both open water and enclosed areas amongst riparian vegetation. Terrestrial insects form a major component of their diet, although dipteran larvae and pupae, and microcrustacea (cladocera & copepods) are also consumed. Recent work suggests that the western minnow feeds at night on freshwater shrimp (Fairhurst, unpub. data., cited *Morgan et al.* 1998).

Plate 16. Western minnow, *Galaxias occidentalis* (photo taken by Glenn Shiell 2004).

Pen & Potter (1991a) studied the reproductive biology of western minnow in the Collie River and stated that one year old males and females are approximately 70 and 75 mm, respectively. After two years males and females attain a length of 90 and 100 mm, respectively (Pen & Potter 1991a). Fish move to small tributaries and feeder streams of wetlands and rivers to spawn amongst flooded vegetation (Allen *et al.* 2002). Spawning is induced by the onset of winter rains (between June and late September) and sexual maturity is attained the following autumn (Allen *et al.* 2002). Some fish survive to spawn in their second year, and a very limited number into a third, fourth and even a fifth year.

In a study of the genetic structure of western minnow in the Canning and North Dandalup River systems, Watts *et al.* (1995) reported populations on the scarp and coastal plain were separate and non-mixing. The basis for differences in genetic structure between populations was thought to be due to isolation of breeding; i.e. scarp populations breeding in tributary creeks on the scarp, whilst coastal plain populations were thought to move into drains and wetlands on the coastal plain (Watts *et al.* 1995).

Thorburn (1999) investigated habitat preferences of native species in the Blackwood River, assessing preferences of species for mesohabitats (i.e. pools, canalised sections, channel habitat and riffled sections) as well as flows, depths and substrate types. The western minnow was found during summer in the highest density in pools (39/10 m^2) (N.B. summer pools were the most abundant feature), and to a lesser extent channel habitat (15/10 m^2), while in winter, it was generally lower in density overall, recorded within pools (1/m^2), riffles (12/10 m^2) and channel habitat (3/m^2) (Thorburn 1999). On average, western minnow was encountered in waters flowing at approximately 0.25 m/s (and up to 2 m/s), and at depths of between 0.6 and 0.7 m, and as deep as 1.6 m. According to Thorburn (1999), the highest density of western minnow in the Blackwood River was encountered in flooded grass. Western minnow did not appear to be associated with any substrate type. Although a majority of individuals were encountered on sand or bedrock, all densities were comparatively low (with bedrock being the substrate on which western minnow was marginally higher than any other) (Thorburn 1999).

Reflecting the need to migrate, disperse and reach spawning habitat, the western minnow has excellent swimming abilities and are able to negotiate low gradient rapids. Morgan (pers. comm.) notes that 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. Storey (pers. obs.) has watched fish jumping through V-notch weirs and 'crawling' up wet rock faces in an attempt to traverse barriers (ARL 1990a). The behaviour of western minnow below a series of different shaped gauging structures was documented, and the ability of fish to traverse the obstacles assessed (ARL 1990a). Where there was a downstream pool, individuals were seen jumping from the pool into the plume from the v-notch and successfully traversing some types of weir. This was observed on 31 Mile Creek (616026) (broad-crested concrete weir 35 cm high with a 25 cm deep pool below), and North Dandalup River (616024) (sharp-crested metal v-notch weir 28cm high flowing into a pool; depth not recorded). Of the fish found in the northern jarrah forest streams, western minnow is probably the strongest swimmer and therefore the least likely to be affected by barriers. However, this species is also the most likely to cover large distances, the others being predominantly territorial (EPA 1987). Of the species found in the East Branch of the Collie, the western minnow, is likely to be the most mobile.

Nightfish (*Bostockia porosa* Castelnau, 1873 (Percicthyidae))

The nightfish (Plate 17) has a distribution from Albany in the east to the Moore River in the north (Allen *et al.* 2002), although populations have recently been found in the Hill River north of this known range (WRM 2005). Within its range, the nightfish occurs in rivers, streams, lakes and pools. It is a solitary, bottom dwelling fish, and, as its name suggests, is more active during the night than during the day. It commonly occurs under ledges, rocks, logs, amongst root mats and inundated vegetation. In the Collie River (Pen & Potter 1990), the majority of fish were one and two years old, although representatives of three and four year olds, and one fish in its sixth year were found. The lengths attained at the end of the 1st, 2nd and 3rd years of life were, respectively, 56 mm, 76 mm and 85 mm for males and 56 mm, 79 mm and 91 mm for females. Most males (61.4%) reach sexual maturity by the end of the 1st year of life, with females becoming sexually mature at the end of the 2nd year. The great majority of fish migrated into tributary creeks when these started to flow in the winter. Spawning occurred in the creeks during late August and early September, when freshwater discharge was still high and temperatures and day length had just begun to increase. Females produce approximately 600 eggs. The nightfish is carnivorous, feeding on a wide range of benthic organisms, particularly ostracods and dipteran larvae, progressing to odonate (dragonfly) larvae, decapods (gilgies, prawns etc) and gastropods (snails) as the fish increases in size.

Plate 17. Nightfish, *Bostockia porosa* (photo taken by Glenn Shiell 2004).

In the Blackwood River, Thorburn (1999) reported that the nightfish was most commonly caught in waters with an average salinity of between 500 - 650 mg/L. This species was additionally shown to be present almost exclusively in tributary sites, and occurred at lower salinities of between 0.5 and 500 mg/L. In the Blackwood, Thorburn (1999) reported that nightfish were commonly encountered at an average depth of 0.6 m, but were found in waters up to 1.6 m deep. Nightfish generally preferred low flows (mean flow rate of 0.29 m/s, and up to 2 m/s) (Thorburn 1999).

Thorburn (1999) noted that like the western pygmy perch, nightfish were associated with structure (or cover), with no individuals being found in open water. The majority of nightfish were found in association with the more complex cover types, in particular, snags and within the bank, while overall, this species was found in highest densities in association with the banks of tributary sites. Nightfish do not show as strong an association for any particular substrate type, but highest densities were recorded from finer substrate types, especially clay and mud, with a majority of fishes caught on mud and sand substrates (Thorburn 1999).

Unfortunately there is little empirical or anecdotal evidence as to the dispersal capabilities of the nightfish. This species appears not to leave the water to negotiate rapids, however, as for western pygmy perch, Morgan (pers. obs.) has observed individuals negotiating waters of only ~1 cm depth with velocities of approximately 2 - 4 m/s.

Western pygymy perch (*Edelia vitatta* Castelnau, 1873 (Nannopercidae))

The western pygmy perch (Plate 18) is also widely distributed in south-west Western Australia, with a range from the Arrowsmith River (300 km north of Perth) to Phillips River (near Hopetoun, east of Albany) (Allen *et al.* 2002). This species is common in rivers, streams, and lakes, and readily re-invades seasonal wetlands via flood-ways and up seasonal creeks/drains. Pygmy perch are often associated with riparian/emergent vegetation and rarely occur in open water. Thorburn (1999) recorded pygmy perch in comparatively high densities in several habitat types in the Blackwood River, especially, snags. In the Blackwood, western pygmy perch were collected in comparable numbers from clay, mud, sand and rock, however conversion into densities resulted in the number of fish per m^2 declining as the substrate become more consolidated, with fish tending towards finer particulate substrates (Thorburn 1999).

Plate 18. Western pygmy perch, *Edelia vittata*, captured from the Lower East Collie (photo taken by Andrew Storey, January 2006).

Western pygmy perch feed on a variety of small benthic invertebrates, particularly dipteran larvae, ostracods, copepods and trichoptera (caddis fly) larvae, and to a lesser extent, terrestrial insects.

Fish move into adjacent flood waters or tributaries in winter to breed. Pen & Potter (1991c) recorded changes in the density of pygmy perch in the main channel of the south branch of the Collie River from a high in January - February to lows over May - December, with a corresponding increase in density in adjacent floodwaters along the banks and in tributary creeks over winter and spring. This species showed a greater preference for the lateral flooded margins rather than the tributary creeks, as opposed to western minnow and nightfish which preferred the tributaries (Penn & Potter, 1991c). Minnows and nightfish have a shorter spawning period, which occurs earlier in the year when tributaries of most rivers would be flowing, however, the western pygmy perch spawns later and often after tributaries have stopped flowing, therefore, use of flooded margins of the main channel prevents spawning failure. Spawning in the Collie is known to take place from late winter to late spring (July-November), and females may spawn more than once in a breeding season (Pen & Potter, 1991c). Shipway (1949) suggested individuals spawn from July up until January in the Canning River, if the food supply was plentiful. Shipway (op cit.) recorded 40 – 50 eggs from ripe females, with number per ovary ranging from 7 – 25. Allen (1989) noted that females lay batches of 20 – 60 eggs at 6 – 8 week intervals, and the adhesive eggs sink and attach to the bottom or to flooded vegetation. The eggs hatch and the larval stage lasts approximately 2 – 3 weeks. Sexual maturity is attained by both sexes at the end of the first year of life. Shipway (1949) reported that the species attains a length of 35 – 46 mm after one year, 54 – 58 mm after two years, 60 – 65 mm after three years, and 70 mm after 4 – 5 years, with a maximum life span of five years. Pen & Potter (1991c) reported that fish reach ~42 mm after one year, 51 mm after two years, 56 mm after three years and 68 mm after four to five years and that the majority of fish live to three years, although fish up to 6 years old have been recorded.

The western pygmy perch is not as strong a swimmer as the western minnow, however, it appears to demonstrate a strong capacity for dispersal. In the Muir/Unicup wetland systems, Storey (unpub dat.) observed pygmy perch moving through shallow water (< 5 cm) up to 1 km away from the nearest wetland. Although the western pygmy perch does not leave the water to negotiate rapids by 'crawling' across rocks and wet surfaces, Morgan (unpub. dat.) has observed individuals negotiating waters of only ~1cm depth with velocities of approximately 2 - 4 m/sec.

Freshwater cobbler (*Tandanus bostocki* Whitley, 1944 (Plotosidae))

Freshwater cobbler (Plate 19) belong to the family Plotosidae, and are commonly referred to as 'eel-tailed catfish'. There are at least 18 freshwater species of Plotosidae, from five genera in Australia (Merrick & Schmida 1984), but only one freshwater catfish, the freshwater cobbler (*T. bostocki*) is found in southwestern Australia. Cobbler are locally common but have a scattered distribution occurring in freshwater streams between the Franklin and Moore Rivers (Allen *et al.* 2002). They have a preference for slow flowing streams, pools, ponds and reservoirs. There is evidence that the distribution of cobbler is becoming increasingly restricted due to habitat loss (vegetation clearing, de-snagging, drain construction), though relatively large populations are believed to exist in several south-west reservoirs (Morgan *et al.* 1998). Diet consists of insect larvae, freshwater shrimp, crayfish, molluscs and small fish.

Cobbler reach a maximum size of approximately 40 cm, and are by far the largest native freshwater species in southwestern WA. They are also the only endemic species targeted by recreational anglers (Morgan *et al.* 2000). Morrison (1988) studied the reproductive biology of freshwater cobbler in detail, Hewitt (1992) studied its biology, and Hutchinson (1992) reported on its spawning sites in the Murray River. Catfish are believed to live for at least nine years. Morrison (1988) noted that fish become sexually mature at around five years of age and weight of at least 500 g. Large cobbler inhabit deeper pools in the main channel of rivers and spawn during summer (Hutchison 1992, Hewitt 1992). Morrison (1988) observed spawning between November - January and females produced approximately 5,000 eggs. The reproductive biology is similar to that of *T. tandanus* (Allen *et al.* 2002), in which the male constructs an oval or circular nest of 0.6 to 2.0 m in diameter. The nest is made of gravel and rocks with a sandy central depression into which the female deposits her eggs. The eggs hatch in about seven days at 19-25 ^{o}C.

Plate 19. Catfish, *Tandanus bostocki*, collected from the Bingham River (photo by Andrew Storey, January 2006).

Thorburn (1999) did not assess habitat preferences of the freshwater cobbler in the Blackwood River however Allen *et al.* (2000) notes they are often found in association with rock, gravel or sand substrates and use in-stream vegetation, LWD, root mounds and bank undercuts for shelter. There are no empirical or anecdotal data as to the swimming ability of this species, or to the migratory behaviour. Based on the general biology of eel-tailed catfish it is assumed that the freshwater cobbler has relatively poor swimming ability and would tend to be relatively sedentary. However, they can display fairly rapid bursts of speed over a few metres (D. Morgan, Murdoch University, pers. com.) and based on size-relationships (10 x body length), they are probably quicker than all the other native species.

Swan-river goby (*Pseudogobius olorum* Sauvage 1880 (Gobiidae))

Although the swan river goby was not collected during the current study, it is known from the lower Collie (WRM 2003a). The swan river goby is common and widely distributed in coastal areas of southern Australia from the Murchison River in the north to Esperance in the southeast. It occurs in estuaries, rivers, and both freshwater and hypersaline lakes (Morgan *et al.* 1998). While the swan river goby is generally associated with coastal water bodies, it does penetrate long distances inland (i.e. Blackwood, Warren, Hay and Kalgan river systems), and occurs in some isolated lakes (e.g. lakes Jasper, Maringup, Towerrinning, Saide, Powell, Moates, Gardner and Angove). Owing to its recent marine origins, the swan river goby is known to tolerate high salinities. It is usually found over mud bottom, and sometimes amongst weeds or adjacent to rocky areas. Diet of this freshwater fish includes algae, fungal mats, bacteria, and invertebrates.

This species spawns in the upper reaches of estuaries where the salinity is typically less than 30% (Allen *et al.* 2002). Spawning occurs during spring and autumn in areas of abundant aquatic vegetation (Allen *et al.* 2002). The life cycle takes less than one year to complete, with young of spring spawning reproducing the following autumn, and *vice versa* for offspring of autumn spawning events. This suggests swan river goby generally reach maturity after only five months. Eggs are laid underneath rocks or logs with males then guarding and fanning them for approximately four days whilst incubation takes place. Females can lay up to 150 eggs. Once hatched, the planktonic larvae are swept downstream into estuaries, from where the juveniles migrate back into the rivers. Some populations are landlocked. This would apply to populations in the Collie River between Wellington Dam and Burekup Weir, because individuals swept downstream would not be able to come back upstream over the Weir.

There are no empirical or anecdotal data as to the swimming ability of this species, or to its migratory behaviour. It is assumed that it has relatively poor swimming ability, and would tend to be relatively sedentary, however, this is purely conjecture based on the general biology of gobies.

Big-headed goby (*Afurcagobius suppositus* Sauvage 1880 (Gobiidae))

Although not recorded as part of the current study, the big headed goby is known from the Collie River system. The Big headed goby has a wide distribution from the Moore River, in the north, to Esperance, in the east. It occurs in estuaries, rivers, streams and coastal lakes and can also penetrate inland waters (i.e. Warren, Scott and Blackwood Rivers (Morgan *et al.* 1998). The big headed goby has a strong preference for heavy cover (Gill & Humphries, 1995). It consumes a wide range of invertebrates, including hemipterans, diptera larvae, bivalves, terrestrial insects, ephemeroptera (mayflies), and trichoptera (caddis flies), and small fish (Young, 1994; cited Morgan *et al.*, 1996). Little is known of the breeding biology of this species, however, Morgan *et al.* (1996) suggest that the life cycle probably lasts two years, with breeding after one year. Males are known to guard the nest which is usually laid under stones or amongst aquatic macrophytes. Breeding likely occurs between late spring and early summer.

As for the Swan River goby, there are no empirical or anecdotal data as to the swimming ability of this species, or to its migratory behaviour. It is assumed that it has relatively poor swimming ability, and would tend to be relatively sedentary, however, this is purely conjecture based on the general biology of gobies.

Mosquitofish (*Gambusia holbrooki* Girard 1859 (Poeciliilinae))

Mosquitofish (Plate 20) were introduced to fresh waters around Perth in 1936 (Mees 1977) in an attempt control mosquitoes. 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). Introduced species can adversely impact native fish fauna through interference competition, increased predation or the introduction of disease. In fact, mosquitofish have been implicated in the decline of several small native species in Australian waters (Myers 1975; Arthington and Lloyd 1989). Modifications to the flow regimes may have significant implications for the dynamics and management of *Gambusia holbrooki* populations.

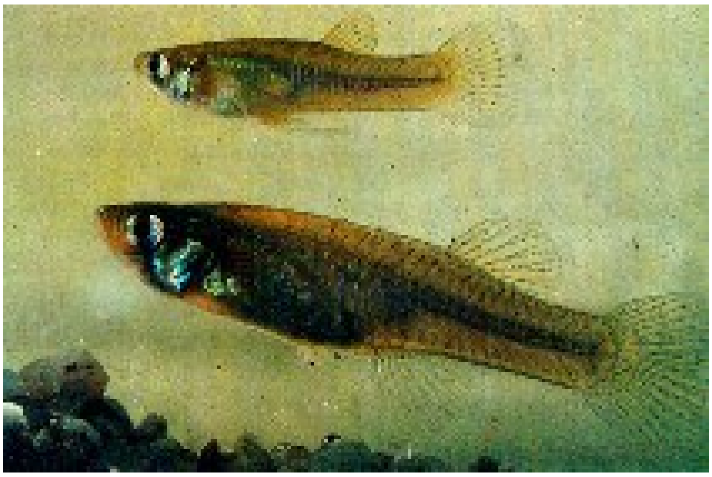


Plate 20. *Gambusia* (photo taken from www.fishbase.org).

Redfin perch (*Perca fluviatilis Linnaeus* 1758 (Percidae))

The introduced redfin perch, *Perca fluviatilis* (Plate 21), was collected during the current study from the lower East Collie and Bingham Rivers. They are native to Eurasia but are now widespread in cooler waters of New South Wales, South Australia, and southwestern Australia (Allen *et al.* 2002). Redfin tend to inhabit slow-flowing waters where there is abundant vegetation. Spawning occurs in spring, although it is also thought to occur in winter in the Collie River, with the females depositing thousands of eggs among vegetation (Allen *et al.* 2002). The egg mass is protected from predation by other fish as it is unpalatable. Eggs hatch between approximately one and three weeks, with the young forming schools before taking up a solitary lifestyle (Allen *et al.* 2002).

Plate 21. The introduced redfin perch, *Perca fluviatilis*, collected from the lower East Collie (photo by Andrew Storey, January 2006).

Redfin perch are aggressive piscivores and can cause the decline of native fish populations through competition and predation. They also predate on invertebrates, such as crustaceans (including freshwater crayfish) and molluscs (Allen *et al.* 2002). Redfin perch are thought to be responsible for the loss of native teleosts from Big Brook and Big Brook Dam (Morgan *et al.* 2002). Allen *et al.* (2002) also suggest they adversely affect pygmy perch populations. If caught, no introduced fish (redfin perch, mosquitofish or rainbow trout) should be returned live to the water. The release of redfin perch into natural waterways should be discouraged.

7.2.7 Water rats

Although not specifically surveyed, water rat feeding platforms were observed along the East Branch at Coolangatta. Water rats (Plate 22) are dependent on aquatic systems, and are known to suffer from heat stress if access to permanent water is lost (Watts and Aslin 1981). They are adapted to an aquatic life and have distinctive broad partially-webbed hind-feet, water-repellent fur, and a thick tail. Water rats are common around coastal Australia and New Guinea, occurring in a wide range of coastal, brackish and freshwater environments (Watts & Aslin 1981). However, the isolated population in southwestern Australia has suffered a substantial decline due to a loss of habitat through salinisation and clearance of riparian vegetation (Lee 1995). They are classified by CALM as a Priority 4 species, indicating they are in need of monitoring (CALM 2005). Within their known range, water rats can be found in rivers, swamps, lakes and drainage channels (Flannery 1995) where they build nests into banks near tree roots or in hollow logs. Therefore, there is a requirement for stable banks, tree roots and large woody debris.

Plate 22. A water rat, *Hydromys chrysogaster* (photo by Bert and Bab Wells).

Water rats are largely carnivorous, feeding on crayfish, mussels, fish, plants, water beetles, water bugs, dragonfly nymphs and smaller mammals and birds. Plants are more commonly consumed in winter or during periods of limited resources (Woollard *et al.* 1978; Harris 1978). Given the predominance of aquatic prey items, their feeding is closely linked with the river system, where they typically forage along the shoreline (Watts and Aslin 1981). They tend to restrict their movements to shallower waters of less than 20 metres depth. Water rat activity is generally obvious since they often take prey to a favourite feeding platform, such as a log, rock, or stump, located close to the water, where remains of its food may be seen.

8 FLOW REQUIREMENTS OF KEY VALUES AND PROCESSES

8.1 Channel Morphology

In-stream flows influence the size, shape and condition of the channel through physical processes such as scouring (Arthington *et al.* 1993). Elevated winter flows are often required to maintain existing (or active) channel dimensions, and prevent the accumulation of sediment and organic debris, and prevent encroachment by riparian vegetation. Disturbances from these events can also be important in structuring benthic macroinvertebrate communities and biofilms, and may have a profound influence on ecosystem function (e.g. primary production, nutrient spiralling and decomposition) (see Resh *et al.* 1988). In addition, scour of river beds, and undercutting of banks is essential for producing and maintaining diversity of habitat, especially for fish. These high winter flows are commonly referred to as 'channel forming' or 'active channel' flows, and are required in winter with the objectives of channel maintenance, riparian vegetation inundation, inundation of higher benches for energy transfer and flushing of pools. It is generally accepted that active channel flows events occur on a 1:2 to 1:3 year frequency for south-west river systems (WRC 2001b).

Unseasonal and/or high velocity flows however, can result in excessive scouring, destabilisation of banks and subsequent increased sediment loads downstream. The erosional power of a system increases disproportionately with its discharge, thus 1:100 year floods or runoff events are extremely important in forming landscapes. Such floods and events carry the largest quantities of sediment and nutrients. Prior to European settlement, natural vegetation provided a high level of resistance to flows throughout south-west river systems. In many catchments, the clearing of vegetation for urban and rural development has made river systems sensitive to flooding, to the extent that 10-year or similar sized floods may now cause catastrophic erosion (Lovett & Price 1999). The associated practices of de-snagging and channelisation resulted in increased current velocity and thus also lead to increased bank and bed erosion, increased sedimentation and more severe flooding of downstream reaches (Lovett & Price 1999).

Channel form on the Bingham River appeared stable, probably reflecting the vegetated catchment and healthy riparian zone, however, the upper East Collie in the vicinity of the proposed diversion scheme was extremely unstable, with severe channel erosion. Not surprisingly, pools downstream were heavily sedimented, with approx. 400 m of the top end of Durdling Pool affected by siltation. This likely reflected deposition of material eroded from the channel upstream. Channel form in Coolangatta reach seems more stable, possibly reflecting lower channel gradient and amelioration of river energy by the sequence of pools.

8.2 Riparian Vegetation

Calculation of EWRs for riparian vegetation normally assumes there is water-dependent (i.e. dependent on water from the river channel) vegetation on the floodplain, which requires regular (annual) inundation to a shallow depth to disperse seed and to saturate soils to promote successful seedset. In these conditions, EWRs usually recommend overbank flooding.

Vegetation of the riparian zone can either intercept groundwater or directly extract channel water. Determination of the extent to which riparian vegetation is reliant on groundwater was beyond the scope of the current study. Since the water-dependent vegetation was within the channel, EWRs for riparian vegetation were calculated assuming a total reliance on surface flows to inundate the riparian zone.

Specific flow volumes to meet riparian vegetation EWRs need to consider duration, frequency and depth of flooding as these will have varying effect on germination, recruitment and successful colonisation by plant species. Changes in biodiversity through succession of one plant assemblage by another is a natural progression, however interruption or loss of any one successional stage can degrade the efficiency of the vegetation system as a whole (Pen 1983).

In Australian riparian zones, greatest numbers of plant species germinate during autumn under water-logged conditions and fewest germinate over summer months (Britton & Brock 1994). Decreased winter flows may thus have greater impact on the germination of fringing vegetation, than increased summer flows.

8.3 Aquatic Fauna

Life histories of aquatic species are intrinsically linked to flow regimes (Bunn 1988). There are two main features of flow regimes that influence aquatic fauna community structure in southwest rivers. These are **seasonality** and **predictability** of flows.

The variation in the degree of seasonality can lead to changes in invertebrate community structure (Bunn *et al*. 1989) and changes in life history patterns. Stream permanence has been found to be an overall determinant of the aquatic invertebrate fauna. Streams with intermittent flows show distinctive aquatic faunal communities compared to permanently flowing streams (ARL 1989; Storey *et al.* 1990). Some macroinvertebrate species are found only in intermittent streams (Bunn *et al.* 1989), while other species show large differences in abundances in permanent compared to intermittent streams (e.g. Bunn *et al.* 1986). Native fish require permanent water (i.e. permanent pools or permanent flows), only colonising seasonal and ephemeral streams during wet season flows.

Analyses of extreme flow events have shown that low-flow events have a far more pronounced effect on the river biota than high-flow events, although in streams of the northern jarrah forest, there is a linkage between near-bed water velocities and macroinvertebrate community structure (ARL 1988a, b, 1990b). The problems associated with low flow include desiccation, de-oxygenation of the water column, and accumulation of leaf leachates (phenols, tannins etc) (Resh *et al.* 1988; Boulton & Lake 1992).

There are marked seasonal changes in the structure and functional organisation of communities in upland streams in south-western Australia (ARL 1986, 1989; Bunn 1986; Bunn *et al.* 1986). This has been attributed to the influence of a highly seasonal and predictable Mediterranean climate with high winter and low summer flows. Some fauna may be influenced by seasonal differences in water temperature, however, it appears that stream flow and/or flow-related variables are the important underlying factors. Flow results in major seasonal differences in benthic organic matter, depth, width and aspects of substrate composition (ARL 1986, 1988c, d; Bunn *et al.* 1986; Storey *et al.* 1991). These seasonal patterns mean that in many systems, aquatic fauna can be grouped into typically 'dry' (summer/autumn) and 'wet' (winter/spring) season communities (Bunn 1986, Bunn *et al.* 1986, Storey *et al.* 1990).

The other major influence on aquatic fauna is the degree of temporal concordance of the flow regime; i.e. the degree to which a flow regime is not only seasonal but also predictable year-to-year (McMahon 1989, Bunn & Davies 1990, Bunn 1995). The high temporal concordance of south-western streams contrasts with streams elsewhere in Australia. Stable flows are a distinctive feature of lowland rivers during the dry season. Species that are susceptible to high and variable flows can synchronise life cycles so that the sensitive stages (e.g. the larvae of crustaceans or pupating stages of some insects) occur only during the dry season. As a consequence, unusually high discharge events during the dry season may be detrimental to the persistence of these species. It is important, therefore, that dry season flows below proposed impoundments remain benign without dramatic changes in flow rate.

8.3.1 Aquatic macroinvertebrates

To further understand the relationships between aquatic macroinvertebrates and their environment, and how changes to the hydrological regime may impact on these relationships, a review of the life history characteristics of all species recorded from this study was undertaken. Some species have been extensively studied, while there is little information on others. A range of sources were referenced for information on macroinvertebrates life histories, including but not limited to; Shipway (1951), Williams (1980), Hynes and Bunn (1984), Bunn *et al.* (1986), Bunn (1988), Boulton (1989), Storey and Edward (1989), Storey *et al.* (1990), Storey *et al.* (1991), Bunn and Davies (1992), Trayler *et al.* (1996), Cartwright (1997), Davis and Christidis (1997), Maguire *et al.* (1999), Edward *et al.* (2000), St Clair (2000), McKie and Cranston (2001), Gooderham and Tsyrlin (2002), Lawrence and Jones (2002), Sutcliff (2003) and the authors' own experience. Taxonomic keys to species were also consulted for relevant information on habitat preferences and distribution.

Spring/summer spawning is a common life history characteristic of many aquatic macroinvertebrates, and those recorded from the study area are no exception (65%; Figure 18a). Such invertebrates include; the amphipod *Austrochiltonia subtenius*, the freshwater crayfish *Cherax cainii*, the chironomids *Kiefferulus martini*, *Paramerina levidensis* and *Procladius paludicola*, the trichoptera *Oecetis* sp. and *Triplectides australis*, the ephemeroptera *Tasmanocoensis tillyardi*, and the dytiscid beetles *Allodessus bistrigatus*, *Limbodessus shuckhardi*, *Necterosoma darwini*, *Necterosoma regulare* and *Sternopriscus multimaculatus*. Very few species collected from the East Collie and Bingham Rivers breed during the wetter winter months (6%) or are likely capable of breeding year-round (11%). Therefore, some spring/summer flows should be maintained to provide breeding habitat.

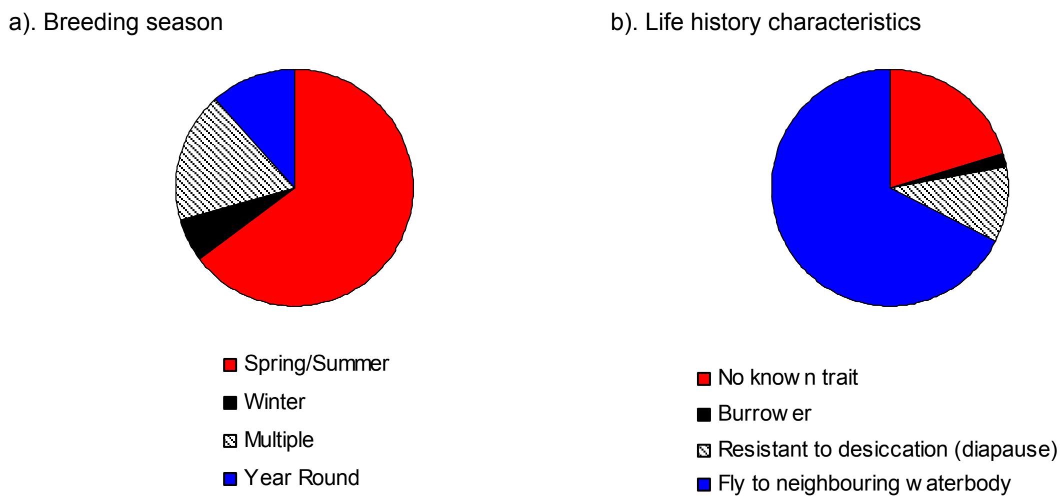


Figure 18. Pie-charts showing a) breeding season of macroinvertebrates collected from the Collie and Bingham Rivers and b) life history characteristics associated with abilities to persist in temporary environments. Only macroinvertebrates for which information was available were included in the analysis.

Only some of the macroinvertebrates recorded from the East Collie and Bingham Rivers have life history traits which would allow them to survive periods of seasonal drying, with the majority likely flying to neighbouring waterbodies (Figure 18b) Some species are capable of burrowing into moist sediments to avoid desiccation (2%), including oligochaetes and gilgies *Cherax quinquecarinatus*. Others are thought to have some resistant stage to their life cycle (usually the egg stage) and may undergo diapause during summer (10%). These include all species of gastropod recorded, cladocera, copepods and ostracods.

Macroinvertebrate diversity is dependent on habitat complexity and diversity since many species are essentially restricted to particular habitats (Brown & Brussock 1991; Humphries *et al.* 1996; Kay *et al.* 1999). A vast number of aquatic invertebrates for example, are associated with complex habitats such as snags, rocks, macrophyte and trailing riparian vegetation. Such invertebrates present in the East Collie and Bingham Rivers included oligochaetes, freshwater crayfish *Cherax quinquecarinatus* and *C. cainii*; the amphipod *Austrochiltonia subtenius*; larvae of many anisopteran dragonfly and damselfly species, most species of chironomid and most caddisfly species. Therefore, it is important to maintain sufficient flows to ensure snags, rocks, macrophytes and some overhanging riparian vegetation remains inundated. This will ensure a diversity of in-stream habitats is maintained.

Riffle zones are also regarded as highly productive habitat for macroinvertebrates (Brown & Brussock 1991). For a perennial system, it is important to maintain coverage of riffles to maintain biodiversity. It is acknowledged that water levels naturally will be lower in summer, and therefore it is unrealistic to inundate the whole width of the riffle during low flow summer months. Therefore, the dual objectives of maintaining an average depth of 5 cm with a 50% coverage of the riffle is considered adequate to maintain a low-flow summer channel over riffle zones and is regarded as the minimum necessary to support benthic invertebrate communities in summer (Storey *et al.* 2001). In winter these flows should be increased to a 100% lateral coverage.

8.3.2 Freshwater crustaceans

Two endemic freshwater crayfish, gilgies *Cherax quinquecarinatus* and marron *Cherax cainii* were recorded from the East Collie River system. Although gilgies are capable of burrowing to avoid summer drying, soils must be moist to ensure their gills remain hydrated. Neither marron nor gilgies have any resistant stage in their life cycle. Therefore, permanent flows or access to pools or shallow groundwater are required to maintain marron and gilgie populations, with EWRs for freshwater crayfish being met by flows set for macroinvertebrates.

8.3.3 Amphibian and reptilian fauna

8.3.3.1 Tortoises

Long-necked tortoises (*Chelodina oblonga*) were recorded from the East Branch of the Collie River in January 2006. Long-necked tortoises inhabit both permanent and seasonal waterbodies and have a wide distribution throughout the south-west. They can migrate relatively long distances overland if local conditions deteriorate (Gerald Kuchling, UWA, pers. comm.) or they can burrow into the sediment and aestivate. Since their diet includes tadpoles, fish, and aquatic invertebrates, tortoises only eat when open water is present. In permanent waters, this species has two nesting periods (September-October and December-January) but in seasonal systems, nesting will only occur in spring. Tortoises generally nest in sandy soils and eggs take up to two hundred days to hatch. EWRs for long-necked tortoises within the East Collie River will be met by those defined for macroinvertebrates and fish.

8.3.3.2 Frogs

The squelching froglet breeds during the wetter months of winter and would require seasonal water in the form of backwaters, shallows, still pools and/or flooded vegetation on the floodplain. In the East Collie River, flows retained to meet EWRs of macroinvertebrates and floodplain inundation flows (refer to section 9.3.5) would be sufficient to ensure survival of frog eggs and tadpoles.

8.3.4 Fish fauna

At least seven native fishes (western minnows, western pygmy perch, nightfish, cobbler, Swan River goby, big headed goby & western hardyhead) and two introduced fish (mosquitofish & rainbow trout) are known to occur in the Collie River. None of these species have adaptations to withstand desiccation. Therefore, there is a requirement for permanent water and the maintenance of summer refuges if fish populations are to be maintained year-round.

Components of the biology of native fish species most likely to be affected by altered flow regimes are migration and reproduction/access to spawning habitat. Western minnows and nightfish migrate up tributaries to spawn during winter months. Thus, maintenance of winter flows is necessary to allow movement of these fish over riffles, snags and other possible barriers. To determine a minimum depth to allow fish passage, WEC (2002) utilised a 'Rule of Thumb' water depth of 20 cm over riffles to provide for passage of western minnow in the Lower Collie River. This value seems to have first arisen from observations in the Angove River (Davies *et al.* 1999) and has been applied in all subsequent EWR studies (see Storey *et. al.* 2001, Streamtec 2002). This minimum depth requirement is likely conservative for western minnow and nightfish but is more realistic for larger, deeper bodied fish, such as the freshwater cobbler. Therefore, a depth of 20 cm over riffles is considered adequate for larger fish species in the East Collie River to maintain current fish diversity. Though minnows, pygmy perch and nightfish have been observed to negotiate waters of only 1 cm depth, they are likely to do so only under duress and such shallow waters are considered unsuitable for spawning and successful recruitment, but a shallow flow threshold of 10 cm over riffles is likely adequate for smaller species.

Predictable winter/spring flooding must also be maintained to ensure breeding success and strong recruitment. In addition, sufficient water is required to inundate trailing riparian vegetation, a favoured spawning habitat of the western minnow during winter. If water levels fall too soon, or fluctuate greatly, eggs may be left dry and desiccate. Flooded vegetation and shallow, flooded off-river areas also provide sheltered, low velocity nursery areas for growing juveniles. Retention of water levels that maintain summer pools can also be important to ensure habitat area is adequate to support populations of fish year-round in the East Collie River.

The mode of delivery of winter flows to provide for fish passage is also a critical issue. It is generally acknowledged that flows should be delivered in pulses to provide sufficient depth to maximise the ability of accumulated fish to traverse natural and man-made obstacles. Generally, fish will wait below a barrier until a spate allows them to negotiate upstream. In the East Collie River there were many natural obstacles such as fallen logs, shallow riffles and small, short water falls. The duration of the higher flows required for fish to negotiate an obstacle is open to conjecture. It is likely that for any individual obstacle, the majority of fish that have accumulated downstream would negotiate the feature within hours of it being passable. Therefore, elevating flows for several hours would probably allow fish to negotiate any individual obstacle. Assuming, there are a series of obstacles sequentially moving upstream (viz. pool-riffle type sequence), then 'passage' flows for several hours may allow fish to negotiate the first obstacle, but, depending upon the distance to the next feature, they may not have sufficient time to reach and pass the next obstacle before flows receded. Therefore, 'passage' flows longer than several hours would be necessary. To maintain fish passage in the East Collie River it would be necessary to maintain the current frequency and duration of fish passage events.

WEC (2002) suggested that increased flows do not need to be continuous, but should be maintained, as pulses to simulate spates, for at least ten days during each of the months of August and September. Streamtec (2002) recommended fish passage flows for the Samson Brook in August, September and October, with 15 flood events in September alone. This aspect of EWRs for fish requires review and further development. Ideally, the number of passage flow events should be based on the natural frequency of events, which will be related to the frequency of rainfall from south-westerly frontal systems from early winter through to spring. The 15 events in September recommended for the Samson Brook seems excessive, with natural events likely occurring at a lower frequency in any one month. Also, native fish start to migrate upstream as soon as winter flows commence (i.e. late May - early June; A.W. Storey, pers. obs.), therefore, passage flows need to start earlier in the winter (i.e. June) and continue through winter. Late winter/spring flows may be lower as fish are able to move downstream over obstacles in considerably less water than required to move upstream.

The introduced mosquitofish was recorded from the system. Pusey *et al.* (1989) suggested that natural winter spates regularly reduce the population density of mosquitofish to low levels, thus permitting the coexistence of this exotic species and small indigenous species with similar habitat and dietary requirements. Conversely, flow regulation, and the absence of large flushing flows in winter in the Canning seemed to favour the prevalence of mosquitofish; a relatively poor swimmer. The breeding strategy of mosquitofish is extremely effective; they bear live young and out-compete native fish, especially in degraded systems. High densities of mosquitofish result in a high incidence of fin damage (fin nipping) in native species (Morgan *et al.* 1998; Storey 2000).

Modifications to flow regimes may have significant implications for the dynamics and management of *Gambusia holbrooki* populations. A combination of flow regimes is required to control mosquitofish. It is suggested that the maintenance of winter spates is necessary to restore/maintain natural habitat characteristics in the lower reaches and provide increased flows which are unfavourable for these poor swimmers. In addition, a period of zero flow days in summer would also be required. These flow recommendations would reduce the suitability of the system for proliferation of the mosquitofish.

In general terms, adult native fish species in south-western Australia appear to be able to withstand relatively high salinity during low flows in summer. However, it is likely that eggs, larvae and juvenile life history stages have a considerably lower salinity threshold. Freshwater species would undoubtedly require fresher water in winter/spring to ensure successful reproduction and recruitment of these more sensitive life stages. This may be in the form of maintaining access to seasonal freshwater tributaries for spawning, or maintaining river channel winter flows comprised of freshwater runoff.

EWRs for fish species can be grouped into four categories:

- Freshwater pools must be maintained in summer to ensure survival of native species.

- Predictable winter/spring flows must be maintained to ensure breeding success and strong recruitment in the western minnow and nightfish. For example, sufficient water is required to inundate trailing riparian vegetation, a favoured spawning habitat of the western minnow during winter.

- Management of the introduced mosquitofish by maintaining winter spates and zero flow days in summer.

- Fish passage flows maintained at the current frequency and duration of fish passage events.

Flows to meet EWRs for fish passage were determined through the use of the channel surveys and Manning's Equation in conjunction with the HEC-RAS hydraulic model and the Flow Events Method. Using hydraulic modelling and time series analysis in the River Analysis Package it was possible to calculate water volumes required to achieve a 10 cm and 20 cm stage height. These depths are considered adequate for the small and deeper bodied species to negotiate most natural features. Occasional, higher natural flood events will provide additional fish passage flows for negotiating larger obstacles. It was estimated that for the rest of the year, the fish EWRs of permanent flows would be adequately met by the baseflow recommended for benthic invertebrates.

8.3.5 Water rats

Water rats are a value known to occur on the East Collie, but for which little is known with respect to water requirements. They tend to breed all year round, and a danger exists of nests being inundated and young drowned by unseasonal changes in water levels. However, this is conjecture as the location of nests relative to water levels is unknown. Their diet appears to be predominantly freshwater crayfish and molluscs, therefore, EWRs to maintain these prey items will provide for their diet. Their habitat appears to be pools and slow flowing reaches, especially with trailing vegetation and snags/woody debris. Therefore flows to maintain riparian zones as a source of large woody debris are important. Similarly, flows to maintain emergent and trailing vegetation are important. Water requirements of water rats will be developed as our knowledge increases, but at this stage we recognise them as a value on the system, and rely on their maintenance through flows to maintain other values.

8.4 Water Quality

Although provisions for EWRs should not be specifically designed as flushing flows to remove poor quality water, a discussion of water quality parameters is warranted, particularly with regard to the nutrient and salinity status of the East Collie River.

Flow regulation can adversely affect downstream water quality through the concentration of nutrients and salt when water levels are reduced. Clearing for agriculture results in increased surface run-off and inflow, and this in turn can lead to higher sediment and nutrient loading, and increased turbidity and salinity in riverine environments. At the same time, regulation by dams or groundwater abstraction reduces the input of good quality water that is low in nutrients, reducing flushing.

Nutrient enriched water bodies are typically characterised by high algal growth. Nitrates (NO_3) and ortho-phosphates (PO_4) are typically the most important forms of nitrogen and phosphorus in terms of algal and macrophytic growth. Excessive and problematic growth is more likely in systems where these nutrients are high and riparian shading is negligible or absent. Ammonia

(NH_3) is another bioavailable form of nitrogen. High levels of ammonia can be toxic to aquatic fauna (fish and macroinvertebrates) through deleterious effects on respiratory systems. Toxicity of ammonia increases as pH and temperature increase.

Elevated nutrient levels have been associated with loss of biodiversity. For example, western minnow, western pygmy perch and nightfish are all absent or greatly reduced in a number in eutrophic water bodies (Morgan *et al.* 1998; Storey 1998, 2000).

