AN AMBIENT WATER QUALITY SURVEY OF THE UPPER WALSH RIVER SYSTEM, NORTH QUEENSLAND

Volume 1: CATCHMENT OVERVIEW

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EXECUTIVE SUMMARY

The Walsh River is a major tributary of the Mitchell River in north Queensland. Since European settlement its catchment area has been subject to a wide variety of land-uses, including cattle grazing, extensive mining and mineral processing (mostly historical), urban development, and in the upper catchment area, intensive irrigated agriculture. In response to community concerns that these human pressures may be impacting on the quality of water and aquatic habitats within the river system, the Northern Gulf Resource Management Group (NGRMG) commissioned the Australian Centre for Tropical Freshwater Research (ACTFR) at James Cook University, to carry out a number of studies relating to water quality and the health of the river system. This report, the fifth in this series, details the findings of an ambient water quality and ecological condition assessment project carried out in the upper Walsh River and selected tributaries (including Two-Mile Creek in the upper Mitchell Catchment), late in the 2006 dry season.

A number of water quality-related issues that could potentially warrant some form of management attention were assessed, however it is very clear that the major problem requiring urgent management attention is the very poor quality of the water being discharged from the intensively developed sub-catchments (particularly Cattle Creek) within the Mareeba-Dimbulah Irrigation Area. Two Mile Creek, a tributary of the upper Mitchell catchment within the irrigation area, is similarly affected by high nutrient levels and is also infested with aquatic weeds. The nutrient concentrations that have been reported in these creeks are two to ten times higher than any level that could be considered acceptable, and historical data indicate that this has been the case for more than a decade. In fact, some of the ammonia concentrations are actually high enough to be acutely toxic to aquatic animals and could cause fish kills under unfavourable (high) pH conditions.

During the dry season survey an unusually large proportion of the nitrogen contained in both creeks was in the form of ammonia, and the concentrations of phosphorus in Two Mile Creek were much higher than other sites. Sewage contains very high concentrations of both ammonia and phosphorus, so the sewage treatment plant is strongly implicated as a contributor to the contamination in Two Mile Creek. The elevated ammonia levels in Cattle Creek on the other hand can most likely be attributed to irrigation tailwater runoff from farms that had recently been treated with an ammonia-based fertiliser such as urea, ammonium nitrate or aqua ammonia. Diffuse runoff from soils and groundwaters generally delivers nitrogen to surface streams in the form of nitrate, so the high ammonia concentrations suggest that much of the runoff in Cattle Creek was being carried there via surface drains – i.e. from readily identifiable point sources that are much easier to manage than diffuse sources. In most cases nutrients are also likely to occur in conjunction with high concentrations of oxygen demanding organic matter (including juice from recently-harvested sugar cane) and this can result in oxygen deficiencies severe enough to cause acute problems such as fish kills, especially if conditions within the receiving waters are unfavourable.

At the time of the survey the oxygen levels in Cattle Creek were barely sufficient to support fish life, even though the stream was being aerated by quite strong flows. It is therefore highly likely that conditions would have deteriorated rapidly as soon as flow rates declined and the water was no longer being so actively aerated. This is an unfortunate situation because one of the surprising findings of this survey (from the perspective of investigators who had only previously seen a few of the poorer sections of the creek) is that Cattle Creek offers a greater diversity of natural aquatic habitats than any of the other sites surveyed (see the site profile and photographs in Volume Two). In fact if some of the more severe water quality problems could be ameliorated without starving the system of water, this creek would have the potential to become a regionally significant biodiversity refuge.

The available data are not detailed enough to be able to confidently assess the downstream impacts of nutrient discharges from Cattle Creek into the Walsh River. At the time of the survey nutrient concentrations in the river were significantly elevated at Algoma, immediately downstream of Cattle Creek, but there was no evidence of equivalent water quality effects further downstream and most of the river sites surveyed during this study were actually in quite good ecological condition. However, under different flow conditions, it is hypothesised that nutrient pulses could potentially be reaching at least as far downstream as Twelve Mile. However, in this region the true test of a river's health is its capacity to cope with the prolonged periods of low flow and contaminant accumulation that occur during unusually dry years, and obviously this could not be assessed during the current study. Since the flows in this section of the river are supplemented with water from Tinaroo Dam, it is less susceptible to the kinds of water quality problems that can develop in natural river reaches where the water may stagnate during droughts.

The concentrations of copper in Cattle Creek (and also Walsh River) were high enough to be ecologically significant. However, the highest copper concentrations were recorded in the Walsh River between the MDIA supplementation point and Cattle Creek, suggesting that the copper may be coming from the irrigation supply and that the inflows from Cattle Creek were actually diluting the copper in the river (and that is the only parameter for which that occurred). All sites that were receiving irrigation water reported concentrations of filterable copper that exceeded ANZECC (2000) guidelines for the protection of aquatic ecosystems, and the site with the highest concentration had significantly lower macroinvertebrate diversity than any other site on the river. These copper concentrations are orders of magnitude too low to affect human water uses, but they are more than high enough to potentially impact on the ecosystem, and their source and full ecological significance should be investigated more closely. Somewhat ironically, given existing

concerns about the potential impacts of abandoned mines, the copper in the irrigation supply is actually the most ecologically significant and widespread occurrence of metal contamination that was encountered during this study.

There are numerous abandoned mines scattered throughout the Walsh River catchment, most of which were not properly rehabilitated. Some of these have already been shown to be discharging metals and other contaminants into waterways (Bartareau *et al* 1998), so the existence of some highly contaminated sites is to be expected. We recorded some significant metal concentrations in the vicinity of mine sites, but these were confined to quite small waterbodies with considerably lower ecological value than the waters in the irrigation area. Moreover, only one such site, Poison Water on Oaky Creek, reported metal concentrations (for zinc, aluminium and copper) high enough to be certain that the water was highly toxic to aquatic life. The Poison Water site aside, none of the water samples collected during this study yielded results that exceeded the ANZECC guidelines for any metal (for which guideline values are available) other than chromium (Cr), copper (Cu) and zinc (Zn). The chromium results appear to simply indicate that natural background concentrations were slightly elevated throughout the region, as there was no evidence of hotspots indicative of a particular contaminant source, no significant chromium enrichment in bottom sediments and no apparent effects on macroinvertebrate diversities or ecological integrity. None of the copper or zinc exceedances were particularly serious either. The zinc concentration at Adder Creek was high enough to have had some measurable impact on the biodiversity at that site, but it was the smallest and arguably the least valuable of the surveyed sites, so this is not considered to be a major issue.

The concentrations of those metals that are not included in the ANZECC guidelines were also generally moderate, however, the water samples collected from Bullaburrah Creek all reported elevated molybdenum levels. The filterable molybdenum value of 96 µg/L at the site located immediately downstream of the Wolfram Camp mine site was quite a significant anomaly given that most sites in the region reported values less than 0.1 µg/L, and concentrations were still quite elevated (27.7 µg/L) further downstream at site WLSH32, the point where Bullaburrah Creek enters the Walsh River. A concentration of 18 µg/L was reported upstream of Wolfram Camp, suggesting that natural background concentrations in this catchment may be elevated; nevertheless, there is obviously some suggestion of possible enrichment downstream of the mining area. These results correlate with those obtained from the sediment investigation which found anomalous molybdenum residues in the bottom sediments of the creek, and assigned a high priority ranking to the WLSH32 site. The ecological significance of these molybdenum values is difficult to gauge as there is very little toxicological information available (which is also why it has no ANZECC guideline value for this metal). However, it is noteworthy that there was a localised reduction in macroinvertebrate diversity in the river at the mouth of Bullaburrah Creek, and there are no obvious water quality anomalies other than the elevated molybdenum levels that could potentially explain this finding.

The results of the water and sediment surveys both show some evidence of slightly elevated metal accumulations in the bottom of dams and weirpools (Butler *et al.* 2007). This appears to simply reflect the inherent capacity for such impoundments to trap sediments and some of the contaminants they contain. None of the levels encountered are high enough to present any threats to human water users but they could be significant enough to have some impact on the ecosystem, especially under unfavourable environmental conditions (such as may develop during droughts for example). The four highest priority sites identified in the sediment study as deserving closer investigation are within, or immediately downstream of impoundments. Two of these, WLSH35 below Collins Weir and WLSH27 Bruce Weir, were surveyed in the current study and both were found to be in quite good ecological condition with high macroinvertebrate diversities. This indicates that metal accumulations are causing no serious or chronic problems, but it would still be worth checking if any episodic problems develop under less favourable environmental conditions.

Overall, despite some significant localised impacts from old mine run-off, these impacts are not far-reaching enough to warrant the substantial expenditure that would need to be invested in order to rehabilitate such sites. The impacts of irrigation run-off are more extensive, more readily amenable to improved management actions, and affect waterbodies with greater ecological value, that this is recommended as a priority area for further management action.

Specific recommendations (explained in more detail in Section 5.2) are:
- Foster the development and implementation of best management farming and irrigation management practices;
- Investigate the feasibility of using artificial wetlands to treat water before it enters creek systems;
- Develop a management plan to protect the ecological values of Cattle Creek;
- Further assess the extent and regional significance of the poor ecological conditions in Two-Mile Creek;
- Consider a nutrient monitoring strategy to assess the extent to which poor quality water discharging from Cattle Creek is affecting the Walsh River;
- Investigate the source of the high copper concentrations in the irrigation supply;
- Investigate high metals concentrations in Bullaburrah creek;
- Check conductivity levels in Cattle Creek frequently enough to provide early detection of any potential rises in surface water salinity;
- Investigate if the high ecological value sites studied here can cope with the ecological conditions encountered in the event of a failed wet season, when natural conditions become more limiting.

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1. INTRODUCTION

The Walsh River is a major tributary of the Mitchell River in north Queensland. Since European settlement its catchment area has been subject to a wide variety of land-uses, including cattle grazing, extensive mining and mineral processing (mostly historical), urban development, and in the upper catchment area intensive irrigated agriculture. In response to community concerns that these human pressures may be impacting on the quality of water and aquatic habitats within the river system, the Northern Gulf Resource Management Group (NGRMG) commissioned the Australian Centre for Tropical Freshwater Research (ACTFR) at James Cook University, to carry out a number of studies relating to water quality and the health of the river system.

This report is the fifth in a series dealing with various water quality-related issues in the Walsh River catchment (see Butler and Burrows 2005, 2006a, 2006b, Butler *et al.* 2007). It details the findings of an ambient water quality and ecological condition assessment project carried out in the upper Walsh River and selected tributaries, late in the 2006 dry season. The project employed unusually intensive field survey techniques including the collection of water samples, macroinvertebrate samples, photographic records and bathymetry data, measurements of the spatial and temporal variability of physicochemical parameters (for periods of up to 24 hours), and assessments of the spatial distribution of important habitat features such as, different substratum types and plant assemblages. Since the study involved a single one-off field survey, it ostensibly affords a snapshot indicative of just one point in time. However, sites were assessed in sufficient detail to obtain a fundamental understanding of how the ecosystem was functioning at the time, and that, in conjunction with knowledge gained through reviews of pre-existing scientific data, personal experience and discussions with local residents, has allowed us to draw some reasonably confident conclusions about the longer term picture.

1.1 Context of this study

Most of the pre-existing data that has been taken into consideration in this report were reviewed and analysed in previous projects and have not been re-iterated here. The first of these studies was a desktop review of pre-existing water quality datasets to determine if they could be used to help answer any of the key questions that have been raised regarding the health of the Mitchell River system (Butler and Burrows 2005). The review concluded that most of the available data, principally water quality data held in the Department of Natural Resources and Water HYDSYS database, had been collected for resource evaluation purposes and were not well-suited to the task of assessing ecological health. Nevertheless, some individual datasets were identified as being worthy of more detailed examination, and this led to the commissioning of subsequent desktop studies.

The second study entailed analysing stream discharge and turbidity records to determine if the dynamics of flow and sediment discharge within the Mitchell River were comparable to some of the more intensively studied rivers in other parts of tropical Australia. The report (Butler and Burrows 2006a) found that the Mitchell behaved quite similarly to some sections of the Burdekin and Herbert Rivers, and concluded that it would be valid and feasible to make inferences about other aspects of the river's natural character based on research findings obtained from studies carried out in these other catchments.

The third study was prompted by public concerns about a potential/emerging salinity hazard in the Cattle Ck section of the Mareeba-Dimbulah Irrigation Area (MDIA) (now known as Mareeba-Dimbulah Water Supply Scheme - MDWSS). It involved analysing stream conductivity (salinity) data collected from numerous sites within the Walsh River catchment (including Cattle Ck) between 1968-2002, plus data obtained from the Mitchell River Condition Study (Ryan *et al.* 2002). The report (Butler and Burrows 2006b) found that there were significant spatial and seasonal salinity variations throughout the catchment, but most of the observed effects could be entirely explained by natural processes such as evapo-concentration. The only anthropogenic effect detected was a general reduction in the dry season salinity levels of supplemented river reaches (due to significant inflows of low salinity water from Tinaroo Dam). The report therefore concluded that there was no suggestion of an existing or imminent salinity problem in the river system, but noted that there was unequivocal evidence of soil salinisation and rising saline water tables in the Cattle Ck subcatchment, and cautioned that problems could develop quite suddenly in the river if these problems were allowed to worsen. Accordingly it recommended establishing an ongoing salinity monitoring program for Cattle Ck but suggested that managers wait until other water quality-related issues have been properly assessed before determining the scope of the program.

The fourth study addressed concerns about potential impacts from the large numbers of abandoned mines scattered throughout the Walsh River catchment, most of which were not properly rehabilitated and some of which have already been shown to be discharging metals and other contaminants into waterways (Bartareau *et al* 1998). This field-based investigation assessed the concentrations of metals in the bottom sediments at a number of sites on the river and within tributaries that drain subcatchments where mining activity has historically been intensive. The study (Butler *et al.* 2007) examined the spatial distribution of metal accumulations throughout the drainage system, focusing mainly on permanent and semi-permanent waterbodies because they are the sites where ecological damage is most likely to occur if metals build up to toxic levels.

Quantitative data analysis methods were used to assign relative hazard ratings to sites based on consideration of both ecological risk and degree of metal enrichment. This dual approach was necessary because some metals can become artificially enriched without becoming toxic, while others can be toxic to sensitive aquatic animals even at naturally-occurring concentrations. Note that, because they constantly take up metals from the water through their gills and body surfaces, animals that live permanently immersed in water are far more sensitive to water-borne metals than are terrestrial animals, including man. Hence the strategy of focusing on ecological risks guarantees that the interests of human water users are very well protected.

The sediment study assessed relative rather than actual risks; and aimed to identify priorities for closer investigation, not management action. Even the highest priority sites may not actually have been suffering from metal-related problems, they are simply the best places to carry out more detailed studies in order to check if the ecosystem is being adversely affected by metal accumulations. The report identified one site (the aptly named Poison Water site on Oaky Ck) that was not assigned a priority rating as it was so obviously contaminated that further investigation would not be worthwhile –managers will need to decide whether that site needs to be rehabilitated or sacrificed. Based on the findings of Bartareau *et al* (1998) it is likely that there are numerous streams and water holes that suffer localised water quality problems due to their very close proximity to old mine sites. However, most of these are small seasonal and/or ephemeral water bodies with limited resource potential and low ecological value, so contamination is not necessarily a major problem provided that it remains localised. Our sediment study was the step towards determining if there is evidence of potential impacts on any of the more valuable water bodies located downstream of such sites.

The sediment report only addressed the issue of metal contamination and recommended against investing any resources into further mine-related monitoring or investigation until other aquatic health issues in the catchment have been properly assessed and prioritised. The main purpose of the current study is to provide a broader overview of water quality related issues in the catchment by assessing sites in sufficient detail to be able to identify the key factors influencing their health.

1.2 Report Layout

The first volume of this report provides an overview of the study and its findings. It details study methods, presents the results of laboratory water quality analyses and macroinvertebrate samples, discusses the results in a broad catchment-wide context, summarises conclusions and recommends priorities for future management attention.

The second volume comprises profiles for each study site. It was not logistically feasible to examine all sites in the same level of detail during this study, and this is reflected in the amount of information provided in the site profiles. Key sites were surveyed intensively for a full day, and the profiles for these sites contain a detailed analysis of the limnology of the water body and a thorough description of aquatic habitat conditions, accompanied by maps, photographic images and graphical displays of field water quality measurements. At the opposite extreme there were some sites that could only be visited briefly to obtain a water sample. The profiles for some of these sites comprise only photographs, nevertheless these still provide very useful indications of prevailing conditions and waterbody type.

Most waterbodies in the study area flow quite slowly, if at all during the dry season, and this creates a situation where instream processes can strongly influence ambient water quality. The information contained in the site profiles was particularly valuable for the current study because it helped investigators to infer whether observed water quality variations could be attributed to internal processes or to external pressures. It also provided a basis for assessing if there was any evidence of general habitat conditions having been impacted by water quality alterations or other anthropogenic pressures. This knowledge underpinned the analysis of between-site water quality variations presented in volume one, and substantially strengthens the confidence of the conclusions that have been drawn.

The site profiles contain valuable baseline information indicative of conditions in the drainage system at the time of the survey, and the site photographs alone will be a valuable resource for assessing if changes have occurred in the future.

2. STUDY DETAILS

2.1 Site Locations

Thirty sites, each located on the Walsh River or one of its tributary streams, were surveyed during this study. Site identifications, map coordinates, survey dates and survey durations are summarised in Table 1 and site locations are mapped in Figure 1. The site profiles in Volume 2 of this report include photographs for each site, and provide detailed locality maps and satellite images for all full-day and half-day survey sites.

2.2 Approach Used

This study sought to collect as much pertinent information as possible about the current condition and intrinsic ecological character of riverine aquatic habitats in the Upper Walsh catchment area. Since sites varied widely in size and regional significance, and were each potentially subject to different kinds of human and natural pressures, it was necessary to employ different survey methods and intensities at different sites in order to obtain the most relevant data possible within available time and budget constraints.

Grab samples of water were collected at all sites for laboratory analysis of nutrients, trace metals and water hardness (which was included because it strongly influences the toxicity of many metals), in order to establish what the regional background levels were at the time of the survey, and to facilitate between-site comparisons. However, as will be seen in Volume 2 and Section 4 of this Volume, the selection of other monitoring parameters and methods varied somewhat between sites. Macroinvertebrate samples for example, were collected at sites located on the main river channel and/or within the irrigation area, but there was insufficient time available to carryout this fairly time-consuming sampling procedure at most other sites.

The largest, and from many perspectives most important, waterbodies in the study area are artificial impoundments created by weirs. Four of these sites (see Table 1) were surveyed intensively for a full day allowing time to assess and map the bathymetry and key habitat features, as well as examining how temperature, pH and dissolved oxygen concentrations varied over the course of a 24 hour period at different places within the water column. (The value and uses of this sort of information is explained in later sections of this report).

It was not feasible to devote this level of effort to all sites, but it was possible to spend a half a day at 8 sites that were considered to be important to this study due to their ecological significance and/or their potential exposure to various human pressures. This allowed sufficient time to map bathymetry and habitat features and examine spatial variations in water quality, but did not provide the opportunity to assess variability over time. Some of these sites encompassed fairly extensive stream reaches and were surveyed by drifting downstream in a canoe in order to determine how conditions varied along the length of the reach.

Most of the remaining sites had to be assessed rapidly. Generally this simply entailed the collection of a spot water sample and site photographs, although in a few cases there was sufficient time available to construct a coarse site map and/or collect a macroinvertebrate sample.

Laboratory water analyses and macroinvertebrate samples were collected to allow direct comparisons between sites. The results for these indicators are discussed in a whole-of-catchment context in Section 4 of this Volume. The field data on the other hand relate to site-specific variables and must be evaluated one site at a time. Hence these data are documented in the individual site profiles in Volume 2. The information contained in the detailed site profiles was critical to the current study as it allowed investigators to gauge whether the between-site variations identified in Volume one were the result of internal processes occurring within individual water bodies or contaminant inflows form upstream.

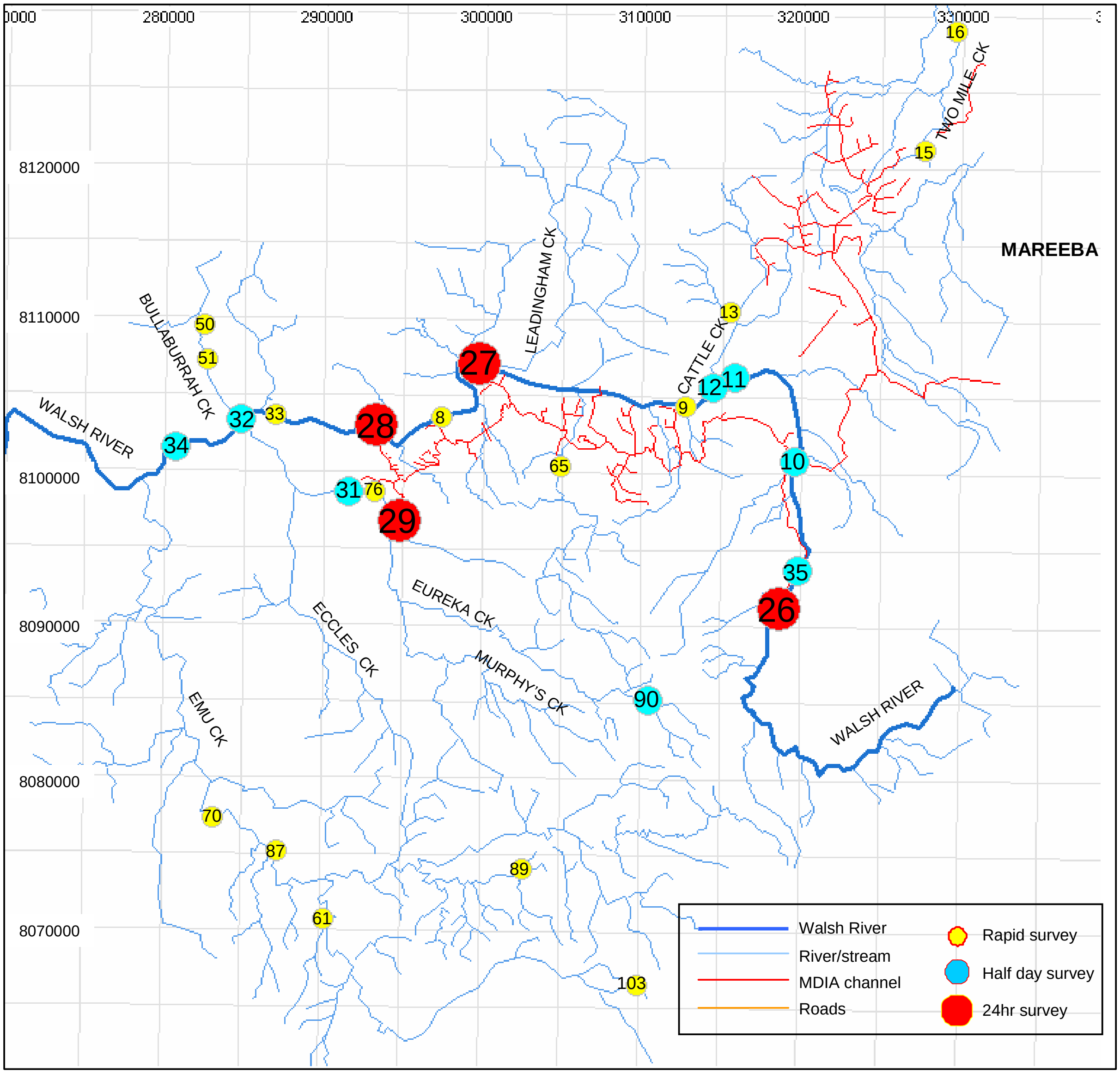
Each volume of the report includes a brief section explaining the scientific basis for interpreting the data contained in the site profiles. However, since interpreting limnological data of this sort is a complex and specialised undertaking, and the report is already very long, interpretive comments have only been provided for the first few site profiles to give readers an idea of how the data have been assessed. Nevertheless, the data for the remaining sites have been carefully appraised and any cases where instream processes are likely to have impacted on the findings of this study are noted in Volume One. Basically if the variables covered in the site profiles are not mentioned in the discussions in Section 4 of this volume, readers can safely assume that they were not considered to be an issue at the time of this survey.