Nutrient status fundamentally influences the photosynthesis and respiration of a waterbody, which determines the oxygen status. Adequate concentrations of dissolved oxygen (DO) are fundamental for the survival of aquatic species and for the maintenance of ecological processes. The DO content of a river is determined by both physical (re-aeration) and biological (metabolic) processes. Community respiration by aquatic plants and animals at night results in a net loss in DO concentration with a DO minimum reached just prior to sunrise. During daylight hours, in-stream primary production (photosynthesis) by algae and macrophytes results in a net gain in DO in the water column. In summer, the solubility of oxygen decreases with increasing water temperatures, while increased light intensity results in higher rates of primary productivity. In rivers where rates of metabolism are high, exceedingly low overnight DO levels can be lethal for aquatic fauna. Sufficient DO over 24 hours is a fundamental requirement of aquatic fauna. DO values of less than 2 mg/L pose considerable "stress" for resident aquatic fauna, particularly fish with high metabolic demand for oxygen. Water quality problems can also arise with the release of nutrients and heavy metals from sediments under conditions of low DO.

Although flows to control water quality issues are not considered an EWR, they may be recommended as "mitigation flows" to prevent water quality problems which may place existing ecological values at a high level of risk.

No nutrient data were sourced or collected for the East Collie River, however, it is likely that there is runoff of nutrients from the surrounding catchment associated with agricultural practices. Most run-off will be in winter, when the system is flowing, and when the lower water temperatures and higher velocities reduce the risk of algal blooms occurring. The higher risk will occur in isolated pools in summer, when the system is not flowing, as observed for the East Collie in and above Wargyl Pool. Concentration of residual nutrients in pools during summer poses a risk to the water quality of these critical refugia for aquatic fauna. The East Branch of the Collie River is not regulated by a dam, therefore it is not possible to manage this situation with mitigation releases. Parts of the East Collie system retain riparian vegetation and are shaded, which will restrict excessive algal growth and therefore minimise the risk from elevated nutrients, should levels be elevated. However, as observed within and above Wargyl Pool, this is not always the situation. Catchment management activities should strive to minimise inputs of nutrients to the system.

Mitigation flows for the East Branch of the Collie River system may be summarised as:

- Natural flows to flush pools in autumn through spring to remove accumulated nutrients.

In addition to the threat from nutrients, elevated salinity also poses a threat to the ecological values of the Collie.

A mitigation flow, as such can not be used to resolve this issue. However, the proposed diversions to reduce salt loads into Wellington Dam will reduce salinity in the East Collie, especially below the confluence with the Bingham, and again below the Harris, where existing fresh inflows from these tributaries will have a greater proportional influence on salinity of main channel flows. If main channel flows are reduced and freshwater tributary inflows are sufficient to reduce main channel salinity below critical salinity thresholds for fauna, then aquatic fauna in the main channel will exist at a lower level of risk from salinity. The management objective for the East Collie River system may be summarised as:

- Reducing main channel salinity levels to a point where freshwater inflows from tributaries (Bingham and Harris) further reduce salinity below critical salinity thresholds for aquatic fauna

8.5 Energy Flows

Stream and river ecosystems are an integral component of the landscape, where ecological boundaries are often the entire catchment. Catchments provide water (surface, groundwater), nutrients and food for aquatic fauna (*e.g.* leaf litter). Therefore, any disruption within the catchment will be translated to impacts to the stream and river ecosystem.

Existing models of ecological processes differ in the interaction between a river and the catchment. The River Continuum Concept (RCC) (Vannote *et al.* 1980) emphasises an upstream-downstream linkage in energy flow, where material derived from forested regions supports downstream ecosystems. Reservoirs inhibit this upstream-downstream linkage in carbon flow. In these circumstances, the input from the riparian zone and tributaries below reservoirs are important to maintain the connectivity between forested and lower reaches.

An alternative model is the Flood Pulse Concept (FPC) (Junk *et al.* 1989) which emphasises the links between the river and its floodplain. These links occur during large flood events and material from the floodplain is transported back into river channels when floods recede.

A third concept, the River Productivity Model (RPM; Thorp & Delong 1994) may also apply to some rivers. This model emphasises the importance of local carbon inputs in providing energy (carbon) to the system. These local inputs consist of in-stream primary production (i.e. autochthonous sources; phytoplankton, benthic algae, other aquatic plants) and direct carbon inputs from the adjacent riparian zone (i.e. allochthonous sources; leaf litter, terrestrial insects). Inundation of in-channel benches is an important mechanism for the movement of leaf litter and terrestrial fauna into the aquatic ecosystem.

Analyses conducted in other south-west river systems (Davies 1993; Davies *et al.* 1998) have shown upland reaches to be reliant on the input of terrestrial carbon from forested lands, whilst Coastal Plain reaches are more dominated by algal-based production. It is likely that carbon derived from the upstream, forested catchments drives the algal-based food webs in the lower reaches on the Coastal Plain however, anthropogenic sources of nutrients also likely support coastal plain food webs.

In the East Collie River, the historic condition would have been the predominance of the RCC in the upper reaches and the FPC on the Coastal Plain. Clearing of vegetation from the upper catchment means that the source of allochthonous material is likely reduced however, regular flooding likely still occurs on the lower systems maintaining river-floodplain connection.

Storey *et al.* (2001 & 2002) and Streamtec (2002) consider that baseflows recommended for macroinvertebrates are adequate to maintain upstream-downstream energy linkages (autumn through to spring). Higher winter flows provided for fish passage and to inundate riparian values will also be adequate to flush the channel and inundate benches and riparian vegetation and thereby maintain floodplain linkages respectively. EWRs to maintain energy linkages therefore, will be met within the EWRs for macroinvertebrate baseflows, combined with fish passage flows and riparian inundation flows.

Energy flows in the East Collie system may be summarised as:

- Winter flows to maintain upstream-downstream linkages and therefore transport of energy/carbon

- Flows to maintain riparian vegetation as an energy source for the RPM

- Flood flows to maintain carbon/energy linkages between the channel and its floodplain by inundating low, medium and high benches.

9 ISSUES, PRESSURES AND ECOLOGICAL FLOW OBJECTIVES

9.1 Risks to the East Branch of the Collie River

Based on the hydrological state of the catchment, current catchment land uses and observed ecological values, a range of management issues and pressures likely to affect the ecological values of the system were identified. Broad key issues are discussed below, and summarised in Table 13.

9.1.1 Loss of permanent pools (water quality and quantity)

The East Collie River supports a number of endemic freshwater fish species, with survival of these species dependent upon permanent pools as refuges during summer, requiring low salinity levels (below a currently unknown threshold) for survival. Although adult fish were taken from relatively high salinity pools on the East Collie River in late summer, it is generally considered that larval and juvenile stages are more sensitive than adults to 'contaminants', including salt. Although specific ecotoxicity data are not available for the response of larval/juvenile fish to salt, a precautionary approach is to ensure flows in winter/spring are fresh to ensure survival of larval/juvenile stages. Changes in hydrogeology, with differential contributions of fresh and saline aquifers to the various pools have the potential to threaten the survival of these species in the East Collie River.

9.1.2 Loss of permanent pools threaten survival of freshwater tortoise

Freshwater tortoises have been recorded from the East Collie River. This species commonly requires summer refugia in the form of permanent pools, although this is not critical as long-neck tortoise can aestivate (Gerald Kuchling, Uni of W.A. pers comm). The juveniles likely use shallow-inundated floodplain areas for growth where they remain away from the faster flows and where the waters are more productive, supporting abundant macroinvertebrate and macrophyte populations which provide food and cover. In perennial systems long-neck tortoise nest in spring and again in summer, but in seasonal systems they nest in spring only (they either aestivate in summer or relocate to other water bodies) (Gerald Kuchling, Uni of W.A. pers comm). Loss of permanent pools may threaten the survival of this species in the East Collie River, and therefore further restrict their distribution. Conversely, permanent flows may lead to a build-up in numbers through increased recruitment, resulting in changes to community structure through increased predation on prey items.

9.1.3 Reduced flows limit fish passage

Changes in flow regime, particularly excessive abstraction to the extent that total discharge and flow period are both reduced may result in inadequate flows over riffles to facilitate passage of migratory fish species (i.e. nightfish and western minnow). Reduced flows also have the potential to affect shallow backwater areas which act as spawning and nursery areas when flooded. Reduced flows may also reduce extent of trailing reeds/rushes which provide habitat for spawning and predator avoidance.

9.1.4 Affect of flow changes on frog populations

Floodplain habitats require occasional flooding to maintain water presence, although local runoff and seeps may maintain water levels in some of these shallow floodplain systems. Although adults can survive some salt, the juvenile stages require freshwater for survival. Changes in flow that affect either winter active channel flows to inundate floodplain habitat, or change the winter/spring salinity have the potential to adversely affect frog populations. Seasonal inundation will also help to flush accumulated salt from these habitats. Therefore, a reduction in flushing would adversely affect communities through the build-up of salt.

9.1.5 Affect of flow changes on macroinvertebrate populations

The East Collie River supports a diversity of aquatic invertebrate species, with some mobile species being able to reinvade the system each year (i.e. Diptera, Lepidoptera, Odonata, Trichoptera, and Ephemeroptera). Seasonal systems, such as the East Collie River, have a characteristic suite of species with a range of adaptations to survive desiccation, and maintaining seasonal flows will be necessary to preserve populations of such species. There are other species, however, which cannot withstand desiccation, require permanent freshwater and take over a year to complete their life cycle (i.e. march flies, tabanids). Changes in flow periodicity and salinity have the potential to adversely affect macroinvertebrate populations.

9.1.6 Adequate scouring required to maintain pools

Many of the ecological values in the East Collie River rely on permanent pools for their continuance. Pools are maintained by having 'channel forming' flows (i.e. bankfull/active channel flows) with sufficient power to scour-out pools and maintain their depth.

Table 13. Issues and pressures affecting hydrology, flow regime and ecology of the East Branch of the Collie River.

Driving Force	Issue	Time of Year	Hydrological/WQ Effects	Potential ecological effects
Salinity	Loss of fresh water in refuge pools for fauna	Summer	Increased salinity in pools in summer	Salinity exceeds thresholds of species, resulting in death
Reduced water table	Loss of permanent pools as refugia for tortoise and fish	Summer	Reduced groundwater levels, reduced flows	Loss of species that require permanent water
Flow diversions	Reduced fish passage flows	Winter	Loss of early/mid winter events that provide fish passage flows	Reduced fish recruitment and loss of species from some parts of the system
Reduced overbank flows	Reduced floodplain inundation and flushing flows to maintain frog habitat	Winter/spring	Reduced inundation and quality of frog habitat	Reduced recruitment success of frogs in floodplain habitats
Reduced flow periodicity	Reduced flow period limiting growth and recruitment period for macroinvertebrates	Winter/spring	Shorter flow period and reduced inundation of critical habitats	Loss of some species of macroinvertebrates
Peak winter	Excessive erosion of the channel with	Winter/spring	Excessively high flows causing	Loss of values dependent on deep, permanent pools

Driving Force	Issue	Time of Year	Hydrological/WQ Effects	Potential ecological effects
flood flows	sedimentation and infilling of pools		channel erosion/reduced peak flows lading to pool infilling	+ loss of species sensitive to increased sediment loads

9.2 EWR Flow Objectives

The ecological flow objectives for key features of the East Collie River were considered to be:

Channel morphology

- Maintain channel forming flows in winter to flush the channel of accumulated sediments and organic detritus and preserve active channel dimensions.

Riparian vegetation

- Inundate the in-channel riparian zone up to the bankfull level, but not above bankfull.
- Inundate the in-channel riparian zone winter (July or August), timed to coincide with naturally-elevated flows.
- Avoidance of excessive flows that cause bank and bed erosion leading to loss of bank-side riparian vegetation (middle and lower reaches).
- Winter base flows for seasonal inundation of riparian vegetation.
- It was considered that flows approximating channel forming flows will inundate benches and existing riparian vegetation within the drainage channel.

Aquatic invertebrates

- Maintain permanent pools during summer as many invertebrates require permanent water to complete their life cycle.
- Maintain winter base flows of 5 cm (0.05 m) stage height over riffles.
- Maintain flow periodicity

Freshwater crayfish

- Maintain permanent low flow (baseflow) to prevent desiccation.
- It was considered that the EWRs for freshwater crayfish would be met in all months by the low flow proposed for aquatic macroinvertebrates.

Freshwater fishes

- Maintenance of permanent pools in summer to ensure survival of native species.
- Maintain current frequency of fish passage flows to allow small and large bodies fish passage upstream across obstacles (10 cm and 20 cm minimum depth respectively), and ensure breeding success and strong recruitment in the western minnow and nightfish.
- Bankfull/overbank flows in winter/spring to inundate backwaters and shallow areas on the floodplain which act as important foraging/nursery habitat for larval and juvenile fish.

Freshwater tortoises

- EWRs for long-necked tortoises will be met by those defined for fish and macroinvertebrates.

Freshwater frogs

- Floodplain inundation flows. Flows retained to meet EWRs of macroinvertebrates would be sufficient to ensure the survival of frog eggs and tadpoles.

Water rats

- Unknown – relationship between flows and nesting sites. Ensure inundation of appropriate habitat, LWD etc, avoid drowning of nest sites

Riparian fauna

- Ensure healthy riparian condition, which will maintain riparian fauna (mammals, reptiles and birds) diversity. Will be maintained by overbank – riparian vegetation flows.

Water quality

- Assess effects of diversion flows on salinity thresholds.

Energy flows

- Winter flows to maintain upstream-downstream linkages for transport of energy/carbon.
- Winter flows sufficient to seasonally inundate higher and lower benches for allochthonous litter transfer. Will maintain carbon/energy linkages between the channel and its benches.
- It was considered that winter baseflows for macroinvertebrates were adequate to maintain upstream-downstream energy linkages (autumn through to spring), and winter medium/high flows to flush the channel and inundate riparian vegetation would be adequate to maintain lateral carbon linkages.
- Winter bench inundation flows to seasonally inundate higher and lower benches for autochthonous litter transfer.

The key ecological features and the flow components to be assessed for each ecological value have been summarised in Table 14 as flow objectives, with the various ecological values grouped under common flow objectives. These were analysed using hydraulic models and the Flow Events Method (FEM).

Table 14. Ecological values and objectives as determined for the East Branch of the Collie River and the Bingham River. Location 1 = Buckingham (Upper East Collie), Location 2 = Coolangatta (Lower East Collie), and Location 3 = Bingham River reach.

Flow Component	Ecological attribute/value	Ecological objective	Location or zone	Season (duration)	Time series (pulse/spells)	Hydraulic metric	Consequence of not meeting objective
Summer base flow	Fish and invertebrate fauna diversity	Maintenance of permanent pools	1 2 3	Summer			Loss of species requiring permanent water.
Summer base flow	Invertebrate diversity	Maintain gravel runs and riffles as biodiversity 'hotspots'	1 2 3	summer	Flow duration	Minimum stage height during summer to maintain current area of gravel runs and riffles to a depth of 0.05 m and 50% lateral coverage	Loss of biodiversity
Summer base flow	Aquatic fauna	Maintain pool volume through summer as a drought refuge for aquatic fauna	1 2 3	summer	baseflow		Loss of pools and species requiring permanent water
Summer low flow	Native fish; aquatic invertebrate fauna diversity; pool water quality	Flows to prevent significant stratification or anoxia in pools	1 2 3	summer	Flow duration	Minimum average discharge of 0.01 m sec-1 in pools to avoid stratification/maintain mixing. Maintain DO levels > 2 mg/L	Pool anoxia with fix kills. Sub-lethal effects to eggs/larvae. Aquatic macroinvertebrates – loss of biota/change in composition to those with strategies to tolerate low oxygen levels.
Fish Passage Flow	Native fish diversity	Provide passage for small bodied fish (i.e. western minnow, pygmy perch and nightfish) and large bodied fish (i.e. cobbler), moving upstream from late autumn through winter and into early spring across obstacles such as shallow riffles and runs.	1 2 3	late autumn through winter and into early spring	events (size and frequency) / duration	Minimum depth over obstacles of 0.10 m for small bodied fish in the upstream reaches of the East Collie and 0.20 m for large bodied fish in downstream reaches of the East Collie. From late May to late August for fish movement	Loss of migratory species from parts of the system if passage restricted. Reduced connectivity.
Stress relief flow	Pool ecology	Maintain oxygen, temp etc, flush contaminants	1 2 3	Late autumn, early winter	Small events	Maintain frequency, timing, duration of early season freshers	Reduced flow period and extended period of summer stress conditions – threats to ecological values in pools
Winter Low	Native Fish	Stage height to ensure	1	Winter	Flow duration	Duration of baseflow	Insufficient flows will leave

Flow Component	Ecological attribute/value	Ecological objective	Location or zone	Season (duration)	Time series (pulse/spells)	Hydraulic metric	Consequence of not meeting objective
Flow		marginal reeds/rushes are trailing & thereby providing fish cover and spawning habitat	2 3			sufficient to inundate trailing vegetation – based on elevation on x-sections.	trailing vegetation above water and not accessible; insufficient continuous duration may expose and dehydrate eggs spawned onto vegetation; increased risk of predation/competition with other fish (introduced) species.
Winter Low Flow	Tortoise and Frogs	Stage height to ensure marginal reeds/rushes are trailing & thereby providing cover.	3	Winter	Flow duration	Duration of baseflow sufficient to inundate trailing vegetation – based on elevation on x-sections.	Loss of biodiversity
Winter Low Flow	Invertebrates	Maintain gravel runs and riffles as biodiversity 'hotspots'.	1 2 3	Winter	Flow duration	Inundate riffles in winter (0.05 m stage height over riffles with 100% lateral coverage)	Loss of biodiversity
Winter Low Flow	Vegetation	Inundate emergent macrophytes and aquatic plants	1 2 3	Winter	Flow duration	Inundate lower benches in winter	Loss of biodiversity
Winter Medium Flow	Process	As for winter low flows					
	Native fish	As for winter low flows					
	Vegetation	Riparian vegetation – main channel bank & emergent vegetation	1 2 3	Winter	Flow duration	Flood lower banks in winter	Change from historic water regime = change in plant community (terrestrialisation) with associated change in structure. Enhanced opportunity for terrestrial weeds (e.g. grasses). Riparian vegetation supplier of LWD as aquatic habitat & material to support detrital food webs. Regulation of nuisance algal growth, through shading.
Winter Medium Flow	Process	Seasonal inundation of benches for allochthonous litter transfer. Predictions of Riverine Productivity Model; seasonal inundation	1 2 3	Winter	Medium wet season events	Inundate lower benches	Detrital material important in food webs. Loss of this material may limit abundance and/or presence of some species.

Flow Component	Ecological attribute/value	Ecological objective	Location or zone	Season (duration)	Time series (pulse/spells)	Hydraulic metric	Consequence of not meeting objective
		& recession 'collects' detrital material in main channel which supports food webs.					
Winter High Flow	Channel morphology	Maintain pools & channel form. Pools provide refugia for fauna in summer & require regular scouring to prevent excessive build up and infilling.	2 3	Winter	Event magnitude/ frequency	Channel forming flows – flows to active channel stage height on a 1:2 – 1:3 year frequency	Loss of pool depth = reduced carrying capacity for fish, loss of summer refugia for fish, tortoise and water rats, greater encroachment by riparian vegetation, higher BOD with associated risk of low DO in summer, loss of benthic fauna due to smothering by fine sediment build up, smothering of snags in pools = reduced habitat.
		Prevent incursion of riparian vegetation into channel. There is a dynamic relationship between flow, sediment deposition & vegetation encroachment on the channel.	1 2 3	Winter	Event magnitude/ frequency	Channel forming flows – flows to active channel stage height on a 1:2 – 1:3 year frequency	Area of active channel will decrease. Peripheral velocities will be reduced resulting in more sediment deposited & weed incursion.
Winter High Flows	Predictions of Flood Pulse Concept; seasonal inundation and recession "collects" detrital material in main channel which supports food webs	Seasonal inundation of higher (and lower) benches for allochthonous litter transfer	1 2 3	Winter	medium wet season events	Inundate higher benches in winter	Detrital material important in food webs. Loss of this material may limit abundance and/or presence of some species.
Overbank flow	Aquatic fauna diversity	Maintain overbank flows to inundate the floodplain and provide shallow floodplain areas for foraging/ nursery habitat for tadpoles and juvenile tortoise - and provide areas in which juvenile stages with poor swimming ability may avoid high flows	3	Winter/ spring	Peak flows	Overbank flows to flood floodplain and inundate/recharge wetlands	Juvenile stages exposed to high flows potentially swept from the system, areas are productive in terms of macrophytes, algae and invertebrates for juveniles to forage on these areas, likely loss of spawning sites (suspected)

Flow Component	Ecological attribute/value	Ecological objective	Location or zone	Season (duration)	Time series (pulse/spells)	Hydraulic metric	Consequence of not meeting objective
Overbank flow	Floodplain wetland maintenance	Maintain overbank flows to inundate wetlands which provide habitat for fish, bugs, tortoise and frogs	3	Winter	Overbank flows – large events of low frequency	Overbank flows to maintain habitats by scouring pools of silt, sediment, nutrients, organics etc	Wetlands will disappear over time if not maintained, with loss of habitat for fauna leading to reduced populations of frogs, tortoise and possibly some invertebrates and fish. Fewer recruitment opportunities for riparian trees, shift away from mesic sedges and rushes.
Overbank flow	Floodplain vegetation	Inundation of floodplain vegetation (i.e. *E. camaldulensis*, *Melaleuca*)	3	Winter	Peak wet season flow duration. Short duration onto floodplain at low frequency (i.e. aseasonal)	Flood floodplain to <= 0.15 m for flow duration in winter.	Reduced wetland species recruitment, possible reduced seed dispersion. Possible enhanced invasion by exotic grasses, terrestrialisation of plant communities (with associated change in structure).

10 FLOW EVENTS MODELLING

10.1 FEM Approach

The Flow Events Method (FEM) was used to estimate ecologically significant flow events for the East Collie River at Buckingham and Coolangatta, and the Bingham River at Palmer. The term 'event' refers to a particular suite of hydrologic conditions identified as significant for maintaining the ecological values of the study reach. FEM comprises two steps: 1) Hydraulic Analysis (HA), which is the derivation of 'habitat rating curves' that show discharge required to achieve desired objectives (*i.e.* stage heights to inundate habitat) and 2) Time Series Analysis (TSA), which is the analysis of the flow record to determine the frequency and duration of ecologically significant flow events, whereby the historic flow record is transformed into records of events using the rating curves. The event record is then analysed for aspects such as flow duration; *i.e.* the percentage of time that an event of a particular magnitude is exceeded or not exceeded, depending upon how the criteria is defined. The analyses can look at the whole flow record, or restrict the analysis to specific seasons (*e.g.* fish passage in winter) (Cottingham *et al.* 2003). Rating curves describe the relationship between ecologically significant flow events (*e.g.* fish passage flows) and discharge. The flow events are defined in terms of specific criteria that can be described in a spreadsheet (*e.g.* minimum depth in the reach of 0.1 m). HA was performed for each reach using the relationships between flow rate and stage height established from the Hec-Ras hydraulic models and the cross-section survey data from the reaches, as summarised above.

As discussed in Section 6, actual flows over a range of stage heights required to calibrate the Hec-Ras model were not surveyed for the current project. For the Coolangatta Reach, calibration was only attempted for the summer baseflow at the time of survey of approx. 0.04 m3/sec and by using elevation of flood debris, estimated to have been deposited in the last 36 mnths, at a flow of approx 19 m^3/sec (calibration gave water levels within 40 cm of flood debris, which was considered reasonable given that the actual peak flow was unknown (i.e. may have been higher than the mean daily flow), and year in which flood debris was deposited was also unknown). For the Buckingham Reach there was zero flow at the time of survey, and there was no observable flood debris on which a high flow calibration could be achieved. Therefore, the Hec-Ras model for Buckingham is currently uncalibrated to any flows. For the Bingham Reach there was no flow at the time of survey, but as for Coolangatta, elevation of flood debris against recent peak flood flows (approx 2.9 m^3/sec) was used to calibrate the Bingham Reach hec-ras model to peak flows only.

In the absence of a series of discharge to water level readings for the reach, a rating curve was derived for each model by running steady-state flow simulations for a range of flow rates covering the range of flows likely for the system (*i.e.* 0.001 – 200 m^3/sec). Therefore, the Hec-Ras models have not been calibrated to a range of flows. However, although the accuracy of the models is not proven, the analyses below are mainly comparative, therefore, relative differences between scenarios for the same event should not be affected.

Original development of the FEM was for application to the Gouldburn River in eastern Australia (Cottingham *et al.* 2003). The Goulbourn is a regulated river and FEM was used to compare regulated flow to modelled natural flow in order to compare differences between the natural and modified regimes (*i.e.* differences in frequency and duration of bench inundation). This is an aspect to which FEM is particularly suited. This approach was also adopted for the East Collie project by comparing the observed flow series (1980 – 2005), against the modelled time series for the three diversion scenarios (5 GL, 10 GL and 15 GL total diversion) for the

Coolangatta and Buckingham reaches (NB the Bingham flows will not be modified, so only current flows were analysed). The modelled series were derived from the observed series on the rules that flows are diverted when discharge is > 5 ml per day (~ 0.057 m^3/sec instantaneous flow), and when salinity was > 2000 mg/l. This principally targets spring and autumn flows, but with diversion continuing through all winter months.

10.2 Hydraulic analysis

Flow events deemed ecologically important, following consideration of the identified ecological values and current scientific knowledge of the water requirements of these values are summarised together with flow recommendations in Table 44 at the end of Section 10.

In the first instance, HA was performed to determine habitat rating curves and critical flow events to support ecological values and processes on each reach. TSA was then performed on each flow event to determine current frequency and duration, and how this may be affected by the diversion scenarios. The flow records were separated into summer and winter seasons based on approximate times when discharge starts to increase/decrease each year, using 16th May as commencement of winter conditions and 1st November as commencement of summer flows. These transition times are approximate, and will vary from year to year depending on annual rainfall pattern.

The ecological values, processes and features to be maintained and therefore modelled by FEM for each reach included:

- Summer baseflow to maintain flows for inundation of runs/riffles for macroinvertebrate habitat, and maintain flow connectivity for downstream energy (carbon) transfer;
- Winter baseflow for inundation of runs/riffles for maintenance of macroinvertebrate biodiversity and flow connectivity for energy transfer;
- Winter fish passage flows for a.) small and b.) large bodied species to allow movement of native fish upstream to reach spawning habitats from late autumn to early spring;
- Flows for inundation of medium benches for energy transfer;
- Flows to connect off-river wetlands/sumps;
- Active channel flows for channel maintenance (flushing accumulated sand, silt and organic debris from pools), including inundation of riparian vegetation, inundation of woody debris and provision of trailing vegetation for fish habitat;
- Winter high flows for top of bank flooding for inundation of floodplain areas and riparian vegetation.

10.2.1 Coolangatta reach

10.2.1.1 Summer baseflow to inundate riffles

The objective of summer baseflows is to maintain inundation of sand/gravel runs as biodiversity ‘hotspots’ for macroinvertebrates, and provide flow connectivity for downstream energy transfer. The Coolangatta Reach of the East Collie River did not have the classic cobble/pebble/gravel riffles between pools *per se*, but was characterised by sand/gravel runs held together by root mats from riparian vegetation. However, these areas will function as riffle zones in that they will have higher velocity flows, and a diversity of flows in summer relative to pools, but may not support such highly diverse macroinvertebrate fauna compared with typical, more heterogeneous cobble/pebble, gravel riffles.

Analysis of flow data indicated that the East Collie at Coolangatta reach was perennial in some years but intermittent in other years, ceasing to flow for periods in summer months in some years. It was therefore appropriate to determine a summer baseflow to maintain inundation of sand/gravel runs as biodiversity 'hotspots' for macroinvertebrates. In perennial systems, a low-flow summer baseflow objective for maintenance of riffles is regarded as flows to achieve 50% lateral coverage of riffles to 0.05 m average depth across the riffle. However, because this reach flows intermittently during summer months in some years, and flows will recede in summer, especially in dry years, the 50% lateral coverage will likely be seldom met. As for the Hill River EWRs (WRM, 2005b), this riffle inundation rule was modified to 25% coverage to 0.05 m depth. Rules to meet riffle objectives are:

- inundation of riffles to a minimum average depth of 5 cm with 25 % coverage of riffle cross-section.

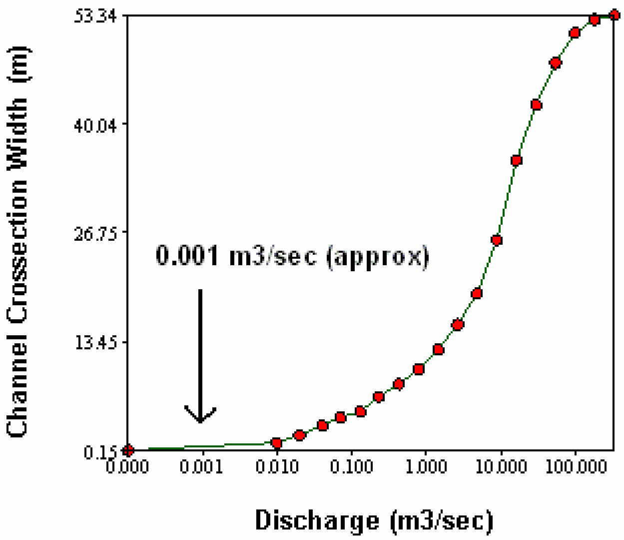


Figure 19. Rating curve for inundation of sand/gravel runs (cross-sections 2, 3, 6, 7, 10 & 11) to 5 cm average depth with 25% habitat coverage on the cross-section (= 1 m cross-section total width) on Coolangatta Reach, East Collie. Change in channel width (y-axis) with increasing discharge (m^3/sec) (x-axis) is shown. The threshold criteria of 0.05 m average depth with 25% coverage of riffles was achieved with a discharge of approx. 0.001 m^3/sec.

Hydraulic analysis was used to calculate discharge required to achieve a mean depth of 0.05 m across sand/gravel runs whilst maintaining 25% coverage of the habitat. Sand/gravel runs were on cross-sections 2, 3, 6, 7, 10 & 11 (cross-sections 13 and 14 were on the gauging station weir structure and so were not considered a riffle). The mean width of the riffle zones, as taken from channel cross-sections, was 4.05 m. Discharge required to achieve 0.05 m average depth with 1 m of channel cross-section inundated (i.e. 25%) was then read from the rating curve. The actual flow threshold was relatively indistinct on the habitat rating curve, probably reflecting the accuracy of the Hec-Ras model at such low flows, but was approximated at 0.001 m^3/sec (Figure 19). It is recommended that this threshold flow is maintained as a minimum baseflow in summer at the current frequency and duration, allowing for periods of zero flow, to maintain ecological values dependent on this flow regime in sand/gravel runs.

TSA performed on this flow event demonstrated that this base flow currently occurs for an average of 71 days each summer, with on average 6 periods of approx 15 days in which flows drop below 0.001 m^3/sec (Table 15). These statistics did not vary under each diversion scenario (Table 15), indicating little influence of diversion scenarios on summer flows. This is not unexpected, since the present rule for commencing diversions is when flows exceed 5 ML per day, which equates to 0.057 m^3/sec, which is substantially higher than the estimated summer baseflow of 0.001 m^3/sec to maintain 25% lateral coverage of sand/gravel runs.

Table 15. Summary statistics for frequency and duration of flows ≥summer baseflows (0.001 m^3/sec) to inundate sand/gravel runs to 25% lateral coverage.

Statistics for summer baseflows	Current	Q5	Q10	Q15
Mean number of events	6.5	6.5	6.5	6.6
Mean duration of events (days)	71.0	71.0	71.0	70.7
Total duration of events (days)	119.4	119.4	119.4	119.4
Mean period between events	15.1	15.1	15.1	14.7
Total period between events	109.7	109.7	109.7	109.8

As noted above, the East Collie at Coolangatta ceases to flow in summer months in some years. These periods of zero flow will influence the ecology of the system. It is therefore important to ensure the duration of zero flow days is not modified by diversion scenarios. Analysis of zero flow days in each month in summer again shows no effect of diversion scenarios on current duration of zero flow days (Figure 20). On average, there were 63 days of zero flow each summer, which did not vary between current and diversion scenarios.

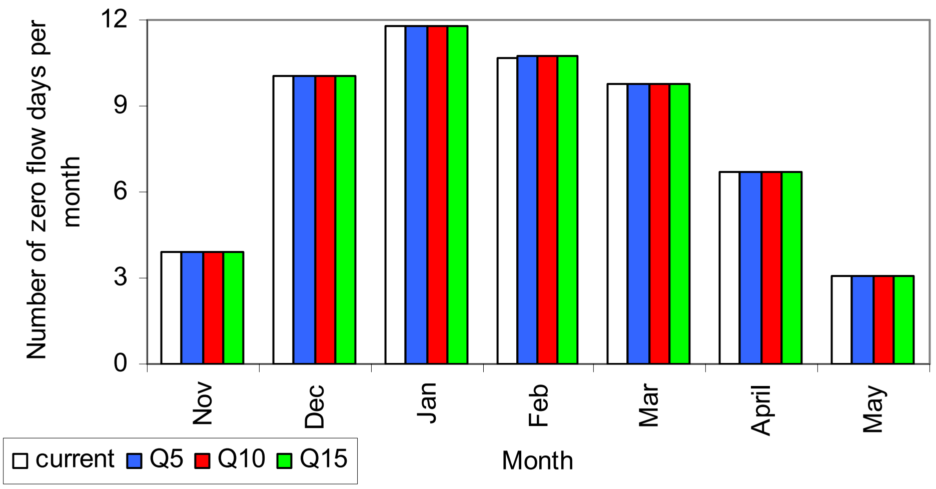


Figure 20. Number of zero flow days in each month in summer under current flows and each diversion scenario for Coolangatta.

10.2.1.2 Winter baseflows on riffles

The objective of winter baseflows is to maintain inundation of sand/gravel runs for macroinvertebrates, and provide flow connectivity for downstream energy transfer in winter. Based on channel morphology, it was considered that the above winter baseflow objectives collectively would be provided for by the minimum depth requirement normally applied to riffle zones, being:

- inundation of sand/gravel runs by a minimum average depth of 5 cm (0.05 m) with 100 % coverage of the sand-run component of the relevant cross-sections.

Hydraulic analysis was used to calculate discharge required to achieve a mean depth of 0.05 m across sand/gravel runs whilst maintaining 100% coverage of the habitat. Sand/gravel runs were on cross-sections 2, 3, 6, 7, 10 & 11. The mean width of the riffle zones, as taken from channel cross-sections, was 4.05 m. Discharge required to achieve 0.05 m average depth with 4.05 m of channel cross-section inundated was then read from the rating curve as 0.01 m^3/sec (Figure 21). It is recommended that this threshold flow is maintained as a minimum baseflow throughout winter to maintain ecological values dependent on sand/gravel runs. The transition from summer to winter baseflows is dependent on annual rainfall pattern, but in general terms should occur in the shoulder months of May/June, with a recession back to summer baseflows in October/November.

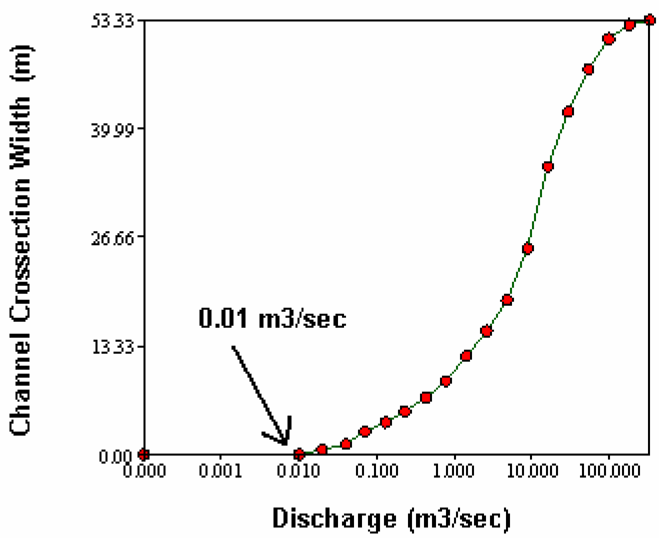


Figure 21. Rating curve for winter inundation of sand/gravel runs (cross-sections 2, 3, 6, 7, 10 & 11) to 5 cm average depth with 100% habitat coverage on the cross-section (= 4.05 m cross-section total width) on Coolangatta Reach, East Collie. Change in channel width (y-axis) with increasing discharge (m^3/sec) (x-axis) is shown. The threshold criteria of 0.05 m average depth with 100% coverage of riffles was achieved with a discharge of 0.01 m^3/sec.

TSA performed on this flow event demonstrated that this base flow was currently achieved throughout the majority of winter in all years (Table 16). Currently, the baseflow occurs for an average of 161 days each winter (maximum of 170 days per winter 'season'), with on average 2 periods of approx 4 days in which flows drop below 0.01 m^3/sec. These statistics for current regime were very similar under each diversion scenario (Table 16). Currently there is a mean minimum winter flow of 0.01 m^3/sec, which decreases to 0.007, 0.007 and 0.006 m^3/sec under 5GL, 10GL and 15GL diversion scenarios respectively, and the current mean winter flow of 2.74 m^3/sec, decreases to 2.64, 2.57 and 2.50 m^3/sec under 5GL, 10GL and 15GL diversion scenarios respectively. Therefore, it appears that the diversion scenarios have minimal effect on proposed winter baseflows. This is not unexpected, since the present rule for commencing diversions each winter is when flows exceed 5 ML per day, which equates to 0.057 m^3/sec, which is five times greater than the estimated winter baseflow to maintain sand/gravel runs. The benign effects of diversions on winter baseflows can be seen in the cumulative probability plot for sand/gravel run inundation, with no apparent loss of inundation, even under the 15 GL diversion scenario, with sand/gravel run inundation achieved for > 200 days on 80% of years (Figure 22).

Table 16. Summary statistics for frequency and duration of winter baseflows to inundate sand/gravel runs.

Statistics for winter baseflows	Current	Q5	Q10	Q15
Mean number of events	2.4	2.9	2.8	2.8
Mean duration of events (days)	100.8	80.5	86.0	84.8
Total duration of events (days)	160.5	160.0	160.1	160.0
Mean period between events	4.2	3.1	3.4	3.3

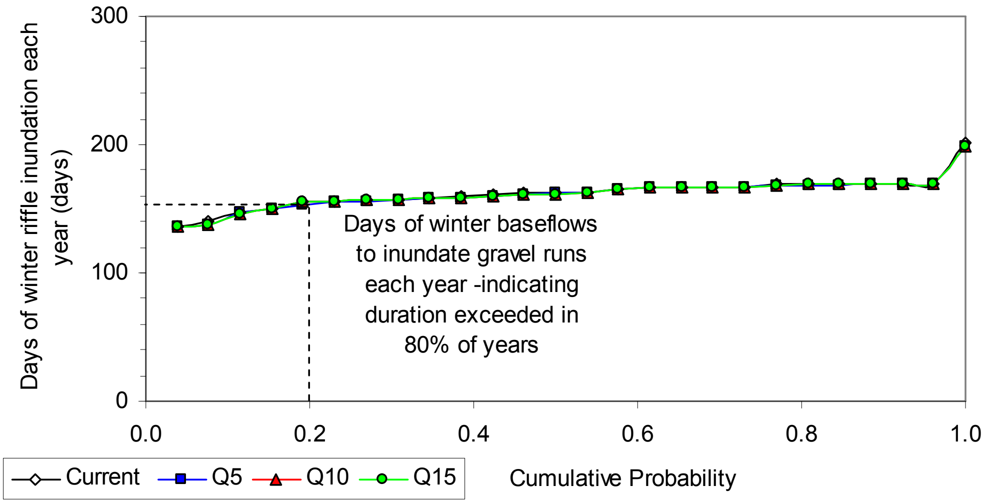


Figure 22. Cumulative probability plot for duration of riffle inundation in winter for Coolangatta, showing effects of each diversion scenario, with minimal effects at the 20% probability level (i.e. days of riffle inundation exceeded in 80% of years).

When broken into months, the mean total duration of sand/gravel run inundation events across years (Figure 23) shows inundation commences in May, with almost continuous inundation through June to October. Comparison of current duration to duration under each diversion scenario shows negligible effects in any month (Figure 23, Table 17).

Table 17. Mean of total duration (days) of sand/gravel run inundation each month, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	97.3%	98.0%	97.3%
Feb	98.0%	98.0%	97.3%
Mar	99.3%	98.7%	98.7%
Apr	99.5%	99.5%	99.0%
May	98.9%	98.9%	98.7%
Jun	99.3%	99.4%	99.6%
Jul	99.9%	100.0%	99.7%
Aug	100.0%	100.0%	100.0%
Sep	99.9%	99.9%	99.9%
Oct	99.8%	99.8%	99.8%
Nov	98.6%	98.8%	98.8%
Dec	100.0%	100.5%	100.0%

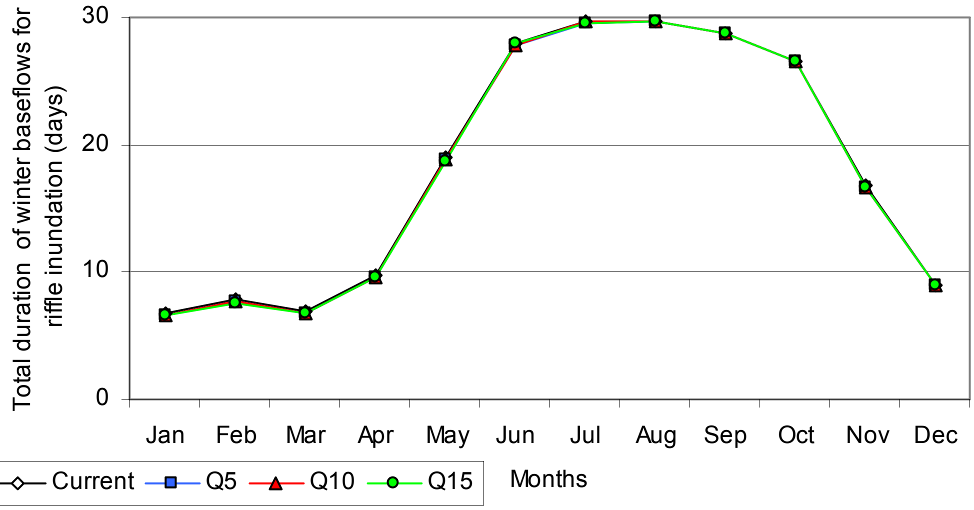


Figure 23. Mean of total duration of sand/gravel run inundation for each month for Coolangatta for current flows and the three diversion scenarios, where discharge exceeded 0.01 m^3/sec.

Similarly, the mean monthly duration of sand/gravel run inundation shows negligible effects of diversion in each month (Figure 24, Table 18). The greatest reduction in any month is 14% from current duration in May under the 15 GL diversion (Table 18).

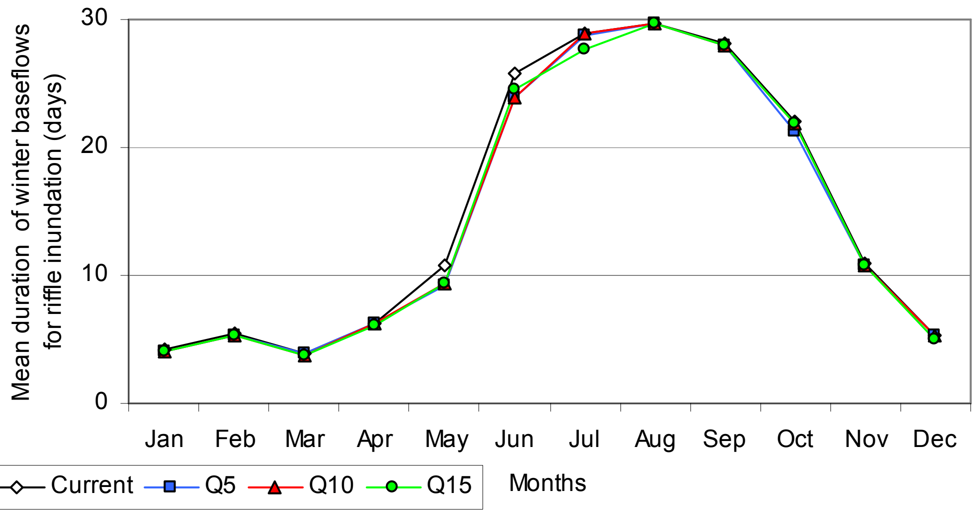


Figure 24. Mean of mean duration of riffle inundation in winter for Coolangatta for each month for current flows and the three diversion scenarios, where discharge exceeded 0.01 m^3/sec.

Analysis of total number of days of sand/gravel run inundation also shows minimal change due to proposed diversion scenarios in any year (Figure 25). Therefore, inundation of sand/gravel runs to 0.05 m depth, with 100% of the habitat covered requires flows of 0.01 m^3/sec, and these occur through the majority of each winter. Implementation of diversion scenarios has little effect on frequency or duration of inundation of this habitat, even at the highest diversions, because the diversions commence at a higher discharge than is required to meet riffle inundation.

Table 18. Mean of mean duration (days) of riffle inundation each month, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	97.0%	97.3%	96.2%
Feb	98.2%	98.2%	96.3%
Mar	99.4%	96.3%	96.3%
Apr	99.6%	99.6%	97.2%
May	86.1%	87.9%	87.3%
Jun	92.8%	92.9%	95.3%
Jul	99.6%	100.0%	95.6%
Aug	100.0%	100.0%	100.0%
Sep	99.3%	99.3%	99.3%
Oct	97.1%	99.5%	99.5%
Nov	98.9%	99.0%	99.0%
Dec	100.1%	100.4%	94.2%

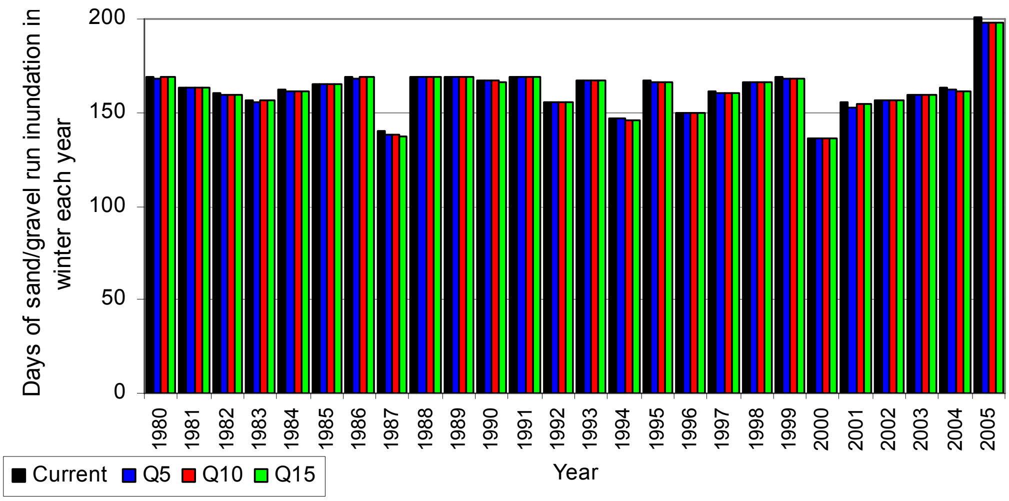


Figure 25. Total number of days of sand/gravel run inundation in winter for Coolangatta in each year of modelled flow record, where discharge exceeded 0.01 m^3/sec.

10.2.1.3 Fish passage flows

Based on fish surveys, fish passage flows are required for both small bodied (western minnow, nightfish, western pygmy perch), and large bodied species (cobbler). The rules used for fish passage in SW WA, based on current scientific knowledge, are 0.1 m minimum depth along the reach (i.e. thalweg depth on shallowest cross-section) for small bodied fish and 0.2 m minimum reach depth for large bodied fish. These minimum threshold depths are considered adequate to allow upstream movement of native fish. The deeper threshold for larger fish will usually occur at a lower frequency than the shallower events for small bodied fish since the former require higher flows, which usually occur less frequently. Therefore, habitat rating curves and TSA were calculated for both rules/flow events.

Fish passage for small bodied species (0.1 m minimum depth)

Hydraulic analysis was performed to determine flows that achieved a minimum depth of 0.1 m for the reach to allow passage of small bodied fish. Analysis produced a reach-level rating curve for Coolangatta (Figure 26) which indicated a discharge threshold of 0.04 m^3/sec, below which fish passage for small species was not possible, and above which passage was possible. Cross-section #2 (a shallow sand/gravel run) was the shallowest point on the reach affecting small fish passage.

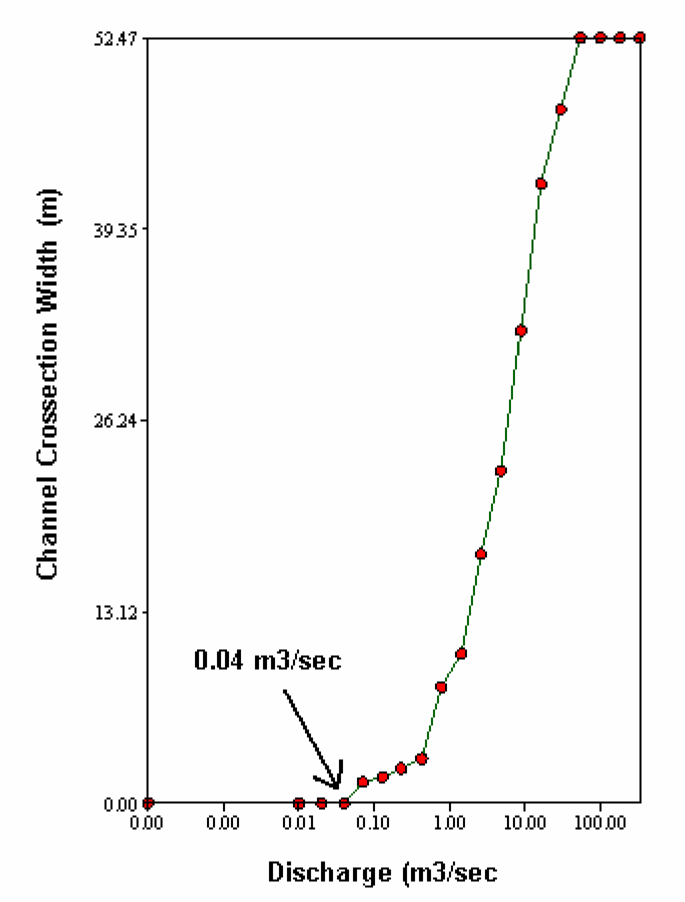


Figure 26. Reach level rating curve for passage of small bodied fish on the East Collie at Coolangatta, showing change in channel width (y-axis) with increasing discharge (m^3/sec) (x-axis). The small bodied fish passage criteria of 10 cm minimum threshold depth was achieved with a discharge of 0.04 m^3/sec, with cross-section #2 being the critical point for flow depth.

TSA was undertaken to assess frequency and duration of fish passage events for current and diversion scenarios to assess effects of diversions on small fish passage. Analysis indicated that, currently, there was an average of 2.9 events which provided fish passage in winter months, and each event lasted on average 69 days, with passage possible for approx 152 days each winter (i.e. small fish passage is possible for most of each winter) (Table 19). Flow diversions resulted in a progressive increase in the number of fish passage events, but a decrease in the mean duration of each event, indicating that reduced flows due to diversions resulted in more but shorter periods of fish passage. However, overall, there was only a minimal reduction in the total duration of fish passage each winter, from 151 days currently, to 149.6 days under 15 GL diversion scenario (Table 19). This can be seen in the cumulative probability plot for small fish passage, with a minimal loss of fish passage, even under the 15 GL diversion scenario (Figure 27).

Table 19. Summary statistics for frequency and duration of small fish passage events.

Statistics for winter fish passage	Current	Q5	Q10	Q15
Mean number of events	2.9	3.9	4.2	4.4
Mean duration of events (days)	69.2	56.2	51.5	44.6
Total duration of events (days)	151.8	150.0	149.8	149.6

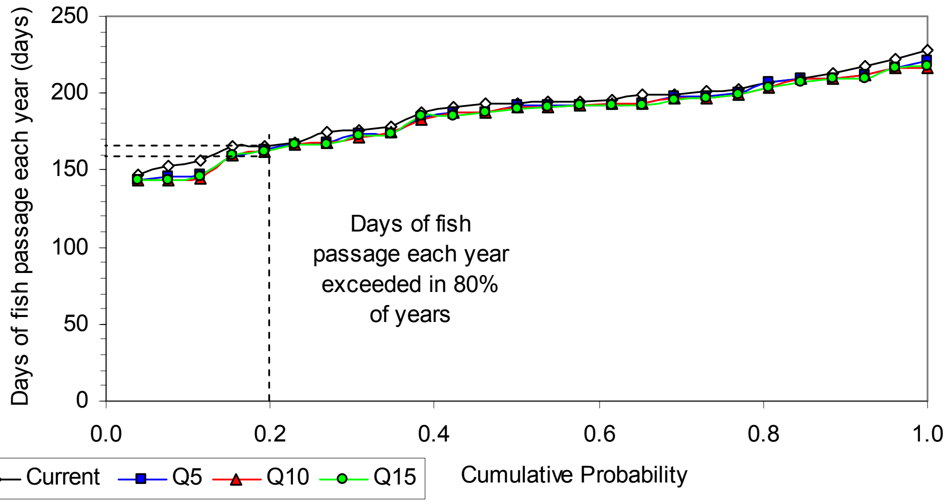


Figure 27. Cumulative probability plot for duration of fish passage for small bodied species of fish at Coolangatta, showing effects of each diversion scenario, with minimal effects at the 20% probability level (i.e. days of fish passage exceeded in 80% of years).

When broken into months, the total duration of fish passage events across years (Figure 28) shows the classic pattern for southwestern WA rivers, whereby fish passage commences in May, increasing through June and July to a peak in August, before declining through spring. Comparison of current duration to duration under each diversion scenario shows negligible effects of diversion on total number of days of fish passage possible each month (Figure 28, Table 20).

Table 20. Total duration (days) of fish passage events each month for small bodied species, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	94.2%	95.4%	94.2%
Feb	91.8%	89.8%	88.8%
Mar	91.2%	88.2%	89.7%
Apr	96.1%	95.1%	94.2%
May	92.1%	90.9%	90.9%
Jun	97.7%	96.8%	96.7%
Jul	99.6%	99.3%	99.2%
Aug	100.0%	100.0%	100.0%
Sep	99.6%	99.6%	99.7%
Oct	99.8%	99.8%	99.5%
Nov	97.5%	97.5%	97.5%
Dec	95.5%	95.5%	96.2%

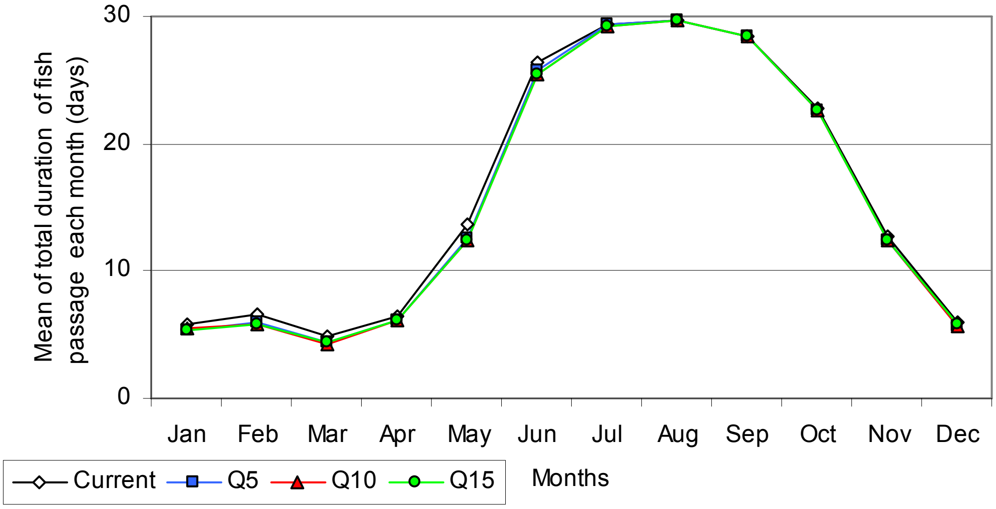


Figure 28. Mean of total duration of fish passage for each month for Coolangatta for current flows and the three diversion scenarios, where discharge exceeded 0.04 m^3/sec.

Similarly, the mean monthly duration of fish passage events across years shows negligible effects of diversion on mean duration of fish passage each month (Figure 29, Table 21). The greatest departure from current occurs in April, May and June, however, reductions in April are probably not particularly significant given natural patterns of fish passage. Even under maximum diversion (Q15 GL), reductions in May (35% lower than current), and June (25% lower than current), are probably acceptable, given they still provide substantial periods of fish passage for the month (Table 21).

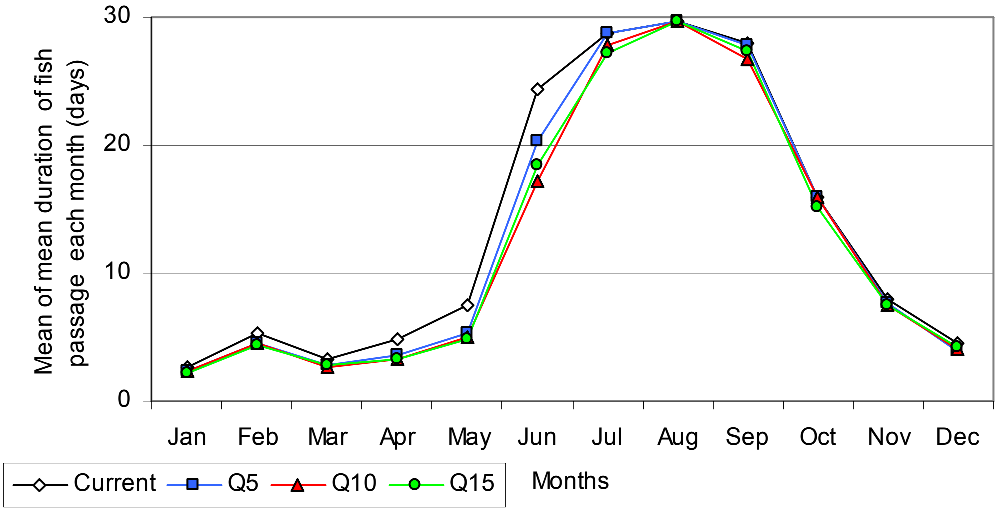


Figure 29. Mean duration of fish passage for each month for Coolangatta for current flows and the three diversion scenarios, where discharge exceeded 0.04 m^3/sec.

Analysis of total number of days of fish passage each year also shows that fish passage has been possible for the majority of each winter in most years, with on average a 2.5% reduction from current in the number of days passage was possible under the 15 GL diversion scenario (Figure 30).

Table 21. Mean duration (days) of fish passage each month for small bodied species, expressed as a percentage of current conditions, for 5 GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	83.7%	84.1%	81.7%
Feb	85.7%	83.8%	81.3%
Mar	86.8%	82.4%	85.7%
Apr	75.8%	70.3%	67.5%
May	70.9%	66.4%	65.4%
Jun	83.4%	70.3%	75.6%
Jul	99.5%	96.5%	94.4%
Aug	100.0%	100.0%	100.0%
Sep	99.2%	95.5%	97.7%
Oct	99.5%	99.5%	94.7%
Nov	96.5%	94.6%	94.2%
Dec	86.1%	88.5%	94.6%

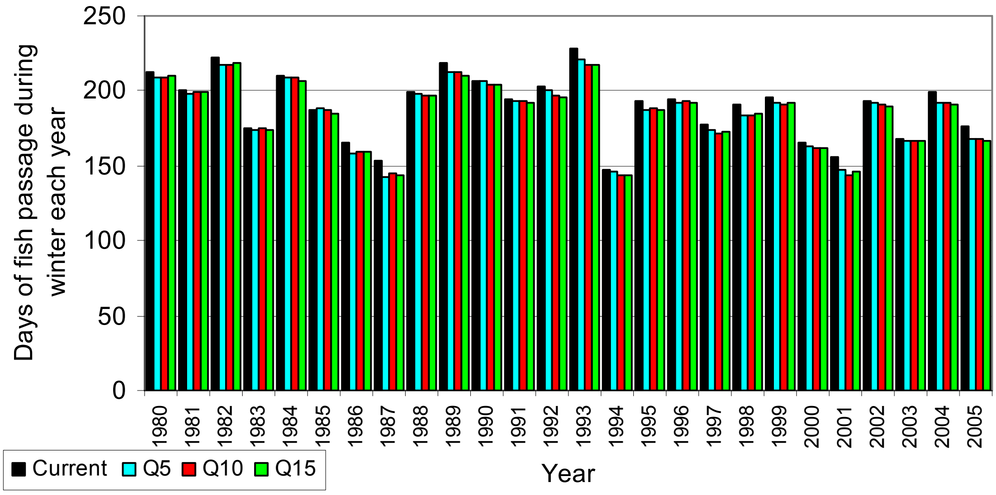


Figure 30. Total number of days of fish passage possible in winter for Coolangatta in each year of modelled flow record, where discharge exceeded 0.04 m^3/sec.

Therefore, passage for small bodied species of fish requires flows of 0.04 m^3/sec, and these occur through the majority of each winter. Implementation of diversion scenarios has little effect on fish passage for small bodied species, even at the highest diversions, with the only appreciable effects being loss of small events in May and June which results in intermittent fish passage (i.e. more events of shorter duration). There is also a marginal reduction in the total duration of fish passage in autumn/early winter, but the reduction is minor, and still leaves adequate duration of fish passage for small bodied species.

Fish passage for large bodied species (0.2 m minimum depth)

Hydraulic analysis was performed to determine flows that achieved a minimum depth of 0.2 m for Coolangatta reach to allow passage of large bodied species of fish. Analysis produced a reach-level rating curve for Coolangatta (Figure 31) which indicated a discharge threshold of 0.23 m^3/sec, below which fish passage was not possible, and above which passage for large bodied fish was possible. Cross-section #2 (a shallow sand/gravel run) was the shallowest point on the reach affecting fish passage.

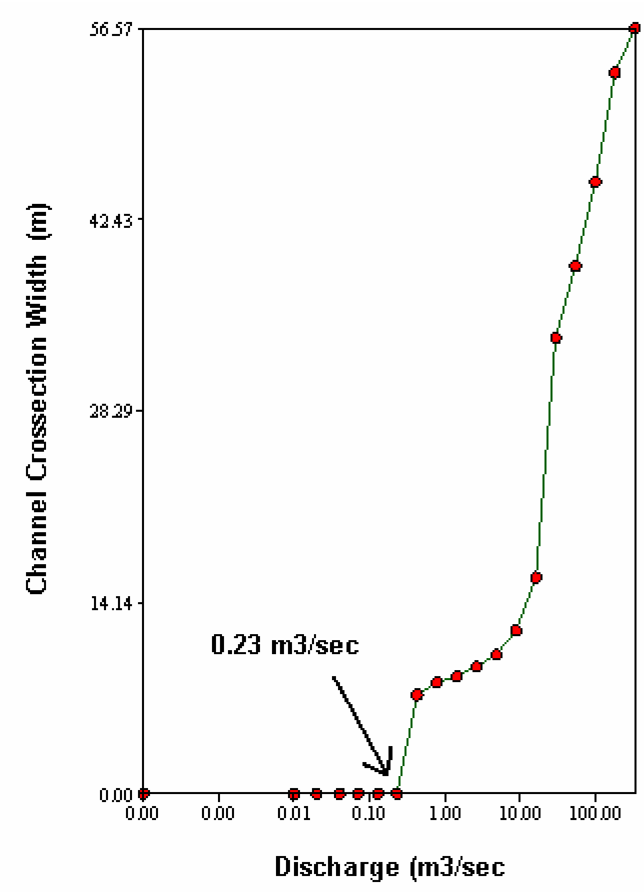


Figure 31. Reach level rating curve for passage of large bodied fish on the East Collie at Coolangatta, showing change in channel width (y-axis) with increasing discharge (m^3/sec) (x-axis). The large bodied fish passage criteria of 20 cm minimum threshold depth was achieved with a discharge of 0.23 m^3/sec, with cross-section #2 being the critical point for flow depth.

TSA was undertaken to assess frequency and duration of fish passage events for current and diversion scenarios to assess effects of diversions on large fish passage. Analysis indicated that, currently, there was an average of 5.4 events which provided fish passage in winter months, and each event lasted on average 34 days, with passage possible for approx 128 days each winter (i.e. large fish passage is possible for most of each winter) (Table 22). Flow diversions resulted in a progressive increase in the number of fish passage events, but a decrease in the mean duration of each event, indicating that reduced flows due to diversions resulted in more, but shorter periods of fish passage. However, overall, there was only a minimal reduction in the total duration of fish passage each winter, from 128 days currently, to 113 days under 15 GL diversion scenario, being a 11% reduction (Table 22). This can be seen in the cumulative probability plot for large fish passage, with a minimal loss of fish passage, even under the 15 GL diversion scenario (Figure 32).

Table 22. Summary statistics for frequency and duration of large fish passage events.

Statistics for winter fish passage	Current	Q5	Q10	Q15
Mean number of events	5.4	5.9	6.5	7.2
Mean duration of events (days)	34.4	28.7	26.2	22.7
Total duration of events (days)	127.8	119.6	115.9	113.2

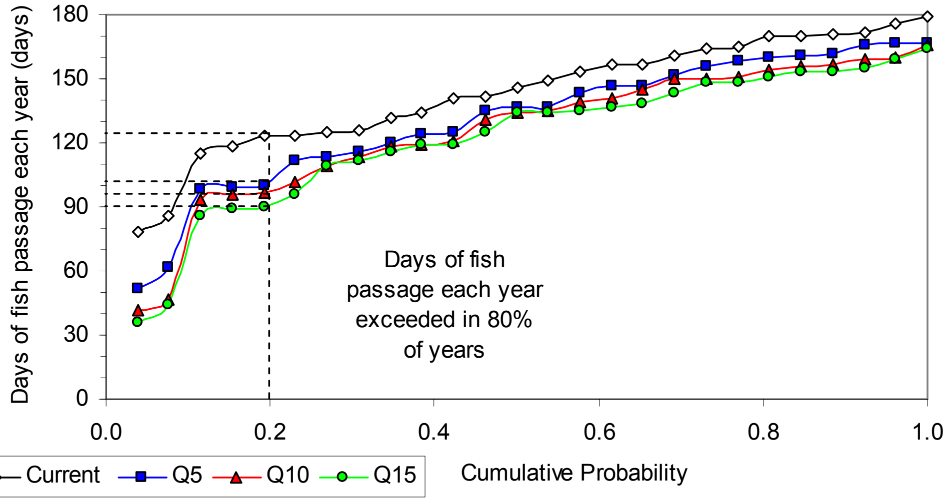


Figure 32. Cumulative probability plot for duration of fish passage for large bodied species of fish for Coolangatta, showing effects of each diversion scenario, with effects at the 20% probability level (i.e. days of fish passage exceeded in 80% of years).

When broken into months, the total duration of fish passage events each month (Figure 33) shows again the classic pattern for southwestern WA rivers, whereby passage for large fish species commences in May, increasing through June and July to a peak in August, before declining through spring. Comparison of current duration to duration under each diversion scenario shows a small effect of diversions on total number of days fish passage is possible each month (Figure 33, Table 23). Relatively large reductions in total days of fish passage occur under the greatest diversion (15 GL) in April (60% fewer days) and May (45% fewer days), however, the extent of fish passage in these months is small (4 – 5 days) as this is not when fish tend to move upstream (Figure 15). There were intermediate losses in extent of fish passage in June (up to 25% fewer days under 15 GL compared with current), but with lesser effects throughout the rest of winter (generally < 10% reduction in days of fish passage) (Figure 33; Table 23).

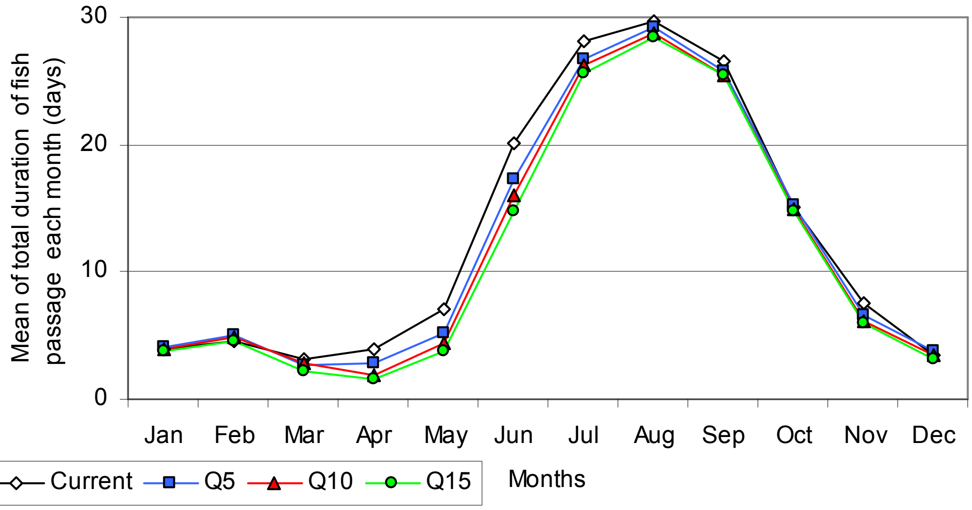


Figure 33. Total duration of fish passage for each month for Coolangatta for current flows and the three diversion scenarios, where discharge exceeded 0.23 m^3/sec.

Table 23. Total duration (days) of fish passage events each month for large bodied species, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	103.1%	96.9%	93.8%
Feb	111.1%	107.9%	101.6%
Mar	84.9%	89.1%	70.0%
Apr	70.8%	45.8%	37.5%
May	73.3%	61.4%	54.3%
Jun	86.2%	79.8%	74.1%
Jul	95.2%	93.3%	90.9%
Aug	98.2%	96.6%	96.0%
Sep	97.0%	95.8%	95.6%
Oct	101.5%	98.9%	98.3%
Nov	87.2%	82.8%	80.2%
Dec	106.1%	96.7%	87.9%

Mean duration of fish passage events each month shows similar effects of diversion (Figure 34, Table 24). The greatest departure from current occurs in April, May, June and July, however reductions in April and May are probably not particularly significant given natural patterns of fish passage. Under maximum diversion (15 GL), June (45% lower than current), and July (20% lower than current), show the greatest reduction in mean duration. Mean duration of each fish passage event in June drops from 13 to 7 days, and in July from 26 to 21 days. These reduced durations are still probably acceptable, given they still provide substantial periods of fish passage on each event and for the month in general (Table 24).

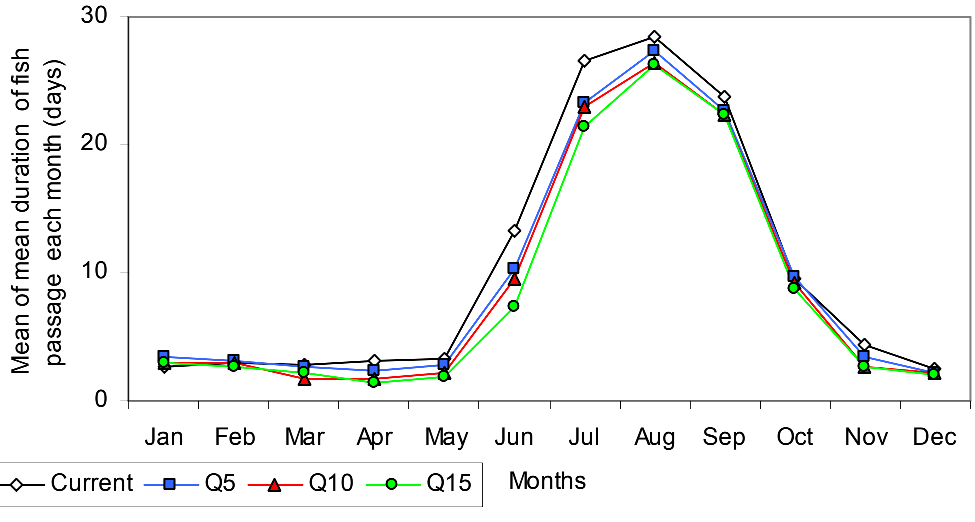


Figure 34. Mean duration of fish passage for each month for current flows and the three diversion scenarios, where discharge exceeded 0.23 m^3/sec.

Analysis of total number of days of fish passage each year also shows that fish passage has been possible for the majority of each winter in most years, with a 10 – 15% reduction from current in the number of days passage was possible under the 5 - 15 GL diversion scenarios (Figure 35). Generally large fish passage was possible for > 100 days per year, except in particularly dry years (i.e. 1987 and 2001), when passage was less than 100 days on current flows, but < 50 days under maximum diversion (15 GL).

Table 24. Mean duration (days) of fish passage each month for large bodied species, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	129.3%	116.5%	111.7%
Feb	102.4%	97.6%	90.5%
Mar	93.3%	59.5%	77.0%
Apr	78.7%	54.3%	43.4%
May	86.6%	65.1%	57.5%
Jun	77.9%	71.4%	55.0%
Jul	87.8%	86.4%	80.8%
Aug	96.0%	92.6%	92.2%
Sep	95.4%	94.0%	93.9%
Oct	101.4%	95.8%	90.5%
Nov	79.6%	61.9%	60.3%
Dec	92.1%	88.7%	84.5%

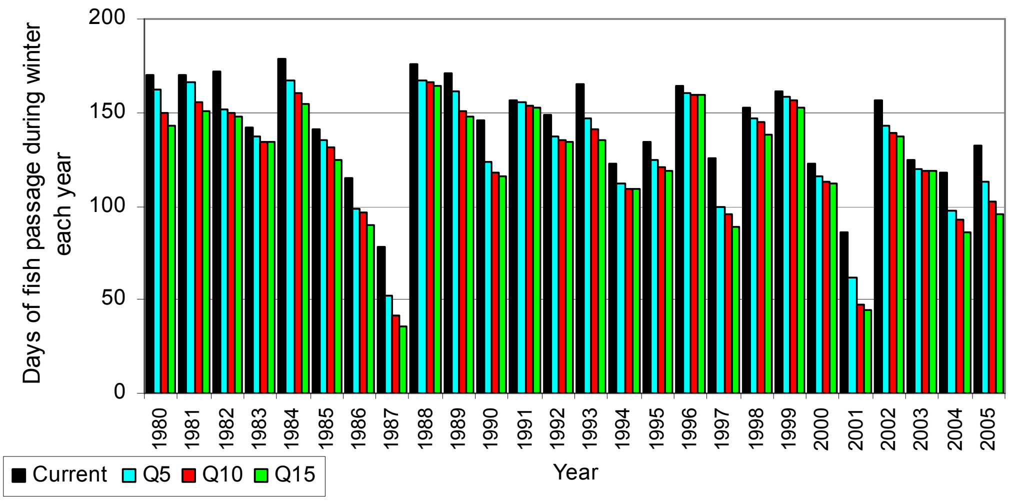


Figure 35. Total number of days of fish passage possible in winter in each year of modelled flow record for Coolangatta, where discharge exceeded 0.23 m^3/sec.

As with passage for small bodied species of fish, passage for large fish species, which requires flows of 0.23 m^3/sec, occurs through the majority of each winter. Implementation of diversion scenarios has a greater effect on large fish passage than on small fish passage, but mainly only at the highest diversion and in June. But even at the highest diversions, the reductions would not be considered major, with long periods of fish passage available for large bodied species.

10.2.1.4 Active Channel Flows

Active channel or channel forming flows typically occur on a 1:2 to 1:3 year frequency in south-west river systems. These flows help maintain the shape of the channel by mobilising sediment, scouring pools and preventing riparian vegetation encroaching into the channel. These flows also inundate trailing vegetation to provide cover and spawning habitat for fish, and provide inputs of autochthonous (algal production) and allochthonous energy (leaf litter/detritus) from banks. Based on channel surveys, it was considered that active channel flows would be sufficient to provide trailing macrophytes as fish habitat, and redistribute leaf litter (carbon) off benches.

The current active channel was determined from cross-sectional surveys (*i.e.* the level on the banks above which vegetation is stable and below which the bank is eroding/bare, and without extensive riparian vegetative growth). Using all channel cross-sections, the average depth of the bed to active channel height, taken as the deepest part of the channel (thalweg depth) to the level approximated as active channel was 1.64 m. Using this stage height (*i.e.* a thalweg depth of 1.64 m), the reach level rating curve produced by hydraulic analysis calculated a discharge of 3.0 m^3/sec to achieve active channel flows (Figure 36).

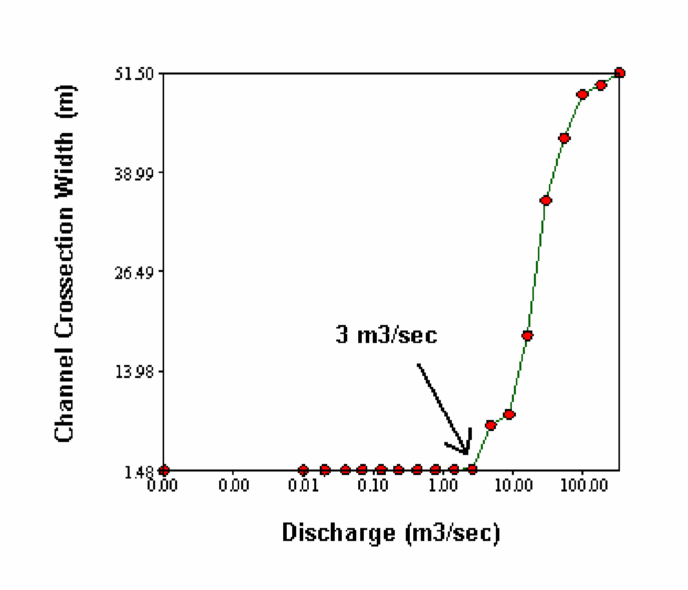


Figure 36. Reach level rating curve for discharge to attain active channel stage height for Coolangatta, with threshold flow of 3.0 m^3/sec required to reach active channel flow.

TSA of the active channel flow event determined that currently in winter there was on average 9 events of 3.0 m^3/sec or greater that reach active channel stage height, and under the 5 GL, 10 GL and 15 GL scenarios this did not change (Table 25). Essentially, the diversion scenarios have negligible effect on the annual frequency of flow events to reach active channel stage height. Each event lasted on average 6.0 days, again, with little change under the 5 GL, 10 GL or 15 GL scenarios (Table 25). Similarly, the total duration of active channel flows each winter showed little change from current under each diversion scenario (Table 25).

This can be seen in the cumulative probability plot for active channel flows, with a negligible loss of these flows, even under the 15 GL diversion scenario (Figure 37).

Table 25. Summary statistics for frequency and duration of active channel flow events.

Statistics for active channel flows	Current	Q5	Q10	Q15
Mean number of events	8.8	8.8	8.9	8.8
Mean duration of events (days)	6.2	6.1	5.8	5.7
Total duration of events (days)	45.5	44.0	42.2	41.0

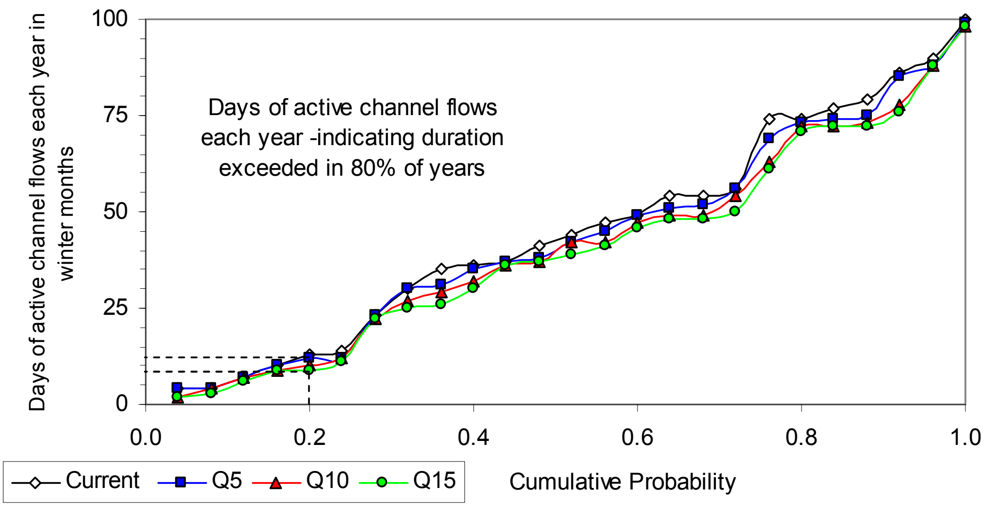


Figure 37. Cumulative probability plot for duration of active channel flows for Coolangatta, showing effects of each diversion scenario, with effects at the 20% probability level (i.e. days of active channel flows exceeded in 80% of years).

When broken into months, the mean number and mean total duration of active channel flows each month under current flows (Figure 38) increased through June and July to a peak in August, before declining through spring. Comparison of current duration with each diversion scenario shows a small effect of diversions on mean number of events (Figure 38) and total number of events each month (Figure 39, Table 26), with generally < 10% reduction in any month, even under the highest diversion (Figure 39, Table 26).

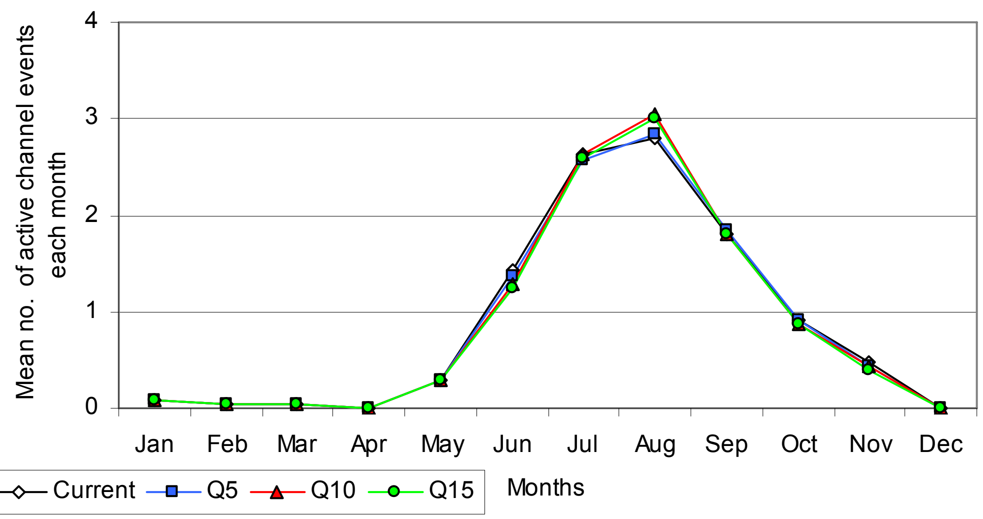


Figure 38. Mean number of flow events each month in winter for Coolangatta to reach active channel stage height, where discharge exceeded 3.0 m^3/sec.

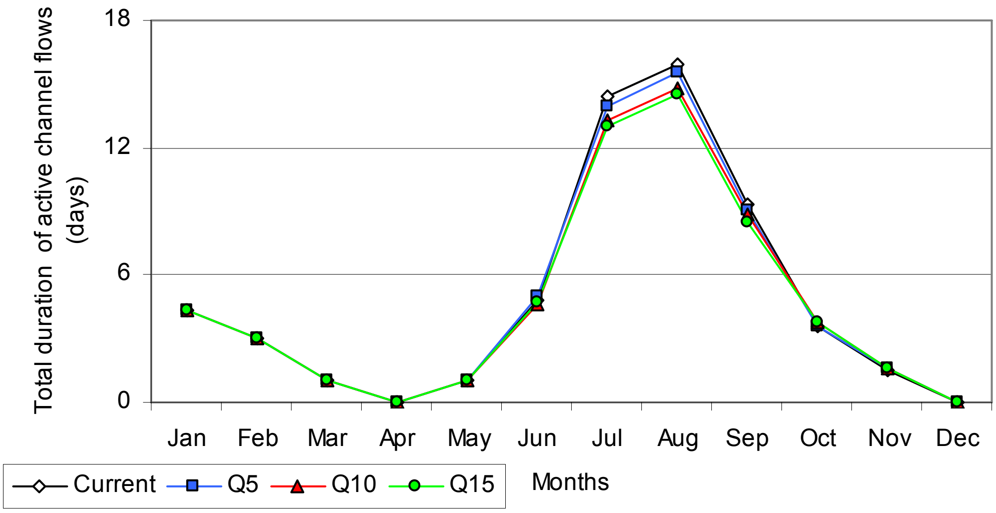


Figure 39. Total duration of flow events each month in winter to reach active channel stage height for Coolangatta , where discharge exceeded 3.0 m^3/sec

Table 26. Total duration (days) of active channel flows each month, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	100.0%	100.0%	100.0%
Feb	100.0%	100.0%	100.0%
Mar	100.0%	100.0%	100.0%
Apr	-	-	-
May	100.0%	100.0%	100.0%
Jun	103.3%	94.8%	96.8%
Jul	96.5%	92.2%	90.2%
Aug	97.2%	93.0%	91.0%
Sep	97.3%	95.1%	91.5%
Oct	100.0%	104.8%	103.0%
Nov	103.7%	103.7%	108.3%
Dec	-	-	-

Mean duration of active channel flows each month showed similar effects of diversion as above (Figure 40, Table 27). The greatest departure from current occurs in August, but the reduction is relatively minor. Some summer events were evident as a result of summer rainfall events, but these had low frequency of recurrence.