For example there is no general discussion of dissolved oxygen status, but it is noted as a contributing factor and/or threat in any cases where the results for a particular contaminant or biodiversity indicator are considered likely to have been affected by oxygen availability.

Table 2.1: Site locations, survey durations and survey dates

Site ID	Site Description	Sub-catchment	Survey Intensity	Survey Date	Latitude	Longitude
WLSH26	Collins Weir	Walsh River	Full day	03/11/06 8:44	17º 15.42'	145º 17.65'
WLSH35	Below Collins Weir	Walsh River	Half day	06/11/06 7:20	17º 15.00''	145º 17.88'
WLSH10	MDIA inflow	Walsh River	Half day	06/11/06 12:40	17º 10.21'	145º 18.25'
WLSH11	Bontaba	Walsh River	Half day	02/11/06 10:00	17º 07.24'	145º 16.10'
WLSH09	Algoma	Walsh River	Rapid	15/11/06 14:50	17º 08.24'	145º 14.39'
WLSH27	Bruce Weir	Walsh River	Full day	07/11/06 10:00	17º 06.72'	145º 07.05'
WLSH08	Dimbulah	Walsh River	Rapid	15/11/06 14:20	17º 08.60'	145º 05.73'
WLSH28	Leafgold Weir	Walsh River	Full day	11/11/06 10:00	17º 08.89'	145º 03.39'
WLSH33	Wongoo	Walsh River	Rapid	31/10/06 9:48	17º 08.52'	144º 59.84'
WLSH32-1	Walsh US Bullaburrah	Walsh River	Half day	15/11/06 9:22	17º 08.59'	144º 58.60'
WLSH32-3	Walsh DS Bullaburrah	Walsh River		15/11/06 10:57	17º 08.85'	144º 58.53'
WLSH34	Twelve Mile Waterhole	Walsh River	Half day	13/11/06 11:00	17º 09.64'	144º 56.28'
WLSH13	Dingo Ck	Cattle Creek	Rapid	14/11/06 13:30	17º 05.11'	145º 16.72'
WLSH12	Cattle	Cattle Creek	Half day	16-11-06 pm	17º 07.56'	145º 15.33'
WLSH16	Pickford Rd	2-Mile Creek	Rapid	17/11/06 14:00	16º 54.97'	145º 23.99'
WLSH15	Cardillo's	2-Mile Creek	Rapid	17/11/06 13:15	16º 59.19'	145º 22.88'
WLSH50	Wolfram US	Bullaburrah Ck	Rapid	31/10/06 8:47	17º 06.60'	144º 55.56'
WLSH51	Wolfram DS	Bullaburrah Ck	Rapid	31/10/06 7:50	17º 04.66'	144º 56.23'
WLSH32	Bullaburrah	Bullaburrah Ck	Rapid	15/11/06 10:37	17º 08.66'	144º 58.73'
WLSH90	Stannary Weir	Eureka Creek	Half day	30/10/06 8:34	17º 18.68'	145º 55.84'
WLSH29	Solanum Weir	Eureka Creek	Full day	09/11/06 10:25	17º 12.26'	145º 04.20'
WLSH76	Eureka DS1	Eureka Creek	Rapid	29/10/06 10:25	17º 11.23'	145º 03.34'
WLSH31	Eureka DS2	Eureka Creek	Rapid	17-11-07 am	17º 11.25'	145º 02.43'
WLSH103	Adder	Emu Creek	Rapid	30/10/06 12:20	17º 28.74'	145º 12.60'
WLSH61	Gregory Dam	Emu Creek	Half day	29/10/06 13:24	17º 25.62'	144º 03.18'
WLSH87	Top Camp	Emu Creek	Rapid	29/10/06 13:00	17º 23.96'	144º 57.72'
WLSH70	Castle Rock	Emu Creek	Rapid	29/10/06 12:00	17º 22.53'	144º 57.72'
WLSH65	Horse US	Horse Creek	Rapid	31/10/06 14:18	17º 11.03'	144º 10.29'
WLSH89	Poison Water	Oaky Creek	Rapid	29/10/06 16:20	17º 24.61'	144º 08.53'

Figure 2.1: Site location map



2.3 Seasonal Context

Sites were surveyed in October or November 2006. This timing was chosen in order to target the late dry season in recognition of fact that many waterways in the local area typically experience stressful conditions at that time of the year due to a lack of natural stream flow, falling water levels and rising water temperatures. However, in practice the conditions encountered during the survey did not prove to be typical of the time of year. In fact most of the tributary sites that had ceased flowing during the middle of the dry season in previous years, were still flowing quite strongly in November 2006. This can presumably be attributed to an unusually late 2005-2006 wet season.

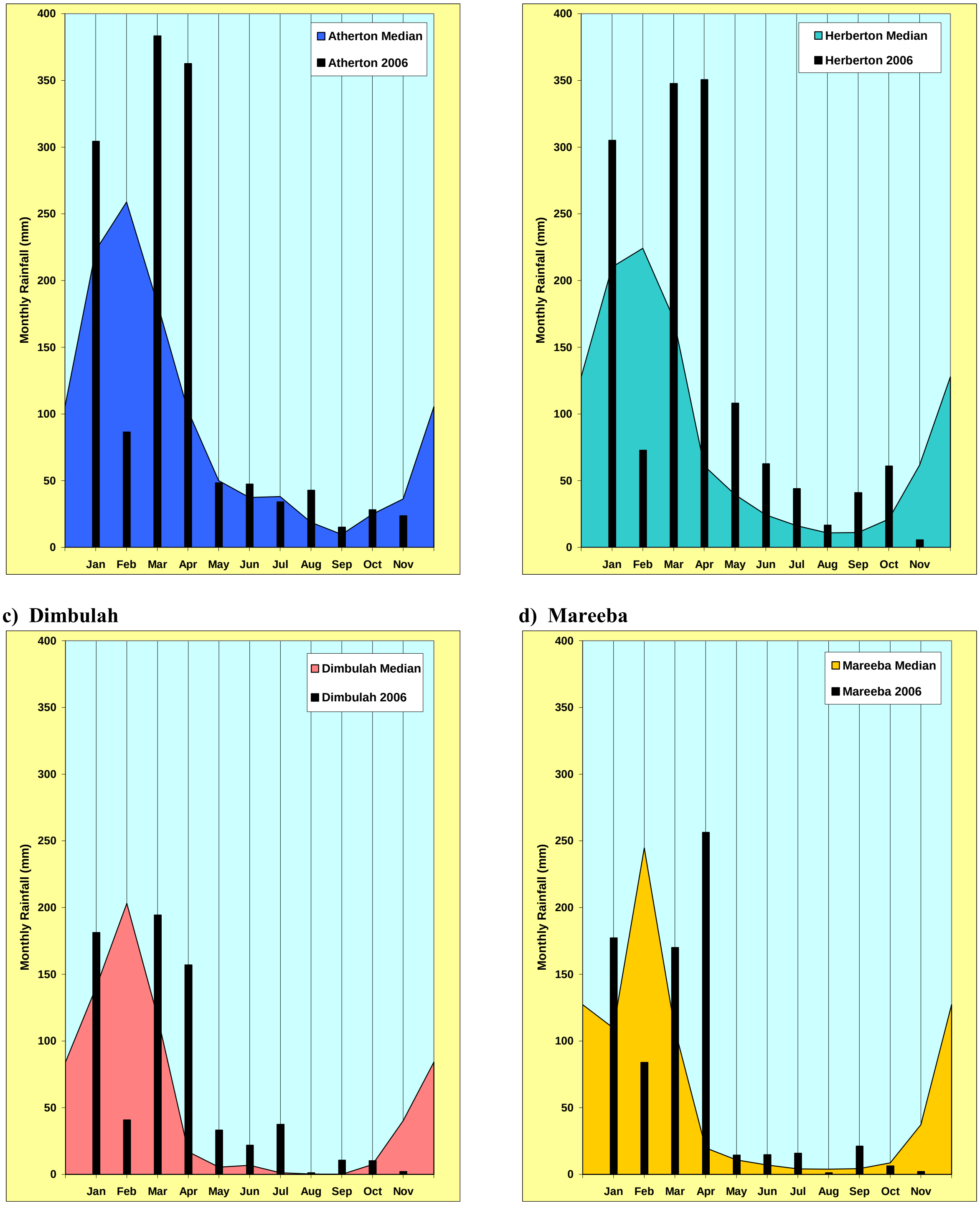
As can be seen in Figure 2.2, the rainfall registered in the region over the months leading up to the 2006 survey was only a little higher than the long term median, but a very large proportion of it occurred in April, towards the end of the wet season. Consequently the baseflows remaining in regional streams in November were unusually high and more characteristic of what would be expected earlier in the dry season, say in July or August.

This is an important factor to take into consideration when evaluating the representativeness of the survey and the significance of the results. Notably, since strong flow maintains a direct link between stream and catchment, it can make it easier to identify the sources of contaminants such as nutrients and metals. However, the ecological impacts of these contaminants may not become fully apparent until flows fall to a minimum, potentially allowing instream processes to begin adversely affecting key parameters such dissolved oxygen and pH.

Figure 2.2: Monthly rainfall for 2006 compared to long term median values for major centres in the study area.

a) Atherton **b) Herberton**

c) Dimbulah **d) Mareeba**



3. SCIENTIFIC BACKGROUND INFORMATION

Subsequent sections of this report interpret the significance of the spatial and temporal variations in physico-chemical parameters (i.e. temperature, pH and DO) recorded at each site. The following text briefly explains the scientific basis for these interpretations.

Vertical temperature profiles (i.e. variations in temperature with water depth) provide an indication of how well the water column is mixed. A significant decline in temperature with depth is referred to as a **thermocline** and indicates poor mixing. The cooler water below a thermocline is denser (i.e. heavier) than the overlying surface water, so it forms a separate discrete water layer (or **strata**). This process is termed **thermal stratification**. Many local waters heat up sufficiently to stratify during the day but cool down enough to mix during the night. Such waters are said to be **diurnally stratified** (diurnal meaning during daylight hours). Regardless of whether this happens, the surface layers of poorly mixed waters usually exhibit pronounced temperature variations over the course of each day due to normal fluctuations in sunlight intensity and air temperature. This cyclical daily variation pattern is called **diel cycling** (diel meaning over a 24 hour period), but is sometimes less correctly termed diurnal cycling.

Water mixing is important because it allows air to be transferred down into the water column to replenish the oxygen consumed by organisms living in the water, a process called **re-aeration**. Very productive waters (i.e. those that contain a lot of living organisms, including plants and microbes) consume oxygen rapidly and don't always mix quickly enough to fully re-aerate deeper waters. Hence it is possible for an oxygen gradient (termed **oxycline**) to develop even in the absence of a thermocline. Other chemical parameters such as carbon dioxide and pH can exhibit similar gradients referred to as **chemoclines**.

DO and pH fluctuations are caused by changes in the balance between photosynthesis and respiration. **Photosynthesis** is the biochemical process that allows plants and algae to grow in the presence of sunlight. It makes plants produce oxygen and consume carbon dioxide, increasing DO and pH levels in the surrounding water. (pH is influenced because carbon dioxide dissolves in water to form carbonic acid). Respiration (i.e. breathing) involves DO uptake and carbon dioxide release and therefore tends to decrease the water's DO and pH levels. Natural waters contain large numbers of organisms (mostly microscopic) that respire constantly. Submerged plants and algae also respire and take up oxygen from the water whenever it is too dark for them to photosynthetically produce their own. Most of the organisms that occupy a waterbody, live on or in the benthos (i.e. at the bottom). Hence oxygen consumption rates are usually highest near the bottom and that is why DO concentrations often decline with depth.

When there is sufficient light present, submerged plants and algae can produce enough excess oxygen to meet the needs of everything else that lives in the water. However, if the water is too dark for photosynthesis to occur (for example in deep water layers), the effects of respiratory oxygen consumption and carbon dioxide production can become evident, and DO and pH levels fall. Since photosynthesis cannot occur at night, DO and pH often exhibit pronounced diel cycling, with concentrations rising to a maximum during daylight hours and falling to a minimum overnight. Waters that are very well-mixed and/or unproductive can maintain high enough re-aeration rates to prevent such fluctuations from occurring. However, many waters in this region are poorly mixed and highly productive, and as a consequence they can experience pronounced diel cycling. Such waters usually rely heavily on the production of excess oxygen by plants and algae to maintain adequate DO concentrations. This is a delicately balanced oxygenation system that can easily fail if plant the plants grow too vigorously and begin to consume too much oxygen at night.

Adequate DO is essential to the health and survival of most aquatic organisms, including most fish. Oxygen deficiency (termed **hypoxia**) can adversely affect fish in different ways depending on the severity and duration of exposure. Very low oxygen levels can cause acute (rapid and severe) lethal reactions such as asphyxiation. Less serious DO deficiencies may not be harmful in the short term but .prolonged exposure can gradually cause a variety of chronic lethal and sublethal effects (such as reduced growth rates and diminished breeding success). DO guidelines for the protection of north Australian fish have recently been developed (Butler and Burrows 2007). These recommend the use of two different **trigger values (TVs)**:

1) an **acute TV (ATV)** of 30% saturation, which is indicative of the concentration below which sensitive species run the risk of acute injury or death, and;
2) a **chronic TV (CTV)** of 75% saturation, the concentration below which sublethal chronic effects can potentially begin to develop.

It should be noted that in northern Australia it is natural for the DO levels in non-flowing fresh waters to fall below the CTV and local fish seldom exhibit any obvious adverse effects unless DO falls to significantly lower levels for prolonged periods. Hence the CTV is not designed to be used in a simple compliance testing role but rather as a benchmark for assessing and comparing the relative hypoxia risks presented by different situations. This is based on the understanding that risks increase in proportion to the severity and duration of a TV breach. Note though that severe hypoxia is an inherent feature of some natural habitats, so low DO values are not necessarily indicative of a problem.

4. RESULTS

Laboratory analysis results are tabulated in appendix 1, and raw macroinvertebrate data are presented in appendix 2. Field data including physico-chemical measurements and records relating to biophysical descriptors including flow, weather conditions, aquatic plant communities and substratum types are too extensive to be tabulated in this report. However, the detailed site descriptions in later sections of this report incorporate plots of all physico-chemical data including vertical profiles and datalogger records, diagrams showing the bathymetry and biophysical make-up of each site, and selected photographic images.

Since most freshwater sites in this region only flow slowly, if at all during the dry season, ambient water quality is strongly influenced by localised instream and riparian factors. This is especially true of physico-chemical parameters such as DO, pH and temperature, which as explained in Section 3, are governed by biophysical processes that function at the individual waterbody scale. Basically this means that two sites with the same contaminant inputs may exhibit quite different water quality characteristics simply because they are physically different enough to process contaminants in distinctive ways. This study has attempted to describe the conditions at key sites in sufficient detail to able to identify these key differences, and discriminate between internal and external influences on water quality and ecological health. These detailed site by site condition assessments are documented in Volume 2 of the report.

Though most water quality variables are to some extent affected by localised site-scale influences, many parameters still function adequately as indicators of larger scale (catchment or drainage system) effects, and these can validly be used to compare between sites. The most reliable of these are electrical conductivity or EC (an indicator of water salinity) and ionic composition (alkalinity, hardness and the concentrations of individual salts), none of which are greatly affected by biological processes. The total quantities of trace metals and nutrient contained in a waterbody can also be a useful indicator of catchment source, although since these substances can be assimilated and/or transformed by biota, and/or deposited in bottom sediments, the significance of the concentrations occurring within the water column (i.e. in water samples) must be interpreted more carefully.

This section focuses on these broader-scale indicators and comprises a parameter-by-parameter examination of water quality variations throughout the upper Walsh River drainage system.

4.1 Dissolved Salts

The concentration and composition of the salts contained in fresh water are not greatly influenced by biological processes. However, salt concentrations can increase substantially over time if there is significant water evaporation (a process termed evapo-concentration), and the rate at which this occurs can vary considerably between sites if they differ with respect to water depth, water residence time or exposure (to sun and wind). Water is constantly evaporating as it flows through a river system so as a rule of thumb salt concentrations generally increase with increasing distance from the water source (i.e. there is usually a salinity gradient along the length of a flowing river reach). In this context the term source refers to any point of water inflow (surface or subsurface) – obviously a new gradient is established every time the intercepts a new source.

Evaporation effects generally intensify over the course of the dry season as flow rates and water levels fall, and reach a maximum in the hot months leading up to the wet season. Salinity gradient can become distinctly non-linear in situations where water depths vary significantly along the length of the river, because evapo-concentration can accelerate enormously in river reaches that contain very shallow water. For similar reasons, salinity gradients are seldom evident in systems that stop flowing for prolonged periods, allowing the river to break up into separate waterholes, each with its own evapo-concentration characteristics.

The figures presented later in this section are designed to facilitate convenient comparisons between sites, subcatchments and watercourses. However, when making such comparisons it is important to remember that most tributary streams only flow intermittently and are generally shallower than river sites. Hence they are more strongly influenced by evaporation and, other factors being equal, are more likely to develop elevated salt concentrations during the dry season. Moreover, some monitoring sites are located within isolated waterholes which can develop their own independent water quality characteristics. The condition of such a site is not necessarily indicative of the quality of the catchment or of the waters that flowed through it earlier in the year.

The salinity status of the upper Walsh and Cattle Ck catchments has been assessed previously by Butler and Burrows (2006). Electrical conductivity (EC) is a reliable indicator of salinity so they analysed historical EC data (1968 to 2002), to determine if there was any evidence that existing soil and groundwater salinity problems in the Cattle Creek catchment were impacting on surface waters. Their main conclusions were as follows:

- All surface water monitoring sites in the upper catchment reported low to moderate EC levels (<300 µS/cm) indicative of salinities well below the concentrations that could adversely affect environmental values (see Figure 4.1);
- Cattle Ck was measurably more saline than the river during the dry season, but the observed concentrations were well within natural expectations and could be explained entirely by evaporation effects, and;
- The main impact that irrigation and river regulation have had on stream salinities is to reduce and stabilise dry season salinity levels in regulated upper reaches of the river.

The latter effect can be attributed partly to constant dilution from inflows of low salinity water from Tinaroo Dam and partly to a reduction in evapo-concentration rates caused by decreased water residence times and increased water depths.

The seasonal reduction in the salinity of the regulated reaches is evident in Figure 4.2, which has been extracted from Butler and Burrows (2006b). This typical example shows how salinity values remained stable and low for the duration of the 1998 dry season. Note that the black lines on the graph encompass all data-points obtained from sites located between Cattle Ck and Flatrock. They illustrate the general seasonal trend at these sites, which was for salinity to remain stable or decline slightly over time. The green lines on the other hand indicate the natural seasonal trend observed at unsupplemented sites further downstream. This is indicative of what would have been expected in the upper river reaches if they had been unregulated and subject to natural evapo-concentration effects.

Figure 4.2: EC values reported at upper Walsh River sites during the 1998 dry season. Refer to text for explanation of reference lines. (Butler and Burrows 2006b).

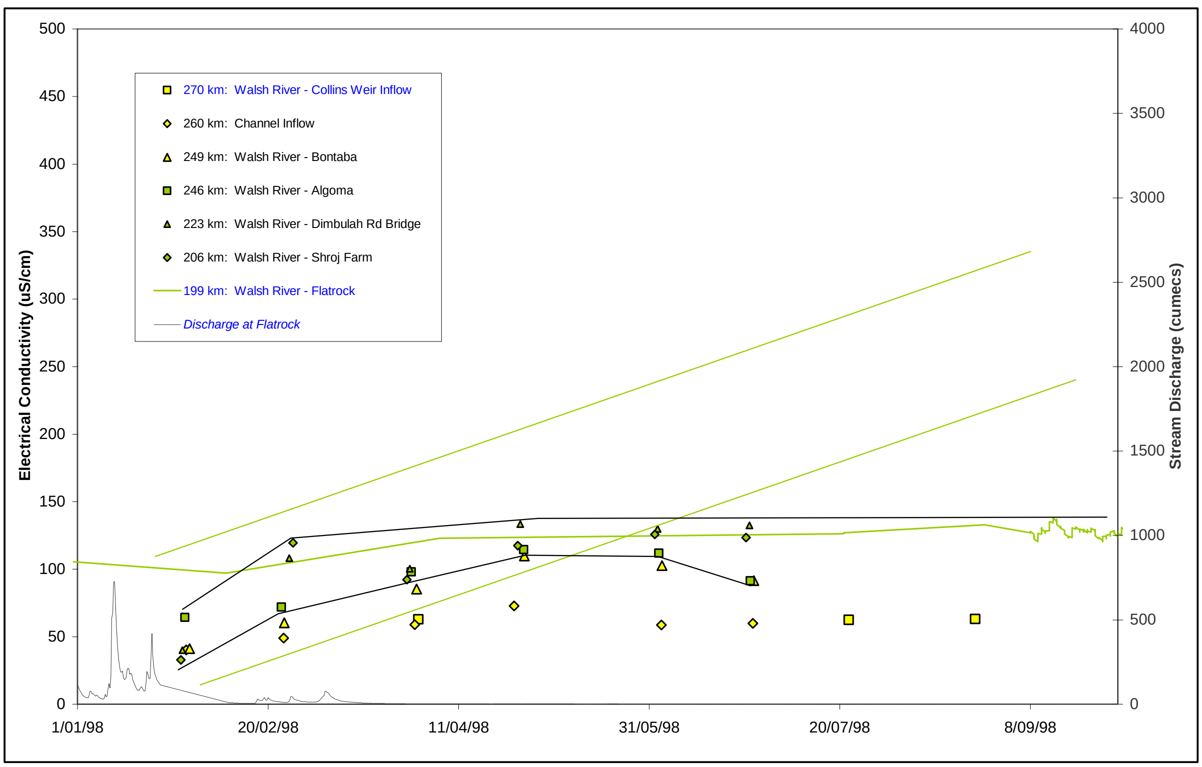
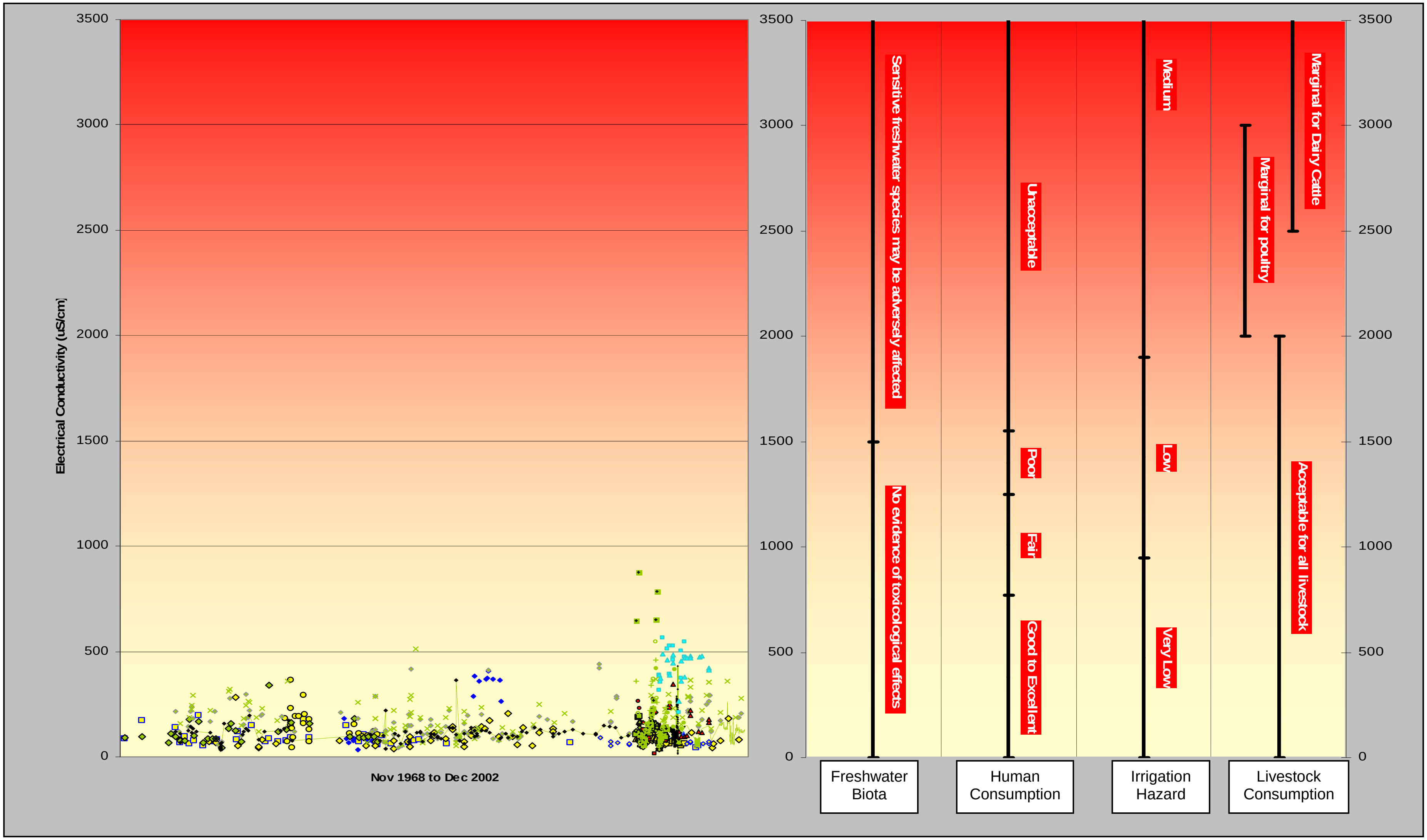


Figure 4.1: Left: Scatter-plot showing temporal variations in EC at monitoring sites in the Walsh catchment between1968 to 2002.
Right: Water quality criteria for the protection of various environmental values. (Taken from Butler and Burrows 2006b).



EC is the most commonly used indicator of water salinity, but due to a field instrument failure EC values were not collected at all sites in this study. Fortunately all samples were analysed for water hardness (the concentration of calcium and magnesium salts) and this is also a reasonable indicator of salinity.

The EC and hardness results obtained during this study are plotted in Figure 4.3. They largely confirm the findings of the previous study and can be summarised as follows:

- All sites for which EC data are available reported values less than 250 µS/cm which is indicative of low to moderate salinity levels well below the concentrations that would normally be expected to threaten environmental values (see Figure 4.1).
- The water hardness at WLSH50 on Bullaburrah Ck (235.6 mg $CaCO_3$/L) was almost 4.4 times higher than any of the sites for which EC values were available. The EC at this site would also have been elevated (a value as high as 800 to 1000 µS/cm being possible). As can be seen on figure 4.1 that is an unusually high salinity concentration for this region, but would still be expected to have only minor adverse effects on the aquatic ecosystem.
- WLSH51 on Bullaburrah Ck also had a moderately elevated hardness level (65 mg $CaCO_3$/L) but this is still consistent with an EC value less than or equal to 250 µS/cm.
- The EC results for Cattle Ck were remarkably similar to the values that have been recorded there in the past (Butler and Burrows 2006b). They strongly support the contention that during the dry season, Cattle Creek is usually significantly more saline than the river. This can be at least partially attributed to the fact that, due to supplementation, salinities in the river at that time of the year are significantly lower than they should be.
- To determine exactly what effect irrigation and farming activities have had on the EC levels in Cattle Creek it would be necessary to carry out much more detailed investigations. However, it is clear from the available data that existing salinity levels are not hazardous, so a more detailed examination would seem unwarranted at this stage.
- As would be expected, inflows of water from Cattle Creek increase the river's EC levels discernibly, but the effect is too subtle to be of any significance to the ecosystem.

The Bontaba (WLSH11) and Algoma (WLSH9) sites on the Walsh River are located immediately upstream and downstream of the point where Cattle Creek enters the river, while WLSH12 is situated in the lowest reach of Cattle Creek. These sites are close enough together to expect the concentrations salt (and many other contaminants) at Algoma to be governed almost exclusively by inputs from the other two sites. Hence the EC of the downstream site should be simple mathematical function of the balance between the discharge rates and salt concentrations at the other two sites. It follows that mass balance calculations can be carried to determine a flow ratio indicative of the proportion of the river discharge that was coming from each upstream site at the time of sampling.

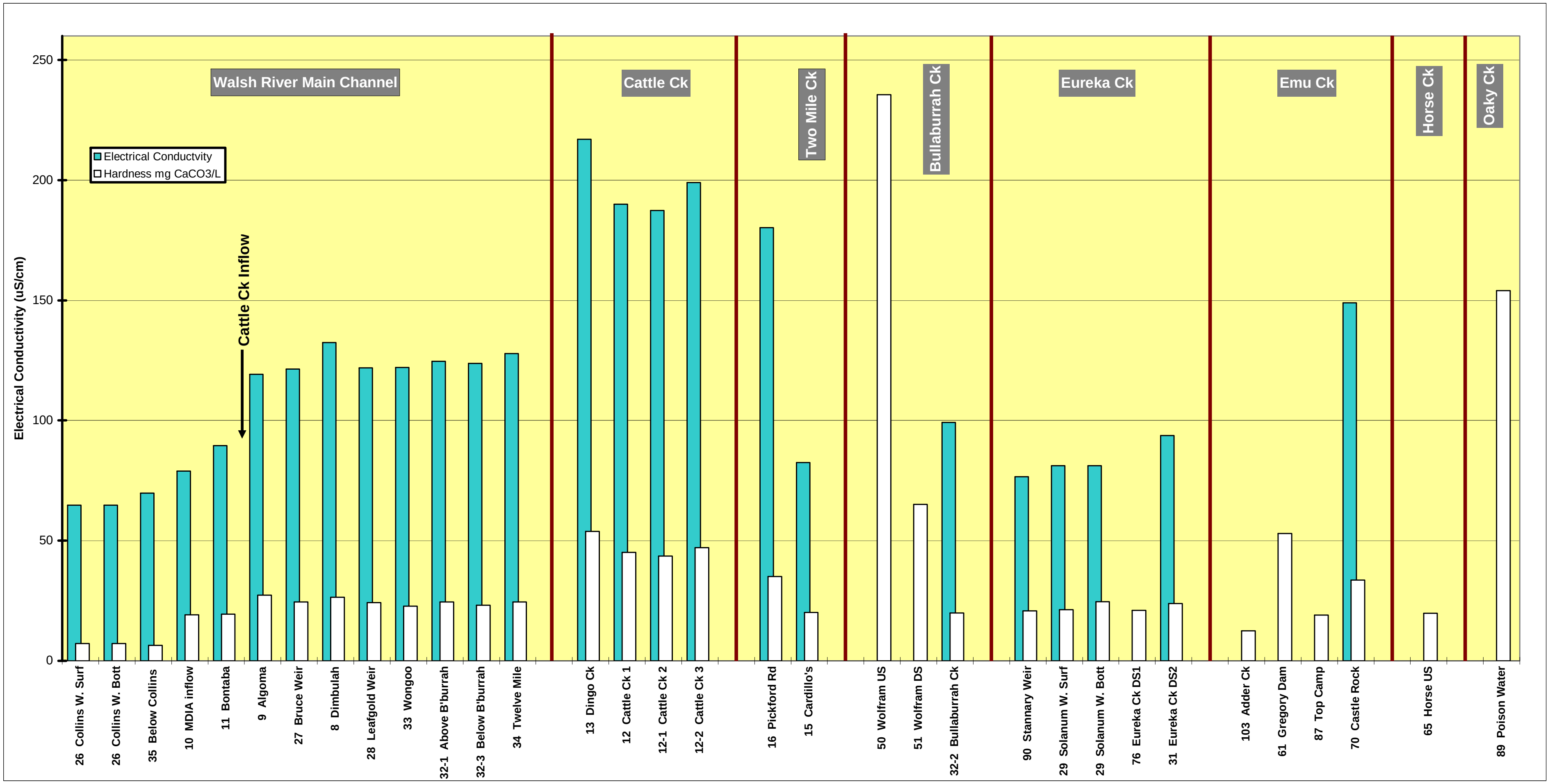
Such calculations can be performed using any water quality parameter, and theoretically, each should yield the same flow ratio estimate within a reasonable experimental error. (Note though that some parameters such as trace metals often occur at such low concentrations that values cannot be measured accurately enough to be used in this way). This constraint aside, mass balance calculations provide a convenient means of detecting anomalous results; if a parameter generates an unexpected flow estimate it can be assumed either that a measurement error has occurred or that the substance in question is behaving non-conservatively (i.e. it is being chemically or physically transformed in some way).

EC, hardness, potassium, chloride, sodium and alkalinity all generally behave conservatively and they are well mixed through the water column (meaning that they can be reliably measured by taking grab samples or spot readings). Accordingly it was not surprising to find that in this case calculations performed using each of these parameters produced very similar flow ratio estimates; the results indicating that, at the time of sampling, the discharge rate of the Walsh River at Bontaba was 2.3 (±0.1) times greater than that of Cattle Creek.

Other water quality parameters such as nutrients do not necessarily behave conservatively. They are seldom well-mixed because they are constantly being assimilated and/or released by organisms that live in different parts of the waterbody and may also interact with sediment particles that can settle to the bottom. Moreover, pulses of nutrients released from intermittent point sources such as farm drains can also take a long time to disperse. These variables make it particularly difficult to obtain representative samples. Hence results obtained from single grab samples are not always totally reliable, and it is common for mass balance equations to yield odd results.

Fortuitously, in this case total Phosphorus (TP) and filterable reactive phosphorus (FRP) both generated flow estimates that were in complete agreement with the results obtained from the salt-related parameters, suggesting that the grab samples had provided quite accurate estimates of ambient concentrations. However, the same cannot be said of nitrogen nutrient species, which yielded flow ratio estimates that were far too high (3.1 to 12.6), indicating that inputs and outputs did not balance. Possible explanations for this are discussed in Section 4.2.2.

Figure 4.3: **Results obtained for EC (µS/cm) and hardness (mg $CaCO_3$/L) during this survey (Oct to Nov 2006). Sites are grouped by catchment/ watercourse and are arranged upstream to downstream (left to right).**



4.2 Nutrients

There are no authoritative ambient nutrient concentration guidelines for riverine habitats in the northern Gulf region of Australia. ANZECC (2000) Australian water quality guidelines (AWQG) and the current Queensland water quality guidelines (QWQG) provide some default trigger values (TVs) that are meant to be applicable to waters located in other parts of the country/state. These have been derived simply by determining the 80th percentile values of data collected from reference sites located within the relevant regions. The QWQG suggest, in the absence of any better alternatives, trying to apply some of these TVs to Gulf streams, but they concede that no-one yet knows if this would be a valid procedure. Based on our previous reviews of the northern Gulf dataset (Butler and Burrows 2005) and the results of the current study we are convinced that would not be an effective approach, and accordingly the AWQG and QWQG default TVs for nutrients have not been used in this report.

Note that the default values are only meant to be used when there are no local data available – the preferred approach is to monitor local reference sites at monthly intervals for a few years and employ the 80th percentile of that dataset as a local TV. There are currently insufficient data available for potential reference sites in this area to be able to apply that approach here. In this case, as an interim measure, 80th percentile "reference values" have been derived by analysing the data collected during this study, after removing sites that were located in the immediate vicinity of farms or known nutrient point sources. The resulting reference values have been plotted on the relevant nutrient graphs to aid interpretation of results.

Elevated nutrient concentrations do not normally have any direct adverse effects on human health and/or water uses, but the ecological imbalances that nutrients cause can seriously undermine the water's desirability for human use. Moreover, nutrients from human contamination sources are symptomatic of the potential for exposure to a variety of related contaminants that can be potentially hazardous to humans. For example pathogenic microbes are often present if the nutrients arise from human or animal wastes (e.g. from sewage treatment plants or septic systems), while nutrients from farm fertilisers may be accompanied by other agricultural chemicals such as pesticides.

Nutrient enrichment, termed eutrophication, can cause excessive growth of aquatic plants and algae and often leads to proliferation of problem species such as toxic blue-green algae, malodorous and/or scum-forming algae, bacterial or fungal slimes and/or invasive weeds. Excessive growth of any species – plant, algae, animal or microbe – can upset the delicately balanced biological oxygenation system described in Section 3, resulting in oxygen depletion and a variety of associated problems including fish kills. Hypoxic (i.e. oxygen depleted) waters also tend to accumulate undesirable contaminants including toxicants such as foul-smelling hydrogen sulphide, ammonia and dissolved metals. The amounts of aquatic life (termed biomass) that the ecosystem can sustain, and hence the quantities of nutrient it can assimilate without suffering from such problems can vary enormously between waterbodies and should ideally be assessed on a site by site basis. Nevertheless, there are very few, if any riverine waterbodies in this region that can maintain healthy conditions in the long term if they are chronically subjected to nutrient concentrations that are far above normal expectations.

Some of the sites sampled in this study reported nutrient values that were two to ten times higher than any level that could conceivably be considered acceptable, and in a few cases concentrations were even high enough to potentially be acutely lethal to sensitive fish species. The significance of extreme concentrations of this sort are not difficult to interpret – they are unequivocally undesirable under any circumstances.

4.2.1 Phosphorus (P)

Analytical results for total phosphorus (TP), total filterable phosphorus (FP) and filterable reactive phosphorus (FRP) are plotted on Figures 4.4 and 4.5. Two plots were necessary because the concentrations at WLSH16 on Two Mile Creek, and to a lesser extent the sites on Cattle Creek, were so high that they could not be displayed on the same scale as other sites. This fact in itself says much about the results.

The 80th percentile reference values for TP and FRP, at 18 and 8 μg P/L respectively, are a little higher than the TVs that are normally obtained if data are collected from pristine sites over a number of seasons, but they lie well within the range of concentration values that are normally encountered in slightly disturbed rivers late in the dry season. Basically, except for the located within or immediately downstream of intensive farming areas, most sites reported quite moderate P concentrations.

The P levels at the Pickford Road site on Two Mile Creek (TP – 2411 μg P/L; FRP – 358 μg P/L) are indicative of quite extreme contamination and presumably relate to the close proximity of the sewage treatment plant. Regardless of whether that is the case, the site clearly warrants closer examination and/or remediation. Note that future investigations should include consideration of microbiological pathogens and/or disinfection byproducts, neither of which has been addressed in the current study.

Figure 4.4: **Results obtained for total phosphorus (TP), total filterable phosphorus (FP) and filterable reactive phosphorus (FRP) during this survey (Oct to Nov 2006). Reference lines are 80th percentile values obtained from a selective analysis of this dataset. The plot is scaled to allow resolution of low values, over-range results are shown on Figure 4.5. Sites are grouped by catchment/ watercourse and are arranged upstream to downstream (left to right).**

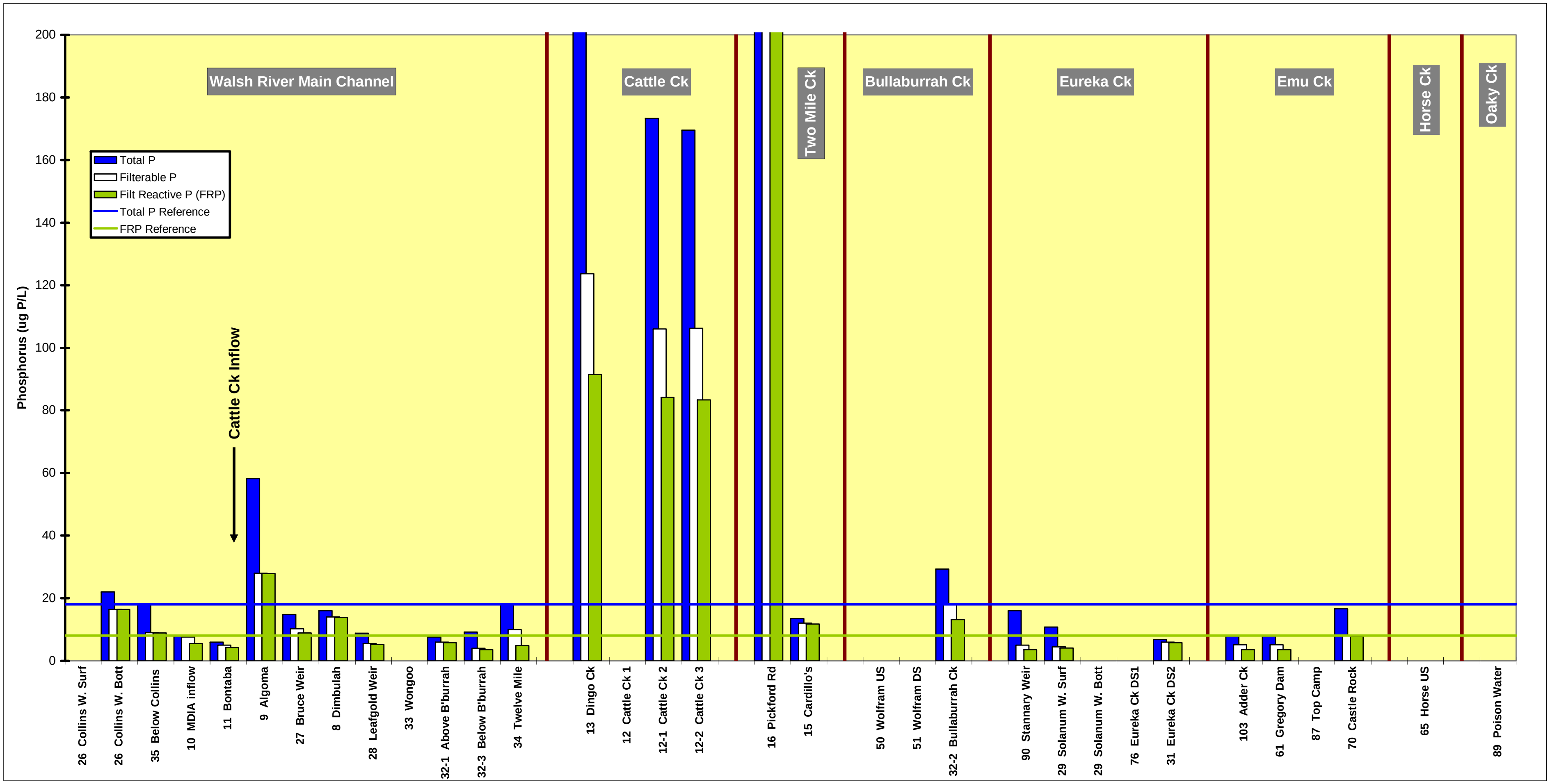
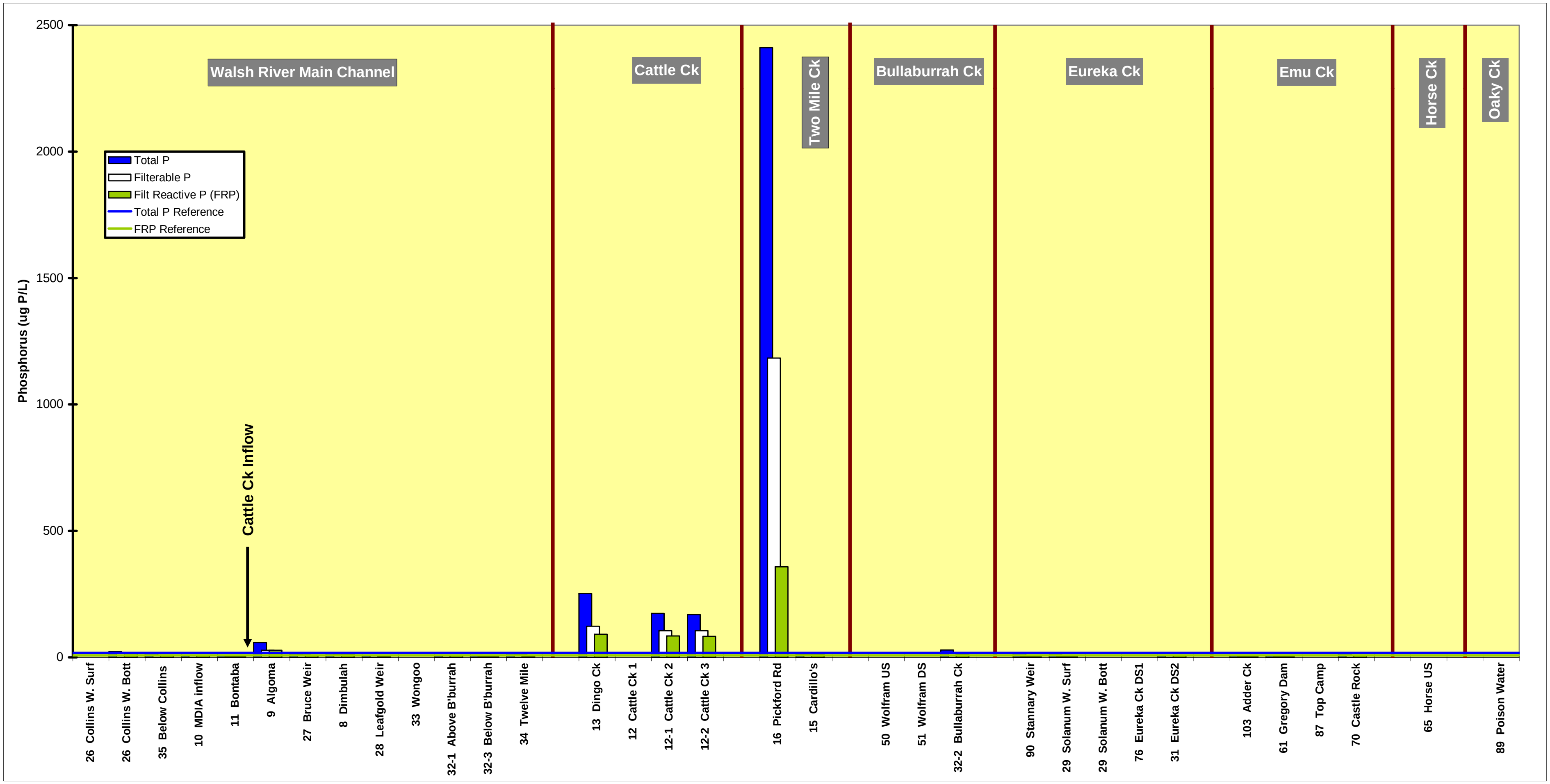


Figure 4.5: A reproduction of Figure 4.4, rescaled to display all high values.