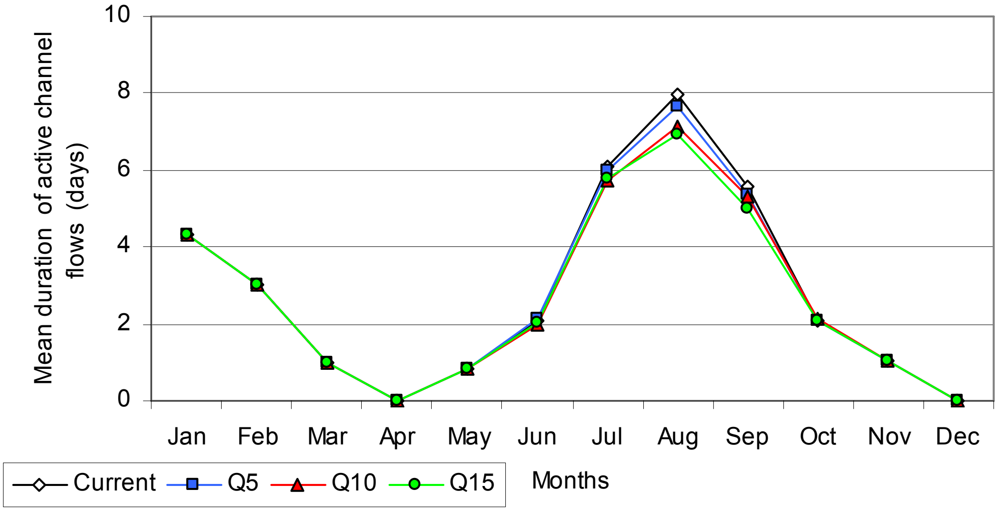


Figure 40. Mean duration of flow events each month in winter to reach active channel stage height for Coolangatta, where discharge exceeded 3.0 m^3/sec.

Table 27. Mean duration (days) of active channel flows each month, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	100.0%	100.0%	100.0%
Feb	100.0%	100.0%	100.0%
Mar	100.0%	100.0%	100.0%
Apr	-	-	-
May	100.0%	100.0%	100.0%
Jun	102.4%	96.5%	98.0%
Jul	98.6%	93.8%	95.0%
Aug	95.8%	89.2%	86.8%
Sep	97.0%	96.1%	89.6%
Oct	100.0%	103.5%	100.3%
Nov	100.6%	100.6%	101.1%
Dec	-	-	-

Analysis of total number of days of active channel flows each year shows high inter-annual variability, with total duration ranging from ~ 100 days in wet winters to 2 days in dry years. This indicates that in some years the duration of active channel flows is greatly reduced due to low rainfall. Generally, even the maximum diversion (15GL) has little influence on the total duration of active channel flows each year, and has minimal effect relative to inter-annual variability (Figure 41).

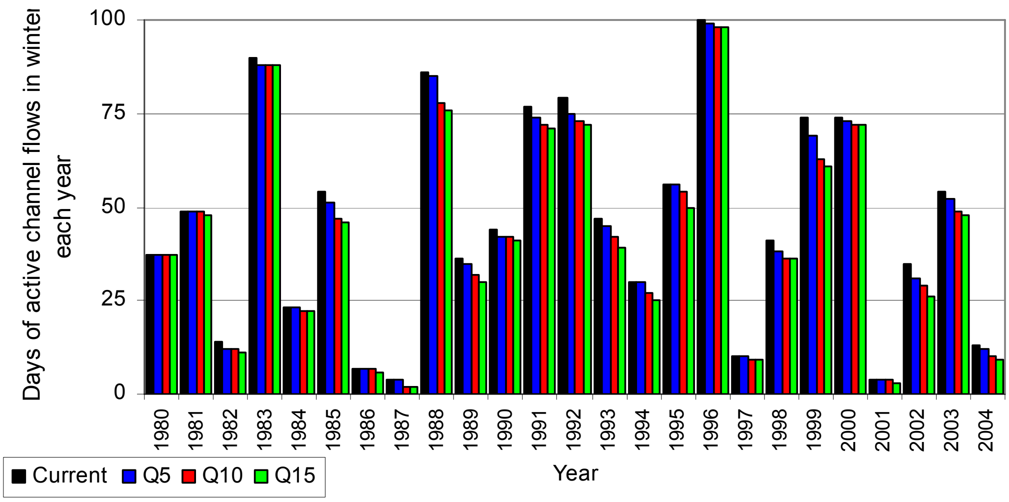


Figure 41. Total number of days of active channel flows in winter in each year of modelled flow record for Coolangatta, where discharge exceeded 3.0 m^3/sec.

Active channel flows of 3.0 m^3/sec do not appear to be greatly affected by any of the diversion scenarios, with minimal reductions in frequency and duration in winter months. This likely reflects the fact that active channel flows occur under relatively high flow events, and any diversions are a relatively small proportion of these flows. As a result, changes are minimal. Compared with well vegetated, undisturbed systems in SW WA, active channel flows for Coolangatta Reach occur at a relatively high frequency and duration. This may reflect inaccuracies of the uncalibrated hec-ras model, or alternatively reflect the disturbed nature of the catchment, with greater and faster run-off as a result of catchment clearing.

10.2.1.5 Flows to inundate medium elevation benches

The objective of winter flows to inundate benches is to provide inputs of allochthonous energy (leaf litter/detritus) from benches into the creek. Flows to occasionally inundate medium elevation benches were determined. Using channel cross-sections with medium level benches (1, 2, 3, 8 and 13) (Figure 42), a rating curve was generated to show the discharge at which benches became inundated, using a rule based on lateral gradient and channel depth. The reach level rating curve produced by hydraulic analysis calculated a discharge of 9 m^3/sec to achieve inundation of medium benches (Figure 43).

TSA of this flow event determined that currently in winter there was on average 2.84 events of 9 m^3/sec or greater to inundate medium benches, and under the 5 GL, 10 GL and 15 GL scenarios this decreased to 2.80, 2.76 and 2.76 events respectively (Figure 44). Essentially, the diversion scenarios have negligible effect on the frequency of flow events to inundate medium benches. Each event lasted on average 3.0 days under the current regime, and this duration did not change appreciably under the 5 GL, 10 GL or 15 GL scenarios (Figure 44). The total duration of bench inundation flows each winter, under current flows, was 12.9 days, decreasing only marginally under the diversion scenarios, to 12.8, 12.7 ands 12.6 days for 5 GL, 10 GL and 15 GL scenarios, respectively (Figure 45 & 46).

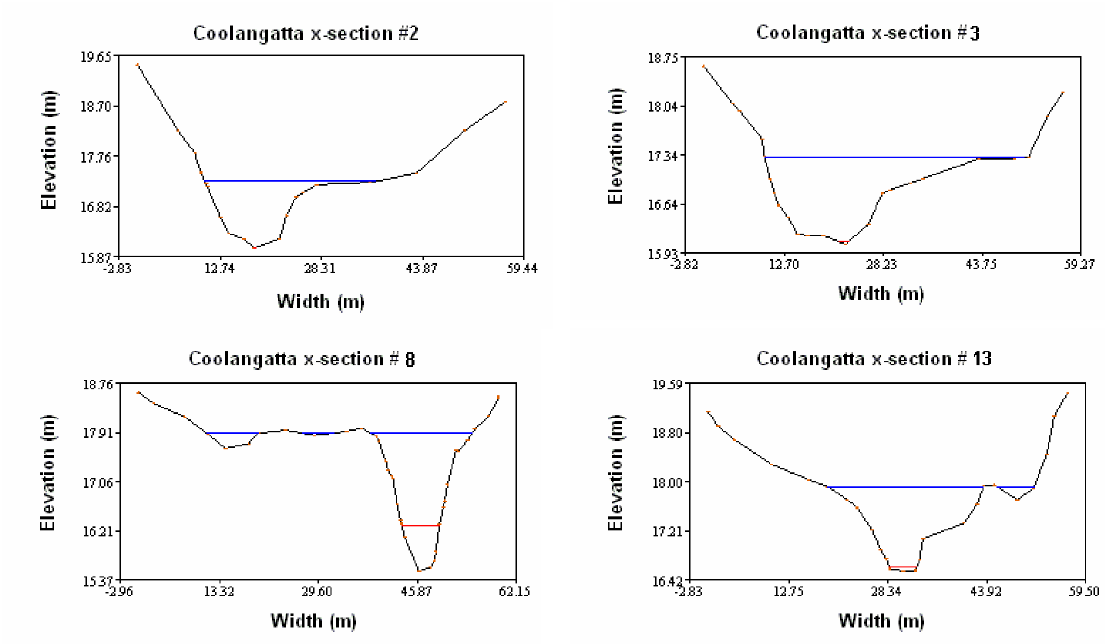


Figure 42. Examples of cross-sections with medium-elevation benches, used to determine habitat rating curve for flows to inundate these benches**.**

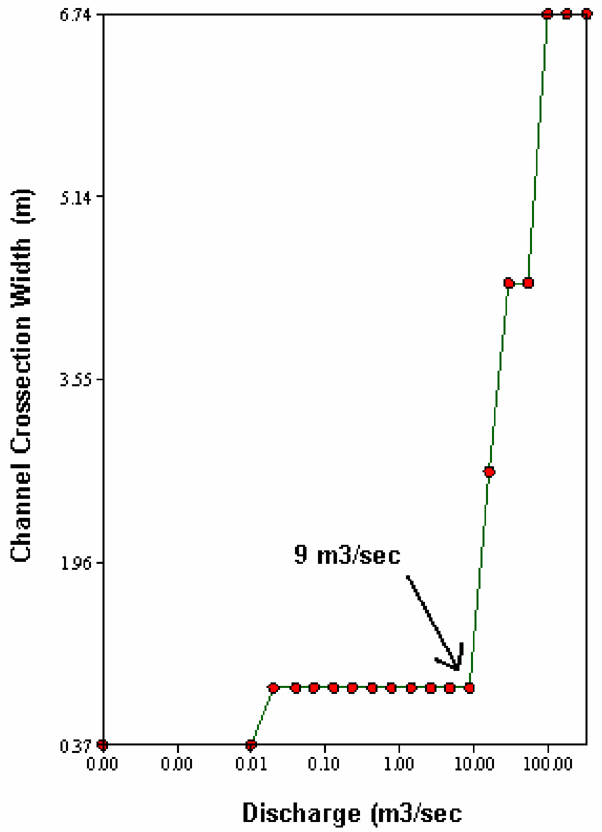


Figure 43. Reach level rating curve for flows to inundate medium benches on Coolangatta reach, showing change in channel width (y-axis) with increasing discharge (m^3/sec) (x-axis). The rule of lateral gradient < 0.01 m/m was used to identify benches, and the rating curve identified a discharge of 9 m^3/sec.

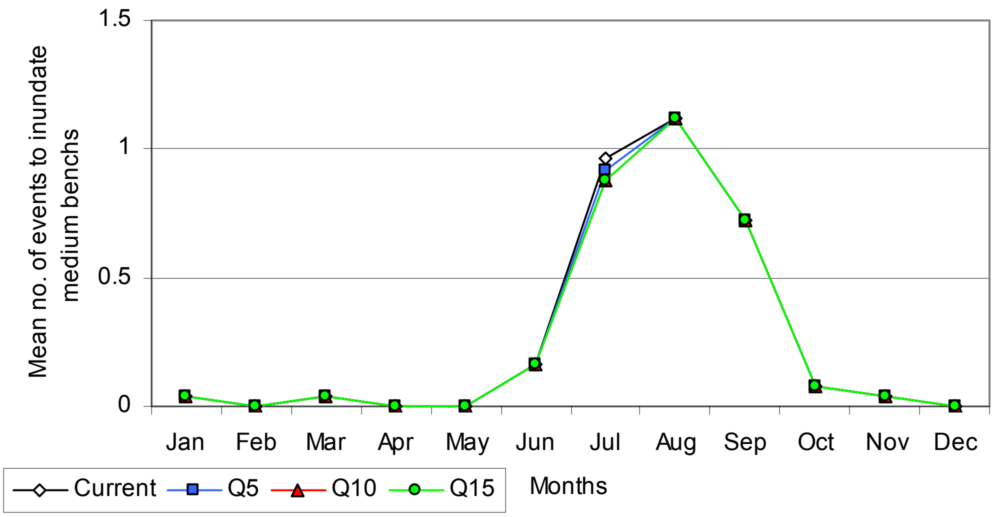


Figure 44. Mean number of flow events each month in winter to inundate medium benches for Coolangatta, where discharge exceeded 9 m^3/sec.

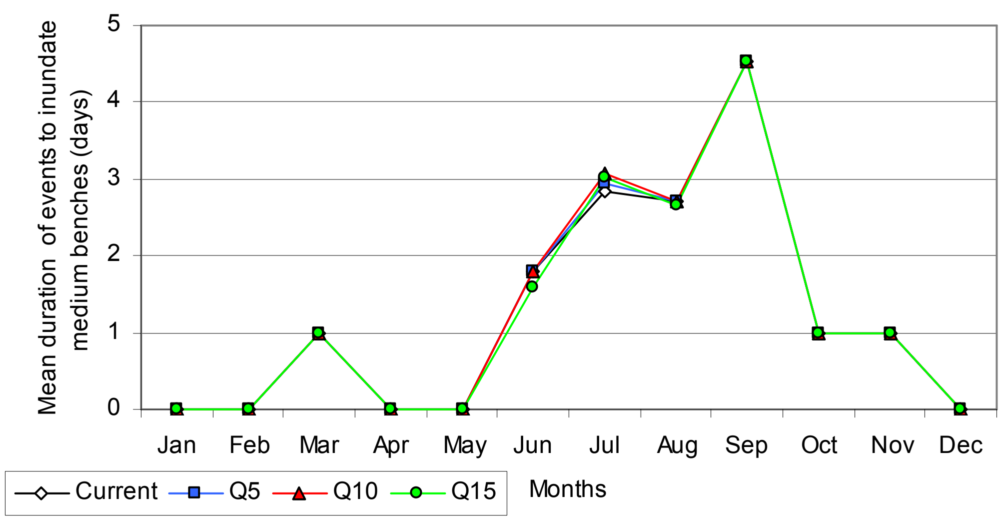


Figure 45. Mean duration of each flow event each month sufficient to inundate medium benches for Coolangatta, where discharge exceeded 9 m^3/sec.

The small effect of the diversion scenarios on frequency and duration of flows required to inundate medium benches is evident in the flow duration curve for winter months, with only a negligible effect of the diversions on flows compared with current flows (Figure 47).

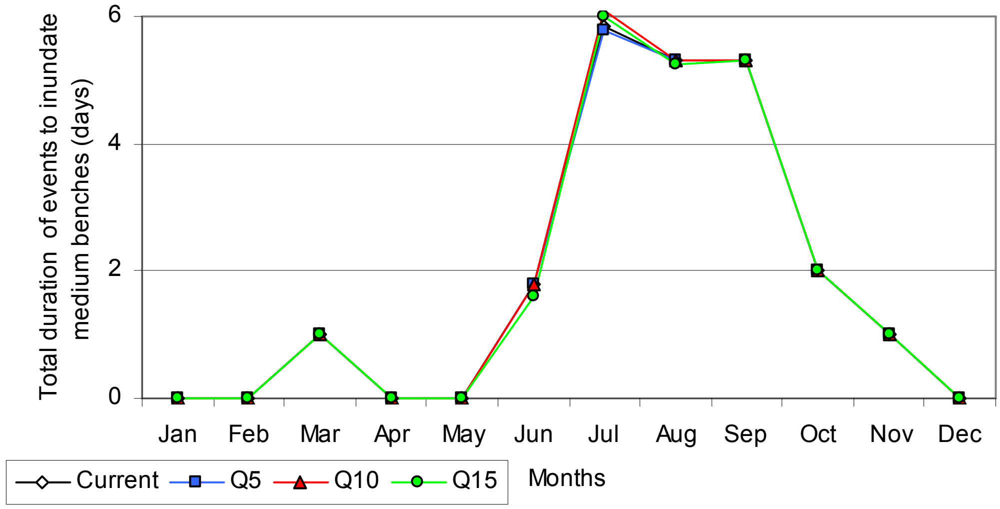


Figure 46. Total duration of flows each month sufficient to inundate medium benches for Coolangatta, where discharge exceeded 9 m^3/sec.

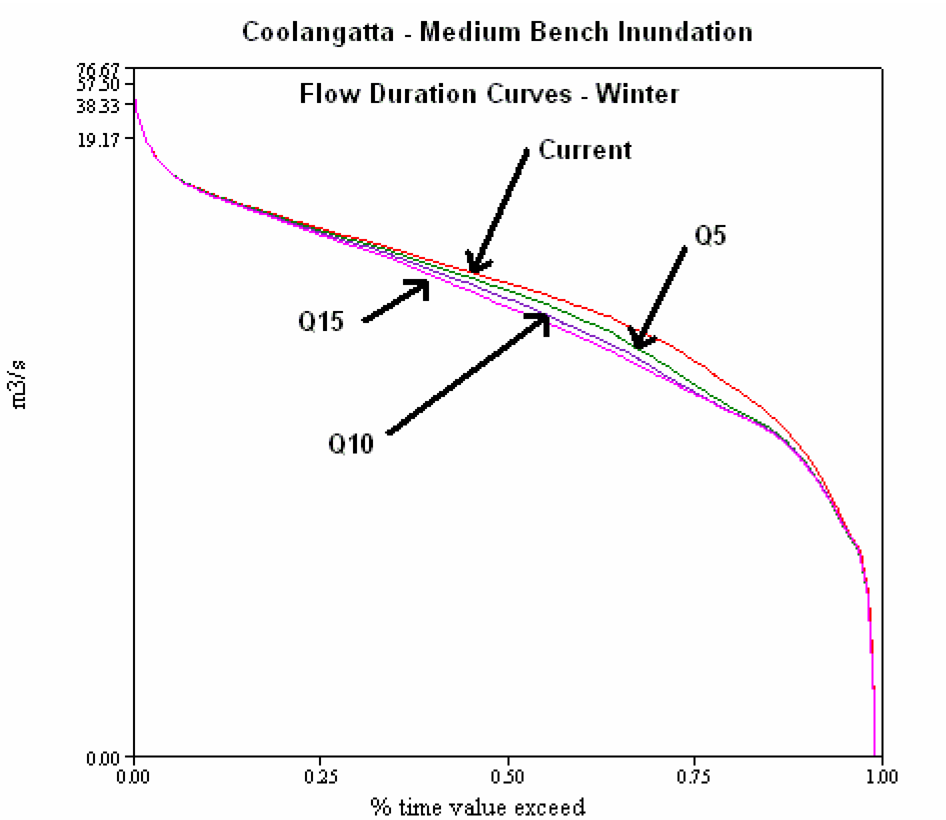


Figure 47. Flow duration curves for winter months, showing minimal influence of diversion scenarios on flows sufficient to inundate medium benches for Coolangatta, where discharge exceeds 9 m^3/sec.

10.2.1.6 Flows to connect off-river wetlands/sumps

The East Collie River at Coolangatta is set in a relatively steep-sided valley, and does not have a floodplain, in the traditional sense of a broad, flat area regularly inundated. However, there was a high flow channel on the lower end of the Coolangatta Reach which consisted of a series of small interconnected sumps/wetlands which likely contained water well into spring (based on water lines, vegetation distribution etc), and so likely provided spawning/breeding/nursery habitat for frogs and tortoise, as well as opportunistic habitat for macroinvertebrates, and feeding habitat for waterbirds (ducks, egrets, herons). The point where the main channel evulsed into the high flow channel was on cross-section #9[6] (Figure 48). HA and TSA were performed to determine the flows required to inundate this channel. RAP indicated that the discharge required to provide flows to this channel was 9 m^3/sec. This coincidentally was the same flow required to achieve inundation of medium elevation benches (above). Therefore, results from TSA for inundation of medium benches also apply to flows to connect this off-river channel and associated sumps/wetlands.

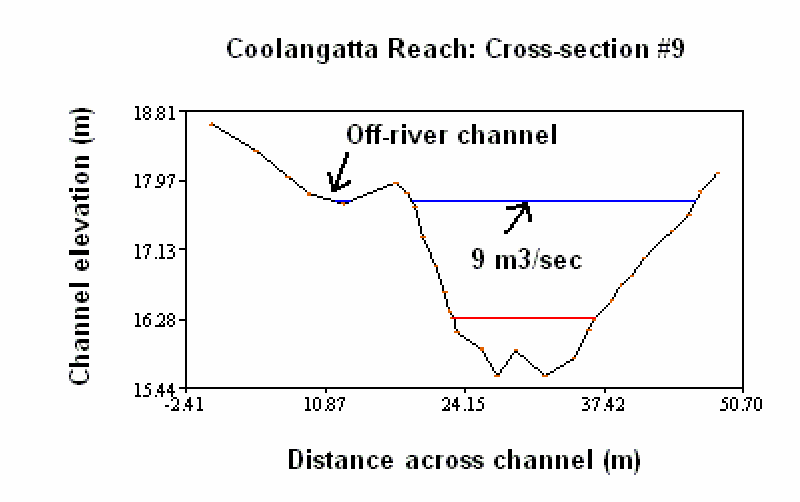


Figure 48. Flow magnitude required to inundate off-river channel and associated sumps and wetlands for Coolangatta, with evulsion point located on cross-section #9, showing water level sufficient to achieve this stage height (9 m^3/sec).

10.2.1.7 Winter High Flows

The objective of winter high flows was to exceed top of bank (TOB) stage heights to start inundation of the floodplain and associated riparian vegetation. The floodplain often supports shallow wetland/sump areas that are likely important feeding and nursery areas for frogs and provide habitat in which juvenile stages with poor swimming ability may avoid high flows. Overbank flows are required to inundate and recharge these wetlands. Similarly, riparian vegetation on the floodplain (*i.e. Eucalyptus rudis* and paperbarks) also requires occasional inundation to a.) disperse seed, and b.) assist seed set, and soak soil profiles to promote successful germination. To achieve the above objectives, a flow above TOB is required. These flows usually occur as a result of larger rainfall events, principally in winter, and occur infrequently and for short durations.

The current TOB stage height was determined from cross-sectional surveys as the level on the cross-sections above which the bank profile started to decline onto a flatter floodplain (i.e. reduced lateral gradient). This can usually be detected in the HA habitat rating curves by a sudden increase in habitat area when flows reach the TOB level. HA demonstrated that the change in area of this habitat occurred at approx 9 m^3/sec, which was the same as that determined for the inundation of lateral wetlands/sumps (see above), which was also the same

[6] NB this cross-section was specifically located at this point for this purpose

flow required to achieve inundation of medium elevation benches. Therefore, results from TSA for inundation of medium benches also apply to flows to achieve floodplain inundation.

10.2.1.8 Pool Maintenance

Coolangatta reach contains long, permanent pools surrounded by a water course that reduces to a very low baseflow. These pools provide critical dry season habitat/refuge for water dependent ecological values. Maintenance of the quantity and quality of water in these pools is therefore critical to sustain the ecological values which are dependent on them (i.e. macroinvertebrates, fish and tortoise). Water levels in pools in seasonally flowing systems are often driven by groundwater influence, either in the form of seeps and springs or through direct surface expression of the groundwater. Determination of riverine EWRs to maintain these pools in summer in a seasonally flowing system is not possible however, consideration should be given to the role groundwater inflows/levels play in maintaining these pools. As has been shown in other studies (A.W. Storey unpub. dat.), there will be a relationship between diversity of ecological values using these pools and water depth. As the pools become shallower, they will start to loose diversity. This is particularly true for fish, and likely also applies to tortoise. At this stage the relationship between depth and diversity is unknown for these pools. However, under a precautionary principle, water depth should be maintained at current levels.

10.2.2 Buckingham reach

10.2.2.1 Summer baseflow to inundate riffles

Analysis of modelled flow data for the East Collie at Buckingham indicated the system was perennial but becoming seasonal in recent years; decreasing minimum flows in early years of the modelled flow record (1980 to 1997), and then became intermittent in recent years, ceasing to flow in summer months for 70, 97, 85 & 134 days in 2002, 2003, 2004 and 2005 respectively. It was therefore appropriate to determine a summer baseflow to maintain extent, depth and duration of inundation of gravel/cobble runs as biodiversity 'hotspots' for macroinvertebrates, allowing for periods of zero flow. In perennial systems, a low-flow summer baseflow objective for maintenance of riffles is usually taken as flows to achieve 50% lateral coverage of riffles to 0.05 m average depth. However, because the East Collie at Buckingham flows intermittently during summer months in some years, and flows will recede in summer, especially in dry years, it is likely that the 50% lateral coverage will be seldom met. Therefore, as applied to Coolangatta reach, this riffle inundation rule was modified to 25% coverage to 0.05 m depth. Rules to meet riffle objectives are:

- inundation of riffles to a minimum average depth of 5 cm with 25 % lateral coverage of riffle cross-section.

Hydraulic analysis was used to calculate discharge required to achieve a mean depth of 0.05 m across gravel/cobble riffles whilst maintaining 25% lateral overage of the habitat. Riffles were on cross-sections 6 (2.5 m width), 8 (3.6 m), 9 (9.5 m), 11 (5.5 m), 12 (7.6 m), & 15 (8.3 m). At the time of survey there was no flow along the study reach, with low water levels in pools exposing riffles between the pools which were assumed to be control points for upstream pools. However, after the channel survey data had been loaded into the each hec-ras model, it was evident that the run/riffle on cross-sections 11 & 12 were not control points for the upstream pool, but would be inundated within the pools located on either side before there was flow along the reach. Therefore, these 'riffle' cross-sections were excluded in the calculations (i.e. they would not act as riffles when the river was flowing). The mean width of the remaining riffle zones (6, 8, 9, & 15), was 5.98 m. Discharge required to achieve 0.05 m average depth with 25% lateral coverage (1.49

m) of channel cross-section inundated for those cross-sections was then read from the rating curve as 0.04 m^3/sec (Figure 49). It is recommended that this threshold flow is maintained as a minimum baseflow in summer at the current frequency and duration, allowing for periods of zero flow, to maintain ecological values dependent on this flow regime in riffles.

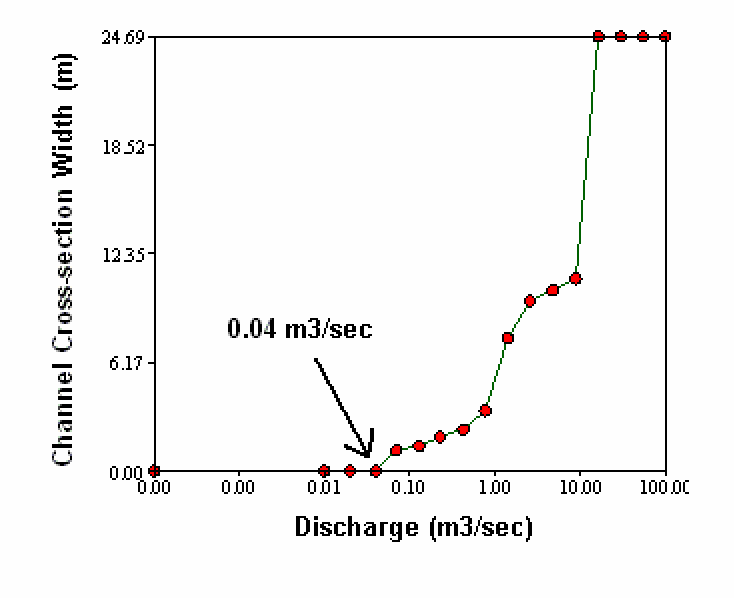


Figure 49. Rating curve for inundation of gravel/cobble riffles (cross-sections 6, 8, 9 & 15) to 5 cm average depth with 25% habitat coverage on the cross-section (= 1.49 m cross-section total width) on Buckingham Reach, East Collie. Change in channel width (y-axis) with increasing discharge (m^3/sec) (x-axis) is shown. The threshold criteria of 0.05 m average depth with 25% coverage of riffles was achieved with a discharge of approx. 0.04 m^3/sec.

TSA demonstrated that this base flow currently occurs for an average of 35 days each summer, with on average 7 periods of approx 29 days in which flows drop below 0.04 m^3/sec (Table 28). It should be noted that these statistics cover the whole modelled flow period (1980 – 2005), and these statistics would vary for the latter half of the period when flows ceased each summer. However, these statistics did not vary under each diversion scenario (Table 28), indicating no influence of diversion scenarios on summer flows. This is not unexpected, since the present rule for commencing diversions is when flows exceed 5 ML per day, which equates to 0.057 m^3/sec, which is higher than the estimated summer baseflow of 0.04 m^3/sec to maintain 25% lateral coverage of riffles.

Table 28. Summary statistics for frequency and duration of flows ≥summer base flows (0.04 m^3/sec) to inundate gravel/cobble riffles to 25% lateral coverage.

Statistics for summer baseflows	Current	Q5	Q10	Q15
Mean number of events	7.0	7.0	7.0	7.0
Mean duration of events (days)	4.9	4.9	4.9	4.9
Total duration of events (days)	34.7	34.7	34.7	34.7
Mean period between events	28.9	28.9	28.9	28.9
Total period between events	138.2	138.2	138.2	138.2

As noted above, the East Collie at Buckingham ceases to flow in summer months, particularly in recent years. These periods of zero flow will influence the ecology of the system. It is therefore important to assess the influence of diversion scenarios on frequency and duration of zero flow days. Analysis of zero flow days in each month in summer again shows no effect of diversion scenarios on duration of zero flow days (Figure 50). The lack of any effect likely reflects that summer flows are generally below the threshold flow for diversions to be initiated.

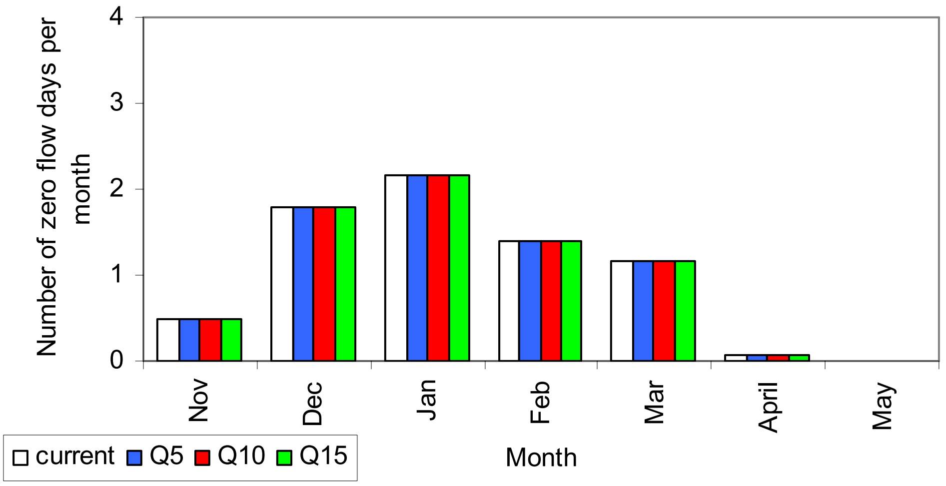


Figure 50. Number of zero flow days in each month in summer under current flows and each diversion scenario (1980 – 2005 modelled flow record) for Buckingham.

10.2.2.2 Winter baseflows to inundate riffles

The objective of winter baseflows is to maintain inundation of gravel/cobble riffles as biodiversity 'hotspots' for macroinvertebrates, and provide flow connectivity for downstream energy transfer. The Buckingham Reach of the East Collie River was characterised by broad, low profile gravel/cobble riffles between longer pools. Although the quality and heterogeneity of these riffles was relatively low compared with less disturbed systems, it is assumed these areas will function as riffle zones in that they will have higher velocity and diversity of flows relative to pools in winter, but may not support such highly diverse macroinvertebrate fauna compared with typical, more heterogeneous cobble/pebble/gravel riffles in forested streams.

Based on channel morphology, it was considered that the winter baseflow objectives collectively would be provided for by the minimum depth requirement normally applied to riffle zones, being:

- inundation of riffles by a minimum average depth of 5 cm (0.05 m) with 100 % lateral coverage of the riffle component of the specific cross-sections.

Hydraulic analysis was used to calculate discharge required to achieve a mean depth of 0.05 m across riffles whilst maintaining 100% coverage of the habitat. Riffles were located on cross-sections 6 (2.5 m width), 8 (3.6 m), 9 (9.5 m), 11 (5.5 m), 12 (7.6 m), & 15 (8.3 m). However, at the time of channel survey there was no flow along the study reach, and low water levels in pools had exposed riffles between the pools, which were assumed to be control points. After the Buckingham channel survey data had been loaded into the hec-ras model, it was evident that the riffle on cross-sections 11 & 12 were not control points for the upstream pool, but would be inundated within the pools located on either side before there was flow along the reach. Therefore, these 'riffle' cross-sections were excluded in the calculations (i.e. they would not act as riffles when the river was flowing, and would overly influence calculation of average depth). The mean width of the remaining riffle zones (6, 8, 9, & 15), was 5.98 m. Discharge required to achieve 0.05 m average depth with 5.98 m of channel cross-section inundated for those cross-sections was then read from the rating curve as 0.07 m^3/sec (Figure 51). It is recommended that this threshold flow is maintained as a minimum baseflow throughout winter to maintain ecological values dependent on riffles. The transition from summer to winter baseflows is dependent on annual rainfall pattern, but in general terms should occur in the shoulder months of May/June, with a recession back to summer baseflows in October/November.

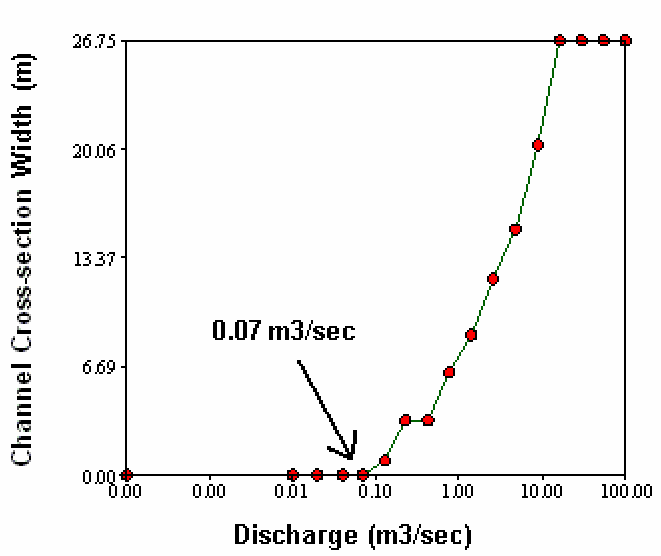


Figure 51. Rating curve for inundation riffles (cross-sections 6, 8, 9 & 15) to 5 cm average depth with 100% habitat coverage on the cross-section (= 5.98 m cross-section total width) on Buckingham Reach, East Collie. Change in channel width (y-axis) with increasing discharge (m^3/sec) (x-axis) is shown. The threshold criteria of 0.05 m average depth with 100% coverage of riffles was achieved with a discharge of 0.07 m^3/sec.

TSA performed on this flow event demonstrated that this base flow was currently achieved throughout the majority of winter in all years (Table 29). Currently, the baseflow occurs for an average of 139 days each winter (maximum of 170 days per winter 'season'), with on average 5 periods of approx 5 days in which flows drop below 0.07 m^3/sec, but does not dry. TSA on the diversion scenarios showed a progressive reduction in the duration of winter riffle inundation under each diversion scenario (Table 29). Diversions were expected to affect riffle inundation at Buckingham since the present rule for commencing diversions each winter is when flows exceed 5 ML per day, which equates to 0.057 m^3/sec, which is less than the estimated winter baseflow to inundate riffles (0.07 m^3/sec) (i.e. diversions will commence at low flows). Therefore, diversions will commence before the riffle inundation rule is met. The effects of diversions on winter baseflows can be seen in the cumulative probability plot for gravel/cobble riffle inundation, with a progressive loss of inundation with each diversion scenario. Under the 15 GL diversion scenario, riffle inundation would be achieved for 70 days in 80% of years compared with ~ 125 days under current flows (Figure 52).

Table 29. Summary statistics for frequency and duration of winter baseflows to inundate riffles.

Statistics for winter baseflows	Current	Q5	Q10	Q15
Mean number of events	5.1	6.8	7.8	8.9
Mean duration of events (days)	39.3	22.7	17.6	13.7
Total duration of events (days)	139.3	118.8	104.0	94.0
Mean period between events	5.2	5.5	5.9	5.9

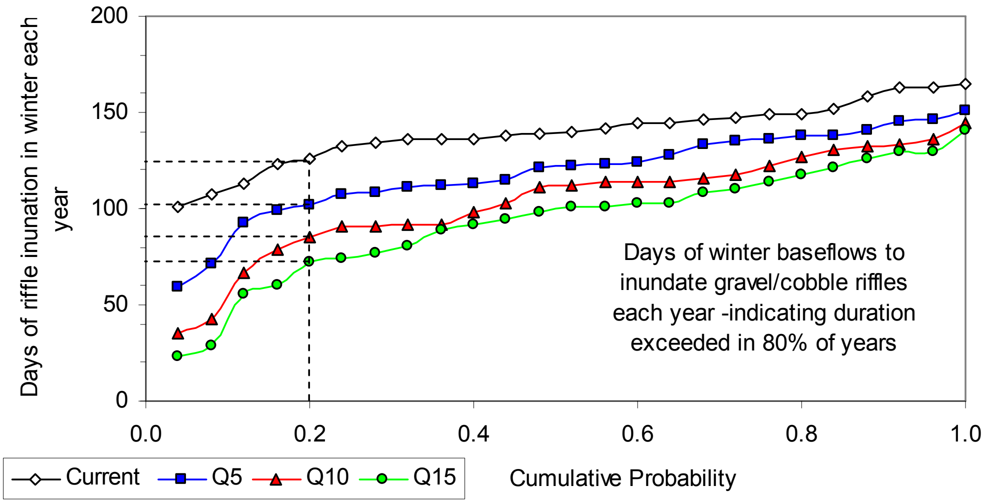


Figure 52. Cumulative probability plot for duration of riffle inundation for Buckingham, showing effects of each diversion scenario, with effects at the 20% probability level (i.e. days riffle inundation exceeded in 80% of years).

When broken into months, the mean total duration of gravel/cobble riffle inundation events across years (Figure 53) shows that inundation commences in May, increasing through June, with almost continuous inundation in July and August, before starting to decrease in September. Comparison of current duration to duration under each diversion scenario shows substantial effects in June and September (up to 40% reduction in period of inundation), with 10 – 15% reduction in July/August under the 15 GL scenario (Figure 53, Table 30).

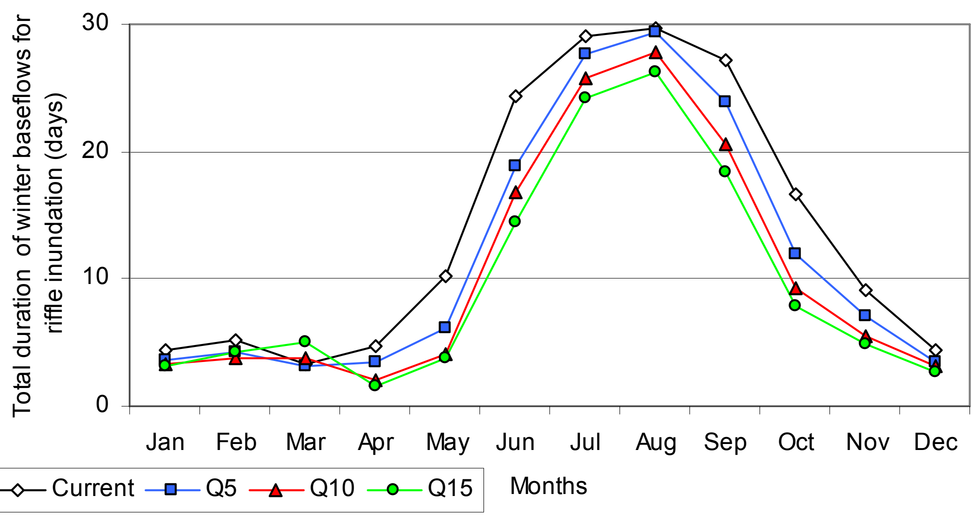


Figure 53. Mean of total duration of riffle inundation for each month for current flows and the three diversion scenarios for Buckingham, where discharge exceeded 0.07 m^3/sec.

Similarly, the mean monthly duration of riffle inundation shows effects of diversion in each winter month (Figure 54, Table 31), with up to 60% loss of period of riffle inundation in June and September, and 30 – 40% reduction in July/August under the 15 GL scenario (Table 31).

Table 30. Mean of total duration (days) of riffle inundation each month, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	80.3%	76.2%	70.6%
Feb	82.1%	72.2%	81.0%
Mar	98.5%	115.4%	153.8%
Apr	73.9%	43.1%	32.3%
May	59.6%	39.6%	36.4%
Jun	77.7%	68.9%	59.4%
Jul	95.2%	88.7%	83.3%
Aug	98.5%	93.3%	88.2%
Sep	88.1%	75.8%	67.6%
Oct	71.5%	55.4%	47.1%
Nov	77.1%	59.8%	53.5%
Dec	77.9%	72.3%	61.7%

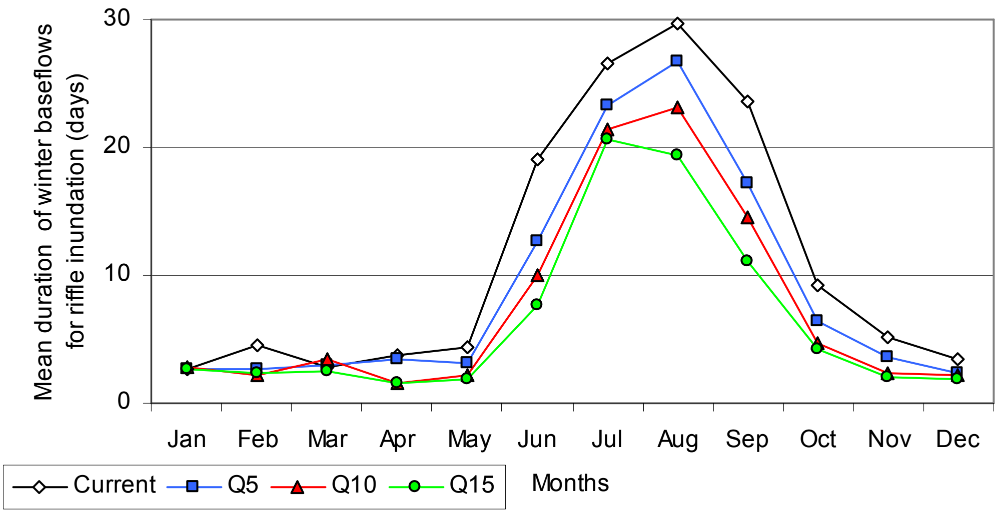


Figure 54. Mean of mean duration of riffle inundation for each month for current flows and the three diversion scenarios for Buckingham, where discharge exceeded 0.07 m^3/sec.

Table 31. Mean of mean duration (days) of riffle inundation each month, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	101.1%	105.2%	98.9%
Feb	58.3%	48.4%	52.9%
Mar	107.4%	125.4%	89.5%
Apr	92.0%	40.3%	40.3%
May	69.3%	47.7%	42.9%
Jun	66.7%	52.8%	40.1%
Jul	87.7%	80.8%	77.8%
Aug	89.7%	77.9%	65.1%
Sep	73.0%	61.3%	46.7%
Oct	69.7%	50.3%	46.2%
Nov	71.4%	46.4%	40.6%
Dec	70.2%	64.1%	55.2%

Analysis of total number of days of riffle inundation also shows substantial changes due to proposed diversion scenarios in any year, with relatively small inter-annual variation due to climate, but with reduced inundation in drier years, such as 1987 and 2001 (Figure 55).

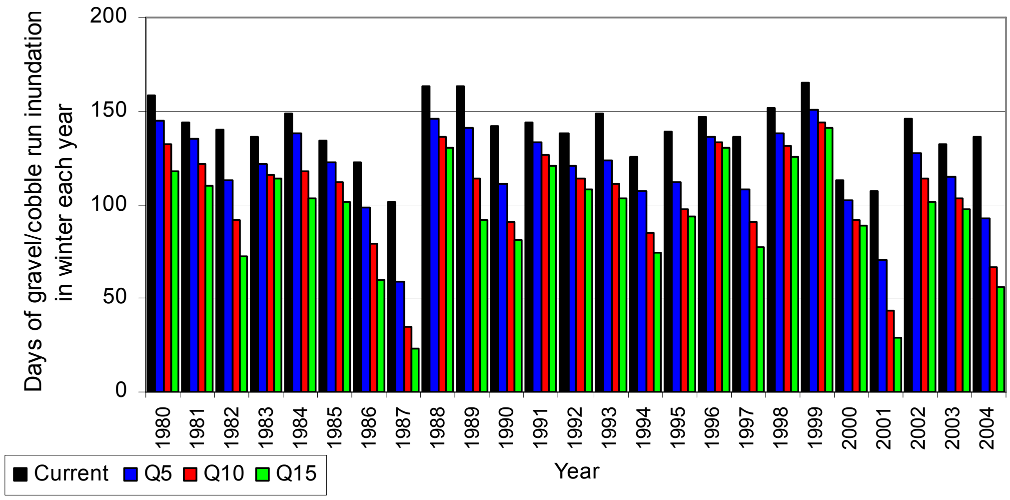


Figure 55. Total number of days of sand/gravel run inundation in winter in each year of modelled flow record for Buckingham, where discharge exceeded 0.01 m^3/sec.

Therefore, inundation of riffles to 0.05 m depth, with 100% lateral coverage of the habitat requires flows of 0.07 m^3/sec, and these currently occur through the majority of each winter. However, because the rule for implementation of diversion scenarios (0.057 m^3/sec) occurs at discharges below the riffle inundation discharge (0.07 m^3/sec), there is a substantial effect of diversions on riffle inundation in winter. This means that riffle areas that normally would be permanently inundated in winter will be either exposed, or will have less than the desired depth of water (< 0.05 m) over this habitat. This poses a threat to the values dependent on riffle habitats, through exposure and dehydration, reduction in habitat area and unknown risk from too shallow inundation (< 0.05 m).

10.2.2.3 Fish passage flows

Based on fish surveys and discussions with local landowners, fish passage flows are required for both small bodied (western minnow, nightfish, western pygmy perch), and large bodied species (cobbler). The rules used for fish passage in SW WA are 0.1 m minimum cross-section depth on the reach (i.e. thalweg depth on shallowest cross-section) for small bodied fish and 0.2 m minimum reach depth for large bodied fish. These minimum threshold depths are considered adequate to allow upstream movement of native fish. The deeper threshold for larger fish since they require higher flows, which usually occur less frequently, will usually occur at a lower frequency than the shallower events for small bodied fish. Therefore, habitat rating curves and TSA were calculated independently for both rules/flow events.

Fish passage for small bodied species (0.1 m minimum depth)

Hydraulic analysis was performed to determine flows that achieved a minimum depth of 0.1 m for the Buckingham reach to allow passage of small bodied fish. Analysis produced a reach-level rating curve for Buckingham (Figure 56) which indicated a discharge threshold of 0.15 m^3/sec, below which fish passage for small species was not possible, and above which passage was possible. Cross-section #9 (a broad, flat, uniform, low-profile riffle zone) was the shallowest point on the reach affecting small fish passage (Figure 56).

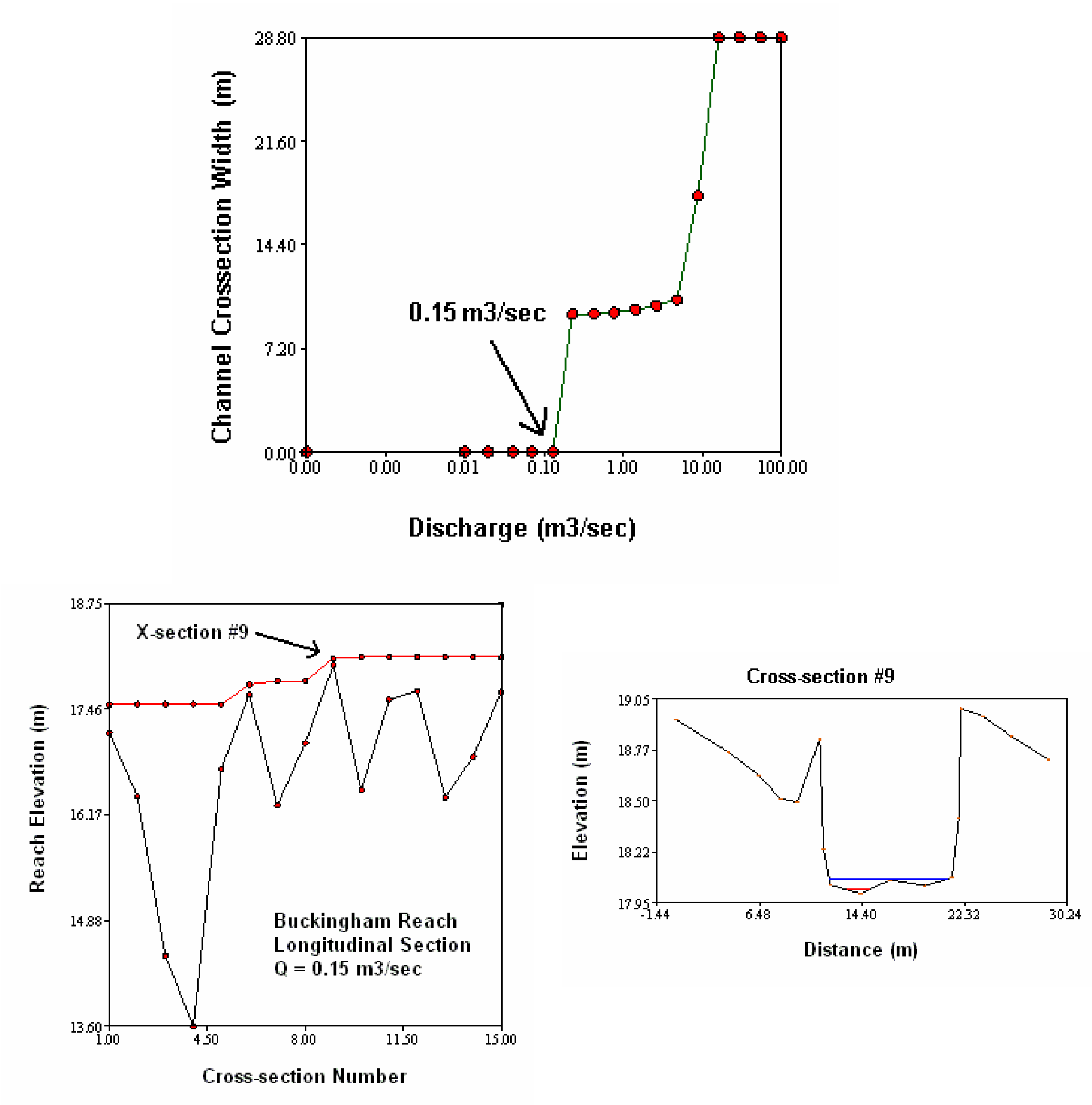


Figure 56. Reach level rating curve for passage of small bodied fish on the East Collie at Buckingham, showing change in channel width (y-axis) with increasing discharge (m^3/sec) (x-axis). The small bodied fish passage criteria of 10 cm minimum threshold depth was achieved with a discharge of 0.15 m^3/sec (TOP), with longitudinal section (BOTTOM LEFT) and cross-section profile (BOTTOM RIGHT) showing cross-section #9, being the critical point for flow depth.

TSA was undertaken to assess frequency and duration of fish passage events for current and diversion scenarios to assess effects of diversions on small fish passage. Analysis indicated that, currently, there was an average of 5.9 events which provided fish passage in winter months, and each event lasted on average 28 days, with passage possible for approx 127 days each winter (i.e. small fish passage is currently possible for most of each winter) (Table 32). Flow diversions resulted in a progressive increase in the number of fish passage events, but a decrease in the mean duration of each event, indicating that reduced flows due to diversions resulted in more but shorter periods of fish passage (i.e. periods that previously provided continuous passage now provide intermittent passage). Overall, there was a reduction in the total duration of fish passage each winter, from 127 days currently, to 91 days under 15 GL diversion scenario (Table 32). This can be seen in the cumulative probability plot for small fish passage, with a progressive loss of fish passage under each diversion scenario (Figure 57).

Table 32. Summary statistics for winter frequency and duration of small fish passage events.

Statistics for winter small fish passage	Current	Q5	Q10	Q15
Mean number of events	5.9	7.4	8.3	9.7
Mean duration of events (days)	28.1	19.1	15.9	12.2
Total duration of events (days)	127.4	110.5	98.8	90.5

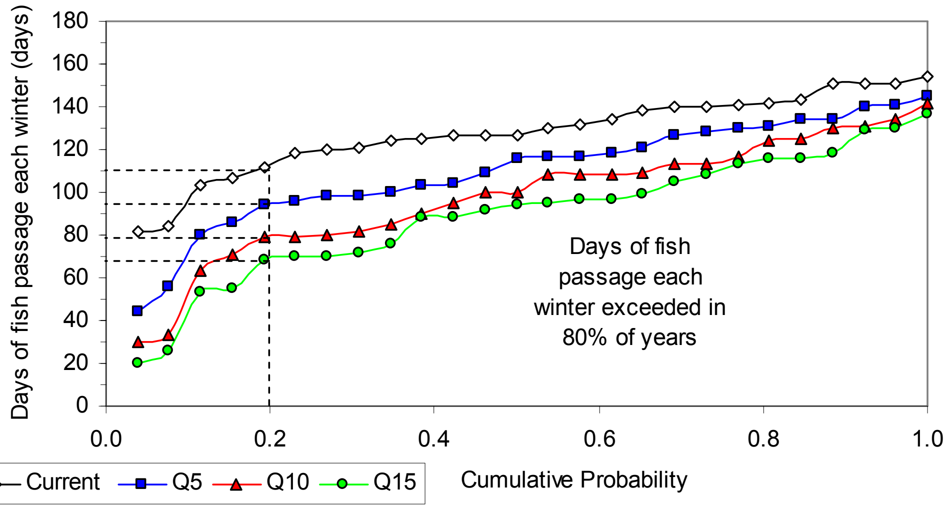


Figure 57. Cumulative probability plot for duration of fish passage for small bodied species of fish for Buckingham, showing effects at the 20% probability level of each diversion scenario (i.e. days of fish passage exceeded in 80% of years).

When broken into months, the mean total duration of fish passage events across years (Figure 58) shows the classic pattern for southwestern W.A. rivers, whereby fish passage commences in May, increasing through June and July to a peak in August, before declining through spring. Comparison of current duration to duration under each diversion scenario shows progressive effects of the diversion scenarios on total number of days fish passage is possible each month (Figure 58, Table 33).

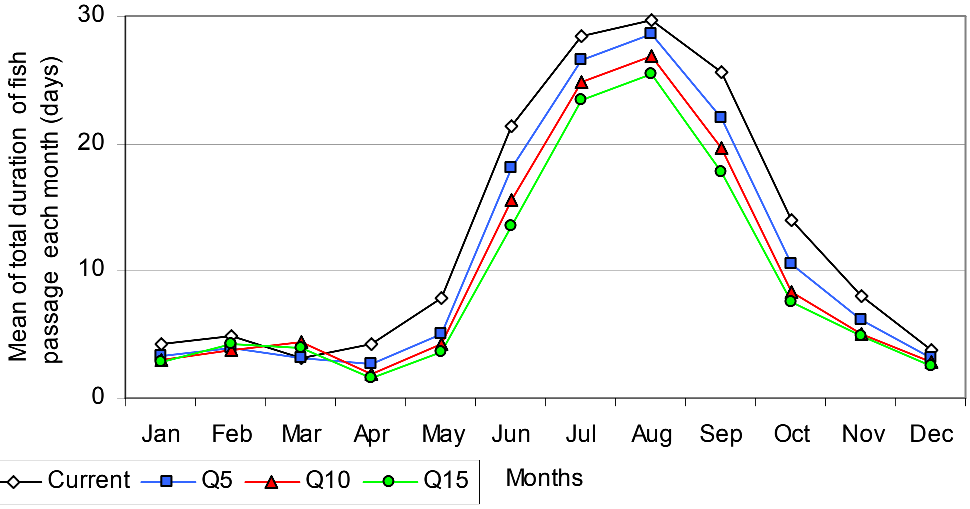


Figure 58. Mean of total duration of fish passage for each month for current flows and the three diversion scenarios for Buckingham, where discharge exceeded 0.15 m^3/sec.

Table 33. Mean of total duration (days) of fish passage events each month for small bodied species, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	77.4%	71.4%	68.5%
Feb	80.4%	77.4%	86.8%
Mar	102.4%	138.7%	128.0%
Apr	62.8%	43.1%	35.3%
May	64.1%	53.7%	46.2%
Jun	84.8%	72.9%	63.1%
Jul	93.4%	87.3%	82.3%
Aug	96.4%	90.7%	86.0%
Sep	86.1%	76.4%	69.1%
Oct	75.2%	59.6%	54.4%
Nov	76.1%	62.9%	60.3%
Dec	79.9%	74.1%	64.1%

There was also a progressive reduction under each diversion scenario in the mean of mean monthly duration of fish passage events across years (Figure 59, Table 34). The greatest proportional reduction from current (i.e. 55 – 65% reduction) occurs in spring and autumn months, however these reductions are based on relatively low total durations, and probably are not particularly significant given most fish passage is in winter. The diversion scenarios resulted in proportionately smaller reductions in fish passage in winter months, however, the actual reduction in days of fish passage were large, with, for example, the current August duration of 28.5 days reducing to 25.2, 21.2 and 17.9 days under 5GL, 10GL and 15GL diversion respectively. Similar reductions occurred in June, July and September (Table 34).

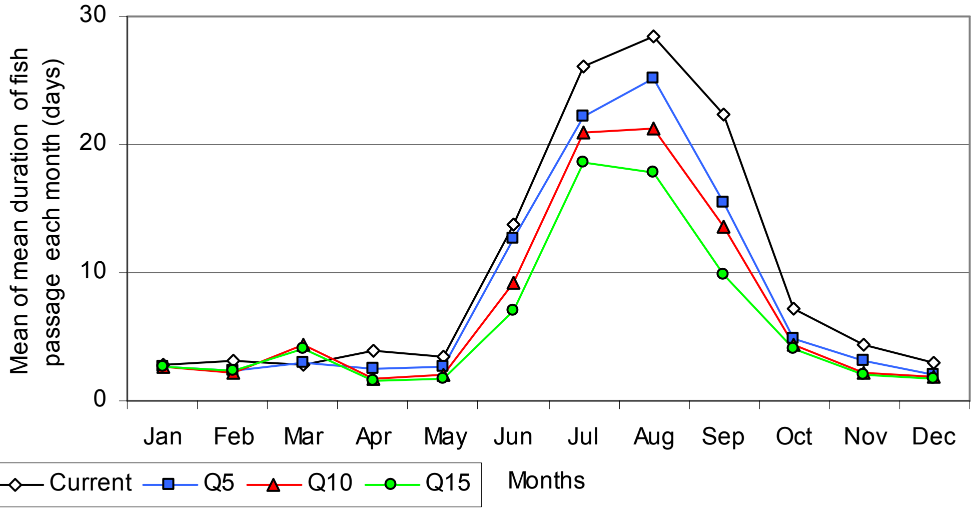


Figure 59. Mean of mean duration of fish passage for each month for current flows and the three diversion scenarios for Buckingham, where discharge exceeded 0.15 m^3/sec.

Table 34. Mean of mean duration (days) of fish passage each month for small bodied species, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	94.3%	94.3%	92.1%
Feb	73.7%	69.2%	75.5%
Mar	106.7%	154.1%	142.2%
Apr	63.5%	42.3%	38.1%
May	76.8%	58.0%	51.9%
Jun	91.4%	66.5%	51.2%
Jul	85.0%	80.0%	71.1%
Aug	88.5%	74.4%	62.8%
Sep	69.4%	60.4%	43.9%
Oct	68.7%	60.3%	56.8%
Nov	69.5%	48.1%	46.1%
Dec	67.8%	65.1%	58.4%

Analysis of total number of days of fish passage each winter also shows that fish passage has been possible for the majority of each winter in most years under current flows, with 15%, 25% and 32% reduction from current in the number of days passage was possible under the 5 GL, 10 GL and 15 GL diversion scenarios respectively (Figure 60).

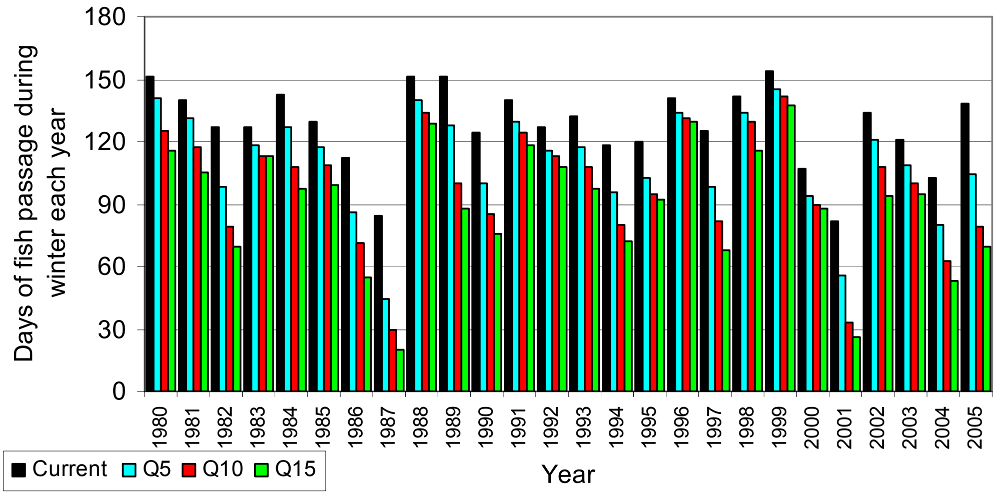


Figure 60. Total number of days of small fish passage possible in winter in each year of modelled flow record for Buckingham, where discharge exceeded 0.15 m^3/sec.

Therefore, passage for small bodied species of fish requires flows of 0.15 m^3/sec, and this currently occurs through the majority of each winter. Implementation of diversion scenarios has progressively increasing effects on fish passage for small bodied species, even at the lowest diversions, resulting in intermittent fish passage (i.e. more events of shorter duration). There is also a proportionately large reduction in the total duration of fish passage in autumn/spring, but the reduction is minor in actual terms, since there is likely minimal fish passage at these times. However, reductions in frequency and duration of winter fish passage would be considered to limit small fish passage along this reach.

Fish passage for large bodied species (0.2 m minimum depth)

Hydraulic analysis was performed to determine flows that achieved a minimum depth of 0.2 m for the reach to allow passage of large bodied species of fish. Analysis produced a reach-level rating curve for Buckingham (Figure 61) which indicated a discharge threshold of 1.50 m^3/sec, below which large fish passage was not possible, and above which passage for large bodied fish was possible. Cross-section #9 was the shallowest point on the reach affecting fish passage (see Figure 61).

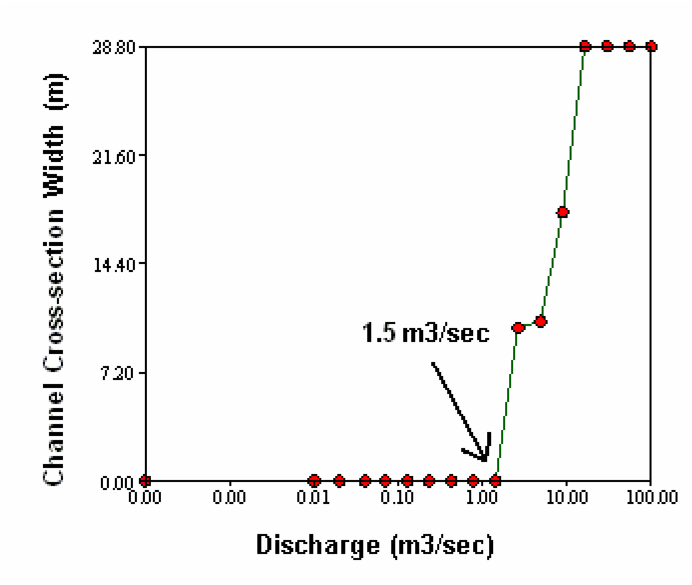


Figure 61. Reach level rating curve for passage of large bodied fish on the East Collie at Buckingham, showing change in channel width (y-axis) with increasing discharge (m^3/sec) (x-axis). The large bodied fish passage criteria of 20 cm minimum threshold depth was achieved with a discharge of 1.5m^3/sec, with cross-section #9 being the critical point for flow depth.

TSA was undertaken to assess frequency and duration of fish passage events for current and diversion scenarios to assess effects of diversions on large fish passage. Analysis indicated that, currently there was on average 9.6 events which provided fish passage in winter months, and each event lasted on average 7.2 days, with passage possible for approx 57 days each winter (Table 35). Flow diversions resulted in minimal change in the number of large fish passage events, a small decrease in the mean duration of each event, and only a medium reduction in the total duration of fish passage each winter, from 57 days currently, to 48 days under 15 GL diversion scenario (Table 35). This can be seen in the cumulative probability plot for large fish passage, with a minimal loss of fish passage, even under the 15 GL diversion scenario (Figure 62).

Table 35. Summary statistics for frequency and duration of large fish passage events.

Statistics for winter large fish passage	Current	Q5	Q10	Q15
Mean number of events	9.6	9.8	9.8	9.3
Mean duration of events (days)	7.2	6.7	6.0	5.9
Total duration of events (days)	57.2	54.4	50.8	47.7

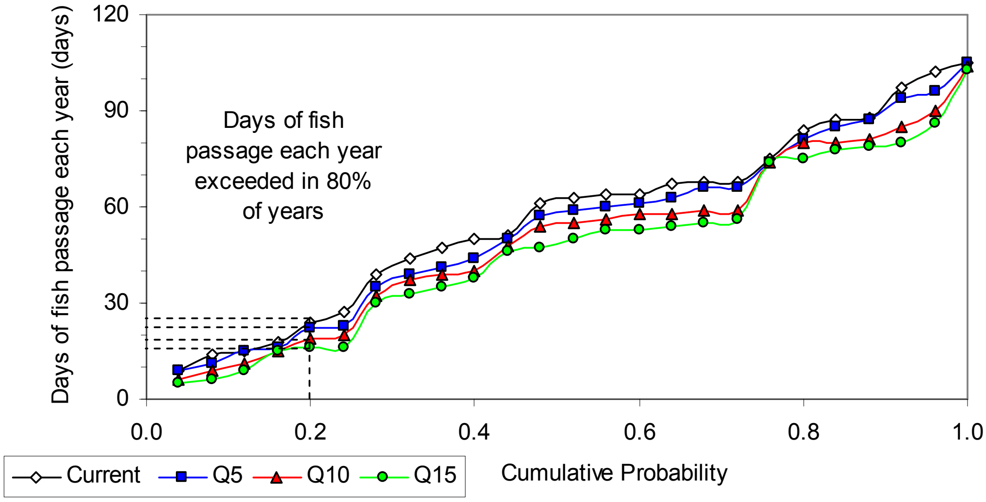


Figure 62. Cumulative probability plot for duration of fish passage for large bodied species of fish, showing effects of each diversion scenario for Buckingham, with effects at the 20% probability level (i.e. days of fish passage exceeded in 80% of years).

When broken into months, the mean total duration of fish passage events each month (Figure 63) shows again the classic pattern for southwestern WA rivers, whereby passage for large fish species commences in May, increasing through June and July to a peak in August, before declining through spring. Comparison of current duration with each diversion scenario shows a small effect of diversions on total number of days of fish passage possible each month (Figure 63, Table 36). Relatively large proportional reductions in total days of fish passage occur under the greatest diversion (15 GL) in April (total loss of fish passage) and May (20% fewer days), however, the actual extent of fish passage in these months is small (1 - 2 days) (Figure 63). There were declines in extent of fish passage in winter months (June – Sept), but with generally these were < 15% reduction in days of fish passage, even under the highest diversion (Figure 63; Table 36).

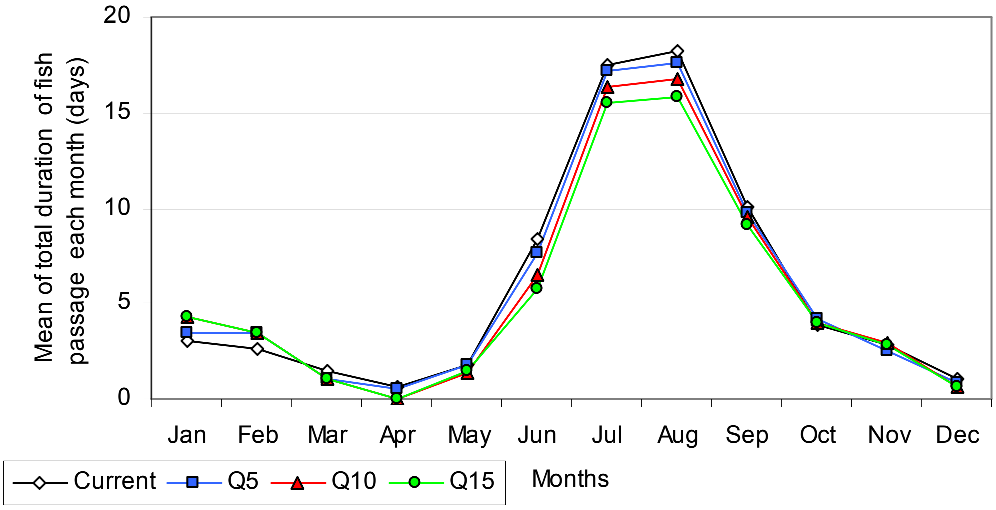


Figure 63. Mean of total duration of fish passage for each month for current flows and the three diversion scenarios for Buckingham, where discharge exceeded 1.5 m^3/sec.

Table 36. Mean of total duration (days) of fish passage events each month for large bodied species, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	116.7%	144.4%	144.4%
Feb	131.2%	131.2%	131.2%
Mar	66.7%	66.7%	66.7%
Apr	75.0%	0.0%	0.0%
May	97.2%	76.4%	79.4%
Jun	91.0%	77.4%	68.8%
Jul	98.2%	93.2%	88.9%
Aug	96.7%	91.9%	86.6%
Sep	96.8%	94.2%	90.5%
Oct	107.2%	102.7%	102.7%
Nov	90.3%	102.5%	98.6%
Dec	80.0%	66.7%	66.7%

Mean duration of fish passage events each month show similar effects of the diversion scenarios (Figure 64, Table 37). The greatest departure from current occurs in April from an average of 0.5 days per month to zero days, however these reductions are probably not particularly significant given natural patterns of fish passage. Under maximum diversion (15 GL), June (27% lower than current), and July (22% lower than current), show the greatest reduction in mean duration. Mean duration of each fish passage event in June drops from 3.5 to 2.5 days, and in July from 8.8 to 6.9 days (Table 37).

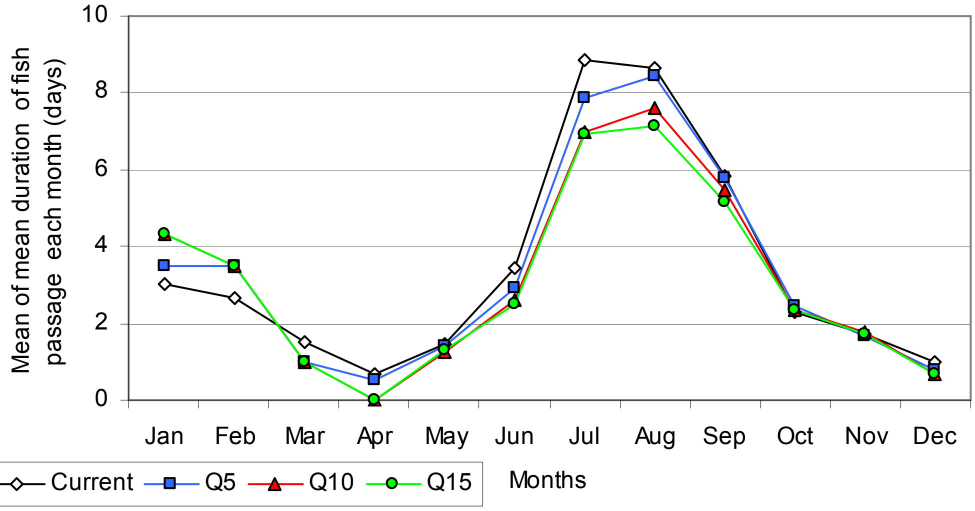


Figure 64. Mean of mean duration of large fish passage for each month for current flows and the three diversion scenarios for Buckingham, where discharge exceeded 1.5 m^3/sec.

Analysis of total number of days of passage for large fish each year shows high inter-annual variability, with total duration ranging from 100 days in wet winters to < 10 days in dry years (Figure 65). This indicates that in some years passage for large fish species is often greatly reduced. Generally, even the maximum diversion (15 GL) has little influence on the total duration of fish passage each year relative to inter-annual variability, however, in dry years, the diversions would reduce large fish passage to extremely low levels (i.e. in 2001, total winter large fish passage would be reduced from 14 days to 5 days) (Figure 65).

Table 37. Mean of mean duration (days) of fish passage each month for large bodied species, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	116.7%	123.8%	144.4%
Feb	131.2%	100.0%	131.2%
Mar	66.7%	100.0%	66.7%
Apr	75.0%	0.0%	0.0%
May	96.6%	88.2%	87.7%
Jun	85.7%	88.9%	73.0%
Jul	89.0%	88.4%	78.3%
Aug	97.4%	90.2%	82.5%
Sep	99.6%	94.6%	89.1%
Oct	107.9%	94.4%	102.3%
Nov	96.9%	108.7%	102.1%
Dec	80.0%	83.4%	66.7%

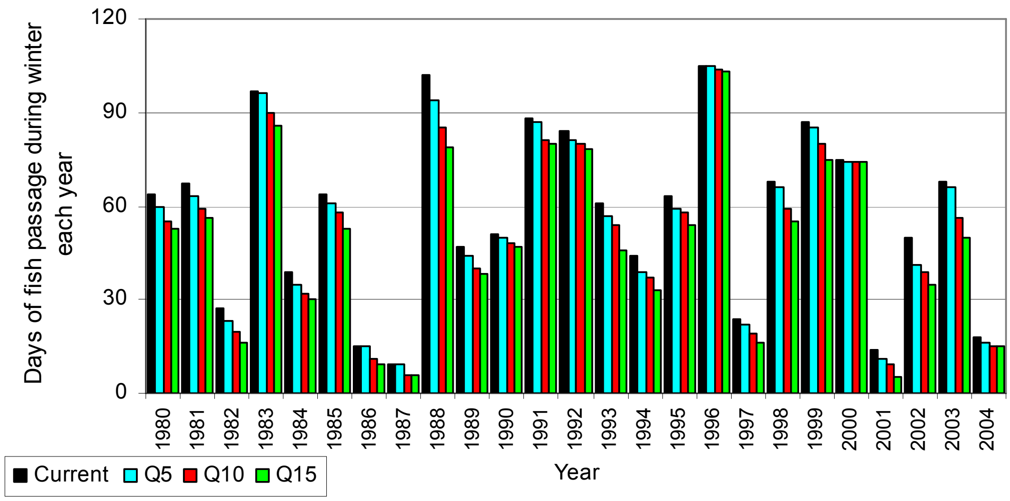


Figure 65. Total number of days of fish passage possible in winter in each year of modelled flow record for Buckingham, where discharge exceeded 1.5 m^3/sec.

Passage for large bodied fish species, which requires flows of 1.5 m^3/sec, occurs less often and for shorter durations than for small bodied species. Because passage for large bodies species occurs under a relatively high discharge, due to the shape of the shallowest riffle zone on the reach (broad and flat riffle), and because diversions have a smaller effect on these high flows compared with flows for small fish passage, the diversion scenarios tend to have a greater effect on small fish passage than on large fish passage, especially at the highest diversion in winter months (June – Sept). At the highest diversions, there are 40 – 50% reductions in some flow statistics which are considered relatively major, and would substantially reduce longitudinal connectivity on the Buckingham Reach. In dry years, the diversions would substantially reduce total duration of large and small fish passage to levels that are not normally experienced in the system.

10.2.2.4 Active Channel Flows

Active channel or channel forming flows are events typically occurring on a 1:2 to 1:3 year frequency in south-west river systems, but their frequency will be altered in cleared catchments with greater and faster run-off. Active channel flows help maintain the shape of the channel by mobilising sediment, scouring pools and preventing riparian vegetation encroaching into the channel. These flows also inundate trailing vegetation to provide cover and spawning habitat for fish, and provide inputs of autochthonous (algal production) and allochthonous energy (leaf litter/detritus) from banks.

The current active channel stage height was determined from cross-sectional surveys as the level on the banks above which vegetation is stable and below which the bank is eroding/bare, and without extensive riparian vegetative growth. Using all cross-sections, with the exception of those in deep pools, the average depth from the bed to active channel stage height, taken as the deepest part of the channel (thalweg depth) to the level approximated as active channel was determined from cross-sections as 1.11 m. Using this stage height (*i.e.* a thalweg depth of 1.11 m), the reach level rating curve produced by hydraulic analysis calculated a discharge of 2.7 m^3/sec to achieve active channel flows (Figure 66).

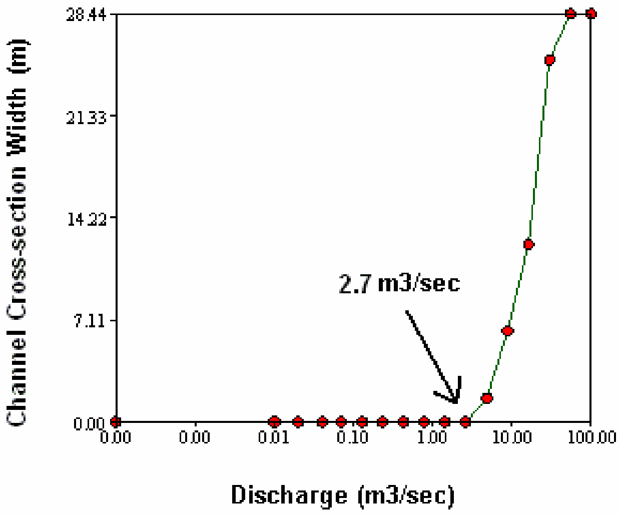


Figure 66. Reach level rating curve for discharge to attain active channel stage height for Buckingham, with threshold flow of 2.7 m^3/sec.

TSA of the active channel flow event determined that currently in winter there was on average 8 events of 2.7 m^3/sec or greater that reach active channel stage height, and under the 5 GL, 10 GL and 15 GL scenarios this decreased very marginally (Table 38). Essentially, the diversion scenarios have negligible effect on the annual frequency of flow events to reach active channel stage height. Each event lasted on average 4.0 days, again, with little change under the 5 GL, 10 GL or 15 GL scenarios (Table 38). Similarly, the total duration of active channel flows each winter showed little change from current under each diversion scenario (Table 38).

This can be seen in the cumulative probability plot for active channel flows, with a negligible loss of these flows, even under the 15 GL diversion scenario (Figure 67).

Table 38. Summary statistics for frequency and duration of active channel flow events.

Statistics for active channel flows	Current	Q5	Q10	Q15
Mean number of events	8.4	8.0	7.8	7.6
Mean duration of events (days)	4.4	4.3	4.1	4.2
Total duration of events (days)	33.6	32.1	30.8	29.6

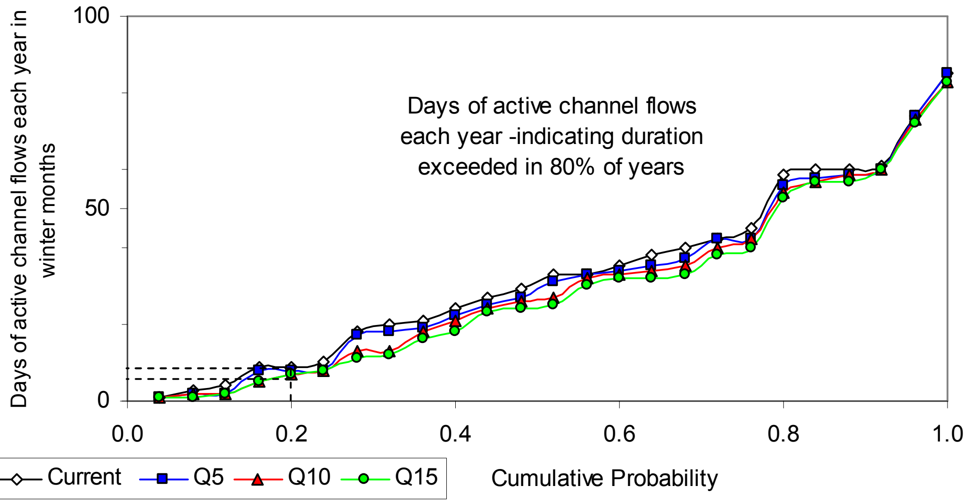


Figure 67. Cumulative probability plot for duration of active channel flows, showing effects of each diversion scenario for Buckingham, with effects at the 20% probability level (i.e. days of active channel flows exceeded in 80% of years).

When broken into months, the mean number and mean total duration of active channel flows each month under current flows (Figure 68) increased through June and July to a peak in August, before declining through spring. Comparison of current duration with each diversion scenario shows a small effect of diversions on mean number of events (Figure 68) and total number of events each month (Figure 69, Table 39), with generally < 10% reduction in any month, even under the highest diversion (Figure 69, Table 39).

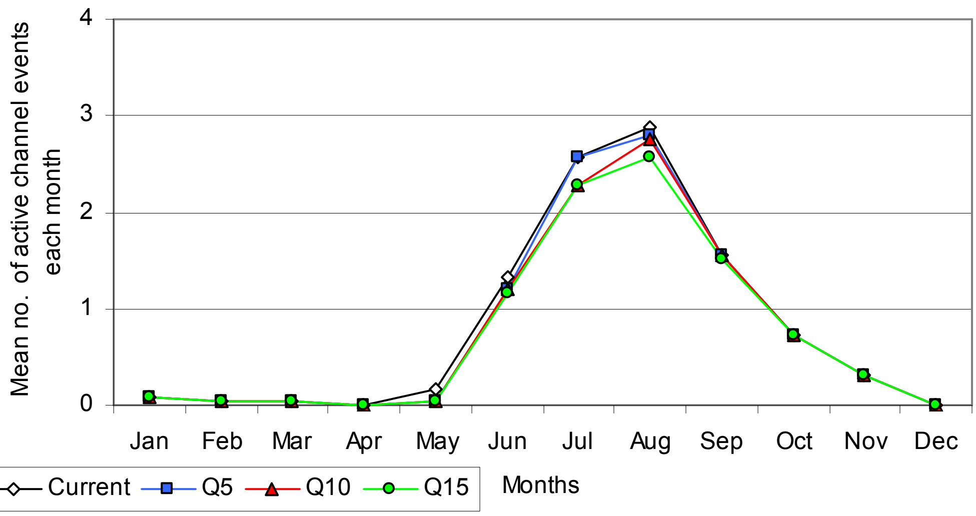


Figure 68. Mean number of flow events each month in winter to reach active channel stage height for Buckingham, where discharge exceeded 2.7 m^3/sec.