Undesirably high P concentrations (TP – 170 to 253 µg P/L) were also reported at all sites on Cattle Creek. The extent to which this impacted on the P status of the river is difficult to gauge from a single set of grab samples. The P concentration of 58 µg P/L at Algoma, immediately downstream of the river's confluence with Cattle Creek, was obviously elevated and high enough to cause problems if it was to persist. The mass balance calculations discussed in Section 4.1 confirm that this elevated concentration was caused by inflows from Cattle Creek. However, the concentrations further downstream show no signs of being affected by these inflows, the P value reported at the next site downstream (Bruce Weir) being almost four times lower than Algoma (15 µg P/L.). It is highly unlikely that anywhere near that much P could have been lost from the water column in the relatively short time taken for the water to travel between these two sites (especially given that any P assimilated by phytoplankton would still be captured in the TP analysis). It is far more likely that at the time of sampling Cattle Creek was carrying a nutrient pulse that had only just reached the river and had not yet arrived at Bruce Weir. As discussed in the next section, the nitrogen results lend further support to this hypothesis.

4.2.2 Nitrogen (N)

Analytical results for total nitrogen (TN) and dissolved organic nitrogen (DON) are plotted in Figure 4.6. Broadly speaking it mirrors the trends that were evident in the P graphs with a few noteworthy exceptions.

The N concentrations at the Pickford Road site, though still undesirably high, were not as elevated as the P levels (i.e. the site had an unusually low N:P ratio). The Cattle Creek sites on the other hand had high N:P ratios. This suggests a high likelihood that the nutrients came from a different kind of source in each case.

Most notably the effects of inflows from Cattle Creek on the Algoma site, which were so consistently evident for all of the parameters discussed in previous sections, were not reflected in the TN and DON values. For example, in order to match the mass balance of the other parameters, the TN value at Algoma would have needed to be at least 580 µg N/L but in actuality it was only 337 µg N/L. This could be the result of sampling error (as mentioned earlier in Section 4 nitrogen can be particularly difficult to representatively sample) but it more likely indicates that, as proposed for P, a nutrient pulse was traveling through Cattle Creek and had only just started to reach the river. The TN concentration at Bontaba (immediately upstream of Cattle Creek) was only 189 µg N/L, so the 337 µg N/L value for Algoma actually represents a significant increase and strongly suggests that N concentrations had already begun to rise.

This is only a hypothesis but if it is correct it would mean that over the next few days or so, TN concentrations at Algoma would have eventually reached levels in the order of 800 µg N/L, which is a considerable concentration for a waterbody of this sort. Moreover, based on the behaviour of other parameters, a considerable proportion of this N would probably have reached the Twelve Mile site.

Dissolved inorganic nitrogen (DIN) analysis results, comprising ammonia, nitrate and nitrite, are plotted on Figure 4.7. DIN species make up a relatively small proportion of the total N pool but they are particularly important because they are the most bioavailable forms of N. In natural waters the organic nitrogen in decomposing organic matter is converted to ammonia by microbes. If there is sufficient oxygen present other microbes rapidly convert this into nitrite and then nitrate. Hence nitrate is normally the predominant form of DIN in healthy well-oxygenated waters, including the groundwaters that drive the natural baseflows in most rivers. A predominance of ammonia can therefore be result of either elevated inputs or an oxygen shortage. Excrement, sewage effluent and many popular fertilisers are major potential sources of ammonia inputs. The ecosystem will converted this to nitrate as rapidly as possible, so ammonia to nitrate ratios generally decline with increasing distance from the ammonia source. However, in waters that are not well-oxygenated this can take a considerable time and/or distance.

It is noteworthy that nitrate and ammonia were co-dominant (i.e. present in approximately equal quantities on average) at the "slightly disturbed" reference sites. In fact both parameters returned the same 80^{th} percentile value and therefore share the same reference line on Figure 4.7. Co-dominance is not uncommon for freshwaters in tropical Australia as high respiration rates, diurnal stratification and diel oxygen cycling (see Section 3) generally limit oxygen availability sufficiently to allow some ammonia to accumulate a little. (The oxygen status of the study sites is discussed in some detail in Volume Two of this report)

The Collins Weir "Bottom" site reported a moderately elevated ammonia concentration of 39 µg N/L, but that sample was collected from the hypoxic (oxygen deficient) bottom stratum of the weir. Moreover TN was not elevated at that site, so the result is clearly indicative of mild ammonia accumulation rather than elevated inputs. The Cattle and Two Mile Creek sites also contained a much higher proportion of ammonia than other sites, but in this case the samples were collected from the aerated surface layer of the water column and TN values were greatly elevated, so the result strongly suggests that a lot of N had been delivered to the creek in the form of ammonia.

Figure 4.6: Results obtained for total nitrogen (TN) and dissolved (filterable) organic nitrogen (DON) during this survey (Oct to Nov 2006). Reference lines are 80th percentile values obtained from a selective analysis of this dataset. Sites are grouped by catchment/ watercourse and are arranged upstream to downstream (left to right).

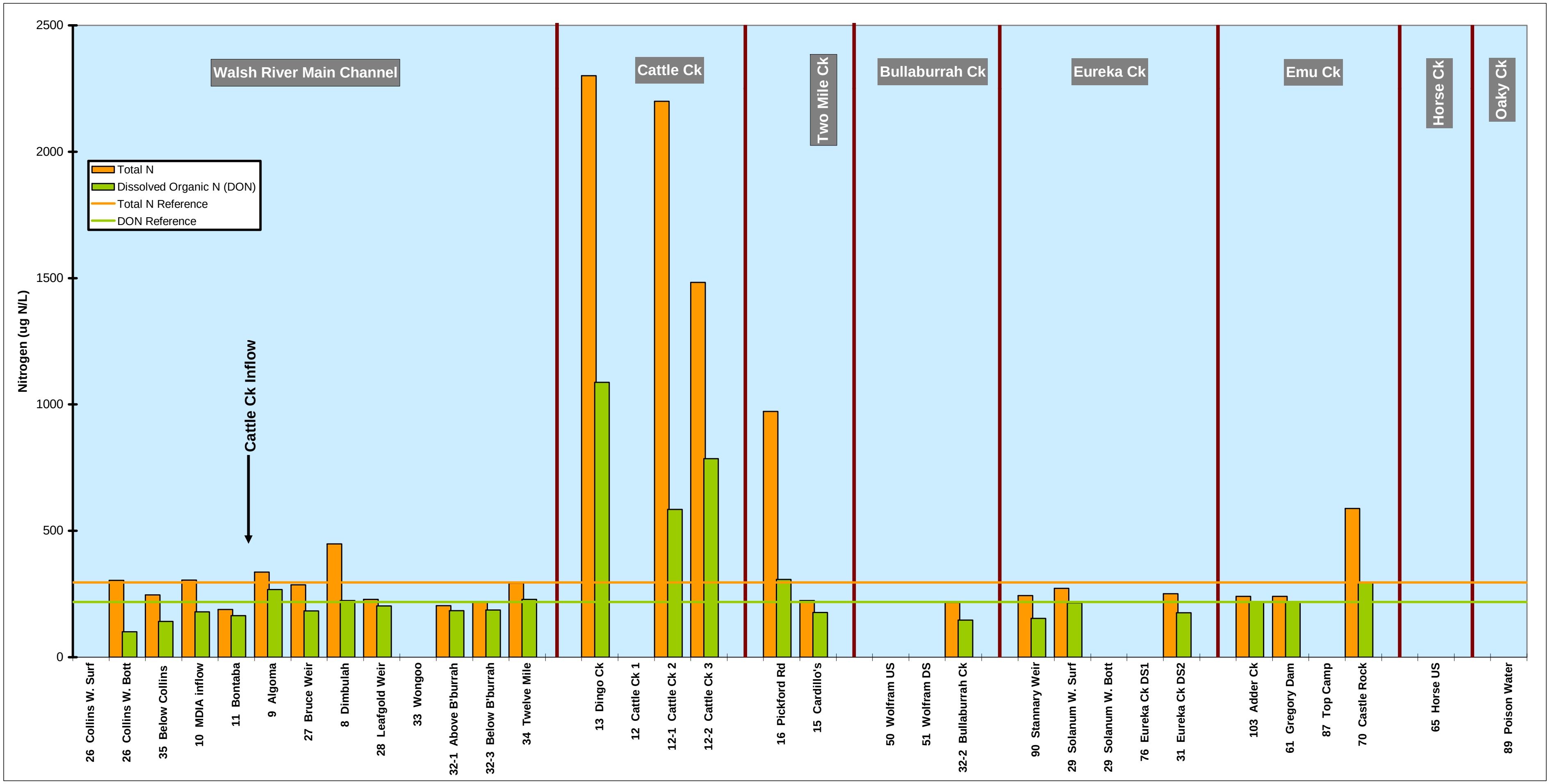
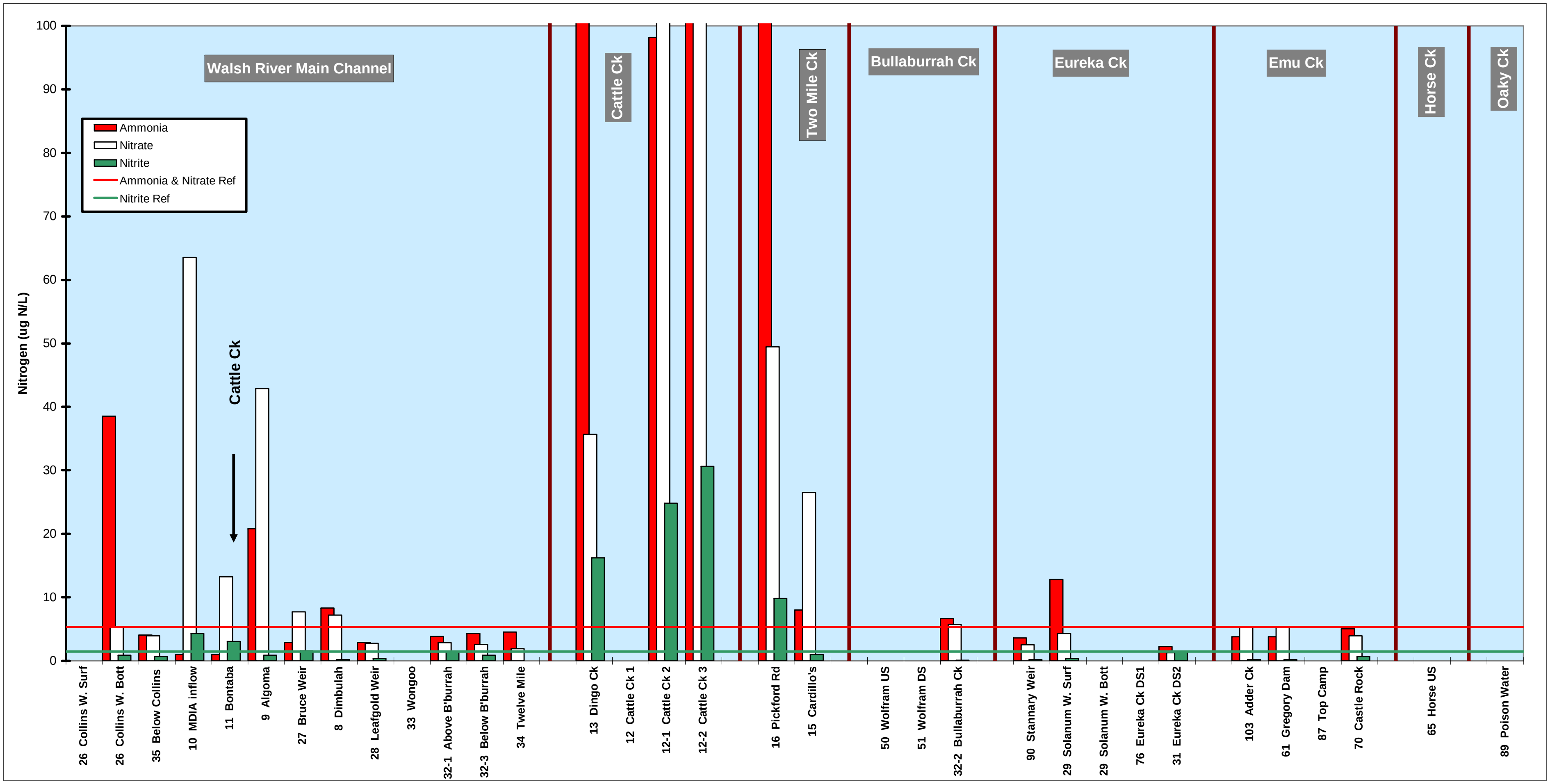


Figure 4.7: Results obtained for ammonia, nitrate and nitrite (dissolved inorganic nitrogen or DIN) during this survey (Oct to Nov 2006). Reference lines are 80th percentile values obtained from a selective analysis of this dataset. The plot is scaled to allow resolution of low values, over-range results are shown on Figure 4.8. Sites are grouped by catchment/ watercourse and are arranged upstream to downstream (left to right).



Moderately elevated DIN concentrations are of concern mainly because they can stimulate excessively rapid growth of plants and algae. The reference lines on Figure 4.7 were provided to help gauge the point at which such problems could potentially start to develop. However, extremely high concentrations of DIN, and especially ammonia, can become acutely toxic to aquatic fauna such as fish. For this reason the AWQG include secondary toxicity-based criteria for ammonia.

The toxicity of ammonia increases enormously with pH, particularly if pH values are greater than 8. Hence the TVs are a function of pH and must be calculated from a formula provided in the AWQG. Unfortunately, as will be seen in subsequent sections of this report and for reasons explained in Section 3, the pH values of most local waters can fluctuate substantially over the course of a normal day and also with water depth.

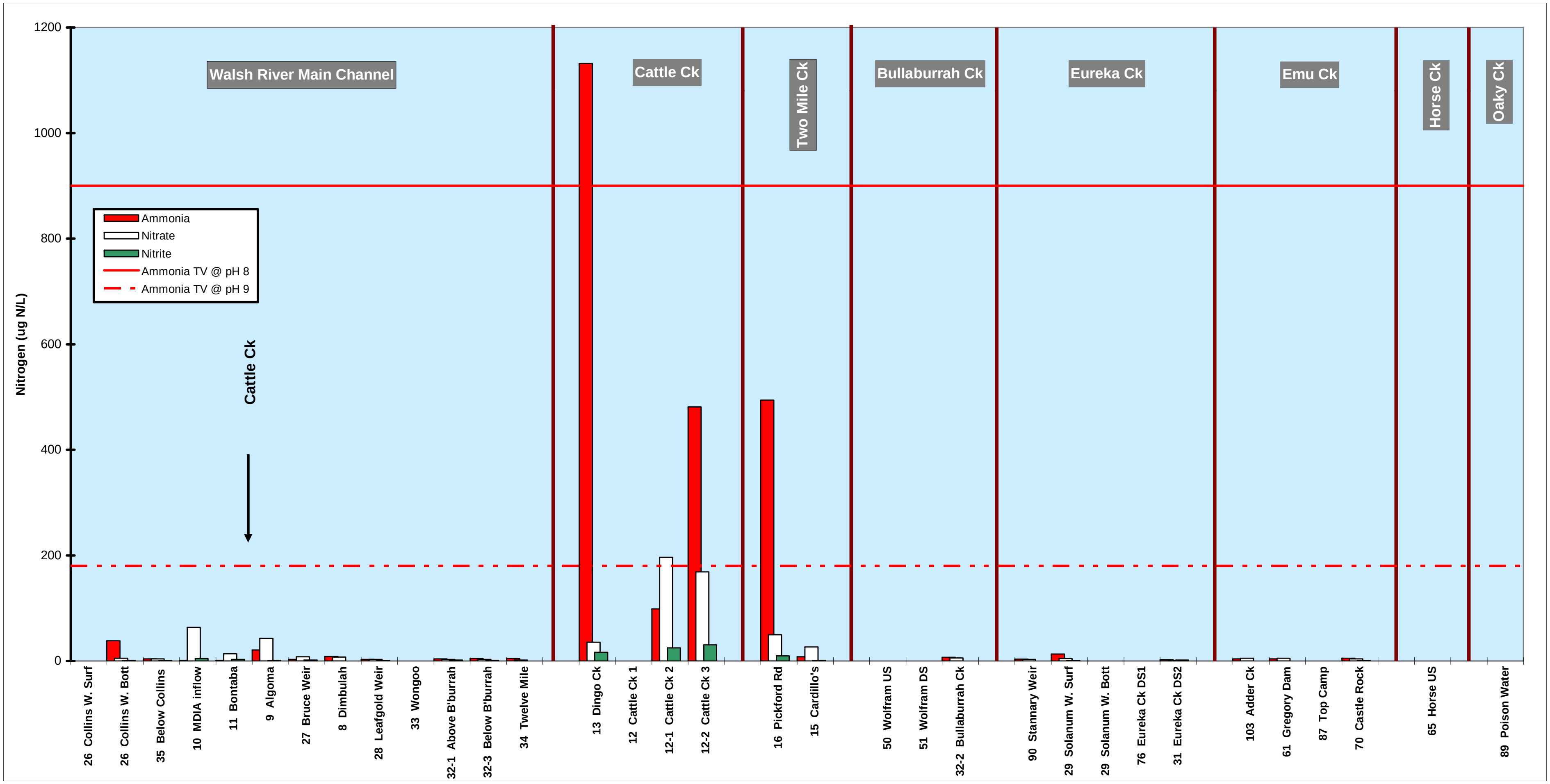
During this survey pH levels generally remained fairly close to neutral, but stream flows were unusually strong for the time of year, so aeration rates were higher than normal and consequently pH cycling was almost certainly being suppressed. Moreover, Cattle Creek had been visited during the previous dry season and on that occasion it had stopped flowing and had broken into a series of isolated waterholes. Under such circumstances pH would be expected to rise to much higher levels during the day. Based on experience with similar sites in other catchments, maximum daily pH values greater than 8.7 are very likely and values higher than 10 are quite possible, given the high nutrient concentrations and consequent potential for accelerated photosynthetic activity. Note that pH values only need to be high for a short time for ammonia to harm fauna. For example in laboratory tests juvenile barramundi were permanently damaged and eventually died after only 6 hours exposure to 700 µg N/L of ammonia at pH 9.

On balance, for the purposes of this study it seemed logical and conservative to employ two TVs for ammonia; one assuming a pH of 8, which is indicative of the maximum pH values that might be expected under the conditions encountered during this study (this is also the default value recommended in the AWQG), and another based on a pH of 9, a value that could easily develop if flow was to cease. These two TVs appear as reference lines on Figure 4.8.

As was the case with phosphorus, the DIN concentrations in Cattle Creek and at Pickford Road on Two Mile Creek were too high to be displayed on the same scale as most other sites, so Figure 4.8 has been re-scaled. The plot shows that the ammonia levels at these sites are unequivocally hazardous to aquatic fauna and have the potential to cause fish kills if pH conditions were to become unfavourable even for a short time. Concentrations this high would be cause for concern even if they were a rare occurrence. However, Ryan *et al* (2002) previously conducted monthly sampling at these sites for two years and encountered elevated nutrient concentrations on numerous occasions, suggesting that the problem is chronic. They also reported a fish kill which at the time they considered was likely to have been caused by unusually low water temperatures; however, in light of more recent toxicological data it appears very likely that ammonia was at least partially responsible.

The DIN values at other sites are not particularly noteworthy although nitrate (64 µg N/L) was noticeably elevated at the MDIA inflow, and as would be expected in view of the results obtained for other parameters, both ammonia (21µg N/L) and nitrate (43µg N/L) were elevated at the Cattle Creek inflow. These concentrations levels are high enough to affect the productivity of the ecosystem and could eventually cause damage if they were to persist, however, they wouldn't directly threaten the survival of aquatic organisms.

Figure 4.8: A reproduction of Figure 4.7, rescaled to display all high values. Note that here the reference lines are TVs indicative of the concentrations at which ammonia could become toxic to aquatic fauna under different pH conditions



4.3 Metals and Metalloids

The concentrations of metals contained in water are at least 1,000 to 10,000 times lower than the concentrations in underlying bottom sediments. Moreover, these concentrations can fluctuate considerably because metals can precipitate, dissolve, settle, re-suspend and/or change chemical form whenever conditions within the water column change. Hence water samples capture only a small and potentially variable proportion of the total pool of metals that are present at a site. The results obtained from the water samples collected during the current study only provide a snapshot indicative of what was happening at the moment when samples were collected and it is not easy to gauge how well this represents the typical state. This constraint is inherent to any study of this kind and cannot be avoided. However, this study has attempted to minimise the uncertainties involved by examining waterbody and ecosystem processes in sufficient detail to be able to make inferences about the longer term significance of findings.

The metals concentrations in bottom sediments tend to be much more stable than in the water column so it is common to rely more heavily on sediment samples when attempting to assess metal status at catchment scales. A preliminary investigation of the spatial distribution of sedimentary metal accumulations throughout the upper Walsh drainage system has already been carried out (Butler and Burrows 2006a). Though only a one-off study that did not examine variability over time, the report provides a fairly reliable basis for identifying potential metal contamination hot spots within the catchment. However, the investigation focused entirely on metals in sediment and did not include the much more detailed ecological assessments that would be required to determine if ecosystems were actually being adversely affected by metals and/or how significant an issue metals were compared to the many other contaminants and pressures that could potentially be impacting on the waterways. The work required to confidently answer such questions is far too laborious and expensive to carryout at all potential monitoring sites – the main purpose of the sediment investigation was to help identify the sites where such an investment would be most justifiable.

The summary of findings from this previous study has been reproduced in Table 4.1. Readers wishing to fully appreciate the contents of this table will need to familiarise themselves with the methodology employed in the study and may wish to consult the brief explanation provided in the text box at the end of this section. Please refer to the original report for further details.

It is important for all readers to recognise that the sediment study assessed relative rather than actual risks; and that the identified priorities relate to the need for further investigation, not management action. Even the highest priority site may not actually have been suffering any serious problems; it is simply the most logical place to start looking in order to check if the ecosystem is being adversely affected by metal accumulations. The only site surveyed that was definitely unacceptably contaminated was the Poison Water site (designated Oaky2 on the table) and it was assigned the lowest priority ranking because the problems at that site were so clearly evident that there would be no point in conducting further assessments.