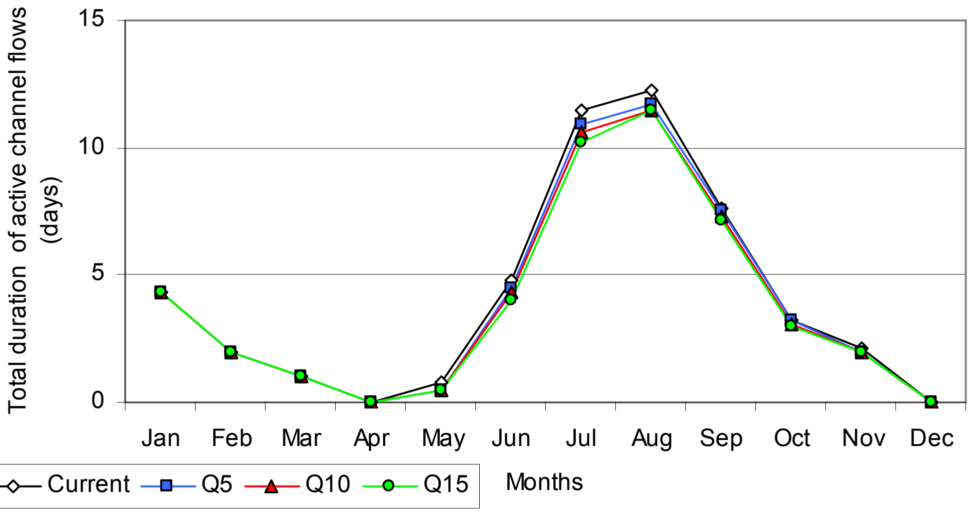


Figure 69. Total duration of flow events each month in winter to reach active channel stage height for Buckingham, where discharge exceeded 2.7 m^3/sec

Table 39. Mean of total duration (days) of active channel flows each month, expressed as a percentage of current conditions, for 5 GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	100.0%	100.0%	100.0%
Feb	100.0%	100.0%	100.0%
Mar	100.0%	100.0%	100.0%
Apr	-	-	-
May	62.5%	62.5%	62.5%
Jun	92.7%	89.0%	82.9%
Jul	95.3%	92.6%	89.2%
Aug	95.7%	93.3%	93.4%
Sep	98.7%	95.4%	93.5%
Oct	100.0%	97.1%	94.3%
Nov	93.3%	93.3%	93.3%
Dec	-	-	-

Mean duration of active channel flows each month showed similar effects of diversion as above (Figure 70, Table 40). The greatest departure from current occurs in May, but from a very low frequency, and so this reduction is probably not particularly significant relative to the importance of active channel flows which occur in winter. Some summer events were evident as a result of summer rainfall events, but these had low frequency of recurrence and are not considered ecologically important.

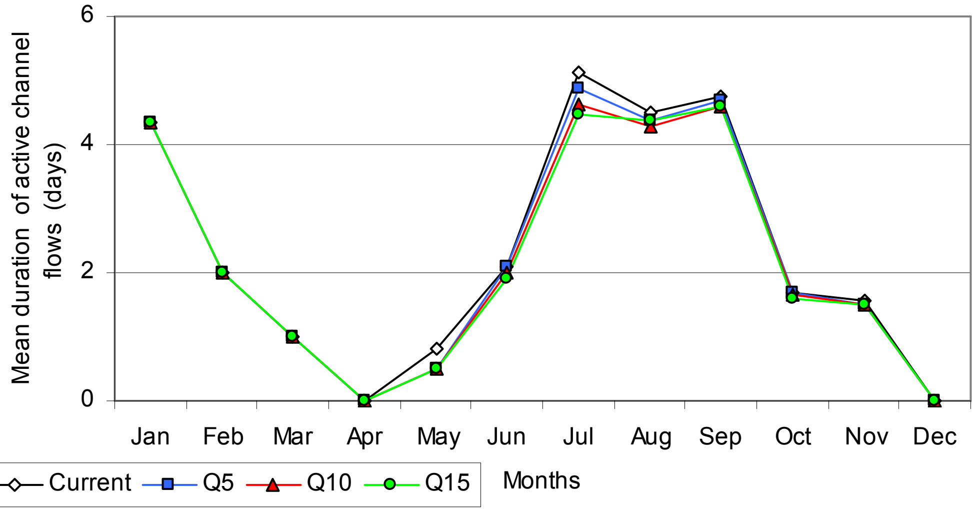


Figure 70. Mean duration of flow events each month in winter to reach active channel stage height for Buckingham, where discharge exceeded 2.7 m^3/sec.

Table 40. Mean of mean duration (days) of active channel flows each month, expressed as a percentage of current conditions, for 5 GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	100.0%	100.0%	100.0%
Feb	100.0%	100.0%	100.0%
Mar	100.0%	100.0%	100.0%
Apr	-	-	-
May	62.5%	62.5%	62.5%
Jun	99.8%	95.4%	90.5%
Jul	95.1%	90.0%	87.1%
Aug	97.2%	95.4%	97.5%
Sep	98.9%	96.8%	96.8%
Oct	100.0%	97.3%	94.6%
Nov	95.5%	95.5%	95.5%
Dec	-	-	-

Analysis of total number of days of active channel flows each year shows high inter-annual variability, with total duration ranging from ~ 85 days in wet winters to 1 day in dry years. This indicates that in some years the duration of active channel flows is greatly reduced due to low rainfall. Generally, even the maximum diversion (15 GL) has little influence on the total duration of active channel flows each year, and has minimal effect relative to inter-annual variability (Figure 71).

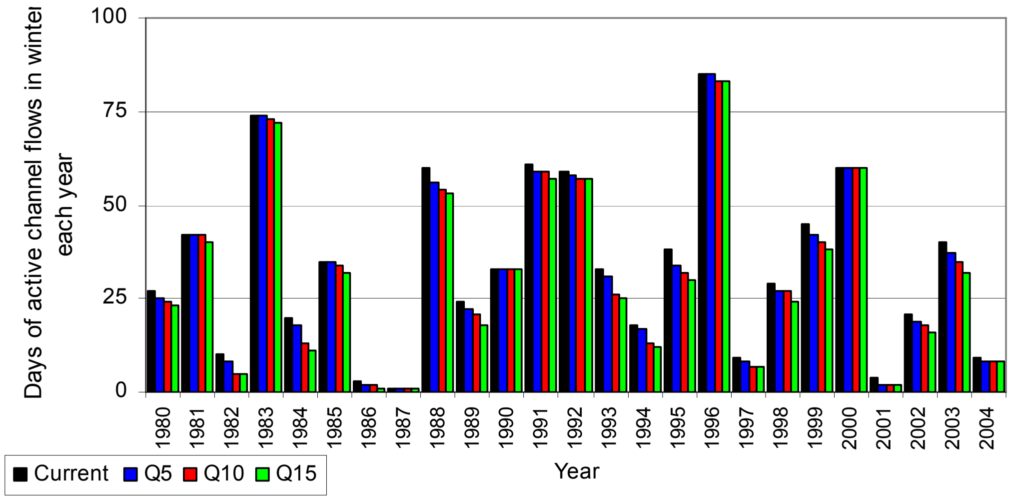


Figure 71. Total number of days of active channel flows in winter in each year of modelled flow record for Buckingham, where discharge exceeded 2.7 m^3/sec.

Active channel flows of 2.7 m^3/sec do not appear to be greatly affected by any of the diversion scenarios, with minimal reductions in frequency and duration in winter months. This likely reflects the fact that active channel flows occur under relatively high flow events, and any diversions are a relatively small proportion of these flows. As a result, changes are minimal. Compared with well vegetated, undisturbed systems in SW WA, active channel flows for Buckingham Reach occur at a relatively high frequency and duration. This may reflect inaccuracies of the uncalibrated hec-ras model, or alternatively reflect the disturbed nature of the catchment, with greater and faster run-off as a result of catchment clearing. The effects of the latter are evident in the Buckingham Reach, with substantial bed and bank erosion, as well as pool infilling from massive sediment deposition. The top end of Duderling Pool has been in-filled by a sand slug, and from survey data it is estimated that approximately 225 m linear length of pool and channel at the top end of the pool have been in-filled by sediment. Much of this erosion and sediment mobilisation has likely been generated in large magnitude, low-frequency events, with large events being more common in recent years. Diversion of these large events may help protect the channel to some extent, however, diversions only have a small effect on these big events, and catchment restoration is the best long-term solution.

10.2.2.5 Flows to inundate medium elevation benches

An objective of winter flows to inundate medium benches to provide inputs of allochthonous energy (leaf litter/detritus) from benches into the river was set for Coolangatta Reach, based on benches identified from channel cross-section surveys. However, the channel at Buckingham was much more uniform and incised than at Coolangatta, alternating between areas of steep eroding banks and sediment deposition, and there were no such benches visible along the representative reach. Therefore, an equivalent flow objective could not be determined, and therefore was not incorporated in the EWR assessment for the Buckingham Reach.

10.2.2.6 Flows to connect off-river wetlands/sumps

On the Coolangatta reach a flow objective was set to inundate a series of small interconnected sumps/wetlands which likely contained water into spring (based on water marks, distribution of vegetation etc), and so likely provided spawning/breeding/nursery habitat for fauna such as frogs and tortoise, as well as opportunistic habitat for macroinvertebrates and feeding habitat for waterbirds (ducks, egrets, herons). However, no such habitats/off-river wetlands or sumps were evident on the Buckingham reach, and therefore an equivalent flow objective could not be incorporated in EWR assessment for the Buckingham reach.

10.2.2.7 Winter High Flows

The objective of winter high flows was to exceed top of bank (TOB) stage heights to start inundation of the floodplain and associated riparian vegetation. The floodplain often supports shallow wetland/sump areas that are likely important feeding and nursery areas for frogs and provide habitat in which juvenile stages with poor swimming ability may avoid high flows. Overbank flows are required to inundate and recharge these wetlands. Similarly, riparian vegetation on the floodplain (*i.e. Eucalyptus rudis* and paperbarks) also requires occasional inundation to a.) disperse seed, and b.) assist seed set, and soak soil profiles to promote successful germination. To achieve the above objectives, a flow above TOB is required. These flows usually occur as a result of larger rainfall events, principally in winter, and occur infrequently and for short durations.

The current TOB stage height was determined from cross-sectional surveys as the level on the cross-sections above which the bank profile started to decline onto a flatter floodplain. This can usually be detected in the HA habitat rating curves by a change in habitat area relating to a change in bank gradient. However, this inflection was not distinct in the Buckingham reach habitat rating curve, reflecting the absence of a well defined TOB/floodplain on all cross-sections. Therefore, TOB was determined using cross-section elevation data. Using all cross-sections, with the exception of those in deep pools, the average depth from the bed to top of bank stage height, taken as the deepest part of the channel (thalweg depth) to the level approximated as top of bank was determined from cross-sections as 2.0 m. Using this stage height (*i.e.* a thalweg depth of 2.0 m), the reach level rating curve produced by hydraulic analysis calculated a discharge of 8.8 m^3/sec to achieve top of bank flows (Figure 72).

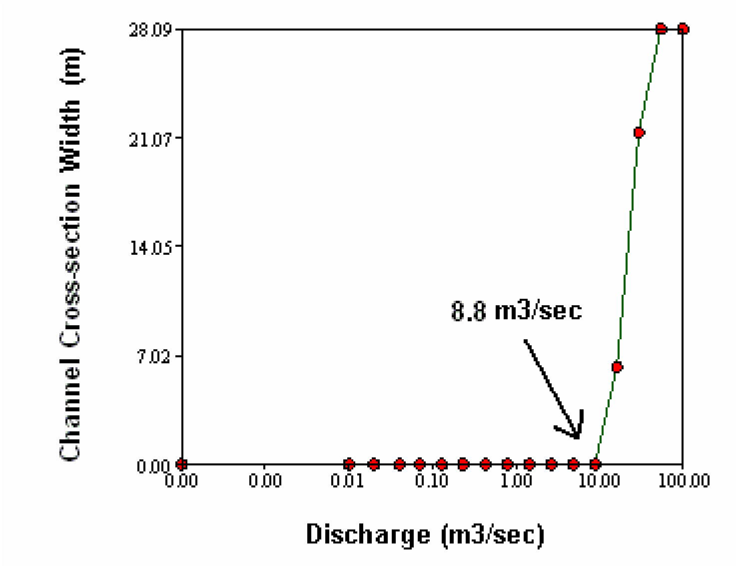


Figure 72. Reach level rating curve for TOB stage height on Buckingham Reach, with floodplain inundation occurring at approximately 8.8 m^3/sec.

TSA of the TOB flow event determined that currently in winter there was on average 1.7 events of 8.8 m^3/sec or greater that reach TOB stage height, and there was no change in this frequency under the 5 GL, 10 GL and 15 GL scenarios (Table 41). Each event had a mean magnitude of 15 m^3/sec, lasted on average 2.6 days, with approx. 8 days of flows above TOB each winter. These flow statistics did not change under the 5 GL, 10 GL or 15 GL scenarios (Table 41).

Table 41. Summary statistics for frequency and duration of TOB flow events.

Statistics for TOB flows	Current	Q5	Q10	Q15
Mean number of events	1.7	1.7	1.7	1.7
Mean magnitude (m3/sec)	15.2	15.2	15.1	15.1
Mean duration of events (days)	2.6	2.6	2.6	2.6
Total duration of events (days)	8.0	8.0	8.0	8.0

When broken into months, the mean number and total duration of TOB flows each month (Figure 73) tended to be highest in winter, although occasional summer rains also provided TOB events in summer months. Comparison of current duration with each diversion scenario was unable to detect any effect of the diversions on mean number of events (Figure 73) or total number of events each month (Figure 74, Table 42), with no reduction in any month, even under the highest diversion (Figure 74, Table 42).

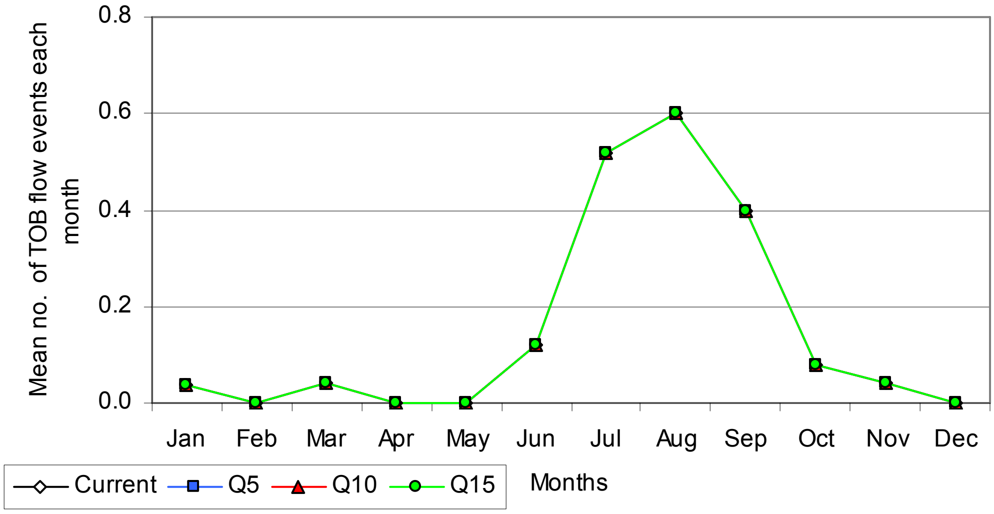


Figure 73. Mean number of flow events each month in winter to reach TOB stage height for Buckingham, where discharge exceeded 8.8 m^3/sec.

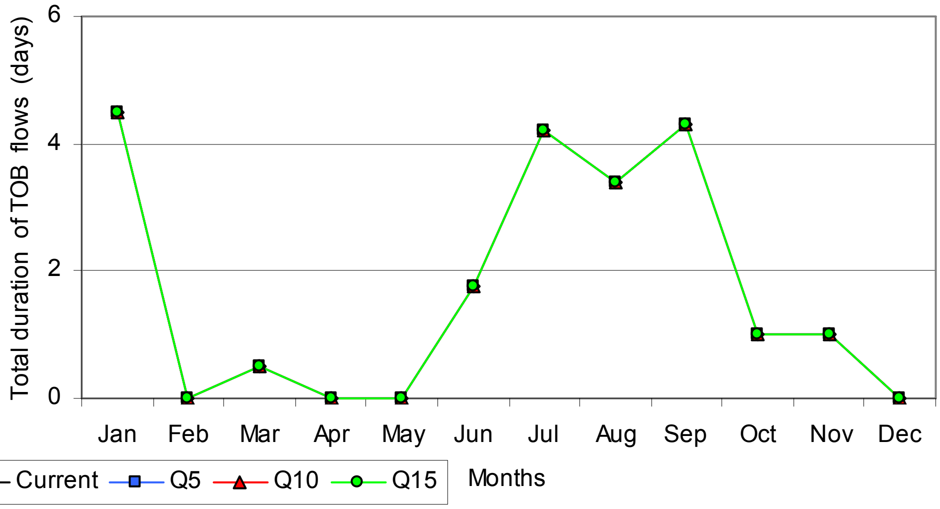


Figure 74. Total duration of flow events each month in winter to reach TOB stage height for Buckingham, where discharge exceeded 8.8 m^3/sec

Table 42. Mean of total duration (days) of TOB flows each month, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	100.0%	100.0%	100.0%
Feb	-	-	-
Mar	100.0%	100.0%	100.0%
Apr	-	-	-
May	-	-	-
Jun	100.0%	100.0%	100.0%
Jul	100.0%	100.0%	100.0%
Aug	100.0%	100.0%	100.0%
Sep	100.0%	100.0%	100.0%
Oct	100.0%	100.0%	100.0%
Nov	100.0%	100.0%	100.0%
Dec	-	-	-

Mean duration of TOB flows each month similarly showed no effects of diversion (Figure 75, Table 43). The greatest departure from current occurred in May, but from a very low frequency, and so this reduction is probably not particularly significant relative to the importance of TOB flows which mostly occur in winter.

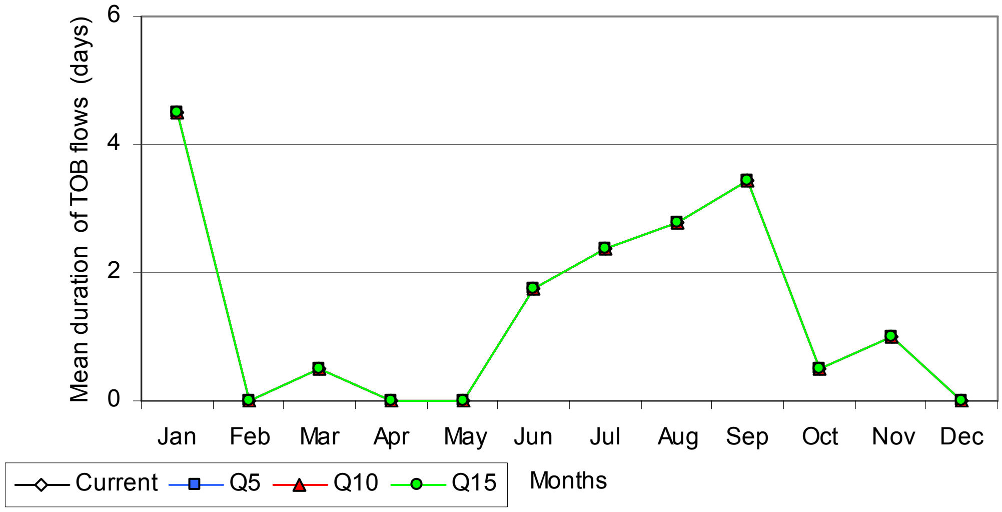


Figure 75. Mean duration of flow events each month in winter to reach TOB stage height for Buckingham, where discharge exceeded 8.8 m^3/sec.

Table 43. Mean of mean duration (days) of TOB flows each month, expressed as a percentage of current conditions, for 5GL, 10 GL and 15 GL diversion scenarios.

Month	Q5	Q10	Q15
Jan	100.0%	100.0%	100.0%
Feb	-	-	-
Mar	100.0%	100.0%	100.0%
Apr	-	-	-
May	-	-	-
Jun	100.0%	100.0%	100.0%
Jul	100.0%	100.0%	100.0%
Aug	100.0%	100.0%	100.0%
Sep	100.0%	100.0%	100.0%
Oct	100.0%	100.0%	100.0%
Nov	100.0%	100.0%	100.0%
Dec	-	-	-

Analysis of total number of days of TOB flows each year shows high inter-annual variability, with total duration ranging from ~ 39 days in wet years to zero days in drier years. This indicates that in some years the duration of TOB flows is greatly reduced due to low rainfall. As indicated above, there was no detectable effect of any diversion scenario on the number of days of TOB flows, and as such, effects of diversions are minimal relative to inter-annual variability (Figure 76).

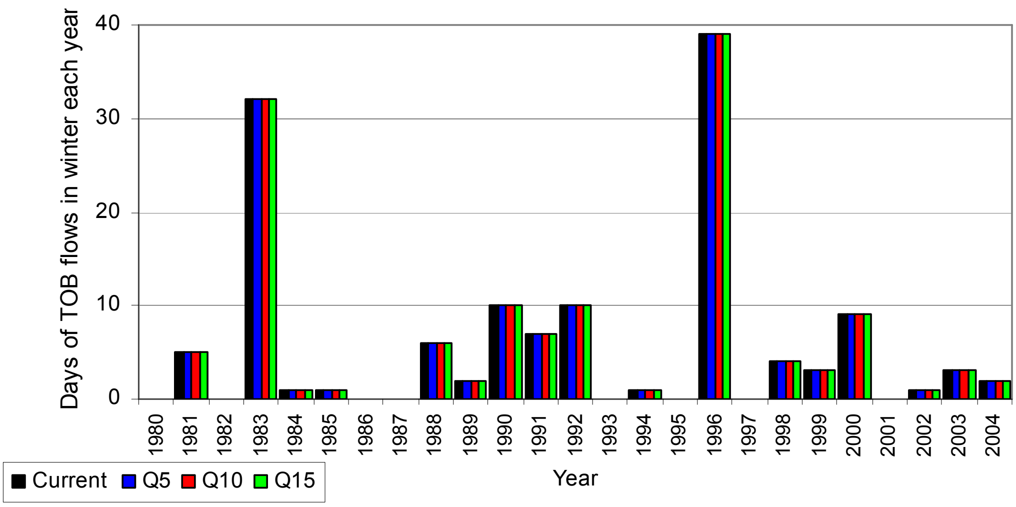


Figure 76. Total number of days of TOB flows in winter in each year of modelled flow record for Buckingham, where discharge exceeded 8.8 m^3/sec.

TOB flows of 8.8 m^3/sec do not appear to be affected by any of the diversion scenarios, with no detectable change in frequency or duration in winter months. This likely reflects the fact that TOB flows are low frequency and high magnitude, and any diversions are small relative to these high flow events.

10.2.2.8 Pool Maintenance

Buckingham reach contains permanent pools surrounded by a seasonally flowing water course. These pools provide critical summer habitat/refuge for water dependent ecological values, as well as deep water habitat during winter. Maintenance of the quantity and quality of water in these pools is therefore critical to sustain the ecological values which are dependent on them (i.e. macroinvertebrates, fish and tortoise). Water levels in permanent pools in seasonally flowing systems are often driven by groundwater influence, either in the form of seeps and springs or through direct surface expression of the groundwater. Determination of riverine EWRs to maintain these pools in summer is not relevant in a seasonally-flowing system however, consideration should be given to the current inter-relationship with groundwater inflows, as well as the current role of artificial supplementation via acidic bore water for these pools. As has been shown in other studies (A.W. Storey unpub. dat.), there will be a relationship between diversity of ecological values using these pools and water depth. As the pools become shallower, they will start to loose diversity. This is particularly true for fish, and likely also applies to tortoise. At this stage the relationship between depth and diversity is unknown for these pools. However, under a precautionary principle, water depth in summer (allowing for infilling by sediment deposition) should be maintained at current levels.

10.2.3 Bingham reach

10.2.3.1 Summer baseflow to inundate riffles

The Bingham River is characterised by high seasonality, with long periods of zero flow in summer months. This is illustrated in the flow duration curves for the system, whereby the river ceases to flow each year, for approximately 50% of the time (Figure 77). This period of zero flow is important in maintaining the current structure of riparian vegetation assemblages, macrophyte communities, invertebrate and fish communities, and preventing the establishment of, and domination by, exotics species such as mosquitofish and redfin perch etc. Increased flow duration with a reduction in the extent of periods of zero flow will likely allow the proliferation of weed species. It will also adversely affect species with adaptations to seasonal flows (i.e. macroinvertebrates with resistant egg stages), with species adapted to perennial flows becoming dominant. Increased flows in summer (i.e. reduced number of zero flow days) would likely indicate groundwater discharge, reflecting increased groundwater levels. This could occur due to land clearing, and flows could be freshwater or saline, depending on the groundwater influence. Therefore, analysis was conducted on the Bingham River flow data to describe the current extent of seasonality in flows. Because analysis indicated the system was seasonal, ceasing to flow for the majority of each summer, it was therefore not appropriate to determine a summer baseflow.

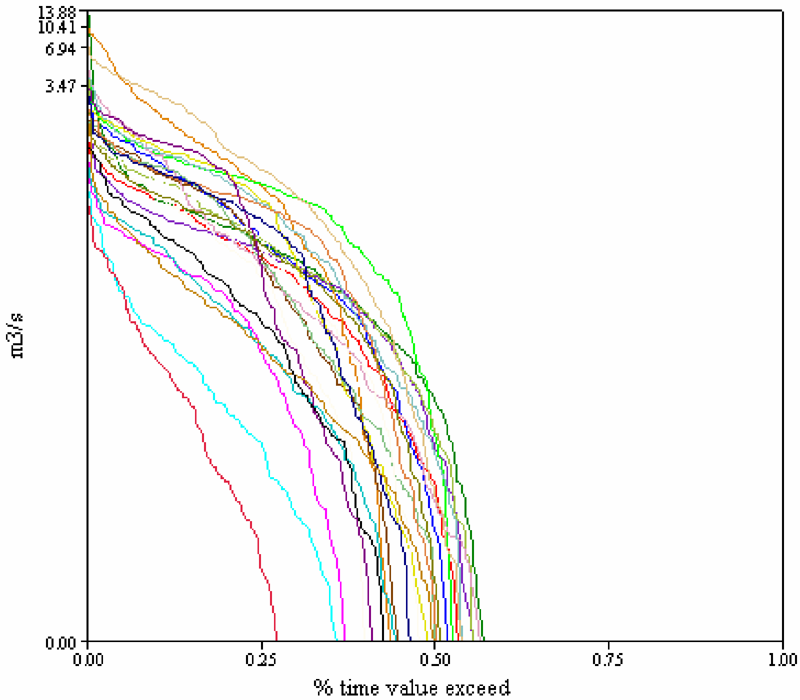


Figure 77. Flow duration curves for each year of record for the Bingham, indicating that it ceases to flow each year approximately 50% of the time.

TSA demonstrated that zero flow days currently occur for an average of 188 days each year (Figure 78), with a monthly average of 26 days of zero flow in each summer month (November to April, incl.). Flows generally commence in May and cease in October/November each year (Figure 79). Therefore, to maintain the current ecology of the Bingham River, this flow periodicity should be maintained. However, the flow record analysed (1980 – 2005) encompasses a low rainfall 'dry period'. It is likely that flow periodicity in a wetter period prior to 1975 may have been different.

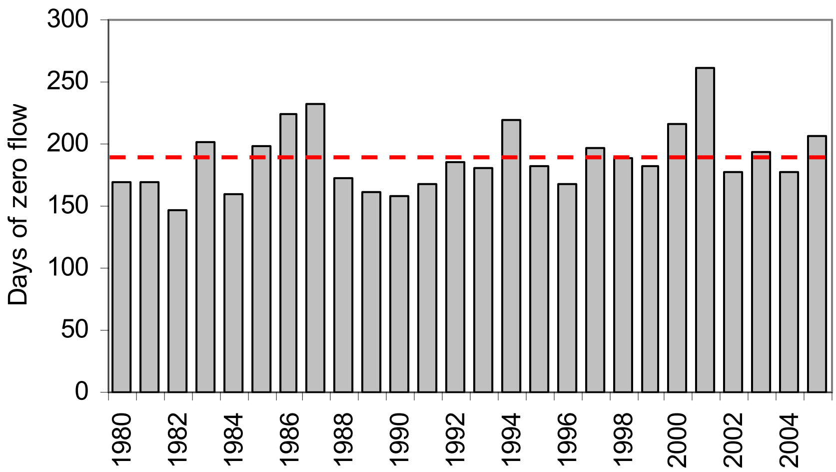


Figure 78. Number of zero flow days in each year on the Bingham River (1980 – 2005), whereby the broken line indicates annual average of 188 days of zero flow.

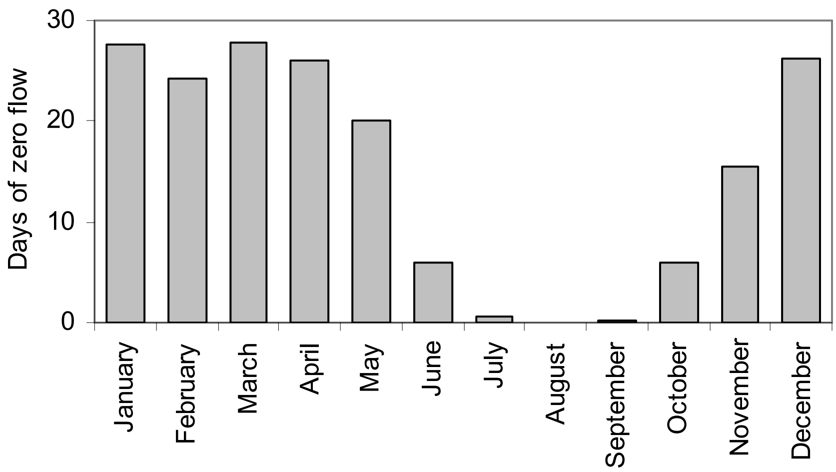


Figure 79. Mean number of zero flow days in each month on the Bingham (1980 – 2005).

10.2.3.2 Winter baseflows to inundate riffles

The objective of winter baseflows is to maintain inundation of gravel/cobble riffles as biodiversity 'hotspots' for macroinvertebrates, and provide flow connectivity for downstream energy transfer. The Bingham River was characterised by sand/clay runs held together by root mats between relatively short and shallow pools, with riparian vegetation growing across the bed of the channel along the length of the reach, except in the pools. Although the quality and heterogeneity of these runs was relatively low compared with more diverse pebble/cobble riffles, it is assumed these areas will function as riffle zones in winter in that they will have higher velocity and diversity of flows relative to pools, but may not support such highly diverse macroinvertebrate fauna compared with typical, more heterogeneous cobble/pebble/gravel riffles in forested streams.

Based on channel morphology, it was considered that the winter baseflow objectives collectively would be provided for by the minimum depth requirement normally applied to riffle zones, being:

- inundation of riffles by a minimum average depth of 5 cm (0.05 m) with 100 % lateral coverage of the riffle component of the specific cross-sections.

Hydraulic analysis was used to calculate discharge required to achieve a mean depth of 0.05 m across riffles whilst maintaining 100% coverage of the habitat. Riffles were located on cross-sections 3 (6 m width), 6 (3.2 m), 8 (3.5 m), 10 (1.5 m), 12 (3.0 m), 13 (5.5 m) & 15 (7.0 m), with a mean width of 4.24 m. Discharge required to achieve 0.05 m average depth with 4.24 m of channel cross-section inundated for those cross-sections was then read from the rating curve as 0.02 m^3/sec (Figure 80). It is recommended that this threshold flow is maintained as a minimum baseflow throughout winter to maintain ecological values dependent on riffles. The transition from summer to winter baseflows is dependent on annual rainfall pattern, but in general terms should occur in the shoulder months of May/June, with a recession back to summer zero flows in October/November.

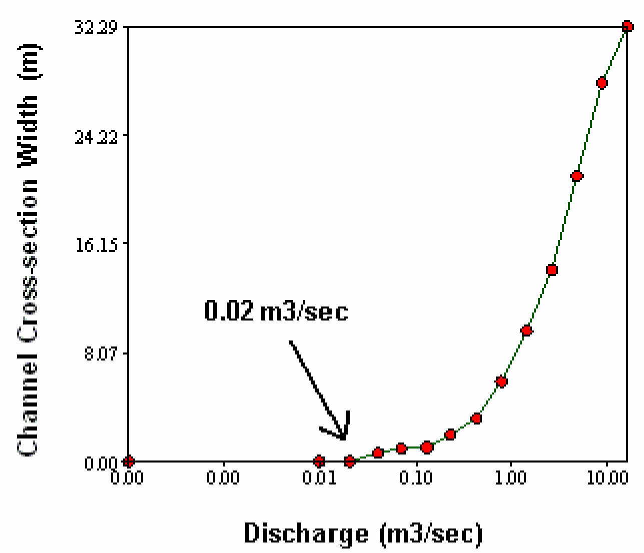


Figure 80. Rating curve for inundation of riffles (cross-sections 3, 6, 8, 10,12, 13 & 15) to 5 cm average depth with 100% habitat coverage on the cross-section (= 4.24 m cross-section total width) on Bingham River in channel width (y-axis) with increasing discharge (m^3/sec) (x-axis) is shown. The threshold criteria of 0.05 m average depth with 100% coverage of riffles was achieved with a discharge of 0.02 m^3/sec.

TSA performed on this flow event demonstrated that this base flow was currently achieved throughout the majority of winter in all years. Currently, the baseflow occurs for an average of 130 days each winter (maximum of 170 days per winter 'season'), with on average 7 events each winter, each lasting approx 22 days in which flows are equal or greater to 0.02 m^3/sec. Winter baseflow occurs for at least 100 days for 80% of the time each winter (Figure 81).

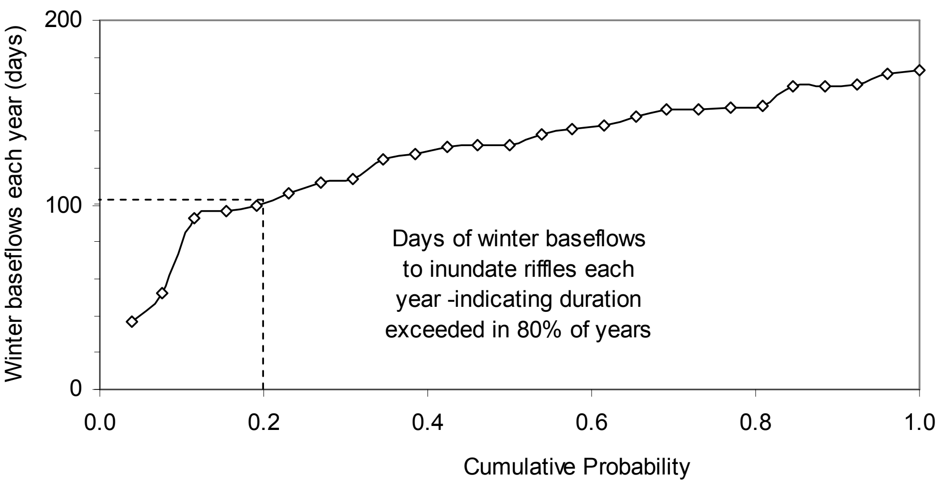


Figure 81. Cumulative probability plot for duration of riffle inundation for Bingham, showing effects of each diversion scenario, with effects at the 20% probability level (i.e. days riffle inundation exceeded in 80% of years).

When broken into months, the total duration of riffle inundation events across years (Figure 82) shows that inundation commences in May, increasing through June, with almost continuous inundation in July, August and September, before starting to decrease in October and November.

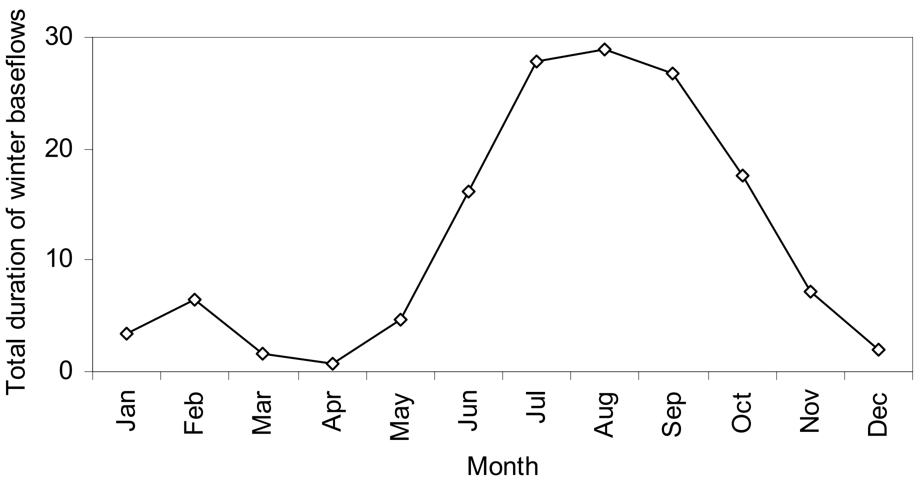


Figure 82. Mean of total duration of riffle inundation for Bingham for each month for current where discharge exceeded 0.02 m^3/sec.

Similarly, the mean monthly duration of riffle inundation shows a similar pattern (Figure 83).

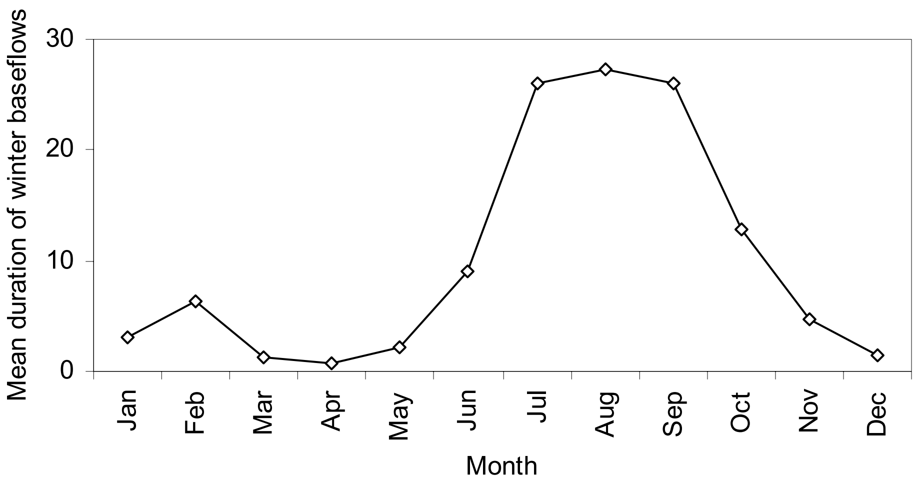


Figure 83. Mean of mean duration of riffle inundation for Bingham for each month for current flows and the three diversion scenarios, where discharge exceeded 0.02 m^3/sec.

Analysis of total number of days of riffle inundation shows relatively small inter-annual variation due to climate, except with reduced inundation in drier years, such as 1987 and 2001 (Figure 84).

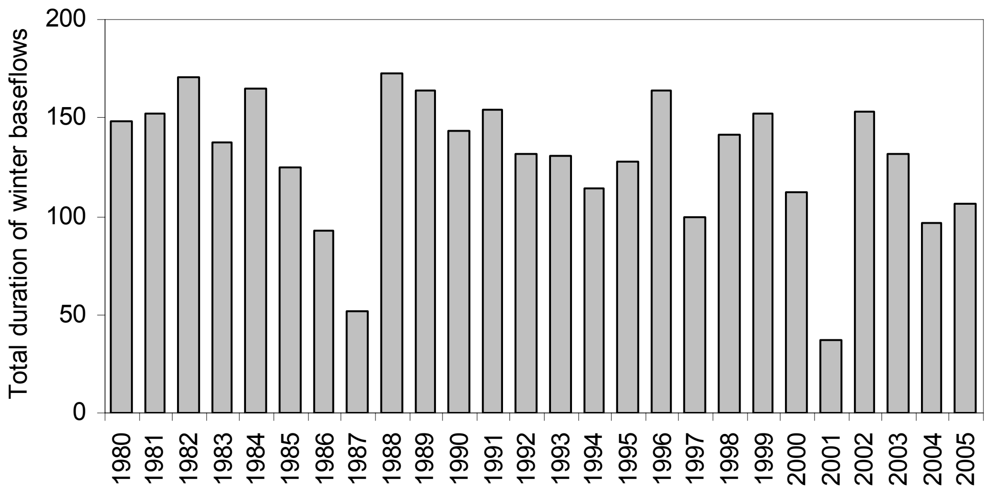


Figure 84. Total number of days of riffle inundation in winter in each year of modelled flow record for Bingham, where discharge exceeded 0.02 m^3/sec.

Therefore, inundation of riffles to 0.05 m depth, with 100% lateral coverage of the habitat requires flows of 0.02 m^3/sec, and these currently occur through the majority of each winter on the Bingham River.

10.2.3.3 Fish passage flows

Based on fish surveys and discussions with local landowners, fish passage flows are required for both small bodied (western minnow, nightfish, pygmy perch), and large bodied species (cobbler). The rules used for fish passage in SW WA are 0.1 m minimum cross-section depth on the reach (i.e. thalweg depth on shallowest cross-section) for small bodied fish and 0.2 m minimum reach depth for large bodied fish. These minimum threshold depths are considered adequate to allow upstream movement of native fish. The deeper threshold for larger fish since they require higher flows, which usually occur less frequently, will usually occur at a lower frequency than the shallower events for small bodied fish. Therefore, habitat rating curves and TSA were calculated independently for both rules/flow events.

Fish passage for small bodied species (0.1 m minimum depth)

Hydraulic analysis was performed to determine flows that achieved a minimum depth of 0.1 m for the Bingham reach to allow passage of small bodied fish. Analysis produced a reach-level rating curve for Bingham (Figure 85) which indicated a discharge threshold of 0.07 m^3/sec, below which fish passage for small species was not possible, and above which passage was possible. Cross-section #13 was the shallowest point on the reach affecting small fish passage (Figure 85).

TSA was undertaken to assess frequency and duration of fish passage events for current flow. Analysis indicated that there was an average of 6.5 events which provided small fish passage in winter months, and each event lasted on average 23 days, with passage possible for approx 106 days each winter (i.e. small fish passage is currently possible for most of each winter). This is illustrated in Figure 86, indicating duration of passage exceeded in 80% of years.

When broken into months, the total duration of fish passage events across years (Figure 87) shows the classic pattern for southwestern W.A. rivers, whereby fish passage commences in May, increasing through June and July to a peak in August, before declining through spring. The data show occasional flows sufficient for fish passage in January and February, but these reflect summer rainfall events, and given the life history of fish species present, would be unlikely to be used for upstream passage.

Mean monthly duration of fish passage events across years (Figure 88) showed a similar pattern to total duration, indicating little variation in extent of passage within each month across years.

Therefore, passage for small bodied species of fish requires flows of 0.07 m^3/sec, and this currently occurs through the majority of each winter.

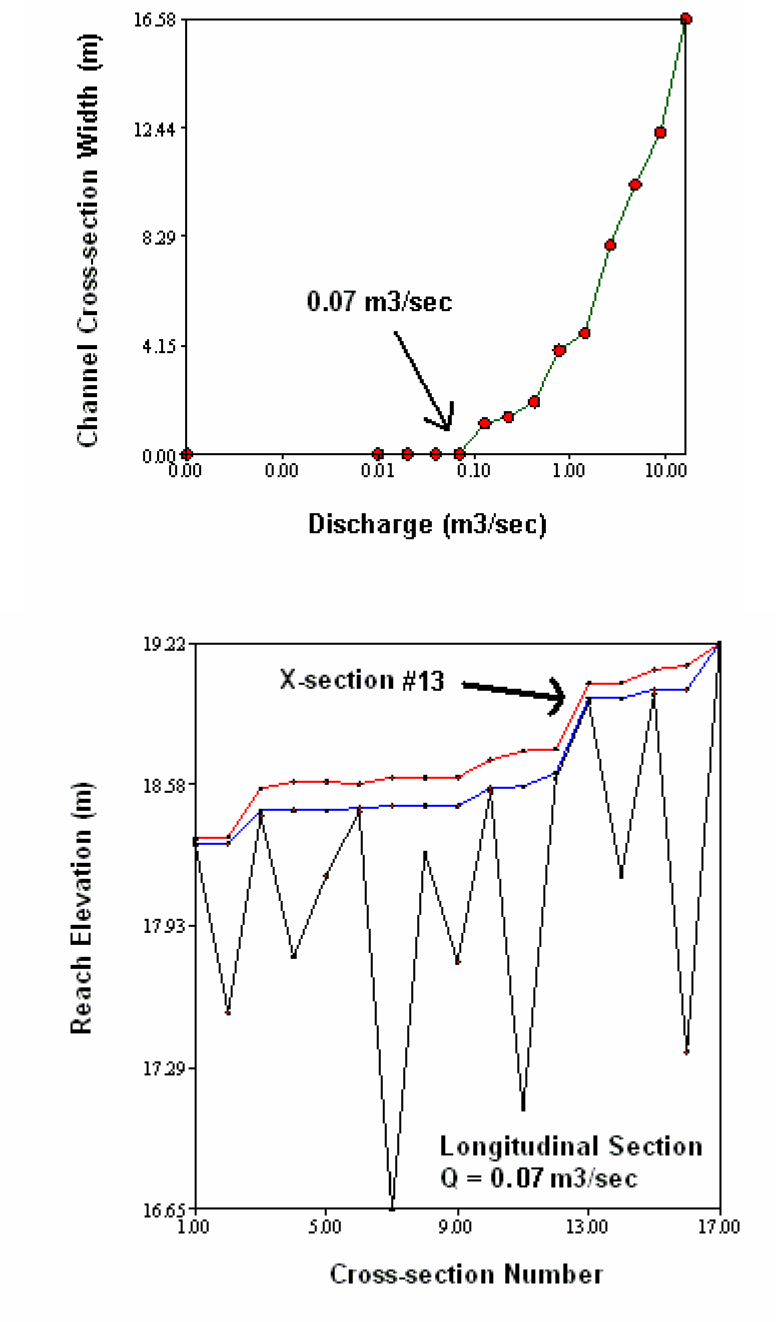


Figure 85. Reach level rating curve for passage of small bodied fish on the Bingham River, showing change in channel width (y-axis) with increasing discharge (m^3/sec) (x-axis). The small bodied fish passage criteria of 10 cm minimum threshold depth was achieved with a discharge of 0.07 m^3/sec (TOP), with cross-section #13 on the longitudinal section (BOTTOM) being the critical point for flow depth.

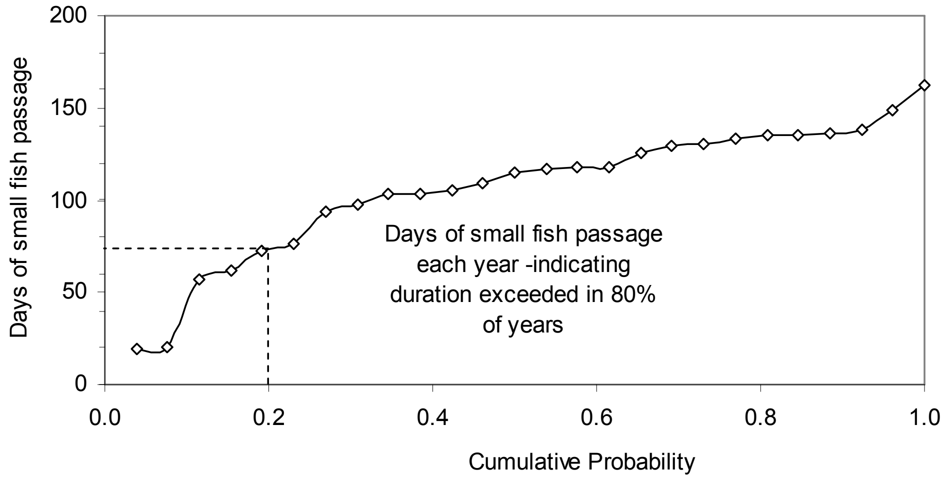


Figure 86. Cumulative probability plot for duration of fish passage for Bingham for small bodied species of fish, showing effects at the 20% probability level of each diversion scenario (i.e. days of fish passage exceeded in 80% of years).

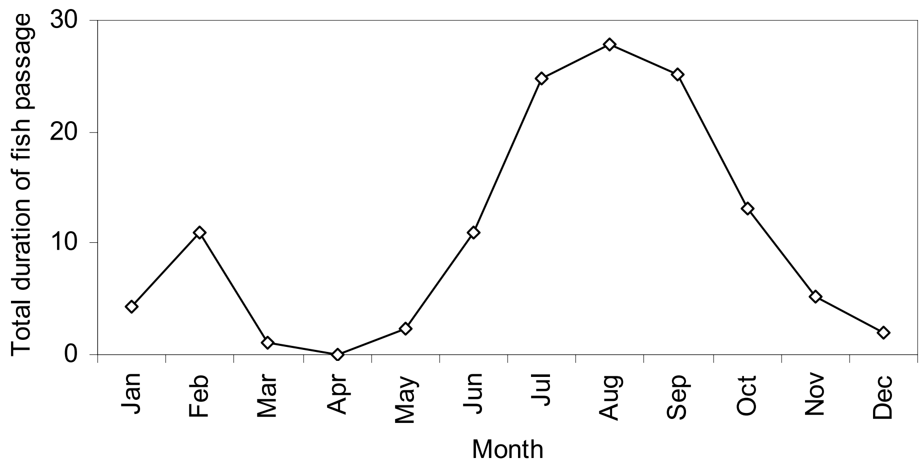


Figure 87. Total duration of fish passage for Bingham for each month for current flows where discharge exceeded 0.07 m^3/sec.

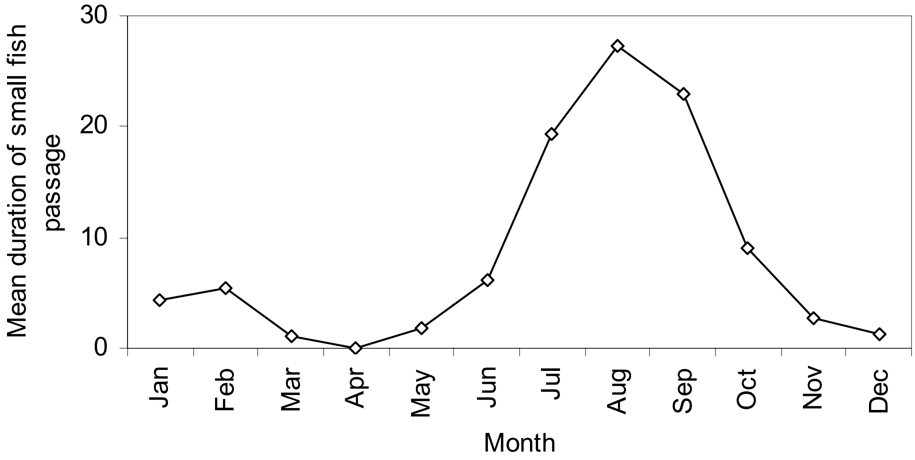


Figure 88. Mean duration of small fish passage for Bingham for each month for current flows where discharge exceeded 0.07 m^3/sec.

Fish passage for large bodied species (0.2 m minimum depth)

Hydraulic analysis was performed to determine flows that achieved a minimum depth of 0.2 m for the reach to allow passage of large bodied species of fish. Analysis produced a reach-level rating curve for Bingham (Figure 89) which indicated a discharge threshold of 0.23 m^3/sec, below which large fish passage was not possible, and above which passage for large bodied fish was possible. Cross-section #15 was the shallowest point on the reach affecting fish passage, although other riffle cross-sections were comparably shallow (see Figure 85).

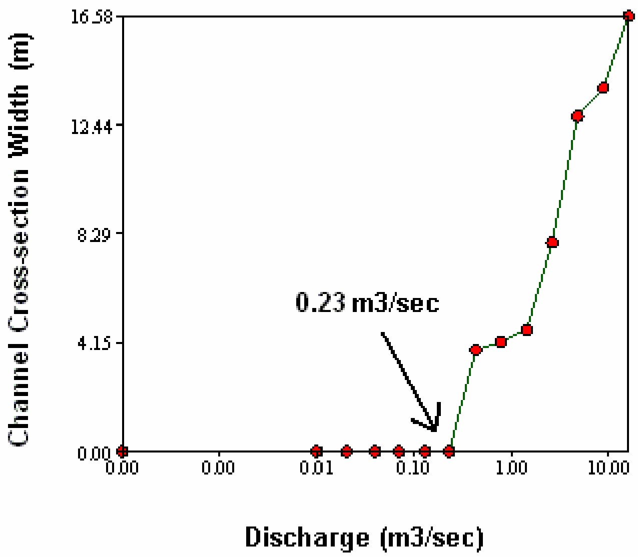


Figure 89. Reach level rating curve for passage of large bodied fish on the Bingham River, showing change in channel width (y-axis) with increasing discharge (m^3/sec) (x-axis). The large bodied fish passage criteria of 20 cm minimum threshold depth was achieved with a discharge of 0.23 m^3/sec, with cross-section #15 being the critical point for flow depth.

TSA was undertaken to assess frequency and duration of fish passage events. Analysis indicated that, currently there was an average 7 events which provided fish passage in winter months, and each event lasted on average 15 days, with passage possible for approx 73 days each winter. The cumulative probability plot for large fish passage shows duration of fish passage exceeded in 80% of years (Figure 90).

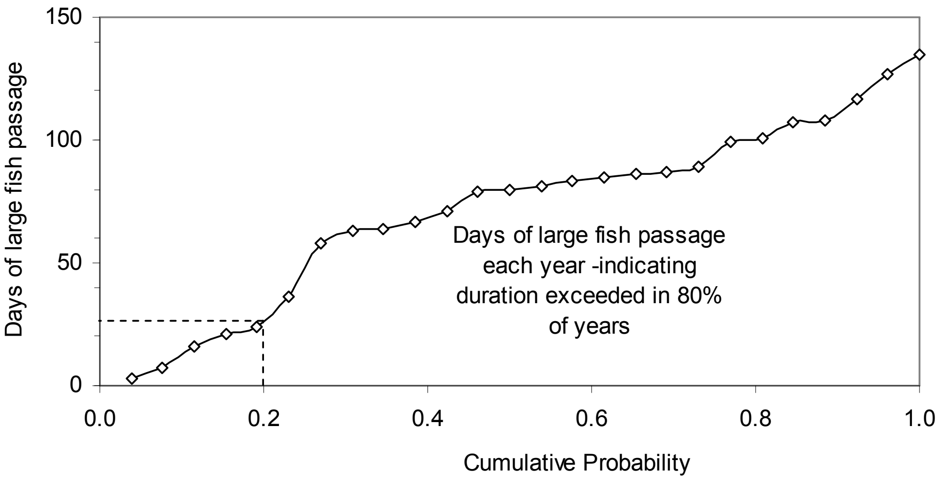


Figure 90. Cumulative probability plot for Bingham for duration of fish passage for large bodied species of fish at the 20% probability level (i.e. days of fish passage exceeded in 80% of years).

When broken into months, the total duration of fish passage events each month (Figure 91) shows again the classic pattern for southwestern WA rivers, whereby passage for large fish species starts to increase in May, June and July to a peak in August, before declining through spring.

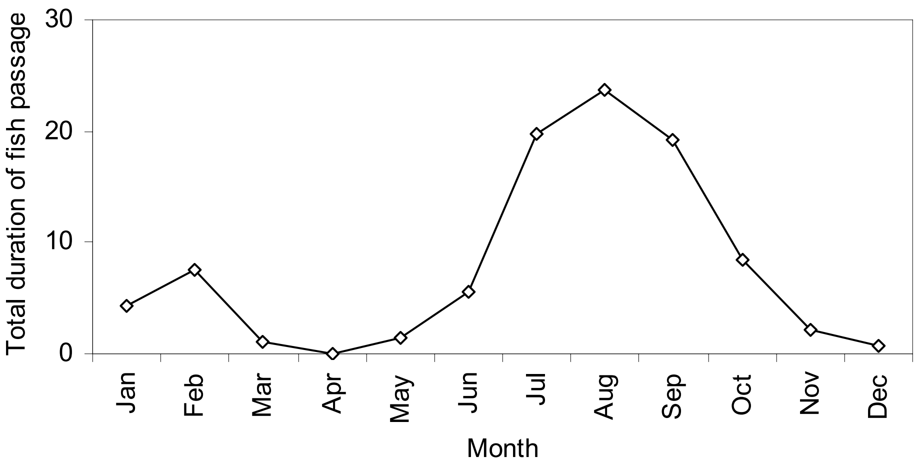


Figure 91. Mean of total duration of fish passage for Bingham for each month for current flows where discharge exceeded 0.23 m^3/sec.

Mean number of fish passage events each month shows a similar pattern, with greatest frequency in June to September, with on average 1.7 passage events each month (Figure 92).

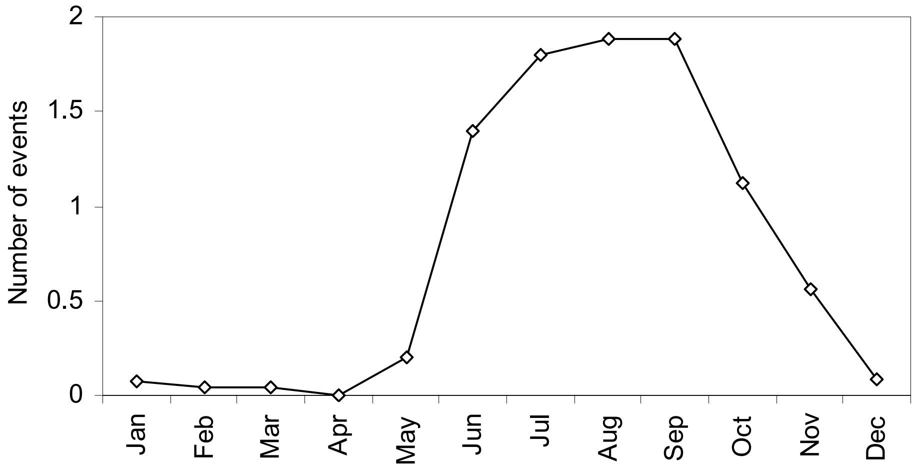


Figure 92. Mean number of large fish passage events for Bingham for each month for current flows where discharge exceeded 0.23 m^3/sec.

Analysis of total number of days of passage for large fish each year shows high inter-annual variability, with total duration ranging from 135 days in wet winters to 3 days in dry years (Figure 93). This indicates that in some years passage for large fish species is often greatly reduced.

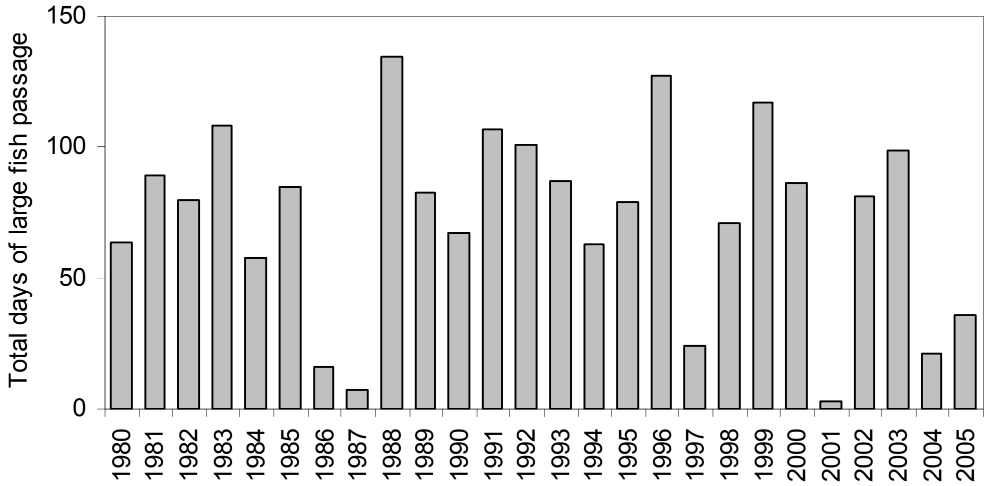


Figure 93. Total number of days of fish passage possible in winter in each year for Bingham where discharge exceeded 0.23 m^3/sec.

Passage for large bodied fish species, which requires flows of 0.23 m^3/sec, occurs less often and for shorter durations than for small bodied species.

10.2.3.4 Flows to inundate medium elevation benches

An objective of winter flows is to inundate medium benches to provide inputs of allochthonous energy (leaf litter/detritus) from benches into the river. This objective was set for Coolangatta Reach, based on benches identified from channel cross-section surveys. However, the channel on the Bingham River was much more uniform than at Coolangatta, consisting of a broad, shallow channel, with no benches visible on cross-sections. Therefore, a specific flow objective for medium elevation benches could not be determined, and therefore was not incorporated in the EWR assessment for the Bingham River. However, the transport of allochthonous material will occur progressively with increasing flows up to active channel stage height and to top of bank stage heights. Therefore, this process will occur as part of these other flow objectives.

10.2.3.5 Active Channel Flows

Active channel or channel forming flows are events typically occurring on a 1:2 to 1:3 year frequency in south-west river systems. Active channel flows help maintain the shape of the channel by mobilising sediment, scouring pools and preventing riparian vegetation encroaching into the channel. These flows also inundate trailing vegetation to provide cover and spawning habitat for fish, and provide inputs of autochthonous (algal production) and allochthonous energy (leaf litter/detritus) from banks.

The current active channel stage height was determined from cross-sectional surveys as the level on the banks above which vegetation is stable and below which the bank is eroding/bare, and without extensive riparian vegetative growth. Using all cross-sections, with the exception of those in deep pools, the average depth from the bed to active channel stage height, taken as the deepest part of the channel (thalweg depth) to the level approximated as active channel was determined from cross-sections as 0.26 m. Using this stage height (*i.e.* a thalweg depth of 0.26 m), the reach level rating curve produced by hydraulic analysis calculated a discharge of 0.23 m^3/sec to achieve active channel flows (Figure 94).

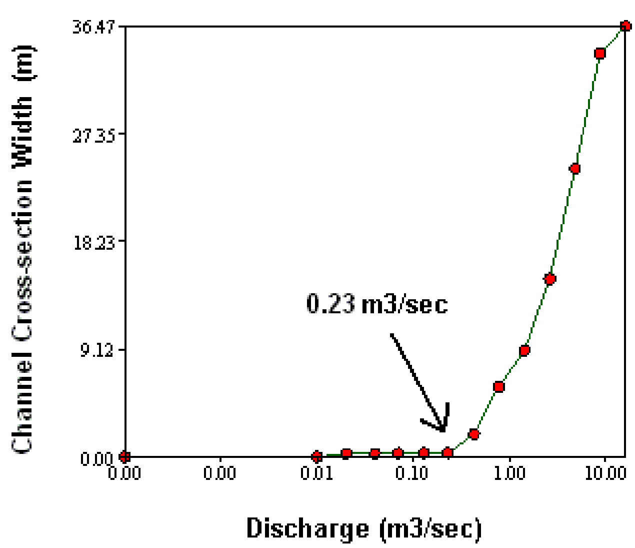


Figure 94. Reach level rating curve for Bingham for discharge to attain active channel stage height, with threshold flow of 0.23 m^3/sec.

This discharge is the same as the large fish passage discharge and therefore any TSA of the active channel flow event would produce the same flow statistics as for the large fish passage flows analysis.

10.2.3.6 Flows to connect off-river wetlands/sumps

The Bingham River at the survey reach is a fairly broad, shallow creekline, with often ill-defined banks and with much riparian vegetation growing across the bed of the channel. On several cross-sections there were small sumps/wetlands which, although isolated from the channel at the time of survey, still contained water in late spring, and so likely provided spawning/breeding/nursery habitat for frogs and tortoise, as well as opportunistic habitat for macroinvertebrates, and feeding habitat for waterbirds (ducks, egrets, herons). Cross-sections 4 and 14 were specifically placed to include these wetlands in the survey (Figure 95). HA was used to determine the flows required to inundate these wetlands/sumps. RAP indicated that the discharge required to provide flows to the wetland on cross-section #4 and #14 were 0.24 and 0.22 m^3/sec respectively, at which flow it was assessed that a sufficient stage height was achieved for flows to pass into these wetlands. This flow was coincidentally approximately the same flow required to allow passage of large bodied fish (and also for Active channel flows) (see above). Therefore, results from TSA for large fish passage also apply to flows to connect these off-river sumps/wetlands.

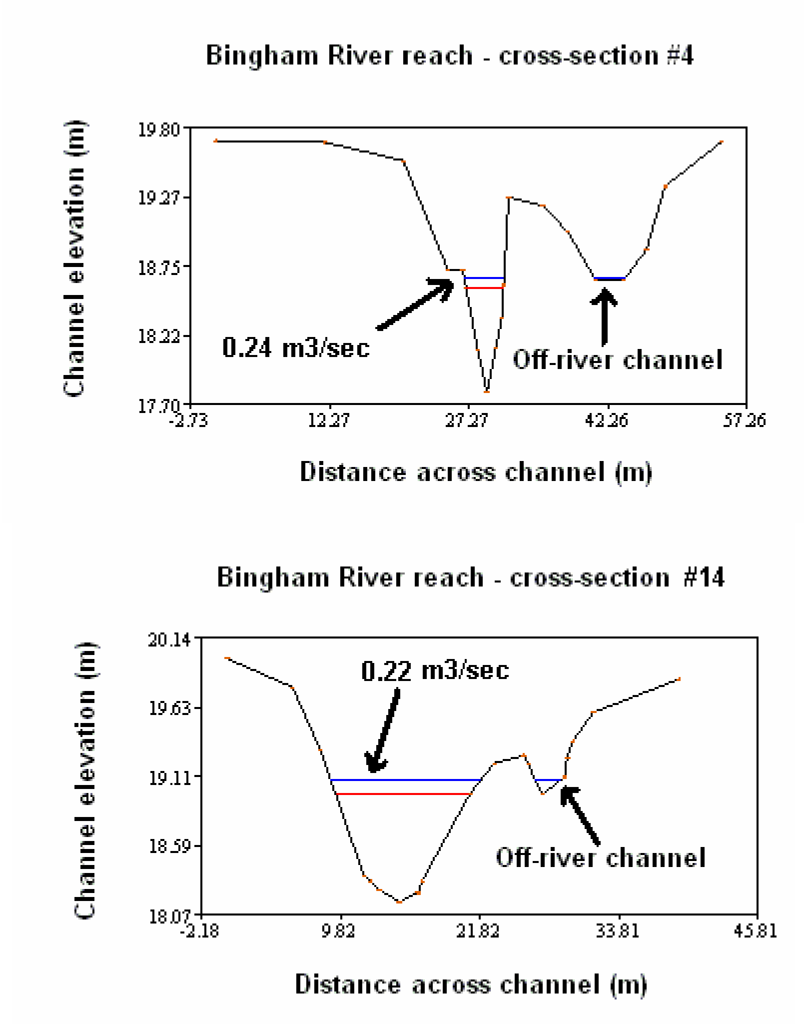


Figure 95. Flow magnitude required to inundate off-river sumps/wetlands located on cross-sections 4 and 14 on the Bingham, showing water levels sufficient to achieve this stage height (~ 0.23 m^3/sec).

10.2.3.7 Winter High Flows

The objective of winter high flows is to exceed top of bank (TOB) stage heights to start inundation of the floodplain and associated riparian vegetation. The floodplain often supports shallow wetland/sump areas that are likely important feeding and nursery areas for frogs and provide habitat in which juvenile stages with poor swimming ability may avoid high flows. Overbank flows are required to inundate and recharge these wetlands. Similarly, riparian vegetation on the floodplain (*i.e. Eucalyptus rudis* and paperbarks) also requires occasional inundation to a.) disperse seed, and b.) assist seed set, and soak soil profiles to promote successful germination. To achieve the above objectives, a flow above TOB is required. These flows usually occur as a result of larger rainfall events, principally in winter, and occur infrequently and for short durations.

The current TOB stage height was determined from cross-sectional surveys as the level on the cross-sections above which the bank profile started to decline onto a flatter floodplain. This can usually be detected in the HA habitat rating curves by a change in habitat area relating to a change in bank gradient. This inflection in the Bingham River reach habitat rating curve showed a rapid increase in channel width at about 1.6 m^3/sec, reflecting a rapid expansion of the wetted area at this discharge. This reflects a rapid increase in wetted perimeter as overbank flows flood relatively flat floodplain above TOB stage height (Figure 96).

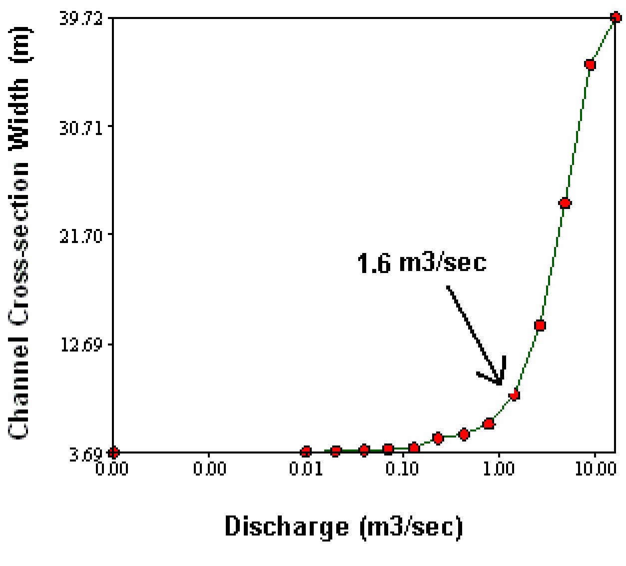


Figure 96. Reach level rating curve for TOB stage height on Bingham River Reach, with floodplain inundation occurring at approximately 1.6 m^3/sec.

TSA of the TOB flow event determined that currently in winter there was on average 2 events of 1.6 m^3/sec or greater that reach TOB stage height each year. Each event had a mean magnitude of 3 m^3/sec, lasted on average 3 days, with approx. 12 days of flows above TOB each winter. This frequency and duration of TOB flows seems relatively high for what is normally a high magnitude-low frequency event. This may reflect the condition of the catchment, the high roughness of the channel along this reach due to the dense riparian vegetation across the channel, or the accuracy/inaccuracy of the uncalibrated hec-ras model.

Analysis of total number of days of overbank flooding each year shows high inter-annual variability, with total duration ranging from 64 days in wet winters to 0 days in dry years, with a median of 2 days and an average of 8 days. These statistics show that the mean duration is heavily influenced by a few large events of long duration (i.e. 1983 and 1996) (Figure 97 & 98). In the 26 year flow record there were 7 years in which there was no overbank flooding (Figure 98).

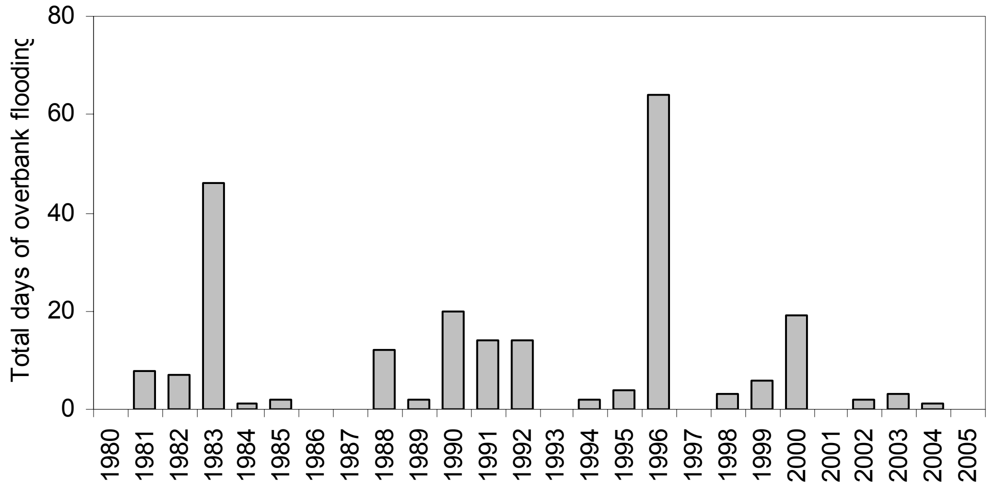


Figure 97. Total number of days overbank flooding in each year for the Bingham where discharge exceeded 1.6 m^3/sec.

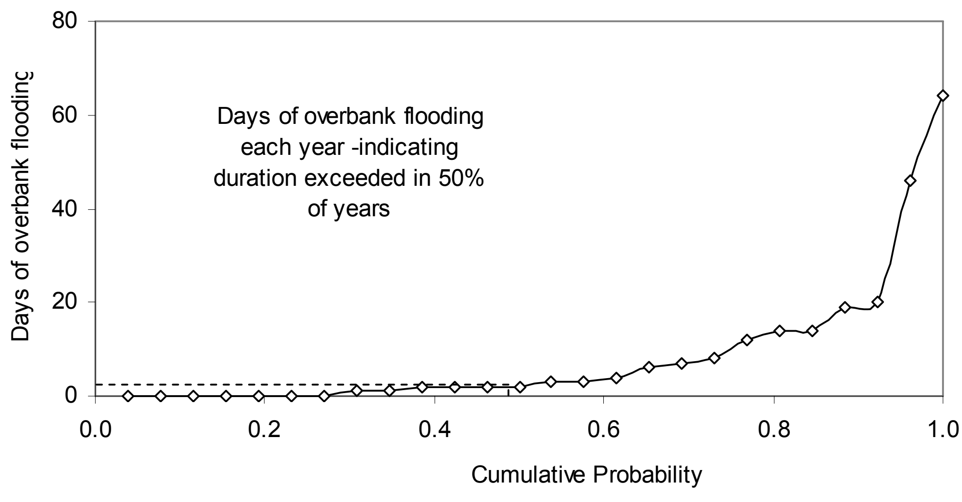


Figure 98. Cumulative probability plot for days of overbank flooding for the Bingham in each year where discharge exceeded 1.6 m^3/sec.

When broken into months, the mean number, total duration and mean monthly duration of overbank flows each month (Figure 99, 100 & 101) were greatest in winter, as would be expected, with a tendency for greater monthly duration towards late winter/early spring (Figure 101).

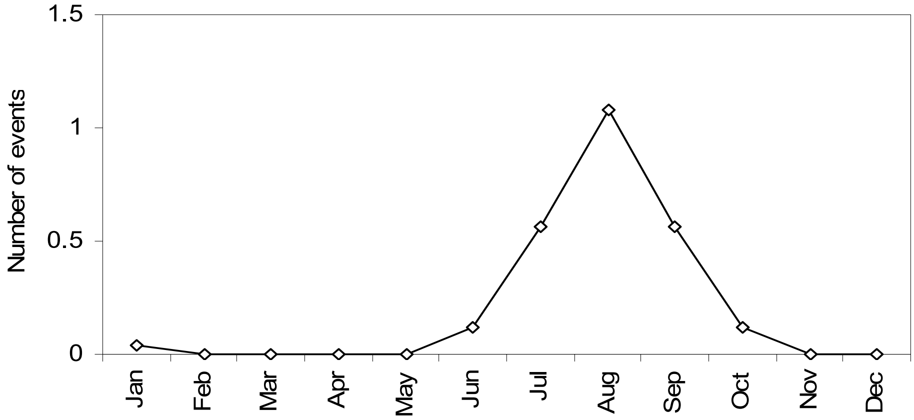


Figure 99. Mean number of flow events each month in winter to reach TOB stage height for the Bingham, where discharge exceeded 1.6 m^3/sec**.**

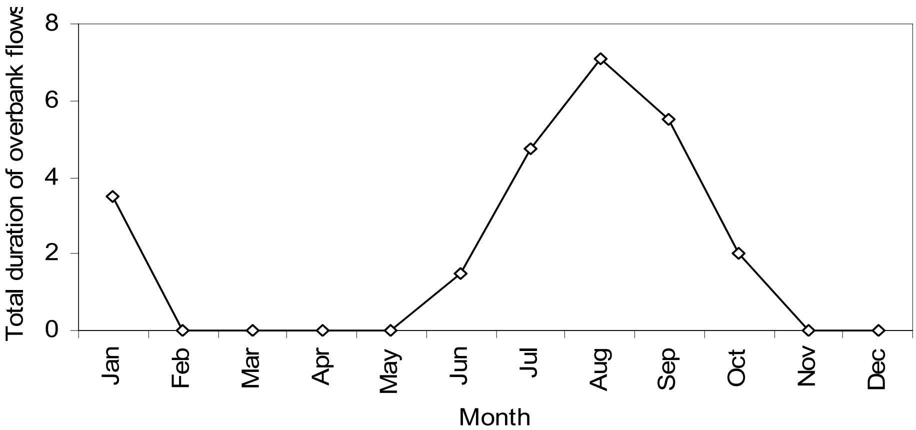


Figure 100. Total duration of flow events each month in winter to reach TOB stage height for the Bingham, where discharge exceeded 1.6 m^3/sec.

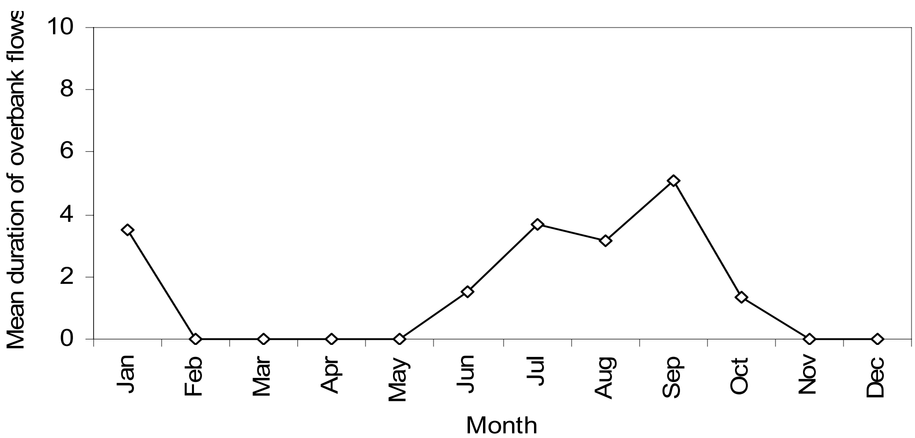


Figure 101. Mean duration of flow events each month in winter to reach TOB stage height for the Bingham, where discharge exceeded 1.6 m^3/sec**.**

HA determined that flows of 1.6 m^3/sec are required to provide overbank flooding in the Bingham River on the survey reach. The frequency and duration of TOB flows seems relatively high for what is normally a high magnitude-low frequency event. This may reflect the condition of the catchment, the high roughness of the channel along this reach, or the level of accuracy of the uncalibrated hec-ras model.

10.2.3.8 Pool Maintenance

The Bingham River at the survey reach contained only a few small, shallow, receding pools in early/mid summer, and the majority of these would likely have dried by the end of summer. There was one large permanent pool upstream of the survey reach which contained cobbler and long-necked tortoise. Permanent pools surrounded by a seasonally flowing water course provide critical summer habitat/refuge for water dependent ecological values, as well as deep water habitat during winter. Maintenance of the quantity and quality of water in these pools is therefore critical to sustain the ecological values which are dependent on them (i.e. macroinvertebrates, fish and tortoise). Water levels in permanent pools in seasonally flowing systems are often driven by groundwater influence, either in the form of seeps and springs or through direct surface expression of the groundwater. Determination of riverine EWRs to maintain these pools in summer is not relevant in a seasonally-flowing system, however, consideration should be given to the current inter-relationship with groundwater inflows.

Table 44. Summary of ecological flow recommendations as determined for the East Collie and Bingham Rivers using the Flow Events Method and River Analysis Package.

Flow Component	Ecological attribute / Value	Season (duration)	Hydraulic metric	Ecological Flow Recommendation COOLANGATTA	Ecological Flow Recommendation BUCKINGHAM	Ecological Flow Recommendation BINGHAM
Summer baseflow	Invertebrates	summer	50% lateral coverage of riffles to a depth of 0.05 m	Discharge required = 0.001 m^3/s Maintain current duration and frequency of summer flows, allowing for periods of zero flow	Discharge required = 0.04 m^3/s Maintain current duration and frequency of summer flows, allowing for periods of zero flow	Maintain zero flow days, 188 days in summer (nov – April)
Winter Base Flows	Invertebrates Native fish Vegetation Process – energy transfer	Winter	100% lateral coverage of riffles to a depth of 0.05 m	Discharge required = 0.01 m^3/s Maintain current duration and frequency of winter flows	Discharge required = 0.07 m^3/s Maintain current duration and frequency of winter flows	Discharge required = 0.02 m^3/s Frequency = 7 events each year Duration = 22 days for each event
Fish Passage Flow (small bodied)	Native fish	Late autumn - early spring	Minimum depth for the reach at; Coolangatta = 0.10 m Buckingham = 0.10 m Bingham = 0.10 m	Discharge required = 0. 04 m^3/s Frequency = 2.9 events each year Duration = 69 days for each event	Discharge required = 0.15 m^3/s Frequency = 5.9 events each year Duration = 28 days for each event	Discharge required = 0.07 m^3/s Frequency = 6.5 events each year Duration = 23 days for each event
Fish Passage Flow (large bodied)	Native fish	Late autumn - early spring	Minimum depth for the reach at Coolangatta = 0.20 m Buckingham = 0.20 m Bingham = 0.20 m	Discharge required = 0.23 m^3/s Frequency = 5.4 events each year Duration = 34 days for each event	Discharge required = 1.50 m^3/s Frequency = 9.6 events each year Duration = 7.2 days for each event	Discharge required = 0.23 m^3/s Frequency = 7 events each year Duration = 15 days for each event
Active Channel Flows	Channel morphology	Winter	Stage height required at; Coolangatta = 1.64 m Buckingham = 1.10 m Bingham = 0.26 m	Discharge required = 3.0 m^3/s Frequency = 9 events each winter Duration = 6 days for each event	Discharge required = 2.70 m^3/s Frequency = 8 events each winter Duration = 4 days for each event	As above (for large fish passage)
Winter Bench Inundation Flow	Native fish Vegetation Process	Winter	Stage height required for channel evulsion/over topping into anabranch channels as defined on cross-sections	Discharge required = 9 m^3/s Frequency = 2.84 events each winter Duration = 3 days for each event	N/A	N/A
Flows to connect off-	Aquatic fauna	Winter	Stage height required for	As above (for winter bench	Discharge required = 2.7	As above (for large fish

Flow Component	Ecological attribute / Value	Season (duration)	Hydraulic metric	Ecological Flow Recommendation COOLANGATTA	Ecological Flow Recommendation BUCKINGHAM	Ecological Flow Recommendation BINGHAM
river waterbodies/ wetland sumps	Wetlands Vegetation		channel evulsion/over topping into anabranch channels as defined on cross-sections	inundation)	m^3/s	passage)
Winter High Flow	Inundate riparian zone/floodplain	Winter	Stage height required at; Buckingham = 2.0 m Bingham = inflection point on habitat rating curve m	As above (for winter bench inundation)	Discharge required = 8.8 m^3/s Frequency = 1.7 events each winter Duration = 2.6 days for each even	Discharge required = 1.6 m^3/s Frequency = 2 events each winter Duration = 3 days for each even

11 EWRS AS A TOTAL ANNUAL VOLUME

For the purpose of determining an allocation plan for the East Collie, it is necessary to know the total environment flows required by the different parts of the system, with the difference between the environmental flows and total system yield being potentially available for consumptive uses/diversion. The EWRs for the above reaches and ecological values are based on summer and winter base flows and a range of events of varying magnitude and duration (i.e. active channel flows) which may or may not occur each year depending on annual rainfall. It is possible to estimate a total environmental flow by totalling the volumes required to achieve each event, multiplied by the number of events required, added to summer and winter base flows for an "average" year. However, this approach can only be a conservative approximation for several reasons:

- The concept of an "average" year is indicative only, with flows in any year dependent upon rainfall, with the EWR (and licensed diversion) ultimately to be dependent upon annual rainfall so that it will mimic natural variability (seasonal and inter-annual) in discharge. As such, aspects such as commencement and termination of winter base flows will vary between years,
- The EWRs apply to the survey reaches, where flow will be a combination of local catchment run-off and groundwater recharge and discharge. The relative proportion of total flow from these sources is unknown, and will vary seasonally and annually.
- The calculated volume in each event will depend on the shape of the flood hydrograph which will vary intra and inter-annually. Therefore it is not possible to definitively define the shape of hydrographs for all events, but approximate shapes for events such as fish passage flows, active channel flows and over bank flows were approximated based on instantaneous flows and previous EWR studies on similar sized channels/catchments.
- Some flows will not occur every year, and other flows (i.e. winter medium or fish passage flows) may occur with varying frequency between years. As a conservative approach, it is assumed they occur every year at the mean annual frequency calculated by TSA in RAP. This frequency will vary between years.
- In years when a higher flow event occurs, it is assumed it will encompass a range of lower flow objectives (i.e. an overbank flow of > 3.0 m^3/sec will also provide a winter medium flow and a fish passage flow). This was provided for in the calculations, however, in years when an active channel flow does not occur, then additional winter medium and fish passage flows will need to be added.

There is also a question over the accuracy of some aspect of the modelling for all three reaches, as the hec-ras models have not been fully calibrated against ratings curves derived from a range of discharge measurements for each reach. Therefore, the accuracy of the models should be confirmed before the estimated annual volumes as accepted and used to set a binding, licensed allocation.

The EWR as an annual total volume for each reach is presented in Table 45. Considering the above caveats/assumptions, a total annual environmental flow of 3.6 GL, 20.0 GL and 24.8 GL was determined for an "average" year for Bingham at Palmer, and East Collie at Buckingham and Coolangatta respectively, which equates to 57%, 68% and 58% of the mean annual flow (1980 – 2005) for each respective reach (Table 45). In wet years the EWR volume is approximately 19% of the annual flow, and in dry years the EWR is 4 x the annual flow (10 x for Bingham), however, it is anticipated the EWR would be varied relative to each winters rainfall.

Table 45. EWRs as an annual total volume for the Bingham at Palmer and East Collie at Buckingham and Coolangatta, calculated by totalling base flows and individual events using magnitude, frequency and duration, determined for each reach using the Flow Events Method and River Analysis Package (see Table 44).