In Table 4.1, the “priority based on risk index” column ranks sites on a ten point scale indicative of the potential ecological risks associated with existing metal residue concentrations; a value of one indicating the highest probability of toxicological impacts, and ten the lowest. Rankings were determined from an index score obtained by comparing the metal concentrations in the samples to Interim Sediment Quality Guideline (ISQG) values provided in the AWQG. Note that ISQG values are available for most of the metals that have historically proven to be problematic around the world, but they are not available for several of the metals analysed in this study. Hence it was not feasible to include all metals in the risk analysis. The “priority ISQG metals” column of the table lists the metals that made the biggest contribution to the overall risk scores at each site – ostensibly they are the metals with the greatest potential to be toxic at existing concentrations.

The “priority based on deviation index” column ranks sites on a ten point scale indicative of the degree of metal enrichment at each site; a value of one (or more correctly, first) indicating that the concentrations of one or more metals were highly anomalous compared to other sites in the study area, and a value of ten (i.e. tenth) indicating that metal concentrations were only very slightly higher than the median values of the overall dataset. Sites with no rank score contained no metal concentrations above the median values. Low rank scores are normally indicative of anthropogenic impacts while low to moderate scores more likely reflect natural differences in catchment geology. Note that these rankings include consideration of all metals irrespective of whether or not ISQG values were available. The “priority metals based on abundance” column of the table lists, in descending order, each of the metals that showed some signs of potential enrichment at each site, metals to the left of a “>” sign being significantly more elevated than those to the right.

Before discussing the results shown in the table, it must be stressed that findings relate only to the issue of ecosystem protection. With the notable exception of the Oaky2 site (which in the current study is referred to as the Poison Water site), none of the metal concentrations encountered at any site during the sediment study provided any indication of potential for adverse affects on humans.

Table 4.1: Priority sites for future investigation of metal status based on an assessment of the metal residues in benthic sediments. Final rankings take into consideration Potential Ecological Value (PEV), Risk Index scores and Deviation Index scores. Sites are sorted in descending order of priority. (Extracted from Butler *et al.* 2007).

Site	Watercourse	Potential Ecological Value (PEV)	Priority based on Risk Index	Priority ISQG Metals	Priority based on Deviation Index	Priority metals based on Relative Abundance*1	Final Priority Ranking
Below Collins Weir	Walsh R.	V. High	2	As>Hg	10	As	1
Jumna Dam (101)	Gibbs Ck	High	2	As>Ag, Pb, Hg	8	Pb, As, W, Sn, Ag	1
Gregory Dam (61)	Upper Emu Ck	High	2	As>Hg		*2	1
Bruce Weir (27)	Walsh R.	V. High	5	Hg, As, Ag	7	Sn, W>Sb	2
Bullaburrah (32)	Bullaburrah Ck	Medium	3	As, Ag	4	Mo>Ag, Bi	2
McGrath Bridge (54)	Lower Emu Ck	High	3	Ag>Hg	9	Ag>Mo	2
Stannary Weir (90)	Eureka Ck	High	4	As, Ag, Hg	9	Ag, W, Sn, Se	3
Wolfram US (50)	Bullaburrah Ck	High	6	Ag, As	7	Mo>Bi, Ag	3
Collins Weir (26)	Walsh R.	V. High	5	Hg, As			4
DS of rehab (76)	Eureka Ck	High	5	As, Hg, As	10	Ag	4
Castle Rock (70)	Lower Emu Ck	High	4	Ag, Hg, As	8	Sb, Ag, Sn, Se, W	4
Mishap (71)	Mishap Ck	Low	2	As>Hg	9	Mo, As, Co, Be	4
Site (58)	Gibbs Ck	Medium	2	Ag>Hg	9	Be, Ag, As, Sb	4
Emuford Dam (73)	Upper Emu Ck	High	6	As, Hg	10	Mo	5
Long Waterhole (74)	Gibbs Ck	High	5	Hg, As, Zn	8	Sb, Sn, Zn, Pb	5
Double Barrel Ck (75)	Gibbs Ck	Medium	4	Zn, Hg, As, Cu	6	Sb, Zn, Cu, Sn, Pb, Tl	5
Sandy Ck (78)	Sandy Ck	Low	3	As>Ag	9	As, Mo, Be	5
Wolfram DS (51)	Bullaburrah Ck	Low	3	Ag>Hg	5	Mo>Ag>Bi	5
Horse Ck (65)	Horse Ck	Medium	4	As, Hg			6
Adder (103)	Upper Emu Ck	Medium	6	As, Ag, Hg	9	Be, Co	6
Solanum Weir (29)	Eureka Ck	High	9				6
Wongoo (33)	Walsh R.	High	8	Hg, As			6
Below Murphys (30)	Eureka Ck	Medium	8	Hg, As			6
Stannary DS1 (91)	Eureka Ck	Medium	7	As, Hg			6
Site (93)	Gibbs Ck	Medium	4	As, Hg	9	Sb, Co, As, W	6
Irvinebank (94)	Gibbs Ck	Low	3	As>Hg	9	As, Cu, W, Sn	6
Top Camp (87)	Lower Emu Ck	Medium	8	Hg			7
Eccles Bridge (86)	Eccles Ck	Medium	9				7
Battery (72)	Upper Emu Ck	Low	9	Hg		*3	8
Oaky2 (88)	Gibbs Ck	Low	1	Zn>>Hg, Sb, Cu, Ni	1	Zn>Sb>Ni, Cu, Se, W, Ba	*4

*1 Metals in red are taken into consideration in Risk Index calculations and those in black are not.
*2 Sample 61 was only analysed for Hg and As.
*3 Sample 72 was analysed only for Hg and cannot be confidently prioritised here
*4 Site 88 is contaminated and is not prioritised for further investigation (see text)

It can be seen that the priority ISQG metal listings for sites located in the Gibbs Ck catchment sometimes included Cu and/or lead (Pb), but over the entire study area it was arsenic (As), mercury (Hg) and to a lesser extent silver (Ag) that most frequently occurred at ecologically significant concentrations. Note, however, that Hg does not appear on any of the relative abundance-based priority lists (i.e. no sites reported anomalous Hg concentrations), and As is only listed at a few sites with high deviation index scores (indicative of very minor enrichment). This appears to suggest that the natural background concentrations of Hg and As throughout the entire catchment area were somewhat elevated, and sufficiently so to potentially affect the ecosystem. This could be the case, however, the study employed specialised sampling techniques that targeted very fine-grained sedimentary materials, and it is conceivable that Hg and As had preferentially accumulated in that sediment fraction creating the false impression that overall concentrations were elevated. Similar considerations apply to Ag, although in that case there is greater between-site variability.

It is noteworthy that Oaky2 which ranked first, was the only site to be assigned a deviation index priority ranking higher than fourth. This is a good result as it indicates that there were no concentration anomalies high enough to suggest severe anthropogenic enrichment. A rank of 4 was allocated to site 32 on Bullaburrah Creek, due mainly to an anomalous molybdenum concentration, the ecological significance of which could not be determined due to a lack of available toxicological information for that metal.

Explanation of the methods and parameters referred to in Table 4.1.

The Potential Ecological Value (PEV) column contains qualitative value ratings that are based mainly on professional judgement. Very High PEV sites are generally large permanent waterbodies thought to be capable of supporting healthy productive and functional, though not necessarily natural, aquatic ecosystems. Low PEV sites on the other hand are mostly small ephemeral waterbodies that do not contain water long enough for aquatic biological communities to become established (in most cases the sediment samples from these sites were collected from dry creek beds). A low PEV site would normally only be prioritised for further investigation if it was thought to present a potential threat to a high value site further downstream.

The AWQG includes interim sediment quality guideline (ISQG) values for a number of commonly occurring metals. These values provide an indication of the concentrations at which metal residues are likely to become harmful to animals that live within the bottom sediments. That means that the values apply to whole sediment (i.e. to the complex mixture of different particles that make up the sediment). Unfortunately the quantities of metals contained within a bottom sediment depend on how much fine particulate material it contains – very fine muds naturally contain much more metal than coarse sands – and since sand and mud can be very unevenly distributed throughout the bottom of a watercourse, the metal concentrations in whole sediment samples can vary wildly. It is so difficult to discriminate between variations caused by these natural effects and those caused by elevated metal inputs, that it is not feasible to use whole sediment samples to assess catchment-wide metal distribution patterns. Instead assessments focus on the fine sediment portion creating a situation where it is valid to assume a direct link between metal inputs and metal concentrations. (This is basically comparing apples with apples).

However, the metal values obtained when using this approach are often considerably higher than would normally be found in whole sediment – ostensibly they provide an estimate of what the concentrations would be if the bottom sediment comprised only fine mud, and although some waterbodies can contain a lot of bottom materials of that kind, most do not. This means that the ISQG values cannot be used to directly assess the ecological significance of the data. Nevertheless the guideline values still provide an indication of the comparative toxicity of different metals and in this case they were used to calculate index values indicative of relative risk. This was done by dividing each metal concentration by its respective ISQG to yield scores that could be directly compared with one another and which could be summed to estimate the cumulative effects of the different metals present. The raw index values provided a quantitative basis for comparing sites noting that any value greater than one indicated that there was some potential risk and a value of two would indicate that there was twice the risk (of adverse effects on benthic fauna).

The raw index scores were then adjusted to reflect inherent differences the fine sediment trapping capacity of different sites (sandy river reaches being scored low and muddy bottomed waterholes being scored high). The adjusted scores when then sorted into descending order and similar values were grouped and assigned the priority rankings shown in the risk index column of the table.

Unfortunately, there are a number of metals for which ISQG values were unavailable. These were assessed by dividing the concentration values for each metal by their respective medians (obtained from statistical analysis of the overall dataset) to yield index values indicative of their degree of concentration enrichment. The resulting values were sorted, grouped and ranked in the same way as the risk index scores.

The current study was designed to complement the sedimentary metal investigations. It has a much broader focus and aims to determine the main factors and/or pressures influencing conditions within the water column at each site. It does not include all of the sites covered in the sediment study but includes some extra sites such as Cattle Creek and Two Mile Creek which were not surveyed during the sediment study because they were not considered to be potentially subject to historical mining impacts. Though sediment metal residues had already been determined at most sites, all water samples collected during the current study were still analysed for metals. This was partly to provide a more complete picture of what was happening in the water column and partly to check for the presence of any metals that might be preferentially remaining in the water instead of accumulating in bottom sediments.

Unfortunately the results of the sediment study were not yet available when the current study was planned, so there was no basis for targeting any specific metals. Accordingly samples were analysed for a broad range of metals and metalloids which can be divided into two groups: aluminium, cadmium, chromium, copper, manganese, nickel, lead, selenium, silver and zinc – elements for which AWQG TVs are available, and;. barium, beryllium, bismuth, cobalt, iron, molybdenum, antimony, tin, thallium and tungsten – elements for which TVs are currently unavailable (although most are known to be toxic at high concentrations). With the notable exception of mercury, these are the main metals that could potentially occur in water samples at detectable and/or toxic concentrations, in this region.

A decision was made not to include Hg analysis because it requires specialised sample preservation procedures and is costly to analyse. It was also reasoned that, since it is a known accumulator, Hg would definitely be picked up in the sediment surveys at any sites where it could be a potential issue. Had the results of the sediment study been available Hg analysis would have been included in order to determine if there was any evidence of elevated background levels in the water column, but at the time there was no reason to suspect that this would be necessary.

Water samples were initially analysed for total metals (without filtration). In cases where the total value for a particular metal was anomalous and/or exceeded the AWQG TVs for ecosystem protection, further analysis was performed on filtered samples. This was done because much of the metal contained in water is usually present in the form of suspended particles that cannot be taken up by organisms and which are not therefore toxic. It is generally only the dissolved forms of metals that are directly toxic to aquatic animals and it is these that need to comply with the TVs. Note though that not all filterable metals are toxic – hence the filterable values may still be over-estimates.

High total metal concentrations are worth knowing about even if they are mainly due to particulate (i.e. non-filterable) metal, because they can be indicative of the potential for future problems to develop, should conditions in the water change and alter the solubility of the metal. Nevertheless high particulate metal levels do not indicate any immediate risk of harm and may simply be the result of high suspended sediment concentrations. Note however, that one site in this study (Poison Water on Oaky Creek) reported total metal concentrations so high that they could not possibly be attributed to suspended sediment. That site was so obviously and severely contaminated that analysis of filterable metals was deemed unnecessary.

Results for the two groups of metals analysed in this study are tabulated in the Appendix. Apart from the exceptions discussed below, most of the reported values fell within the concentration ranges expected of healthy waterways and comfortably complied with the AWQG ecosystem protection TVs. (Note that the ecosystem protection TVs are far more stringent than the limits recommended for human water uses, so compliance guarantees that human interests are well protected).

Poison Water reported extreme exceedances of the guidelines for a number of metals, some of which are shown in subsequent Figures. The main anomaly not shown on graphs is an aluminium value of 53,300μg/L, which is more than 100 times higher than the maximum concentration that would be considered safe for aquatic life. (This is suggestive of acid sulphate runoff because the sulphuric acid generated when sulphide minerals are exposed to oxygenated water often dissolves and mobilises large amounts of aluminium from surrounding soils.)

The only metals that exceeded TVs at any other sites were chromium (Cr), copper (Cu) and zinc (Zn).

The chromium results are plotted on Figure 4.9. Chromium (Cr) occurs in a number of different inter-convertible oxidation states with quite different chemical and toxicological properties. Natural waters comprise mainly Cr(III) and Cr(VI), the latter being about three times more toxic than the former. Hence there are two different TVs shown on Figure 4.9. The relative proportions of each form depend on pH and organic matter, can change over time, and are difficult to confidently predict. Nevertheless, based on experience with other natural waters it is likely that the less toxic Cr(III) would have predominated under the conditions encountered on site during this study.

The results show that Cr concentrations in the northern (Cattle, Two Mile and Bullaburrah Creeks) and to a lesser extent eastern (Upper Walsh River) subcatchments were somewhat elevated, while levels in the catchments to the south (Eureka, Emu, Horse and Oaky Creeks) were more moderate and closer to the background concentrations that would normally be expected. The elevated levels in Cattle Ck do not implicate irrigation water as a source, because the concentration at the MDIA inflow was one of the lowest values reported. The consistency of the Cr distribution pattern suggests that most of the elevated concentrations may simply reflect natural differences in the geology of the catchments. In mineralised catchment areas it is not uncommon to find that natural metal concentrations are high enough to potentially play some role in determining the structure of the natural biological communities that inhabit local streams, and that could potentially be the case here.

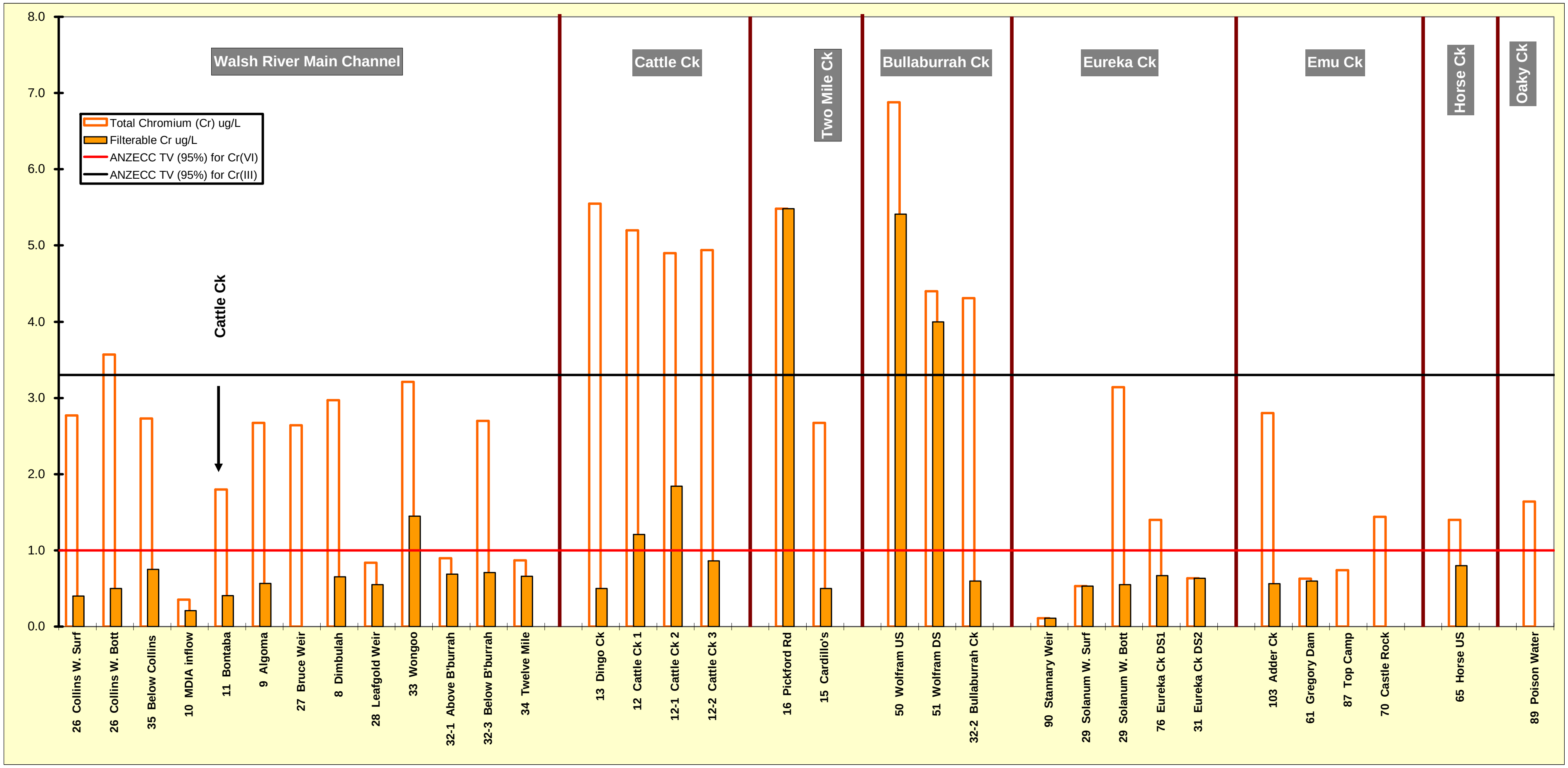
The total Cr levels in Cattle Creek exceeded the TVs suggesting that there is enough Cr in the system for toxicological symptoms to start developing if conditions in the water column were to change unfavourably in the future, as could be the case if flows slowed down or ceased. However, the filterable values were moderate enough to suggest that the Cr would not have been toxic at the time that the samples were taken. (The Cr(VI) TV was slightly exceeded in a few samples but it is highly unlikely that all of the Cr would have been present in that form). A similar situation existed in the river, except that total values were a little lower at these sites suggesting that there was less potential for adverse effects to develop if conditions in the water column were to change.

The filterable Cr at Pickford road on Two Mile Creek exceeded the Cr(III) TV suggesting that the water may have been toxic to some aquatic animal species. Notably, the total Cr levels at this site were not significantly higher than Cattle Creek – the exceedance occurred mainly because all of the Cr was filterable. This suggests that the poor water quality conditions at this site favoured dissolution of the metal, and hence that the Cr toxicity hazard was brought on by other problems such as elevated nutrient concentrations rather than elevated metal inputs.

The Wolfram Creek sites (Bullaburrah subcatchment) also exceeded the default TVs shown on Figure 4.9. However, water hardness substantially reduces Cr toxicity and the default TVs only apply to waters with hardness levels up to 30 mg $CaCO_3$/L. The water at the Wolfram Ck sites was significantly harder than that (253 and 63 mg $CaCO_3$/L, respectively) and although the scientific literature contains insufficient toxicity data to be able to precisely quantify the effects, there are strong indications that these hardness levels would have been adequate to prevent the Cr from becoming toxic in this case. The "elevated" Cr concentrations at these sites are actually only a few micrograms per litre higher than other study sites and may just reflect the subcatchment's distinctive geology.

Significantly, the sediment investigation found no evidence of significant chromium accumulation at any site in the study area. Hence there appears to be little risk of problems developing in the future due to dissolution and/or release of Cr from the sediment. On balance the Cr results appear to be indicative of slightly elevated background concentrations.

Figure 4.9: **Results obtained for total and filterable chromium (Cr) during this survey (Oct to Nov 2006). The reference line is the default trigger value (TV) for 95% protection of aquatic ecosystems (ANZECC 2000). Sites are grouped by catchment/ watercourse and are arranged upstream to downstream (left to right).**



Copper (Cu) results are plotted on Figure 4.10. The stand-out result, bearing in mind that the next highest value recorded at any other site was 29 µg/L, was the concentration of 7,100 µg/L at Poison Water on Oaky Creek, a site which as stated previously is unequivocally highly contaminated. The next highest result came from the bottom water layer of Collins Weir. However, nearly all of that Cu was particulate (and presumed non-toxic), so the results really just indicate that the deeper waters in the weir were trapping some suspended sediment. This is an inherent trait of weirs and impoundments in general and does not usually cause problems unless concentrations in the bottom sediments become enriched with the metal. The highest Cu concentrations in the Eureka and Emu Creek catchments were also recorded in impoundments (Stannary Weir and Gregory Dam, respectively), and are probably symptomatic of the same natural physical process.

The sediment study found no evidence of Cu enrichment in Collins or Stannary Weirs so the elevated particulate Cu levels in the water column do not seem to be indicative of any potential problem. In fact there was no evidence of highly anomalous Cu accumulations in the sediments of any study site (other than Poison Water), although the concentrations in Gibbs Creek were generally significantly higher than the other watercourses.

As is the case with Cr, water hardness can substantially reduce Cu toxicity (by orders of magnitude). The Eureka, Emu and Horse Creek sites recorded a number of minor TV exceedances, but these can be attributed mainly to the fact that the waters were so soft (i.e. they lacked the water hardness needed to protect the ecosystem from Cu toxicity) – the reported concentrations were actually not particularly high. This is a natural characteristic of many northern catchments and is not a cause for particular concern.

In contrast the results obtained from sites located within or downstream of the irrigation area are particularly worthy of discussion and warrant closer examination in the future.

Most of the sites that receive irrigation water exceeded the TV, and the highest Cu value was recorded at the MDIA inflow on the Walsh River. As will be seen later, the macroinvertebrate diversity at the MDIA site was significantly lower than other sites on the river and this could be due to Cu toxicity effects. At the time of sampling the inflows from Cattle Creek were actually diluting the Cu in the river (which is the reverse of what was happening for all other parameters) and concentrations in the river gradually declined with increasing distance from the inflow point. This is consistent with particulate Cu gradually settling out of the water column and/or dissolved Cu being taken up by biota, which would mean that the metal was accumulating within the system at the time. The sediment study found no evidence of anomalous sedimentary Cu accumulations at the time of sampling, but based on the water quality results, the possibility that higher levels could sometimes develop, cannot be discounted. The risk of this happening would be greatest if there was a poor wet season and the river was not flushed out well enough to remove the fine sedimentary materials that accumulate on the bottom during the dry season.

There is insufficient information available to be able to determine the source and/or full ecological significance of the elevated Cu concentrations in the irrigation supply, but these matters should definitely be investigated more closely. It is understood from discussions with local stakeholders that Cu has been introduced into the irrigation supply at times to control algal growth. If that is case the acceptability of the practice should be carefully assessed.

Zinc (Zn) results are plotted in Figure 4.11. Once again it can be seen that the levels reported at Poison Water (47,100 µg/L) were thousands of times higher than both the TV and the values reported at most other sites. The Zn concentrations in Adder Creek were much lower than that, but they were still far in excess of the default TV. Zn is yet another example of a metal with hardness-dependent toxicity, and the TV applies to waters with a hardness of 30 mg $CaCO_3$/L. The hardness levels at Adder Creek were low enough (12.5 mg $CaCO_3$/L) to suggest that Zn could have been toxic to sensitive species even if it occurred at concentrations below the default TV, so the reported concentration is particularly significant and is likely to have adversely affected the biodiversity at that site. Somewhat surprisingly, there was no evidence of elevated Zn levels in the bottom sediments at this site.

There was nothing to suggest that anthropogenic Zn enrichment had occurred at any other study sites. A few sites reported moderately elevated total Zn concentrations, but these were minor, appeared random, and comprised mainly particulate Zn, so the values could easily have been caused by some minor re-suspension of bottom sediment. The total Zn concentrations at sites located within the Eureka Creek catchment were generally higher than other sites, but filterable Zn levels were quite low and did not significantly exceed the TV. Moreover, the previous study found that the concentrations of zinc in bottom sediments at these sites were quite moderate, so the slightly elevated total values in the water column appear to be largely inconsequential.

Note that Gibbs Creek was the only catchment that reported elevated sedimentary Zn residues, and Poison Water on Oaky Creek is the only site sampled during the current study that was located in that catchment. Hence the generally moderate Zn levels reported here are broadly consistent with the findings of the previous study.

Figure 4.10: **Results obtained for total and filterable copper (Cu) during this survey (Oct to Nov 2006). The reference line is the default trigger value (TV) for 95% protection of aquatic ecosystems (ANZECC 2000). Sites are grouped by catchment/watercourse and are arranged upstream to downstream (left to right).**

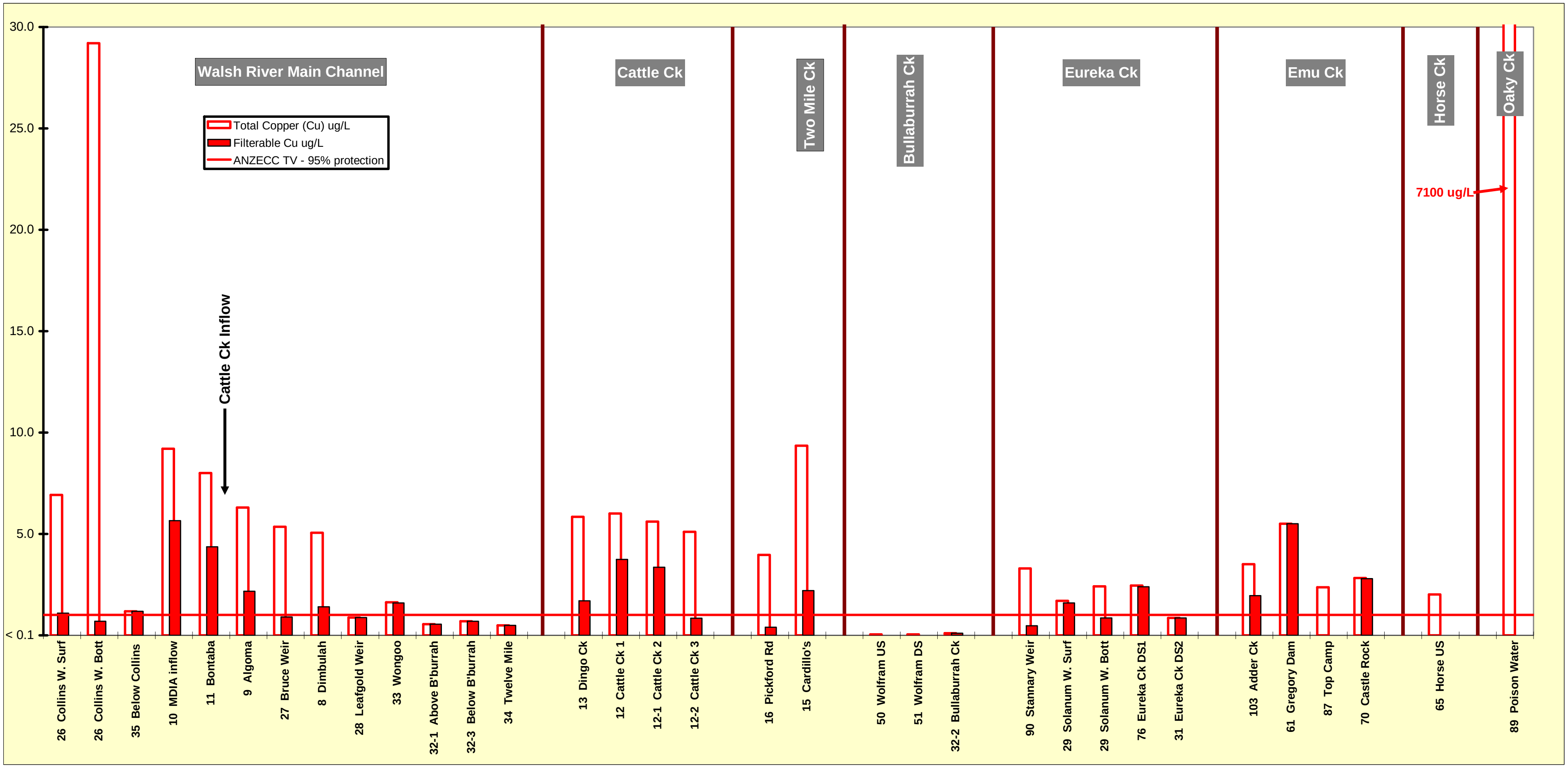
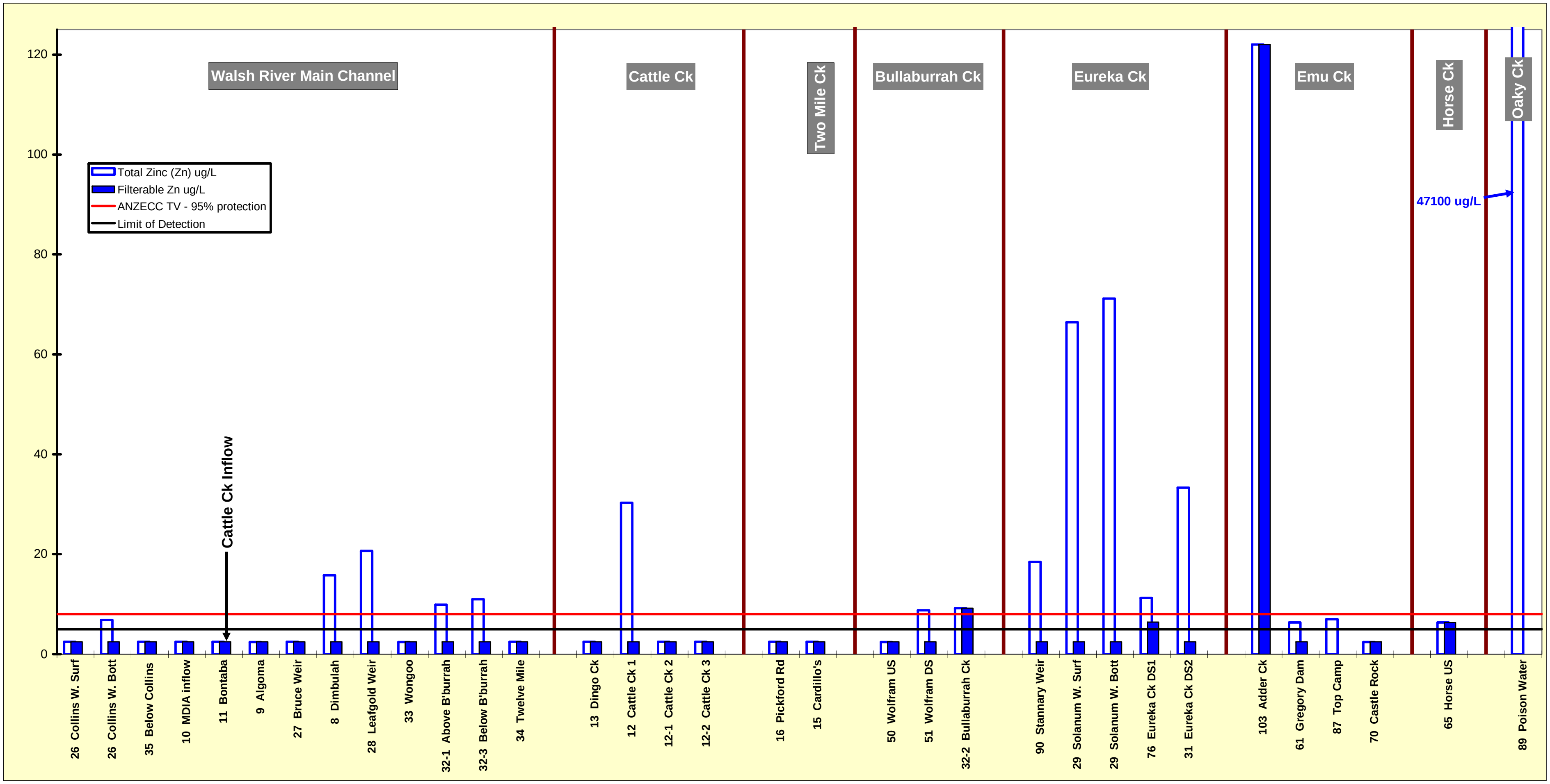


Figure 4.11: **Results obtained for total and filterable zinc (Zn) during this survey (Oct to Nov 2006). The reference line is the default trigger value (TV) for 95% protection of aquatic ecosystems (ANZECC 2000). Sites are grouped by catchment/ watercourse and are arranged upstream to downstream (left to right).**



The only other noteworthy metal results are the consistently elevated molybdenum (Mo) concentrations reported in Bullaburrah Creek. The Mo concentrations at most sites in the study area were <0.1 µg/L and the maximum value recorded for any site outside of Bullaburrah Creek was 3 µg/L. Accordingly the filterable Mo value of 96 µg/L reported at WLSH51, immediately downstream of the Wolfram Camp mine site, is quite a significant anomaly. Perhaps more significantly, concentrations were still significantly elevated (27.7 µg/L) at WLSH32, the point where Bullaburrah Creek enters the Walsh River. A Mo value of 18 µg/L was reported at site 50, upstream of Wolfram, suggesting that natural background concentrations in this catchment may be elevated; nevertheless, there is obviously some suggestion of enrichment downstream of the mining area. These results reflect the findings of the sediment metal investigation which found anomalous molybdenum residues in the bottom sediments of this creek, and assigned a fairly high priority ranking of two to site 32.

The ecological significance of these Mo values is difficult to gauge due to a lack of available toxicological information for this metal. However, as will be seen in the next section of this report, there was a localised reduction in macroinvertebrate diversity at the point where Bullaburrah Creek enters the river, and the elevated Mo concentrations appear to be the most likely potential cause.

4.4 Macroinvertebrates

Macroinvertebrates comprise all of the aquatic invertebrates such as shrimps, shellfish, worms, aquatic insects and insect larvae that are big enough to be seen without the aid of a microscope. The structure and diversity of macroinvertebrate communities are commonly used to assess the health of waterways, and a wide variety of different parameters and statistical methods of varying complexity can be used to accomplish that objective. This section of the report discusses the results obtained for two of the simplest measures of diversity: the number of Taxa present (i.e. the number of different taxonomic families and/or orders represented at each site) and the Shannon-Weaver Diversity Index (S-W Index). The S-W Index takes the numbers of individuals in each taxonomic group into account and provides a mathematical measure of how evenly spread the population is. In general high numbers for both parameters are indicative of good health; however, some eutrophic waters may actually achieve unusually high values for a while before eventually crashing.

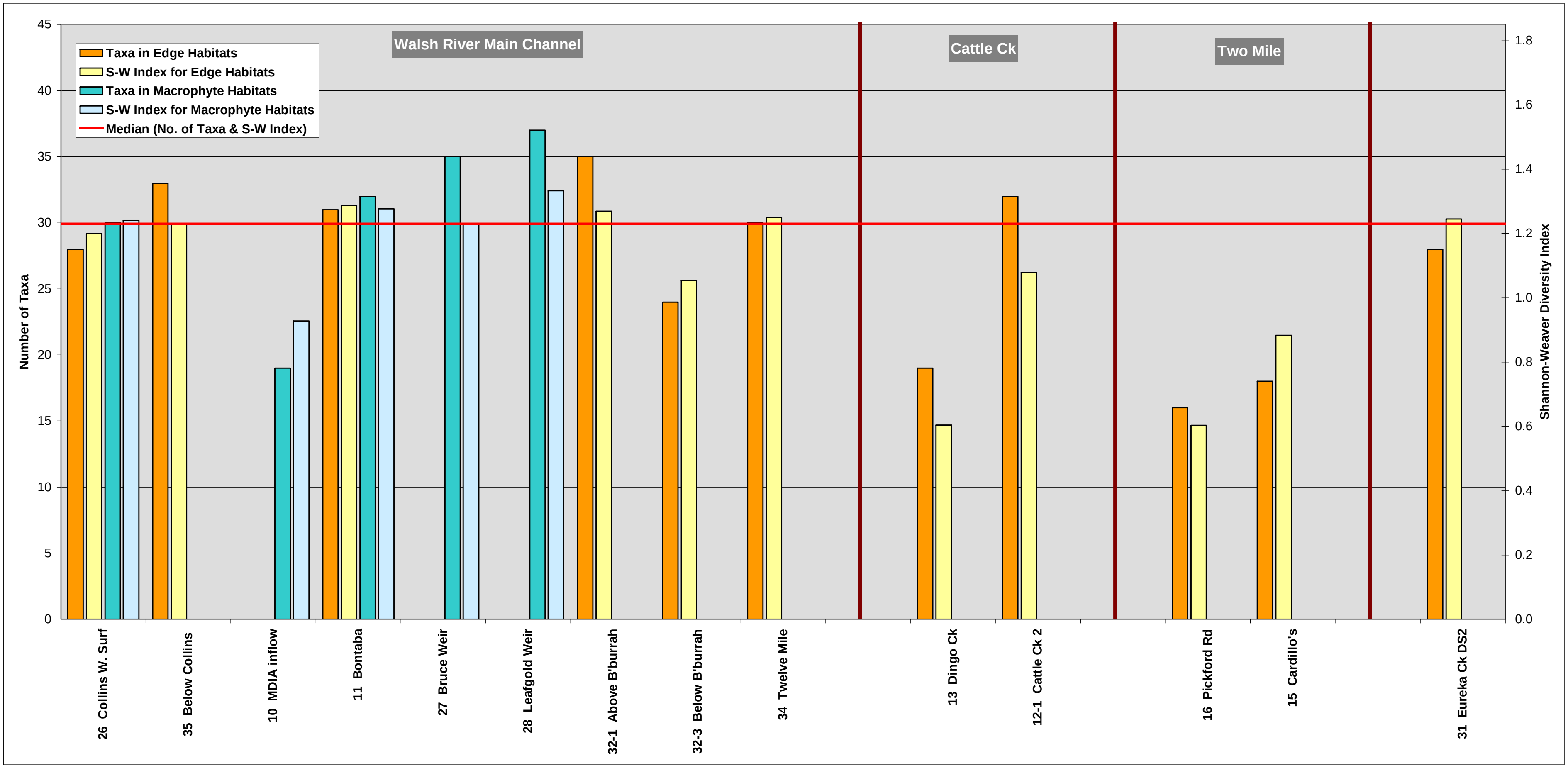
The taxa counts and S-W Index values obtained from the macroinvertebrate samples collected during this study are plotted on Figure 4.11. It can be seen that the Pickford Road site on Two Mile Creek reported the lowest diversities, followed by Cardillo's (WLSH15) and Dingo Creek (WLSH13). These findings were to be expected given the relatively poor water quality and habitat conditions at these sites.

The poor water quality conditions at Cattle Creek are not strongly reflected in the macroinvertebrate data, although the S-W Index was a little low. However, as is from the site profile in Volume two of this report, this site is unlikely to have been able to sustain that level of diversity for much longer because, due to the eutrophic conditions in the creek, oxygen consumption rates were becoming excessive. The stream was flowing quite strongly at the time of sampling but was just barely providing enough aeration to maintain DO concentrations at the critical point below which biodiversity would be expected to rapidly deteriorate. Any subsequent increases in stream biomass (and therefore oxygen consumption) or decreases in stream flow, both of which are likely to have occurred before wet season rains arrived, would very likely have resulted in a serious oxygen deficiency and a sharp reduction in biodiversity.

Most of the Walsh River sites reported moderate to high values indicative of good conditions, but the diversity of the MDIA site was quite low, especially considering that the samples were collected from macrophyte stands which normally contain slightly more diverse invertebrate communities than edge habitats. The elevated Cu concentrations at this site seem to be the only factor that could potentially explain this finding, noting that Cu can be toxic not only to invertebrates, but also to the algae that many invertebrates feed upon, and potentially to some of the macrophytes that provide them with habitat.

The edge habitats at WLSH32-3, a site situated immediately downstream of the river's confluence with Bullaburrah Creek, also reported slightly low values. The elevated inputs of Molybdenum discussed in the previous section appear to be the most likely source of this effect. However, it is worth noting that surface inflows from Bullaburrah Creek ceased at some stage during the two weeks prior to sampling, and at such times it is possible for subsurface inflows (through the basal sands) to exert some influences on the receiving environment. Subsurface waters can be quite poorly oxygenated and in the absence of any accompanying surface inflows (which would provide dilution and aeration) there is potential for macroinvertebrates living on the sands at the infiltration point to be adversely affected. Such effects are generally very temporary and localised, but they could have been coincidently encountered on this occasion.

Figure 4.12: Macroinvertebrate results obtained from triplicate samples collected at selected sites during the Oct to Nov 2006 survey. Sites are grouped by catchment/ watercourse and are arranged upstream to downstream (left to right).



5. CONCLUSIONS AND RECOMMENDATIONS

5.1 Conclusions

This report identifies and discusses a number of water quality-related issues that could potentially warrant some form of management attention. However, based on the findings of this study, reviews of pre-existing water quality data and consideration of previous reports, it is very clear that there is one major problem that demands particularly urgent management attention, and that is very poor quality of the water being discharged from the intensively developed subcatchments within the MDWSS. The very high nutrient concentrations in Cattle Creek are a particular concern because they indicate the likely presence of a variety of other agricultural contaminants, have the potential to threaten the integrity of the Walsh River, and seriously undermine the potentially significant ecological value of the creek itself. The generally degraded state of Two Mile Creek, which is highly eutrophied and infested with aquatic weeds, is also a significant cause for concern as it may potentially threaten the integrity of the upper Mitchell River.

The effects of elevated nutrient concentrations can vary somewhat with circumstances often making it difficult to gauge what levels would be acceptable or tolerable for a particular system. However, the nutrient concentrations that have been reported in these creeks are two to ten times higher than any level that could conceivably be considered acceptable, and historical data indicate that this has been the case for more than a decade. In fact some of the ammonia concentrations are actually high enough to be acutely toxic to aquatic animals and could cause fish kills under unfavourable (high) pH conditions. The significance of extreme concentrations of this sort is not difficult to judge – they are unequivocally undesirable under any circumstances.

During the dry season survey an unusually large proportion of the nitrogen contained in both creeks was in the form of ammonia, and the concentrations of phosphorus in Two Mile Creek were much higher than other sites. Sewage contains very high concentrations of both ammonia and phosphorus, so the sewage treatment plant is strongly implicated as a contributor to the contamination in Two Mile Creek. The elevated ammonia levels in Cattle Creek on the other hand can most likely be attributed to irrigation tailwater runoff from farms that had recently been treated with an ammonia-based fertiliser such as urea, ammonium nitrate or aqua ammonia. Diffuse runoff from soils and groundwaters generally delivers nitrogen to surface streams in the form of nitrate, so the high ammonia concentrations suggest that much of the runoff in Cattle Creek was being carried there via surface drains – i.e. from readily identifiable point sources that are much easier to manage than diffuse sources.

The available data are not detailed enough to be able to confidently assess the downstream impacts of nutrient discharges from Cattle Creek into the Walsh River. At the time of the survey nutrient concentrations in the river were significantly elevated at Algoma, immediately downstream of Cattle Creek, but there was no evidence of equivalent water quality effects further downstream. However, the data analysis in Section 4 suggests the strong possibility that at the time of sampling there was a pulse of nitrogen travelling down Cattle Creek that had only just started to reach the river. If this hypothesis is correct it would mean that significant pulses of contaminated water from Cattle Creek could be periodically passing through the river. Moreover, based on the behaviour of other parameters, a significant proportion of the contaminants contained in the pulses could be reaching at least as far downstream as Twelve Mile. It must be stressed though, that this is just a hypothesis, and one that could only be tested by implementing quite an intensive monitoring program.

Most of the river sites surveyed during this study were actually in quite good ecological condition and there was no evidence of impacts from nutrient pulses. However, there are grounds to suspect that the conditions encountered on this occasion were not typical. Firstly, as noted in Section 2, the 2006 wet season was quite late and regional baseflows were unusually high for the time of year. Secondly, the substrata at most sites (see the site profiles in Volume 2) showed evidence of recent intensive scouring by high water flows suggesting that most of the plants and nutrients which would have accumulated during the previous year, were probably flushed away late in the wet season, leaving little time for them to re-establish. In this region the true test of a river's health is its capacity to cope with the prolonged periods of low flow and contaminant accumulation that occur during unusually dry years, and obviously this could not be assessed during the current study. Since the flows in this section of the river are supplemented, it is less susceptible to the kinds of water quality problems that can develop in natural river reaches where the water may stagnate during droughts, but a persistent low flow of water can create other problems because it constantly introduces new nutrients into the system.

In most cases nutrient enrichment (termed eutrophication) adversely affects aquatic ecosystems by stimulating excessive growth of aquatic plants which can eventually destroy aquatic habitats and/or cause acute water quality problems such as oxygen deficiencies, algal blooms or unhealthy pH levels. However, Cattle Creek is narrow and well-shaded by riparian vegetation, so there are only a few places where there is sufficient light available for aquatic plants to grow vigorously. (This reduces the creeks susceptibility to weed invasion and is one of the major benefits gained from an intact riparian zone.) In this system the main effect of the elevated nutrients (apart from the threat of ammonia toxicity) is to promote the growth of microbes and the organisms that feed on them. These all consume oxygen, so it was not surprising to find that at the time of the survey the oxygen levels in Cattle Creek were barely sufficient to support fish life, even though the stream was being aerated by quite strong flows. It is therefore highly likely that conditions would have deteriorated rapidly as soon as flow rates declined and the water was no longer being so actively aerated.