Volumes	Bingham @ Palmer	East Collie @ Buckingham	East Collie @ Coolangatta
EWR as Annual Total Volume (GL)	3.60	20.00	24.76
Mean Annual Flow (MAF 1980 - 2005; GL)	6.33	29.36	42.87
Maximum MAF 1980 – 2005 (GL)	22.70	90.93	135.86
Minimum MAF 1980 – 2005 (GL)	0.34	5.12	6.61
EWR as % of MAF	56.8%	68.1%	57.8%
EWR as % of Maximum MAF	15.8%	22.0%	18.2%
EWR as % of Minimum MAF	1071.5%	390.9%	374.5%

12 FUTURE STUDIES/MONITORING

As discussed above, it must be assumed that aspects of the modelling are inaccurate until more calibration is undertaken. To confirm the validity of the calculated flows, it is recommended that monitoring of each reach is conducted to relate flows to ecological and hydrological objectives. Subsequent monitoring then assesses the effectiveness of the EWRs in maintaining ecological values at a low level of risk. A tiered approach to monitoring is recommended, whereby Tier 1 monitoring would test whether the recommended flows achieve the desired stage heights (*i.e.* flow to inundate riffles in winter, flow to give desired depth for fish passage over obstacles), Tier 2 monitoring would then be considered as Compliance Monitoring, used to test whether the specified flows are actually released/provided, and Tier 3 monitoring would assess overall effects of the environmental flows on the ecological condition of the system (*e.g.* changes in riparian vegetation, loss of species of aquatic macroinvertebrates, changes in population structure of native fish species). An adaptive context would require revision of flows/management actions in response to results of monitoring. For example, if flows are insufficient to cover riffles in winter (Tier 1), then recommended flows would need to be increased. The monitoring tiers effectively reflect monitoring priorities with Tier 1 monitoring taking highest priority. Tier 1 monitoring would also clarify the above-discussed issue over flows and hydrology.

Tier 1 Monitoring: Flow Objective Monitoring

Flow objectives to test are:

- Is summer baseflow adequate to inundate riffles to 5 cms average depth with average of 25% (50%) lateral coverage;
- Is winter baseflow adequate to inundate riffles to 5 cms average depth with average of 100% lateral coverage;
- Is the recommended fish passage flow adequate to give a minimum depth threshold for the reach of 10 cms for small bodied fish and 20 cms for large bodied fish;
- Is the recommended active channel flow adequate to provide a stage height equivalent to active channel stage height;
- Is the recommended winter medium flow event sufficient to inundate lower benches for energy transfer.
- Is the recommended Top Of Bank flow sufficient to achieve TOB flows.

Monitoring will require measurement of discharge at each survey reach under the varying flow conditions (*e.g.* seasonally and during flood and drought events). Stage height (depth) must be measured for each discharge to assess whether the objectives were met, with depth measured relative to the actual objectives (i.e. depth on riffles, benches etc).

Tier 2 Monitoring: Compliance Monitoring

The aim of Tier 2 compliance monitoring is to determine if the frequency, duration and magnitude (volume) of the various flows requested under the EWR are actually delivered. Tier 2 monitoring would only occur after any adjustments have been made following Tier 1 monitoring (i.e. were requested flows adequate to meet objectives). If agreed flows are not met, then the amount of water released/left in the system after abstraction/diversion will need to increase (or decrease) and Tier 2 monitoring repeated.

Tier 3 Monitoring: Ecosystem Health Monitoring

The ultimate aim of monitoring is to assess whether the calculated EWR flows maintain the observed ecological values at a low level of risk. This is achieved by monitoring the values and processes to be maintained.

Ecosystem Health objectives to test are:

- Has there been any change in macroinvertebrate community composition, species diversity and occurrence of rare/restricted distribution species following implementation of the revised flow regime;
- Has there been a change in the composition of native (& introduced) fish that may be indicative of increasing/decreasing population health;
- Has there been a change in the distribution of aquatic and riparian plants in and adjacent to the channel in response to the revised flow regime and/or changes in salinity.
- Do average seasonal salinity levels remain the same;
- Are fish able to migrate up through the study reach when the required flows are delivered;
- Has abstraction resulted in accumulation of sediment in pools or exposure of riffle areas due to inadequate flow?

Assessing these objectives will require the design of specific short-term programmes to measure attributes such as pool dissolved oxygen levels over 24 hrs during low flows, fish passage through a reach and bed/bank erosion/aggradation. Other Tier 3 objectives will require the design of on-going, low frequency monitoring programmes to sample macroinvertebrate assemblages (to species level), fish communities and population structure and composition and distribution of aquatic and riparian vegetation. Designs must be standardised for on-going monitoring purposes, using accepted methodologies, with frequency of application determined by anticipated rate of change in the attribute being monitored. Initially, monitoring may have a high frequency (*i.e.* annual for fish and macroinvertebrates), but then sampling may be reduced/stopped once there is confidence in there being no observed effect.

13 CONCLUSIONS

This study presents a comparative analysis of the likely risks to ecological values through modifying flows in the East Branch of the Collie River at Buckingham. The effects of the three diversion scenarios (5GL, 10GL and 15GL) on the magnitude, duration and frequency of ecologically important flow events for Buckingham and the downstream Coolangatta reaches are examined in detail, with similar analyses conducted on natural flows in the Bingham River for reference.

Analyses for the Buckingham Reach assessed changes in flow conditions in the relatively short reach between the diversion off-take point and the confluence with the Bingham River tributary. Ecological values surveys demonstrated that ecological values and condition of the East Collie in the region of the Buckingham Reach in general are degraded due to salinity effects, land clearing/loss of riparian vegetation and channel erosion. Channel erosion in particular is extreme, and this likely indicates that the catchment upstream is generating more flows that pre-European times. A result of this channel erosion is severe sedimentation of pools downstream. Therefore, reduction in peak flows may be beneficial to the system. Analyses indicated that the three diversion scenarios have a progressive and significant effect on magnitude, duration and frequency of medium to small events in the Buckingham area (i.e. what would be considered significant effects on flows such as fish passage flows in autumn/winter). However, effects on peak flows are minor, principally due to the small proportion of peak flows that can be diverted due to infrastructure limitations. Effects on hydrology would be regarded as significant, however, because of the degraded condition of the reach and it's ecology, it may be argued that these effects are not of great concern, and may be justifiable in light of perceived benefits discussed below.

Flows at the downstream Coolangatta Reach show a balance between the upstream diversions and inflows from the Bingham River, whereby inflows from the Bingham River ameliorates loss of discharge from the upstream diversions by adding significant inflows. Essentially, the Bingham inflows become a greater proportion of flows in the river at Coolangatta. The added benefit of this effect of the diversions is that freshwater inputs from the Bingham River assist in ameliorating riverine salinity, with the Bingham comprising a greater proportion of flows after diversions than before. Analysis of flows for the Coolangatta Reach therefore indicates EWRs and flow/ecological conditions applicable to the East Collie downstream as far as Harris River inflows.

Analyses demonstrated that implementation of the diversion scenarios (5GL, 10GL and 15 GL) will have:

- little influence on summer flows,
- minimal effect on proposed winter base flows,
- little effect on frequency or duration of inundation of sand / gravel run habitats,
- a small reduction in the total duration of fish passage each winter,
- a small reduction in the total duration of bench inundation flows each winter,
- a small effect on flows that connect off river channels, sumps and wetlands,
- a minimal effect on the magnitude, duration and frequency of active channel flows, and
- no detectable effect on the magnitude, duration and frequency of top-of-bank winter high flows.

Therefore, it appears that the three diversion scenarios have minimal effect on the magnitude, duration and frequency of medium to small events in the Coolangatta area, and similarly, effects on peak flows are hardly detectable. Where there are detectable effects, the percentage change is greatest for the higher diversion scenario (15GL) and effects are progressively less for the 10GL and 5GL scenarios. It appears that the slight reductions in some hydrological metrics (i.e. fish passage flows, bench inundation flows etc) are likely off-set by the benefits to water quality (reduced salinity) for the Coolangatta reach, and for the East Collie from here as far downstream as the confluence with the Harris River. Although not assessed in this study, freshwater inputs from the Harris will help to further alleviate salt-stress on aquatic fauna and flora between Harris confluence and the confluence with the South Collie.

Given that salinity is a major threat to ecological values of rivers in south-western Australia, and salt levels in many rivers exceed critical thresholds for survival of aquatic fauna leading to localised extinctions of sensitive taxa, the ability to reduce salt levels, even at the expense of marginally lower flows will likely benefit the fauna. Reduced salinities as a result of the planned diversions may therefore allow for the continued survival of aquatic fauna and flora in the section of the East Collie between the Bingham River confluence and the South Collie confluence. Assuming the proposal to restore the upper catchment of the East Collie is implemented, in the long term there will be an improvement in water quality, and therefore habitat suitability for aquatic fauna in the upper East Collie. Ecological attributes (i.e. fish, macroinvertebrates, crustaceans, tortoise, water rats, etc) that have survived in the lower reaches of the East Collie as a result of reduced salinity may then recolonise the Buckingham Reach and reaches further upstream from this downstream refuge zone.

REFERENCES

Allen GR, 1989. *Freshwater Fishes of Australia.* Neptune City: T.F.H. Publications.

Allen GR, Midgley SH & Allen M, 2002. *Field Guide to the Freshwater Fishes of Australia.* Western Australian Museum, Perth WA.

ANZECC/ARMCANZ, 2000. Australian and New Zealand guidelines for fresh and marine water quality. www.ea.gov.au/water/nwqms/index.html

ARL, 1986. Canning Catchment 1984/85 Results and Recommendations. Stream Biological Study, Aquatic Research Laboratory, The University of Western Australia, Report 3, pp 50. Report to the Water Authority of Western Australia.

ARL, 1988a. Biological Energy Flow in a Lower River Ecosystem. Stream Fauna Study, Aquatic Research Laboratory, The University of Western Australia, Report 19, pp 20. Report to the Water Authority of Western Australia.

ARL, 1988b. Biological Energy Flow in a Lower River Ecosystem: Seasonal Differences. Stream Fauna Study, Aquatic Research Laboratory, The University of Western Australia, Report 21, pp 22. Report to the Water Authority of Western Australia.

ARL, 1988c. Canning Reservoir Catchment Results and Recommendations. Stream Fauna Study, Aquatic Research Laboratory, The University of Western Australia, Report 10, pp 60. Report to the Water Authority of Western Australia.

ARL, 1988d. North Dandalup Results and Recommendations. Stream Fauna Study, Aquatic Research Laboratory, The University of Western Australia, Report 9, pp 77. Report to the Water Authority of Western Australia.

ARL, 1989. Canning Reservoir Catchment; Lower Canning River Catchment; Serpentine/Gooralong Brook Catchments; Dirk Brook Catchment; North Dandalup Pipehead Dam Catchment Results and Recommendations 1988. Stream Fauna Study, Aquatic Research Laboratory, The University of Western Australia, Report 15, pp 111. Report to the Water Authority of Western Australia.

ARL, 1990a. The effect of gauging station structures on the upstream movement of *Galaxias occidentalis* Ogilby in streams of the northern jarrah forest. Unpublished report to the Water Authority of Western Australia. Report ARL 018 p 23.

ARL, 1990b Habitat differences in Upland Forested Streams. Stream Fauna Study, Aquatic Research Laboratory, The University of Western Australia, Report 20, pp 46. Report to the Water Authority of Western Australia.

ARL, 2005. Ecological Water Requirements of Marbling Brook – Intermediary Assessment. Unpublished report to Marbling Catchment Group by Aquatic Research Laboratory, The University of Western Australia. May 2005.

Arthington AH & Lloyd L, 1989. Introduced Poeciliidae in Australia and New Zealand. IN: G.K. Meffe and F.F. Snelson (eds). *Evolution and Ecology of Livebearing Fishes (Poeciliidae).* Prentice-Hall, New York. pp 333 – 348.

Arthington AH, Bunn SE & Gray M, 1993. Stream Hydrology and Flow Management in the Tully-Millstream Hydro-electric Scheme Area. Report to the Wet Tropics Management Agency, Qld. 80 pp.

Austin CM & Knott B, 1996. Systematics of the freshwater crayfish genus *Cherax* Erichson (Decapoda: Parastacidae) in south-western Australia: electrophoretic, morphological and habitat variation. *Australian Journal of Zoology* **44**: 223-258.

Bailey P, Boon P, & Morris K 2002 Australian Biodiversity - Salt Sensitivity Database. Land and Water Australia. http://www.rivers.gov.au/research/contaminants/saltsen.htm

Bamford MJ, Watkins D (1983) Kemerton wetlands: the vertebrate fauna and its regional significance. Perth, W.A.

Barnes RD, 1987. *Invertebrate Zoology.* Saunders College Publishing, Philadelphia.

Bayly IAE 1972 Salinity tolerance and osmotic behaviour of animals in athalassic saline and marine hypersaline waters. Annual Review of Ecology Systematics 3: 233-268.

Beatty SJ & Morgan DL, 2005. Monitoring the adequacy of Environmental Water Provisions for fish and crayfish communities of Samson Brook, Harvey River and Harris River. Unpublished report to the Water Corporation. August 2005.

Beatty S, Morgan D & Gill H, 2005. Role of life history strategy in the colonisation of Western Australian aquatic systems by the introduced crayfish *Cherax destructor* Clark, 1936. *Hydrobiologia* **549**: 219-237.

Beresford Q, Bekle H, Phillips H & Mulcock J, 2001. The Salinity Crisis. University of Western Australia Press, Perth, Western Australia.

Boulton AJ, 1989. Over-summering refuges of aquatic macroinvertebrates in two intermittent streams in central Victoria. *Transactions of the Royal Society of South Australia* **113**: 23 – 34.

Boulton AJ & Lake PS, 1992. The ecology of two intermittent streams in Victoria, Australia. II. Comparisons of faunal composition between habitats, rivers and years. *Freshwater Biology* **27**: 99- 121.

Britton DL & Brock MA, 1994. Seasonal germination from wetland seedbanks. *Australian Journal of Marine and Freshwater Research* **45**: 1445-1457.

Brown AV & Brussock PP, 1991. Comparisons of benthic invertebrates between riffles and pools. *Hydrobiologia* **220**: 99 - 108.

Bunn SE, 1985. Community structure and functional organization of streams of the northern jarrah forest, Western Australia. Unpublished Ph.D. thesis, The University of Western Australia.

Bunn SE, 1986. Spatial and temporal variation in the macroinvertebrate fauna of streams of the northern jarrah forest, Western Australia: functional organisation. *Freshwater Biology* **16**: 621-632.

Bunn SE, 1988. Processing of leaf litter in a northern jarrah forest stream, Western Australia: seasonal differences. *Hydrobiologia* **162:** 201- 210.

Bunn SE, 1995. Biological monitoring of water quality in Australia: workshop summary and future directions. *Australian Journal of Ecology* **20**: 22-227.

Bunn SE, Edward DHD & Loneragan NR, 1986. Spatial and temporal variation in the macroinvertebrate fauna of streams of the northern jarrah forest, Western Australia: community structure. *Freshwater Biology* **16**: 67-91.

Bunn SE, Davies PM & Edward DHD, 1989. The association of *Glacidorbis occidentalis* Bunn & Stoddart (Gastropoda: ?Hydrobiidae) with intermittently- flowing forest streams in south-western Australia. *Journal of the Malacology Society of Australia* **10**: 25-34

Bunn SE & Davies PM, 1990. Why is the stream fauna of the south-west of Western Australia so impoverished? *Hydrobiologia* **194**:169-176.

Bunn SE and Davies PM, 1992. Community structure of the macroinvertebrate fauna and water quality of a saline river system in south-western Australia. *Hydrobiologia* **248**: 143-160.

CALM, 2005. Records held in CALM's rare fauna files. Western Australian Department of Conservation and Land Management.

Cartwright DI, 1997. A Key to Species of Late Instar larvae of Australian Trichoptera (Families Dipseudopsidae, Glossosomatidae, Polycentropodidae, Pschyomyiidae, Ecnomidae, Philopotamidae and Tasimiidae). Melbourne Water, Western Treatment Plant, Werribee, Victoria.

Christensen P, 1982. The distribution of *Lepidogalaxias salamandroides* and other small freshwater fishes in the lower southwest of Western Australia. *Journal of the Royal Society of Western Australia* **65**: 131-141

Cottingham P, Stewardson M, Crook D, Hillman T, Roberts J, & Rutherford I, 2003. Environmental flow recommendations for the Goulburn River below Lake Eildon. Canberra, Cooperative Research Centre for Freshwater Ecology in association with CRC for Catchment Hydrology.

Cowan WL, 1956. Estimating hydraulic roughness coefficients. *Agricultural Engineering* **37**: 473-475.

Creagh S, Davies PM, Chandler L, and Lynas J, 2003. Aquatic ecosystems of the Upper Harvey River Catchment. Samson Brook, McKnoe Brook, Wokalup Creek, Wellesley Creek and Harvey River: fauna monitoring surveys November 2003. Streamtec Report ST 11.1/03. Unpublished report to the Water Corporation, Western Australia.

CSIRO, 2001. Climate change scenarios for the Australian region. Climate Impact Group: CSIRO Division of Atmospheric Research.

Dallas HF & Day JA, 1993. The effect of water quality variables on river ecosystems: a review. Freshwater Research Unit, University of Cape Town, Rondebosch. Report to the Water Research Commission, Pretoria. August 1993.

Davies PM, 1993. Ecosystem Ecology of Upland Streams of the Jarrah Forest, Western Australia. Unpublished PhD Thesis, Department of Zoology, The University of Western Australia. 236 pp.

Davies PM, Bunn SE, Arthington A & Creagh S, 1998. Environmental Water Requirements for Lowland River Systems on the Swan Coastal Plain. Report to Water & Rivers Commission. September 1998.

Davies PM, Storey AW & Creagh S, 1999. Environmental Water Requirements of Angove Creek and Limeburners Creek. Report 19/99. Unpublished report to the Water Corporation of Western Australia. Perth, Western Australia.

Davis J & Christidis F, 1997. A guide to wetland Invertebrates of Southwestern Australia. Western Australian Museum, Perth, Western Australia.

Davis JA, Rosich RTS, Bradley JS, Growns JE, Schmidt LG & Cheal F, 1993. *Wetlands of the Swan Coastal Plain. Volume 6. Wetland classification on the basis of water quality and invertebrate community data.* Environmental Protection Authority and Water Authority of Western Australia.

Department of Environment, 2003. Stream and catchment hydrology. River Restoration Report, number RR19. Department of Environment, Perth.

Edward DHP, Gazey P & Davies PM, 1994. Invertebrate community structure related to physico-chemical parameters of permanent lakes of the south coast of Western Australia. *Journal of the Royal Society of Western Australia* **77**: 51-63.

Edward DH, Storey AW & Smith MJB, 2000. Assessing river health in south-west Australia: comparison of macroinvertebrates at family level with chironomidae at species level. *Verhandlungen Internationale Vereinigung Limnologie* **27**: 2326: 2335.

EPA, 1987. The Effects of Gauging Station Control Structures on Native Fish Migration in Freshwater Streams of South West Australia. Environmental Protection Authority Bulletin 282.

Flannery T, 1995. Mammals of New Guinea. Reed Books, Chatswood, NSW.

Froend RH 1987 Effects of salinity and waterlogging on the vegetation of Lake Toolibin, Western Australia. Australian Journal of Ecology 12: 281-298.

Gill HS & Humphries PJ, 1995. An experimental evaluation of habitat chooice in three species of goby. *Records of the Western Australian Museum* **17**:231-233.

Gill HS & Potter IC, 1993. Spatial segregation amongst goby species within an Australian estuary, with a comparison of the diets and salinity tolerance of the two most abundant species. *Marine Biology* **117**:515-526.

Gooderham J & Tsyrlin E, 2002. The Waterbug Book: a Guide to Freshwater Macroinvertebrates of Temperate Australia. CSIRO Publishing, Collingwood, Victoria.

Halse SA, Scanlon MD & Cocking JS, 2002. Australia-Wide Assessment of River Health: Western Australia Bioassessment Report (WA Final Report). Monitoring River Health Initiative Technical Report no 7, Commonwealth of Australia and Department of Conservation and Land Management, Canberra and Wanneroo.

Halse SA, Rupreht JK & Pinder AM, 2003. Salinisation and prospects for biodiversity in rivers and wetlands of south-west Western Australia. Australian Journal of Botany 51: 673-688.

Harris WF, 1978. An ecological study of the Australian water rat (*Hydromys chrysogaster* Geoffroy) in southeast Queensland. MSc thesis. University of Queensland, Brisbane.

Hart B, Bailey, Edwards P, Hortle K, James K, McMahon A, Meredith C, & Swadling K 1991 A review of salt sensitivity of Australian freshwater biota. Hydrobiologia 210: 105-144.

Hart BT, Lake PS, Webb JA & Grace MR, 2003. Ecological risk to aquatic systems from salinity increases. *Australian Journal of Botany* **51**: 689-702.

Hawking JH & Theischinger G, 1999. *Dragonfly Larvae (Odonata): A Guide to the Identification of Larvae of Australian Families and to the Identification and Ecology of Larvae from New South Wales*. CRCFE Identification Guide No. 24., Cooperative Research Centre for Freshwater Ecology, Thurgoona.

Hewitt MA, 1992. The biology of the south-west Australian catfish *Tandanus bostocki* Whitely (Plotosidae). BSc (Hons.) Thesis, Murdoch University.

Hill AL, Semeniuk CA, Semeniuk V & Del Marco A, 1996. Wetlands of the Swan Coastal Plain Volume 2. Wetland Mapping, Classification and Evaluation. Water Authority of Western Australia and W.A. Environmental Protection Authority.

Holdich DM & Lowery RS, 1988. *Freshwater Crayfish: Biology, Management and Exploitation.* Croom Helm London. pp 1-6.

Horrigan N, Choy C, Marshall J and Recknagel F (2005) Response of stream macroinvertebrates to changes in salinity and development of a salinity index. Archives of Environmental Contamination and Toxicology 56: 825-833

Humphries PJ, Davies PE & Mulcahy ME, 1996. Macroinvertebrate assemblages of littoral habitats in the Macquarie and Mersey Rivers, Tasmania: implications for the management of regulated rivers. *Regulated Rivers: Research and Management* **12**: 99-102.

Hutchings R, 1987. A review of the Australian freshwater crayfish fauna with reference to aquaculture. *Freshwater Crayfish* **7**: 13-18.

Hutchinson MJ, 1991. Distribution patterns of Redfin Perch, *Perca fluviatilis* Linnaeus and western Pygmy Perch *Edelia vittata* Castelnau in the Murray River System, Western Australia. *Records of the Western Austraian Museum* **15**: 295-301

Hutchinson MJ, 1992. Probable spawning site of the freshwater cobbler Tandanus bostocki located in the Murray River Western Australia. *Western Australian Naturalist* **19**: 25-30.

Hynes HBN, 1970. *The Ecology of Running Waters.* University of Toronto Press, Toronto. Ontario, 555 p.

Hynes HBN & Bunn SE, 1984. The stoneflies (Plecoptera) of Western Australia. *Australian Journal of Zoology* 23: 92-107. Hart, B.T., & McKelvie, I.D. (1986). Chemical Limnology in Australia. [In] P. De Decker & W.D. Williams (eds) *Limnology in Australia.* CSIRO/DR Junk Publishers. pp 3-32.

IUCN, 2004. Red List of Threatened Species. International Union for Conservation of Nature and Natural Resources. http://www.redlist.org/

Junk WJ, Bayley PB & Sparks RE, 1989. The flood-pulse concept in river-floodplain systems. [In] D.P Dodge (ed) Proceedings of the International Lareg River Symposium (LARS). *Candadian Special Publication of Fisheries and Aquatic Sciences* **106**:110-127.

Kay WR, Smith MJB, Pinder AM, McRae JM, Davis JA & Halse S, 1999. Patterns of distribution of macroinvertebrate families in rivers of north-western Australia. *Freshwater Biology* **41**: 299-316.

Kay W, Scanlon M & Halse S, 2000. AusRivAS in Western Australia. *Rivers for the Future* **11**: 32-36.

Kefford BJ, Papas P, Nugegoda D 2003 Relative salinity tolerance of macroinvertebrates from the Barwon River; Victoria, Australia. Marine and Freshwater Research 54: 755-765.

Kite JM, Davies PM & Stone RR, 1998. Environmental water provisions for regulated rivers in Western Australia, Bulletin #108, Australian Committee on Large Dams.

Lawrence C & Jones C, 2002. *Cherax.* In: D. M. Holdich (Ed.) *Biology of Freshwater Crayfish.* Oxford: Blackwell Sciences, pp. 635-669.

Lee AK, 1995. The action plan for Australian rodents. Australian Nature Conservation Agency, Canberra.

Lovett S & Price P, 1999. Riparian Land Management Technical Guidelines, Volume One. Principals of Sound Management. Land & Water Resources Research & Develoment Council. Canberra.

Lynas J, Storey AW, Armstrong K, Prince J & Knott B, 2006. Invasion by the exotic crayfish, *Cherax destructor* Clark (Parastacidae) into habitats of local crayfish near Perth, Western Australia. Freshwater Crayfish **15**: 176-188.

Maguire GB, Cassells G & Lawrence C, 1999. Farming Marron. Unpublished report by Fisheries Western Australia, No. 2.

Marchant NG, Wheeler JR, Rye BL, Bennet EM, Lander NS & MacFarlane TD, 1987. Flora of the Perth Region. Part One. Western Australian Herbarium, Department of Agriculture, Western Australia.

Mattiske EM & Havel JJ, 1998. Vegetation mapping in the South West of Western Australia. Department of Conservation and Land Management.

McKie B & Cranston PS, 2001. Colonisation of experimentally immersed wood in south-eastern Australia: responses of feeding groups to changes in riparian vegetation. *Hydrobiologia* **452**: 1-14.

McMahon TA, 1989. Understanding Australian Streamflow – Implications for Instream Ecology. In C. Teoh (ed) "Proceedings of the Specialist Workshop on Instream Needs and Water Uses". Australian Water Resources Council, Canberra, ACT. pp. 33-1 - 33-11.

Mees GF, 1977. The status of *Gambusia affinis* (Baird and Girard) in south-western Australia. *Records of the West Australian Museum* **6**: 27-31.

Merrick JR & Schmida GE, 1984. Australian Freshwater Fishes. Netley, South Australia: Griffin Press Ltd.

Mitchell BD & Richard K., 1992. Macroinvertebrate communities in two salt affected tributaries of the Hopkins River, Victoria. *International Journal of Salt Lake Research* 1: 81-102.

Morgan DL and Gill HS, 2000. Fish associations within the different inland habitats of lower south-western Australia. *Records of the Western Australian Museum* **20**:31-37.

Morgan DL, Gill HS & Potter IC, 1995. Life cycle, growth and diet of Balston's pygmy perch in it's natural habitat of acidic pools in south-western Australia. *Journal of Fish Biology* **47**: 808-825.

Morgan DL, Gill HS & Potter IC, 1998. Distribution, identification and biology of freshwater fishes in south-western Australia. Records of the Western Australian Museum. Supplement No. 56. Western Australian Museum, Perth.

Morgan DL, Hambleton SJ, Gill HS & Beatty SJ, 2002. Distribution, biology and likely impacts of the introduced redfin perch (*Perca fluviatilis*) (Percidae) in Western Australia. *Marine and Freshwater Research* **53**: 1211-1221.

Morgan DL, Thorburn DC & Gill HS, in press. Salinisation of south-western Western Australian rivers and the implications for inland fish fauna – the Blackwood River, a case study Pacific Conservation Biology.

Morrison PF, 1988. Reproductive biology of two species of plotosid catfish, *Tandanus bostocki* and *Cnidoglanis macrocephalus*, from south-western Australia. PhD thesis, The University of Western Australia.

Morrissy NM, 1983. Crayfish research and industry activities in Australia, New Guinea and New Zealand. *Freshwater Crayfish* **5**: 534-544.

Morrissy NM, Caputi N & House RR, 1984. Tolerance of marron (*Cherax tenuimanus*) to hypoxia in relation to aquaculture. *Aquaculture* **41**: 61-74.

Myers GS, 1975. *Gambusia*, the fish destroyer. *Australian Zoologist* **13**: 102.

Nessimian JL, 1997. Functional categorisation of macroinvertebrates in a sand dune marsh at Rio de Janeiro State, Brazil. *Revista Brasileira de Biologia* **57**: 135-145.

Newbury RW & Gaboury MN, 1994. Stream Analysis and Fish Habitat Design. Newbury Hydraulics Ltd, Gibsons, BC. 256 pp.

Nicholl R & Horwitz P, 2000. Evaluating flagship species for ecosystem restoration; a case study involving freshwater crayfish and a river in south-western Australia. In: *Nature Conservation 5: Nature Conservation in Production Environments: Managing the Matrix* (Craig J, Mitchell N and Saunders O. eds), pp. 557-564. Surrey, Beatty and Sons, Chipping North, NSW.

Nielson DL, Brock MA, Rees GN & Baldwin DS, 2003. Effects of increasing salinity of freshwater ecosystems in Australia. *Australian Journal of Botany* **51**: 655-665.

Pen LJ, 1983. Peripheral Vegetation of the Swan and Canning Estuaries 1981. Department of Conservation and Environment Western Australia & Swan River Management Authority Bulletin No. 113. July 1983.

Pen LJ & Potter IC, 1990. Biology of the Nightfish, *Bostockia porosa* Castelnau, in a south-western Australian river. *Australian Journal of marine and freshwater Research* **41**: 627-645.

Pen LJ & Potter IC, 1991a. Biology of the Western Minnow Galaxias occidentalis Ogilby Teleostei Galaxiidae in a south-western Australian River I. Reproductive biology. *Hydrobiologia* **211**:77-88.

Pen LJ & Potter IC, 1991b. Biology of the Western Minnow *Galaxias occidentalis* Ogilby Teleostei Galaxiidae in a south-western Australian River II. Size and age coposition, growth and diet. *Hydrobiologia* **211**:89-100.

Pen LJ & Potter IC, 1991c. Biology of the Western pygmy perch *Edelia vittata* and comparisons with two other teleost species endemic to south-western Australia. *Environmental Biology of Fishes* **31**: 365-380.

Pen LJ & Potter IC, 1991d. Reproduction, growth and diet of *Gambusia holbrooki* (Girard) in a temperate Australian River. *Aquatic Conservation: Marine and Freshwater Ecosystems* **1**: 159-172.

Pen LJ, Potter IC & Calver MC, 1993. Comparisons of the food niches of three native and two introduced fish species in an Australian River. *Environmental Biology of Fishes* **36**: 167-182.

Pen L, Scott M 1995 Stream Foreshore Assessment in Farming Areas. Blackwood Catchment Co-ordinating Group. National Landcare Program. Department of Agriculture WA. June 1995.

Pinder AM, Halse SA, McRae JM & Shiel RJ, 2004. Aquatic invertebrate assemblages of wetlands and rivers in the Wheatbelt region of Western Australia. *Records of the Western Australian Museum* Supplement 67: 7-37.

Poff L, Allan J, Bain M, Karr J, Prestegaard K & Richter B, 1996. The natural flow regime: a paradigm for river conservation and restoration. *Bioscience* **47**: 769-784.

Pusey BJ & Bradshaw SD, 1996. Diet and dietary overlap in fishes of temporary waters of southwestern Australia. *Ecology of Freshwater Fishes* **5**:183-194.

Pusey BJ & Edward DHD, 1990. Structure of fish assemblages in waters of the southern acid peat flats, south-western Australia. Aust. *Journal of Marine and Freshwater Research* **41**:721-734

Pusey BJ & Morrison PF, 1989. The diet of rainbow trout in Wungong Dam, Western Australia. *West Australian Naturalist* **18**:37-40.

Pusey BT, Storey AW, Davies PM & Edward DHD, 1989. Spatial variation in fish communities in two South-western Australian river systems. *Journal of the Royal Society of Western Australia* **71**: 69-75.

Resh VH, Brown AV, Covich AP, Gurtz ME, Li HW, Minshall W, Reice SR, Sheldon AL, Wallace JB & Wissmar RC, 1988. The role of disturbance in stream ecology. *Journal of the North American Benthological Society* **7**: 433-455.

Richter B, Baumgartner J, Wigginton R & Braun D, 1997. How much water does a river need? *Freshwater Biology* **37**:231-249.

Riek EF, 1967. The freshwater crayfish of Western Australia (Decapoda: Parastacidae). *Australian Journal of Zoology* **15**: 103 -121.

Rutherford J, & Kefford B 2005 Effects of salinity on stream ecosystems: improving models for macroinvertebrates. CSIRO Land and Water Technical Report 22/05. Canberra: CSIRO.

Sarti NL, 1994. Preliminary Report of the Aquatic Fauna of Bungendore Park and Wungong Brook. Unpublished report to Bungendore Park Management Committee.

Searle DJ & Semeniuk V, 1985. The natural sectors of the inner Rottnest Shelf coast adjoining the Swan Coastal Plain. *Journal of the Royal Society of Western Australia* **67**: 116-136.

Seddon G, 1972. *Sense of Place*. University of Western Australia Press, Nedlands, Australia

Shipway B, 1949. Notes on the natural history of the pygmy perch (*Nannoperca vittata*). *West Australian Naturalist* **2**: 1-9.

Shipway B, 1951. The natural history of the marron and other freshwater crayfishes of south-western Australia. Part 1. *Western Australian Naturalist* **3**: 7 -12.

SKM, CRCFE, Freshwater Ecology & Lloyd Environmental Consultants, 2002. The FLOWS Method – A Method for Determining Environmental Water Requirements in Victoria. Unpublished report by Sinclair Knight Merz, the Cooperative Research Centre for Freshwater Ecology, Freshwater Ecology (NRE) and Lloyd Environmental Consultants to the Department of Natural Resources and Environment, Victoria.

Smith BJ, 1996. Identification keys to the families and genera of bivalve and gastropod molluscs found in Australian inland waters. Cooperative Research Centre for Freshwater Ecology Identification Guide No. 6. Presented at the Taxonomy Workshop held at the Murray-Darling Freshwater Research Centre, Albury. 20-21 February, 1996.

Stewardson M, 2001. The Flow Events Method for Developing Environmental Flow Regimes. *In* I. Rutherfird, F. Sheldon, G. Brierley & C. Kenyon (Eds) "Third Australian Stream Management Conference: the Value of healthy Streams." Conference Proceedings, Volume II. Brisbane, Queensland, 27-29 August, 2001. Cooperative Research Centre for Catchment Hydrology. ISBN 1 876006 80 3.

Stewardson MJ & Cottingham P, 2002. A demonstration of the flow events method: environmental flow requirements of the Broken River. *Australian Journal of Water Resources* **5**: 33-48.

Stewardson M & Gippel CJ, 2003. Incorporationg flow variability into environmental flow regimes using the Flow Events Method. *River Research and Applications* **19**: 459-472.

St Clair RM, 2000. Preliminary Keys for the Identification of Australian Caddisfly Larvae of the Family Leptoceridae. Comparative Research Centre for freshwater Ecology Identification Guide No. 27. Presented at the taxonomy workshop held at the Murray Darling Freshwater Research Centre, Albury. 8-9 February 2000.

Storey AW, 1998. Fish and fish habitat survey of the Canning River and its tributaries. Unpublished report to the Upper Canning / Southern-Wungong Catchment Management Team, pp 43.

Storey AW, 1999. Baseline monitoring of aquatic macroinvertebrates in the Canning River – summer 1999. Unpublished report to the Water & Rivers Commission pp 25.

Storey AW, 2000. Ecological monitoring of the Canning and Southern- Wungong Rivers: baseline data for assessing effectiveness of EWRs Draft unpublished report to the Water & Rivers Commission.

Storey AW & Davies PM, 2002. Preliminary Environmental Water Requirements for Gingin and Lennard Brooks. Unpublished report to the Water & Rivers Commission, pp 59.

Storey AW & Edward DHD, 1989. Longitudinal variation in community structure of Chironomidae (Diptera) in two south-western Australian river systems. *Acta Biologica Debrecen Oecologica Hungarica* **3**: 315-328.

Storey AW, Bunn SE, Davies PM & Edward DH, 1990. Classification of the macroinvertebrate fauna of two river systems in south-western Australia in relation to physical and chemical parameters. *Regulated Rivers: Research and management* **5**: 217-232.

Storey AW, Edward DH & Gazey P, 1991. Surber and kick sampling: a comparison for the assessment of macroinvertebrate community structure in streams of south-western Australia. *Hydrobiologia* **211**: 111-121.

Storey AW, Halse SA & Shiel RJ, 1993. Aquatic invertebrate fauna of the Two Peoples Bay area, southwestern *Australia. Journal of the Royal Society of Western Australia* **76**: 25-32.

Storey AW, Davies PM & Creagh S, 2001. Preliminary Environmental Water requirements for the Caning River system. Unpublished report to the Water & Rivers Commission, pp 57

Streamtec, 1998. Harvey Basin Water Allocation Plan Environmental Water Requirements. Report for the Proposed Harvey Basin Surface Water Allocation Plan. Report ST 18/97. Unpubl. report to the Water & Rivers Commission. Perth, Western Australia.

Streamtec, 1999. Lower Collie River including Henty Brook Environmental Water Requirements. Unpublished report to the Water & Rivers Commission by Streamtec Pty. Ltd. Report # 21/99.

Streamtec, 2002. Ecological Water Requirements of Samson Brook: Samson Brook to the confluence with the Harvey River. Unpublished report by Streamtec Pty Ltd to Water Corporation.

Sutcliff K, 2003. The conservation status of aquatic insects in south-western Australia. PhD Thesis, Murdoch University, School of Environmental Biology.

Thorburn DC, 1999. The broad and fine scale habitat associations of the fish fauna of the Blackwood River and it's tributaries. B.Sc. (Hons) thesis. Murdoch University.

Thorp JH & Delong MD, 1994. The riverine productivity model: an heuristic view of carbon resources and organic matter processing in large river systems. *Oikos* **70**: 305-308.

Trayler KM, Davis JA, Horwitz P & Morgan D, 1996. Aquatic fauna of the Warren bioregion, south-west Western Australia: does reservation guarantee preservation? *Journal of the Royal Society of Western Australia* 79: 281-291.

Tyler MJ, Smith LA & Johnstone RE, 2000. *Frogs of Western Australia*. Revised Edition. Western Australian Museum.

Vannote RL, Minshall GW, Cummins KW, Sedell JR & Cushing CE, 1980. The River Continuum Concept. *Canadian Journal of Fisheries and Aquatic Sciences* **38**: 130-137.

Varma S, 2002. Hydrogeology and groundwater resources of the Collie Basin, Western Australia. Water and Rivers Commission, Resource Science Division. Hydrogeological Record Series Report HG5.

Velasquez SM & Miserendino ML, 2003. Allochthonous organic matter and functional organisation of macroinvertebrates at habitat scale in Patagonian mountain rivers. *Ecologia Austral*. **13**: 67-82.

Watts CHS & Aslin HJ, 1981. The rodents of Australia. Angus and Robertson, Sydney.

Watts RJ, Storey AW, Hebbert DR & Edward DH, 1995. Genetic and morphometric variability in populations of the western minnow, *Galaxias occidentalis* Ogilby in streams and rivers of the Darling Range and Swan Coastal Plain. *Journal of Marine and Freshwater Research* **46**:769-777.

WEC, 2002. Welker Environmental Consultancy: Lower Collie River and Henty Brook Preliminary Environmental Water Provisions - Discussion Paper. Unpublished report to Water & Rivers Commission pp 56.

WEC, 2004. Preliminary Ecological Water Requirements Study. Bickley/Munday brooks and Churchman Brook. Unpublished report by Welker Environmental Consultancy to Water Corporation.

WEC & Streamtec, 2000. Environmental Water Requirements Study. Harris River and East Branch of the Collie River (downstream of the confluence) to the South Branch. Welker Environmental Consultancy & Streamtec Pty Ltd. Unpublished report to the Water Corporation. Perth Western Australia.

WEC & Streamtec, 2001. Environmental Water Provisions. South Branch of the Collie River downstream from western 5 open cut. Welker Environmental Consultancy & Streamtec Pty Ltd. Unpublished report to the Water and Rivers Commission, Perth Western Australia.

Wells MR, 1989. Land capability study of the shires of Mandurah and Murray. Land Resources Series No. 2. Western Australian Department of Agriculture.

Williams WD, 1980. *Australian Freshwater Life: The Invertebrates of Australian Inland Waters*. The MacMillan Company of Australia Pty. Ltd., South Melbourne, Australia.

Wollheim WM & Lovvorn JR, 1995. Salinity effects on macroinvertebrate assemblages and waterbird food webs ion shallow lakes of the Wyoming High Plains. *Hydrobiologia* 310: 207-223.

Woollard P, Vestjens WJM, Maclean L 1978 The ecology of the eastern water rat *Hydromys chryogaster* at Griffith, NSW: food and feeding habits. Australian Wildlife Research **5**: 59-73.

WRC, 2000. Draft Environmental Water Provisions Policy for Western Australia, report # WR4, Water Reform Series, Water and Rivers Commission.

WRC, 2001a. Salinity Situation Statement: Collie River. Water Resource Technical Series, Report No. WRT29. Water & Rivers Commission, Perth Western Australia.

WRC, 2001b. Stream Channel Analysis. Water 7 Rivers Commission Restoration Report No. RR 9

WRC, 2002. Simple fishways. Water Notes: Water notes for river management, WN26, January 2002.

WRM, 2003a. Wellington Below – EWR Review: Fish community including migration requirements. Unpublished report to the Water and Rivers Commission by Wetland Research and Management. March 2003.

WRM, 2003b. Preliminary Ecological Water Requirements for Drakesbrook. Unpublished report to Harvey Water by Wetland Research and Management. June 2003.

WRM, 2005a. Ecological Water Requirements of Augustus River – Intermediary Assessment. Unpublished report to Worsley Alumina Pty. Ltd. by Wetland Research and Management. June 2005.

WRM, 2005b. Ecological Water Requirements of Hill River – Intermediary Assessment. Unpublished report to Department of Environment by Wetland Research and Management. June 2005.

WRM, 2005c. Re-assessment of the nature conservation values of the Byenup-Muir peat swamp system. Unpublished DRAFT report to Department of Conservation and Land Management by Wetland Research and Management. April 2005.

WRM, 2006a. Burekup Project: Baseline Aquatic Biology and Water Quality Study. Unpublished report by Wetland Research & Management to Iluka Resources Ltd. April 2006.

WRM, 2006b. Cloverdale Project: Baseline Aquatic Biology and Water Quality Study of the Ludlow River and Tiger Gully. Unpublished report by Wetland Research & Management to Iluka Resources Ltd. April 2006.

WRM, 2006c. Yoganup Project: Baseline Aquatic Biology a Study. Unpublished report by Wetland Research & Management to Iluka Resources Ltd. May 2006.