This is an unfortunate situation because one of the surprising findings of this survey (from the perspective of investigators who had only previously seen a few of the poorer sections of the creek) is that, due to its largely intact riparian vegetation and complex geomorphology, Cattle Creek offers a greater diversity of natural aquatic habitats than any of the other sites surveyed (see the site profile and photographs in Volume Two). In fact if some of the more severe water quality problems could be ameliorated without starving the system of water, this creek would have the potential to become a regionally significant biodiversity refuge.

The severity of the problems that can develop in larger and/or more open waterbodies that receive sufficient sunlight for plants to grow (e.g. large sections of the river) depends on numerous random variables, including the intensity and frequency of stream flushing, and the types of aquatic plants that are present in the system. Therefore the impacts of nutrient inputs can vary widely between sites and over time. For example a well-flushed stream may be able to tolerate eutrophication for many years until a drought develops and there is insufficient wet season flow to wash away aquatic plants, allowing them to become fully established for the first time. Dense stands of rooted aquatic plants can reduce the flushing efficiency of the stream to the point where they can no longer be washed away when normal flows are restored, and if this happens they can become a permanent problem. Infested waterbodies of this sort often suffer from periodic oxygen deficiency problems, especially towards the end of dry spells when water temperatures are high and flow rates are low.

Eutrophic waters that are exposed to sunlight are particularly vulnerable to invasion by introduced weeds because they support such rapid plant growth that conditions within the waterbody only need to become favourable to the weeds for a short time in order for them to become firmly entrenched. Extensive sections of Two Mile Creek have already been infested by paragrass and hymanachne, and are in need of weed control management attention. However, if nutrient concentrations remain as elevated as they are currently, the weeds are likely to return so quickly that controlling them would be a substantial challenge and a major ongoing drain on resources.

Elevated nutrient concentrations do not normally have any direct adverse effects on human health and/or water uses, but the ecological imbalances that they cause can seriously undermine the water's desirability for human use. Moreover, nutrients from human contamination sources are symptomatic of the possible (and in some cases likely) presence of a variety of associated contaminants that are potentially hazardous to the ecosystem and/or human water users. For example pathogenic microbes could be present if the nutrients have arisen from human or animal wastes (e.g. from sewage treatment plants, septic systems or manure), while nutrients from farms may periodically be accompanied by other agricultural chemicals such as pesticides.

In most cases nutrients are also likely to occur in conjunction with high concentrations of oxygen demanding organic matter and this can result in oxygen deficiencies severe enough to cause acute problems such as fish kills, especially if conditions within the receiving waters are unfavourable. Note that many fish species die very quickly in deoxygenated water, so this only needs to happen very occasionally for short periods at a time to have long-lasting consequences. Also note that sugar is a particularly aggressive consumer of oxygen, so the juice spilled on sugarcane fields during harvesting can be particularly damaging if it is washed into waterways before it has had time to breakdown in the soil.

It has been well-established that there are emerging salinity hazards in several sections of the Cattle Creek catchment but to date there is no evidence that the salinity levels in the creek itself have been adversely affected. In fact even though the salinity of Cattle Creek is measurably higher than that of the river (which has been artificially reduced by inflows of low salinity Tinaroo Dam water), all of the reported concentrations are well below the levels that could adversely affect the environmental values of the water. Hence surface water salinity is not currently considered to be a major issue, although continued monitoring would still be advisable as this situation could change quite rapidly in the future, especially if watertables in the catchment continue to rise.

The concentrations of copper in Cattle Creek (and also Walsh River) were high enough to be ecologically significant. However, the copper does not appear to originate from the creek's catchment but rather from the irrigation supply; the highest copper concentrations being recorded in the Walsh River between the MDIA supplementation point and Cattle Creek. Hence the inflows from Cattle Creek were actually diluting the copper in the river, and that is the only parameter for which that occurred. Notably all sites that were receiving irrigation water reported concentrations of filterable copper that exceeded ANZECC (2000) guidelines for the protection of aquatic ecosystems, and the site with the highest concentration had significantly lower macroinvertebrate diversity than any other site on the river. The copper concentrations in question are orders of magnitude too low to affect human water uses, but they are more than high enough to potentially impact on the ecosystem.

There is insufficient information available to be able to determine the source and/or full ecological significance of the copper in the irrigation supply, but these matters should definitely be investigated more closely. It is understood from discussions with local stakeholders that copper has been introduced into the irrigation supply at times to control algal growth. If that is the case the acceptability of the practice should be carefully assessed.

Somewhat ironically, given existing concerns about the potential impacts of abandoned mines, the copper in the irrigation supply is actually the most ecologically significant and widespread occurrence of metal contamination that was encountered during this study.

Some significant metal concentrations were recorded outside of the irrigation area, but these were generally localised occurrences in the vicinity of mine sites, and were confined to quite small waterbodies with considerably lower ecological value than the waters in the irrigation area. Moreover, only one such site, Poison Water on Oaky Creek, reported metal concentrations (for zinc, aluminium and copper) high enough to be certain that the water was highly toxic to aquatic life.

There are numerous abandoned mines scattered throughout the Walsh River catchment, most of which were not properly rehabilitated. Some of these have already been shown to be discharging metals and other contaminants into waterways (Bartareau *et al* 1998), so the existence of some highly contaminated sites like Poison Water was to be expected. The key question to consider in the current study was whether or not these impacts are far-reaching enough to warrant the substantial expenditure that would need to be invested in order to rehabilitate such sites. Accordingly in this case most of the survey sites were chosen on the basis that they were the largest, most permanent aquatic refugia in their respective subcatchments that could potentially be impacted by mine site runoff. These are the sites where contaminants from mine sites would have the greatest potential to impact on the aquatic ecology of the region, rather than just a localised section of an intermittent watercourse.

The Poison Water site was one of the least ecologically important of the sites surveyed and there was no evidence of problematic metal accumulations in the more valuable sites further downstream. Based on this finding it is doubtful that attempts at rehabilitation could be justified on cost-benefit grounds.

The above-mentioned site aside, none of the water samples collected during this study yielded results that exceeded the ANZECC guidelines for any metal (for which guideline values are available) other than chromium (Cr), copper (Cu) and zinc (Zn). The chromium results appear to simply indicate that natural background concentrations were slightly elevated throughout the region, as there was no evidence of hotspots indicative of a particular contaminant source, no significant chromium enrichment in bottom sediments and no apparent effects on macroinvertebrate diversities or ecological integrity. None of the copper or zinc exceedances were particularly serious either. The zinc concentration at the Adder Creek was high enough to have had some measurable impact on the biodiversity at that site, but it was the smallest and arguably the least valuable of the surveyed sites, so this is not considered to be a major issue.

The concentrations of those metals that are not included in the ANZECC guidelines were also generally moderate, however, the water samples collected from Bullaburrah Creek all reported elevated molybdenum levels. The filterable molybdenum value of 96 μg/L at the site located immediately downstream of the Wolfram Camp mine site was quite a significant anomaly given that most sites in the region reported values less than 0.1 μg/L, and concentrations were still quite elevated (27.7 μg/L) further downstream at site WLSH32, the point where Bullaburrah Creek enters the Walsh River. A concentration of 18 μg/L was reported upstream of Wolfram Camp, suggesting that natural background concentrations in this catchment may be elevated; nevertheless, there is obviously some suggestion of possible enrichment downstream of the mining area. These results correlate with those obtained from the sediment investigation which found anomalous molybdenum residues in the bottom sediments of the creek, and assigned a high priority ranking to the WLSH32 site. The ecological significance of these molybdenum values is difficult to gauge as there is very little toxicological information available (which is also why it has no ANZECC guideline value for this metal). However, it is noteworthy that there was a localised reduction in macroinvertebrate diversity in the river at the mouth of Bullaburrah Creek, and there are no obvious water quality anomalies other than the elevated molybdenum levels that could potentially explain this finding.

The previous sediment investigation found that there were some significant residues of copper, lead and/or zinc in the bottom sediments of sites located in the Gibbs Ck catchment, but, of all the metals for which sediment quality guidelines are available, it was arsenic, mercury and to a lesser extent silver that most frequently occurred at ecologically significant concentrations throughout the study area. There were a few anomalous arsenic and silver values indicative of localised enrichment, but in general these metals were consistently high, suggesting that natural background concentrations might be somewhat elevated. However, the study employed specialised sampling techniques that targeted very fine-grained sedimentary materials, and it is conceivable that these metals had preferentially accumulated in that sediment fraction creating the false impression that overall concentrations were elevated. Notably, normal macroinvertebrate diversities were recorded at the sites in question, suggesting that the existing metal residue levels were not causing any significant toxicological effects.

The results of the water and sediment surveys both show some evidence of slightly elevated metal accumulations in the bottom of dams and weirpools. This appears to simply reflect the inherent capacity for impoundments to trap sediments and some of the contaminants they contain. None of the levels encountered are high enough to present any threats to human water users but they could be significant enough to have some impact on the ecosystem, especially under unfavourable environmental conditions (such as may develop during droughts for example). The four highest priority sites identified in the sediment study as deserving closer investigation are within, or immediately downstream of impoundments. Two of these, WLSH35 below Collins Weir and WLSH27 Bruce Weir, were surveyed in the current study and both were found to be in quite good ecological condition with high macroinvertebrate diversities. This indicates that metal accumulations are causing no serious or chronic problems, but it would still be worth checking if any episodic problems develop under less favourable environmental conditions.

5.2 Recommendations

- *Foster the development and implementation of best farming and irrigation practices in MDWSS.*

  This would require the development of strategies to ensure adoption by irrigators, but it would also entail the implementation of investigative plot-scale monitoring projects to determine the efficacy of various land and water management tactics. Monitoring would ideally involve participation by landholders with the assistance of scientists to design programs, provide training and interpret findings. It would be preferable to collect samples and send them to a reputable laboratory for analysis rather than attempting to use field analysis kits. Some samples could be taken manually but we strongly recommend the use of low cost mechanical rising stage samplers which can automatically collect a sample whenever flows increase to a pre-designated level. (Landholders can then retrieve samples at their leisure).

  Although the primary focus would be on nutrients, it would be highly advisable to check other key parameters, especially pesticides, herbicides and oxygen-demanding organic matter. Monitoring of these additional parameters can become very expensive and should undertaken in a very selective manner in order to gain maximum cost-benefits. For example samples should only be routinely analysed for these parameters when there are grounds to suspect that they could be present in high concentrations. Samples should only be analysed for the appropriate environmental residues of the chemical that has been applied – generic scans can be useful for initial orientation work but they should generally be avoided in this kind of targeted investigative work. The BOD (Biochemical Oxygen Demand) of runoff should be checked periodically (at different times of the year and stages of the cropping cycle) at all monitoring sites, and always in the first post-harvest runoff from canefarms. Note that BOD can have much more drastic effects on freshwater ecosystems than nutrients because it can starve the water of vital oxygen.

- *Investigate the feasibility of using constructed wetlands to treat agricultural runoff before it enters natural watercourses.*

  On farm management is the most effective means of minimising the impacts of agricultural runoff on the aquatic environment and every effort should be made to ensure that farmers use the smallest possible quantities of fertilisers and chemicals, and employ every known measure to keep them on the farm. However, the results of the current study suggest that the existing nutrient levels in Cattle Creek would need to be more than halved just to achieve marginally acceptable water quality, and it is highly doubtful that such a substantial reduction could be achieved in the foreseeable future even if best farming practices are adopted over the entire catchment. It may be therefore be necessary to treat the runoff in order to achieve the desired water quality outcomes, and the only cost-effective means of doing this may be to pass the water through constructed wetlands specifically designed for the purpose.

  Depending on land availability and suitability these could be small structures designed to capture the runoff from individual plots and farms, or larger versions designed to service a whole subcatchment. Wetlands treat water slowly and it is unlikely to be feasible to build one large enough to be able to retain the runoff from major storm events long enough to significantly improve its quality. In fact the disturbances caused by high flows can actually damage the wetland and reduce its treatment capacity; hence most current designs incorporate high flow by-passes that allow the runoff from big events to flow directly into the drainage system.

  The contaminants contained in these large event stormwaters will pass rapidly through the river system and will usually be carried long distances getting substantially diluted in the process. Some of them may ultimately be deposited in the floodplain wetlands and estuaries in the lower end of the river, but their effects are likely to be quite subtle compared to the overt water quality problems currently being experienced in upper catchment. In other words the potential effects of the contaminants carried by major high-volume storm flows cannot be totally ignored but they are currently much less of an issue than the contaminants introduced to the river system by tailwater releases and/or runoff from small-scale rain events during periods of low flow when there is very limited capacity to dilute or disperse them.

Properly designed wetlands of a realistic size have the capability of removing a significant proportion of the nutrients and sediment carried by these smaller, but potentially more damaging, runoff events. However, it is important to recognise that Cattle Creek, and in fact most of the tributary streams in this region, become very susceptible to oxygen deficiency problems during periods of low flow. Some wetlands can actually de-oxygenate the water in the process of removing nutrients, so it will be important to ensure that designs incorporate measures to prevent this from occurring and/or to re-oxygenate the water prior to release.

It may also prove feasible and desirable to recycle the wetland effluents back onto farms, in which case de-oxygenation would be less of a problem. However, as discussed below, the consequences of such tactics will need to be assessed carefully with due regard to the water requirements of the aquatic ecosystems we are attempting to protect – we would argue that turning the creek into a series of stagnant pools filled with potentially putrid water is not the best environmental outcome that can be achieved here. Some might suggest that this would just be restoring the stream to its natural hydrological state, but that is not a valid point of view because when it was in its natural state it did not have to contend with inputs of agricultural contaminants.

- *Develop a management plan to enhance and protect the ecological values of Cattle Creek, which are potentially very high.*

  Reducing the amounts of irrigation tailwater runoff entering the creek by fostering the adoption of improved water conservation measures and/or recycling practices could be the quickest way of minimising the transfer of contaminants from Cattle Creek to the Walsh River during the dry season. However, this would not prevent poor quality stormwaters from reaching the creek, and in the absence of flow supplementation this could actually worsen the ambient water quality problems in Cattle Creek. Reducing the flows in the creek would certainly degrade its ecological value because it currently relies heavily on flow to aerate the water and maintain aquatic habitats.

  The creek currently supports a remarkable diversity of flow-reliant hydraulic habitat types and has the potential to support an equally diverse aquatic fauna. In fact the capacity of this system to cope with poor quality water and serve as an effective biodiversity refuge would be considerably enhanced if artificial flows could be provided constantly enough to ensure that it does not stagnate. Accordingly it would be well worth assessing the feasibility of obtaining an environmental flow allocation for the creek. This might simply be a matter of using Cattle Creek to distribute some of the irrigation water that is already being delivered to the Walsh River and may not actually use much additional water.

- *Commission an investigation to determine the extent and regional significance of the degraded conditions in Two Mile Creek, assess the feasibility of water quality remediation and/or weed removal and control, and evaluate the status of those sections of the upper Mitchell River system that are potentially subject to impacts from irrigation, intensive agriculture or urban development.*

- *Decide if it is necessary to implement an intensive nutrient monitoring program to properly assess the extent to which discharges of poor quality water from Cattle Creek are impacting on the Walsh River.*

  There is more than enough evidence to be sure that the quality of the water being discharged from Cattle Creek is in need of remediation and will have been having some impacts on the river. However, the precise nature and extent of these impacts has not been fully quantified. Since water quality and environmental conditions both fluctuate considerably over time this would require the implementation of quite an intensive and costly monitoring program, and whether or not such a significant investment would be advisable depends largely on how Cattle Creek is to be managed.

  If it proves feasible to use irrigation water to maintain flows in Cattle Creek we would recommend investing in works aimed at minimising tailwater releases and improving farm runoff quality, instead of monitoring. However, if supplementation of Cattle Creek does not prove to be a viable alternative, intensive monitoring of the river would be strongly recommended in order to determine if the potential benefits of tailwater minimisation would be sufficient to justify effectively sacrificing much of the potential ecological value of Cattle Creek.

- *Investigate the source of the elevated copper concentrations in the MDWSS irrigation supply, and implement monitoring to determine if it is a persistent or recurrent problem.*

- *Commission an investigation to determine the ecological significance of metals, and especially molybdenum, at site WLSH32 on Bullaburrah Creek and sites WLSH32-1 to WLSH32-3 on the Walsh River.*

- *Check the electrical conductivity levels in Cattle Creek frequently enough to ensure early detection of any potential rises in surface water salinity.*

  The monitoring frequency required to accomplish this objective may increase over time if watertables in the catchment continue to rise. Currently a few well-timed spot readings every few years or so would probably suffice. Ideally this would be done on the tail of the hydrograph of major wet season flow events when there is greatest potential for throughflows and groundwater inflows. It would be particularly advisable to ensure that this is done after any unusually large and/or prolonged rainfall events as these have the potential to make the watertable rise, potentially bringing saline groundwaters to the surface.

- *In the event of a failed wet season and/or unusually dry year, commission an investigation to determine if the priority sites identified in this report are capable of coping with existing water and sediment quality conditions when faced with potentially stressful natural environmental conditions.*

  The key sites, in order of priority are: WLSH8 (Algoma); WLSH27 (Bruce Weir); WLSH26 (Collins Weir); WLSH36 (Below Collins Weir); WLSH101 (Jumna Dam on Gibbs Ck); WLSH61 (Gregory Dam on Emu Ck); WLSH54 (McGrath Bridge on Emu Ck), and; WLSH90 (Stannary Weir on Eureka Ck).

  If the resources are available it would also be advisable to include other sites on the Walsh River in order to gather baseline data which would be a valuable aid for assessing future changes in the condition of the river.

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APPENDIX 1: Water Analysis Results

Table A1.1 Physico-chemical laboratory water quality results

Sample Label	Site Identification	Sub catchment	Lab pH	Conductivity (µS/cm)	Alkalinity (mg $CaCO_3$/L)	Total Hardness (mg $CaCO_3$/L)	Total Suspended Solids (mg/L)	Total Dissolved Solids (mg/L)	Chlorophyll-a (µg/L)	Phaeophytin (µg/L)	Sodium (mg/L)	Potassium (mg/L)	Calcium (mg/L)	Magnesium (mg/L)	Sulphate (mg/L	Chloride (mg/L)	Bicarbonate (mg/L)
26	Collins Weir top	Walsh River	6.89	65	10.7	7.2	2.2	34	5.42	3.4	8.9	2.0	1.4	0.9	3	12	13
35	Below Collins Weir	Walsh River	6.87	70	10.8	6.4	3.7	126	2.67	1.1	9.2	2.0	1.4	0.7	3	12	13
10	MDIA inflow	Walsh River	8.19	79	24.2	19.2	1	40	2.67	0	7	1.9	3.4	2.6	3	8	29
11	Bontaba	Walsh River	7.84	90	25.7	19.4	3	45	<0.2	2	8	2.1	3.5	2.6	4	9	31
9	Algoma	Walsh River	7.43	119	33.5	27.2	2	60	1.79	2	10	3.9	5.3	3.4	5	12	41
27	Bruce Weir	Walsh River	7.35	121	32.1	24.5	1.3	62	1.34	0.4	11.8	3.2	4.7	3.1	5	15	39
8	Dimbulah	Walsh River	7.53	132	35.5	26.5	2.6	68	1.34	1.5	13	3.4	5.0	3.4	6	16	43
28	Leafgold Weir	Walsh River	7.39	122	32.6	24.3	3.1	61	2.99	2.3	12.2	3.2	4.6	3.1	4	14	40
33	Wongoo	Walsh River	7.36		33.3	22.7	3.3		1.17	1.3	13.2	3.3	4.3	2.9	4	15	41
32-1	Walsh US Bullaburrah	Walsh River	7.37	125	33.5	24.4	3.4	63	2.14	1.2	12.8	3.3	4.5	3.2	5	14	41
32-3	Walsh DS Bullaburrah	Walsh River	7.51	124	32.0	23.1	2.6	60	1.5	1.4	12.6	3.2	4.3	3.0	4	14	39
34	Twelve Mile Waterhole	Walsh River	7.45	128	33.9	24.5	8.4	64	3.52	2.2	12.8	3.1	4.7	3.1	6	14	41
16	Pickford Rd	2-mile Ck	7.04	180	50.1	35.0	5.5	88	1.07	1.0	17.7	6.1	7.6	3.9	6	17	61
15	Cardillo's	2-mile Ck	7.56	83	25.3	20.1	3.7	39	1.07	0.4	6.9	1.9	3.6	2.7	2	7	31
13	Dingo Ck	Cattle Ck	7.09	217	61.0	53.8	10.0	98	4.7	3.3	12.9	11.5	12.0	5.8	< 0.2	19	74
12-1	Cattle1	Cattle Ck	-	-	-	45.0	-	-	-	-	13	8.9	9.3	5.3	-	-	-
12-2	Cattle2	Cattle Ck	7.06	187	42.6	43.6	3	96	5.34	2	13	9	9.2	5.0	13	21	52
12-3	Cattle3	Cattle Ck	7.19	199	50.6	47.0	2.3	95	0.53	3.6	13.1	9.8	10.1	5.3	6	20	62
50	Wolfram US	Bullaburrah Ck	-	-	-	235.6	-	-	-	-	50.9	4.9	45.2	29.8	-	-	-
51	Wolfram DS	Bullaburrah Ck	-	-	-	65.1	-	-	-	-	29.9	5.3	14.2	7.2	-	-	-
32	Bullaburrah	Bullaburrah Ck	6.87	99	33.8	19.9	8.0	49	1.23	2.9	10.3	2.5	5.0	1.8	1	8	41
90	Stannary Weir	Eureka Ck	7.33	77	28.5	20.8	1.8	1	2.14	2.0	6.2	1.4	3.2	3.1	2	3	35
29	Solanum Weir top	Eureka Ck	-	81	24.9	21.3	2.7	40	1.74	1.6	6.4	2.0	3.9	2.8	3	7	30
29	Solanum Weir bottom	Eureka Ck	7.00	-	-	24.6	-	-	-	-	6.3	2.0	4.9	3.0	-	-	-
76	Eureka DS1	Eureka Ck	-	-	-	21	-	-	-	-	-	-	-	-	-	-	-
31	Eureka DS2	Eureka Ck	7.26	94	29.2	23.8	6.7	46	3.2	2.5	7.6	2.1	4.6	3.0	3	8	36
103	Adder	Emu Ck	-	-	-	12.5	-	-	-	-	9.3	1.0	2.2	1.7	-	-	-
61	Gregory Dam	Emu Ck	-	-	-	53	-	-	-	-	-	-	-	-	-	-	-
87	Top Camp	Emu Ck	-	-	-	19	-	-	-	-	-	-	-	-	-	-	-
70	Castle Rock	Emu Ck	-	-	-	33.6	3.8	-	-	-	-	-	-	-	-	-	-
65	Horse US	Horse Ck	-	-	-	19.7	-	-	-	-	23.0	3.9	4.6	2.0	-	-	-
89	Poison Water	Oaky Ck	-	-	-	154	-	-	-	-	-	-	-	-	-	-	-

Table A1.2 Nutrient results

Sample Label	Site Identification	Sub-catchment	Total Nitrogen (µg N/L)	Particulate Nitrogen (µg N/L)	PN/TN (%)	Total Filterable N (µg N/L)	Dissolved Organic N (µg N/L)	Ammonia (µg N/L)	Nitrite (µg N/L)	Nitrate (µg N/L)	Oxidised Nitrogen (µg N/L)	NO_X/TN (%)	Total Phosphorus (µg P/L)	Particulate P (µg P/L)	Total Filterable P (µg P/L)	Dissolved Organic P (µg P/L)	Filterable Reactive P (µg P/L)	TN:TP (Ratio)	DIN:DIP (Ratio)
26	Collins Weir	Walsh River	304	159	52.2	145	101	39	1	5	6.2	2.0	22	6	16	< 1	16	30.6	6.0
35	Below Collins Weir	Walsh River	246	95	38.7	151	142	4	1	4	4.6	1.9	18	9	9	< 1	9	30.3	2.2
10	MDIA inflow	Walsh River	305	57	18.8	248	179	1	4	64	68	22.2	8	0	8	2	5	84.2	27.8
11	Bontaba	Walsh River	189	7	3.7	182	165	1	3	13	16.3	8.6	6	1	5	< 1	4	69.7	9.0
9	Algoma	Walsh River	337	5	1.5	332	268	21	1	43	44	13.0	58	30	28	< 1	28	12.8	5.1
27	Bruce Weir	Walsh River	287	91	31.7	196	184	3	2	8	9.3	3.2	15	5	10	1	9	42.9	3.0
8	Dimbulah	Walsh River	448	209	46.6	240	224	8	0	7	7.4	1.6	16	2	14	< 1	14	62.0	2.5
28	Leafgold Weir	Walsh River	229	20	8.8	209	203	3	0	3	3.1	1.4	9	3	6	< 1	5	57.8	2.6
33	Wongoo	Walsh River	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
32-1	Walsh US Bullaburrah	Walsh River	204	12	5.8	192	184	4	1	3	4.3	2.1	8	2	6	< 1	6	59.2	3.1
32-3	Walsh DS Bullaburrah	Walsh River	221	27	12.2	194	186	4	1	3	3.5	1.6	9	5	4	< 1	4	53.4	4.7
34	Twelve Mile Waterhole	Walsh River	293	58	19.9	235	229	5	0	2	1.9	0.7	18	8	10	5	5	35.6	2.9
16	Pickford Rd	2-mile Ck	972	111	11.4	861	308	494	10	49	59	6.1	2,411	1227	1,184	826	358	0.9	3.4
15	Cardillo's	2-mile Ck	224	11	4.9	213	177	8	1	27	27.5	12.3	13	1	12	< 1	12	36.9	6.7
13	Dingo Ck	Cattle Ck	2300	29	1.3	2271	1087	1132	16	36	52	2.3	253	129	124	32	91	20.1	28.6
12	Cattle1	Cattle Ck	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
12-1	Cattle2	Cattle Ck	2199	1294	58.9	904	585	98	25	196	221	10.1	173	67	106	22	84	28.1	8.4
12-2	Cattle3	Cattle Ck	1483	17	1.1	1467	786	481	31	169	199	13.4	170	63	106	23	83	19.3	18.0
50	Wolfram US	Bullaburrah Ck	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
51	Wolfram DS	Bullaburrah Ck	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
32	Bullaburrah	Bullaburrah Ck	216	57	26.5	159	146	7	0	6	5.8	2.7	29	11	18	5	13	16.3	2.1
90	Stannary Weir	Eureka Ck	245	84	34.4	160	154	4	0	3	2.7	1.1	16	11	5	1	4	33.8	3.8
29	Solanum Weir	Eureka Ck	272	40	14.7	232	215	13	0	4	4.6	1.7	11	6	5	< 1	4	55.6	9.4
76	Eureka DS1	Eureka Ck	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
31	Eureka DS2	Eureka Ck	251	71	28.1	181	176	2	1	1	2.7	1.1	7	1	6	< 1	6	81.5	1.9
103	Adder	Emu Ck	241	13	5.6	228	219	4	0	5	5.4	2.2	8	3	5	2	4	66.5	5.6
61	Gregory Dam	Emu Ck	241	13	5.6	228	219	4	0	5	5.4	2.2	8	3	5	2	4	66.5	5.6
87	Top Camp	Emu Ck	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
70	Castle Rock	Emu Ck	588	281	47.8	307	297	5	1	4	4.6	0.8	17	9	8	< 1	8	78.1	2.7
65	Horse US	Horse Ck	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
89	Poison Water	Oaky Ck	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

Table A1.3 Results for ANZECC and ARMCANZ (2000) guideline-allocated metals (μg/L). Results are listed as "total conc. (filtered conc.)", i.e. filtered concentrations are shown in parentheses.

Sample Label	Site Identification	Silver (Ag)	Aluminium (Al)		Cadmium (Cd)	Chromium (Cr)		Copper (Cu)		Manganese (Mn)		Nickel (Ni)		Lead (Pb)		Selenium (Se)	Zinc (Zn)	
26	Collins Weir top	<0.05	74	(74)	< 0.5	2.8	(0.4)	6.9	(1.1)	26	-	< 0.1		0.41	(0.14)	< 1	< 5	
26	Collins Weir bottom	<0.05	976	(976)	< 0.5	3.6	(0.5)	29	(0.7)	661	-	< 0.1		4.18	(0.28)	< 1	7	(<5)
35	Below Collins Weir	<0.05	43	(43)	< 0.5	2.7	(0.8)	1.2	(1.2)	47	-	0.36	(<0.1)	0.54	(0.17)	< 1	< 5	
10	MDIA inflow	<0.05	41	(<0.5)	-	0.2	(0.2)	9.2	(5.7)	53	(16)	< 0.1		0.10	(<0.05)	< 1	< 5	
11	Bontaba	<0.05	38	(<0.5)	-	1.8	(0.4)	8.0	(4.4)	21	(12)	< 0.1		0.17	(<0.05)	< 1	< 5	
9	Algoma	<0.05	43	(<0.5)	-	2.7	(0.6)	6.3	(2.2)	107	(57)	< 0.1		0.18	(<0.05)	< 1	< 5	
27	Bruce Weir	<0.05	54	(54)	< 0.5	2.6	(0.5)	5.4	(0.9)	47	-	< 0.1		0.59	(<0.05)	< 1	< 5	
8	Dimbulah	<0.05	52	(<0.5)	-	3.0	(0.7)	5.1	(1.4)	51	(21)	< 0.1		0.23	(<0.05)	< 1	16	(<5)
28	Leafgold Weir	<0.05	52	(52)	< 0.5	0.8	(0.5)	0.9	(0.9)	59	-	< 0.1		0.26	(<0.05)	< 1	21	(<5)
33	Wongoo	<0.05	51	(3)	-	3.2	(1.5)	1.6	(1.6)	45	(36)	0.94	(<0.1)	0.18	(<0.05)	< 1	< 5	
32-1	Walsh US Bullaburrah	<0.05	62	(62)	< 0.5	0.9	(0.7)	0.6	(0.6)	64	-	< 0.1		0.24	(<0.05)	< 1	10	(<5)
32-3	Walsh DS Bullaburrah	<0.05	68	(68)	< 0.5	2.7	(0.7)	0.7	(0.7)	59	-	0.52	(0.35)	0.32	(<0.05)	< 1	11	(<5)
34	Twelve Mile Waterhole	<0.05	65	(65)	< 0.5	0.9	(0.7)	0.5	(0.5)	67	-	< 0.1		0.61	(0.1)	< 1	< 5	
16	Pickford Rd	<0.05	241	(241)	< 0.5	5.5	(5.5)	4.0	(0.4)	319	-	< 0.1		0.30	(0.07)	< 1	< 5	
15	Cardillo's	<0.05	98	(98)	< 0.5	2.7	(0.5)	9.4	(2.2)	65	-	< 0.1		0.47	(<0.05)	< 1	< 5	
13	Dingo Ck	<0.05	404	(404)	< 0.5	5.6	(0.5)	5.8	(1.7)	390	-	1.65	(1.65)	0.79	(0.17)	< 1	< 5	
12	Cattle1	<0.05	580	(7)	-	5.2	(1.2)	6.0	(3.8)	384	(384)	1.21	(0.37)	0.82	(<0.05)	< 1	< 5	
12-1	Cattle2	<0.05	82	(11)	-	4.9	(1.8)	5.6	(3.4)	251	(251)	0.834	(0.38)	0.27	(<0.05)	< 1	< 5	
12-2	Cattle3	<0.05	109	(109)	< 0.5	4.9	(0.9)	5.1	(0.8)	299	-	1.10	(1.1)	0.49	(0.14)	< 1	< 5	
50	Wolfram US	<0.05	9	(<0.5)	-	6.9	(5.4)	< 0.1		60	(11)	0.91	(<0.1)	< 0.05	(<0.1)	< 1	< 5	
51	Wolfram DS	<0.05	18	(5.1)	-	4.0	(4)	< 0.1		167	(30)	0.21	(<0.1)	0.22	(<0.05)	< 1	9	(<5)
32	Bullaburrah	<0.05	42	(42)	< 0.5	4.3	(0.6)	0.1	(0.1)	184	-	1.07	(0.14)	0.32	(<0.05)	< 1	9	(9)
90	Stannary Weir	< 0.05	63	(63)	< 0.5	0.1	(0.1)	3.3	(0.5)	66	-	< 0.1		< 0.05		< 1	<5	
29	Solanum Weir top	<0.05	88	(88)	< 0.5	0.5	(0.5)	1.7	(1.6)	71	-	< 0.1		1.66	(0.08)	< 1	66	(<5)
29	Solanum Weir bottom	<0.05	86	(86)	< 0.5	3.1	(0.6)	2.4	(0.9)	786	-	< 0.1		1.15	(0.1)	< 1	71	(<5)
76	Eureka DS1	<0.05	31	(6.9)	< 0.5	1.4	(0.7)	2.8		112	-	< 0.1		0.50	-	< 1	< 5	
31	Eureka DS2	<0.05	40	(40)	< 0.5	0.6	(0.6)	0.9	(0.9)	54	-	< 0.1		0.71	(<0.05)	< 1	33	(<5)
103	Adder	<0.05	247	(9.1)	-	2.8	(0.6)	3.5	(2.0)	1020	(659)	0.37	(0.15)	0.51	(<0.05)	< 1	12	(12)
61	Gregory Dam	<0.05	89	(2)	< 0.5	0.6	(0.6)	2.0	(2.0)	302	(189)	< 0.1		0.63	(<0.05)	< 1	6	(<5)
87	Top Camp	< 0.05	57	-	< 0.5	0.7	-	2.4		144	-	< 0.1		< 0.05		< 1	<5	
70	Castle Rock	<0.05	31	(<0.5)	-	1.4	-	2.8	(2.8)	112	-	< 0.1		0.50	(<0.05)	< 1	< 5	
65	Horse US	<0.05	15	(15)	< 0.5	1.4	(0.8)	< 0.1		59	-	< 0.1		0.35	(0.06)	< 1	< 5	(<5)
89	Poison Water	0.3	53300	-	< 0.5	1.6	-	7,610	-	14200	-	< 0.1		< 0.05		< 1	47,100	

Table A1.4 Results for metals (μg/L) with no recommended guidelines. Filtered concentrations are in parentheses.

Sample Label	Site Identification	Barium (Ba)		Beryllium (Be)		Bismuth (Bi)	Cobalt (Co)		Iron (Fe)		Molybdenum (Mo)		Antimony (Sb)		Tin (Sn)	Thallium (Tl)		Tungsten (W)	
26	Collins Weir top	8	-	< 0.1		< 0.05	< 0.1		795	-	< 0.1		< 0.05		< 0.1	< 0.05		28	-
26	Collins Weir bottom	36	-	0.427		< 0.05	3.06	(<0.1)	15600	-	< 0.1		< 0.05		< 0.1	< 0.05		6	-
35	Below Collins Weir	14	-	< 0.1		< 0.05	0.12	(<0.1)	693	-	< 0.1		< 0.05		< 0.1	< 0.05		3	-
10	MDIA inflow	8	(7.7)	< 0.1		< 0.05	0.11	(<0.1)	196	(<100)	< 0.1		< 0.05		< 0.1	< 0.05		46	(<0.1)
11	Bontaba	9	(9.2)	< 0.1		< 0.05	<0.1		296	(112)	< 0.1		< 0.05		< 0.1	< 0.05		47	(<0.1)
9	Algoma	21	(21)	< 0.1		< 0.05	0.6	(0.5)	767	(241)	< 0.1		< 0.05		< 0.1	< 0.05		19	(<0.1)
27	Bruce Weir	16	-	< 0.1		< 0.05	0.29	(<0.1)	430	-	0.214		< 0.05		< 0.1	< 0.05		63	-
8	Dimbulah	16	(16)	< 0.1		< 0.05	0.3	(0.2)	474	(187)	0.4		< 0.05		< 0.1	< 0.05		26	(<0.1)
28	Leafgold Weir	19	-	< 0.1		0.685	0.31	(<0.1)	360	-	< 0.1		0.174		< 0.1	< 0.05		3	-
33	Wongoo	19	(14)	< 0.1		< 0.05	0.42	(0.2)	447	(170)	< 0.1		< 0.05		< 0.1	< 0.05		1	(<0.1)
32-1	Walsh US Bullaburrah	18	-	< 0.1		< 0.05	0.34	(<0.1)	485	-	< 0.1		< 0.05		< 0.1	< 0.05		41	-
32-3	Walsh DS Bullaburrah	18	-	< 0.1		< 0.05	0.30	(<0.1)	487	-	< 0.1		< 0.05		< 0.1	< 0.05		6	-
34	Twelve Mile Waterhole	19	-	< 0.1		< 0.05	0.24	(<0.1)	376	-	0.109		< 0.05		< 0.1	< 0.05		3	-
16	Pickford Rd	36	-	< 0.1		< 0.05	1.05	(<0.1)	1440	-	< 0.1		< 0.05		< 0.1	< 0.05		69	-
15	Cardillo's	14	-	< 0.1		< 0.05	0.14	(<0.1)	328	-	< 0.1		< 0.05		< 0.1	< 0.05		1	-
13	Dingo Ck	75	-	< 0.1		< 0.05	5.42	(<0.1)	1700	-	0.125		< 0.05		< 0.1	< 0.05		24	-
12	Cattle1	71	(71)	< 0.1		< 0.05	2.45	(2)	2320	(840)	< 0.1		< 0.05		< 0.1	< 0.05		17	(<0.1)
12-1	Cattle2	58	(58)	< 0.1		< 0.05	1.78	(1.6)	1430	(602)	< 0.1		< 0.05		< 0.1	< 0.05		15	(<0.1)
12-2	Cattle3	65	-	< 0.1		< 0.05	2.45	(<0.1)	1560	-	< 0.1		< 0.05		< 0.1	< 0.05		16	-
50	Wolfram US	86	(63)	< 0.1		< 0.05	0.26	(0.1)	280	(144)	18	(18)	0.083	-	< 0.1	< 0.05		4	(0.3)
51	Wolfram DS	44	(31)	< 0.1		< 0.05	0.49	(0.2)	531	(248)	96	(96)	0.317	(0.2)	< 0.1	< 0.05		2	(<0.1)
32	Bullaburrah	39	-	< 0.1		< 0.05	0.93	(<0.1)	3510	-	27.7	-	< 0.05		< 0.1	< 0.05		4	-
90	Stannary Weir	5	-	< 0.1		< 0.05	< 0.1		< 100	-	< 0.1		< 0.05		< 0.1	< 0.05		< 0.1	-
29	Solanum Weir top	12	-	< 0.1		< 0.05	0.18	(<0.1)	501	-	< 0.1		< 0.05		< 0.1	< 0.05		24	-
29	Solanum Weir bottom	82	-	< 0.1		< 0.05	1.18	(<0.1)	2040	-	< 0.1		0.066	-	< 0.1	< 0.05		1	-
76	Eureka DS1	13	-	< 0.1		< 0.05	0.15	(<0.1)	228	-	< 0.1		0.201		< 0.1	< 0.05		3	-
31	Eureka DS2	13	-	< 0.1		< 0.05	0.11	(<0.1)	321	-	< 0.1		< 0.05		< 0.1	< 0.05		129	-
103	Adder	11	(9.5)	1.54	(1.1)	< 0.05	3.59	(2.2)	698	(<100)	< 0.1		0.052	(0.05)	< 0.1	0.335	(0.27)	46	(<0.1)
61	Gregory Dam	26	(14)	0.176	(<0.1)	< 0.05	0.34	(<0.1)	1170	(175)	3	(3)	< 0.05		< 0.1	0.054	(<0.05)	1	(<0.1)
87	Top Camp	10	-	< 0.1		< 0.05	< 0.1		< 100		< 0.1		< 0.05		< 0.1	< 0.05		< 0.1	
70	Castle Rock	13	(8.81)	< 0.1		< 0.05	0.15	(<0.1)	228	(<100)	< 0.1		0.201	(0.12)	< 0.1	< 0.05		3	(<0.1)
65	Horse US	16	-	< 0.1		< 0.05	0.15	(<0.1)	2100		< 0.1		< 0.05		< 0.1	< 0.05		1	-
89	Poison Water	38	-	< 0.1		< 0.05	< 0.1		< 100		< 0.1		< 0.05		< 0.1	< 0.05		< 0.1	

APPENDIX 2: Raw Macroinvertebrate Data

Habitat	bottom	edge	bottom	riffle	bottom	edge	riffle	edge	bottom	edge	bottom	edge	bottom	macrophyte	bottom	macrophyte	bottom	macrophyte	edge	edge	edge	edge	riffles	edge	bottom	edge	macrophyte	bottom	macrophyte	riffle	edge	edge	bottom	macrophyte
Family/Taxa	**WLSH35a**	**WLSH35b**	**WLSH35b**	**WLSH35**	**WLSH34**	**WLSH34**	**WLSH34**	**WLSH32-1**	**WLSH32-2**	**WLSH32-3**	**WLSH31**	**WLSH31**	**WLSH28**	**WLSH28**	**WLSH27**	**WLSH27**	**WLSH26**	**WLSH26**	**WLSH26**	**WLSH16**	**WLSH15**	**WLSH13**	**WLSH12**	**WLSH12**	**WLSH11**	**WLSH11**	**WLSH11**	**WLSH10**	**WLSH10**	**WLSH10**	**WLSH09**	**WLSH08**	**WLSH29**	**WLSH29**
Aeshnidae																																		
Amphipterygidae																																		
Ancylidae																								1	1									
Anisoptera														1																				
Atyidae				1																	13													
Austrocorduliidae		1						1																		1								
Baetidae	2	19	8	1	1		72	5	21	2		5		3	2	47	7	27	8	2	40		1	3	18	18	9	58	216	31	2	4	1	128
Belostomatidae																																		
Caenidae	6	6	39	13	46	23	43	16	1	61	8	1	47	37	55	1	21	8	4		63			4	21	17	4	85	41	31	43	64	2	19
Calamoceratidae		1						2							8		1									2	1		1					
Calocidae																																		
Calopterygidae																																		
Carabidae																1																		
Ceratopogonidae		2	2	3	4		4	2		2		2	3		3	1	4	3	5		6		9	2		4	2	5		1	5	8	4	3
Chaoborinae																																		
Chironomidae	23	63	63	82	28	11	25	18	41	41	14	21	58	46	56	12	89	55	119	10	68	18	179	103	25	49	39	41	80	31	110	45	84	50
Cirolanidae																																		
Cladocera		46	13	4		5		2	21	1	2	27	9	23	1	30	3	47	16	1	1			11	3	10	16	7	51	4				17
Coenagrionidae		3							7	1		19	1	35		102		9	13		3	1		1		24	16		86			4		8
Collembola																													1					
Conchostraca		6						1					1	2																				
Copepoda		2	12					3	26	1	1	15	5	13	1	10	7	53	13	9	2			9	5	7	4	4	11	1				16
Corbiculiidae	3			3		1							1		4									5										
Cordulephyidae																																		
Corixidae	2	2	10			1		1	10		4	2	2	1		51	8		17		48	225		2	7	2		26	43	1	16	1	4	255
Corduliidae/ Libellulidae						1																												
Corydalidae																																		
Culicidae			1						7							7											2							2
Curculioniidae																											1							
Cyclopoida																																		
Diphlebiidae																																		
Dolichopodidae																																		
Dytiscidae		6			1	35		17	53	17	4	8		1		9	2	6	11	2		10		1	1	17	10	2	3		6	3		9
Ecnomidae	3		3	7		1		1		1	3			1	2		7	3	4				12	2					4	3	14	1	5	
Elmidae		1	2	11	2	1	36	5		6		2	6	4	1		1	1	1		1		7	5	3	8	2	2	1	8	14	22		
Empididae																																		
Gelastocoridae																																		
Gerridae	2	3				1				2		1				2		5	4								1		3		1			
Glossomatidae																																		
Gomphidae	1	3		6	9	17	23	8	8	8	1		4	2										1	1		1		1	2	1	2		
Gordiidae																																		
Gripoterigidae																																		
Gyrinidae									1													1		1		1			1					
Hapliplidae																																		
Hebridae															1	19				1							2		19		1			4
Heliopsychidae																																		
Hemicorduliidae, Urothemistidae, Libellulidae	1	17	7	11		22	31	18	43	12	5	7	5	10	2	45	2	4	5	2	5		10	4	11	24	7	35	34	35	5	13		14
Hirudinea													1																					
Hydra		12	1	1				2				4	6	9		5		8							1	2	2							
Hydracarina	1	4	11	2	2	9	5	5	11	3	11	19		13	4	20	7	11	11	1					6	34	13	5	40	4	18	25	4	119
Hydraenidae		4				16	1	4		3	1			6		1			2	2				1		2			1			1		
Hydrobiidae																																		
Hydrochidae																																		

Family/Taxa	WLSH35a	WLSH35b	WLSH35b	WLSH35	WLSH34	WLSH34	WLSH34	WLSH32-1	WLSH32-2	WLSH32-3	WLSH31	WLSH31	WLSH28	WLSH28	WLSH27	WLSH27	WLSH26	WLSH26	WLSH26	WLSH16	WLSH15	WLSH13	WLSH12	WLSH12	WLSH11	WLSH11	WLSH11	WLSH10	WLSH10	WLSH10	WLSH09	WLSH08	WLSH29	WLSH29
Habitat	**bottom**	**edge**	**bottom**	**riffle**	**bottom**	**edge**	**riffle**	**edge**	**bottom**	**edge**	**bottom**	**edge**	**bottom**	**macrophyte**	**bottom**	**macrophyte**	**bottom**	**macrophyte**	**edge**	**edge**	**edge**	**edge**	**riffles**	**edge**	**bottom**	**edge**	**macrophyte**	**bottom**	**macrophyte**	**riffle**	**edge**	**edge**	**bottom**	**macrophyte**
Hydrometridae								1						1								1												
Hydrophilidae	1	20				18	4	12	21		1	13		10	1	2		5	7	3		4	1	4		4	1	2	24	2	6	7		2
Hydropsychidae	6			97			8								1						1	1	23	4						14				
Hydroptilidae								3															1											
Hygrobiidae																																		
Hyriidae																																		
indet.		6	1	4		1					1		2				1									7	1	1	15					
Isopoda																											1							
Isostictidae		1						1			1		1						3					3		2			1		6	1		
Leptoceridae	1	23		1		3	1	56	2	4	1	3		5			2	6	7		6	1		21		25	11	1	24	7	18	20		
Leptophlebiidae	7	16	5	20	11	18	25	32	15	21	6	2	7	20	1		9	1	7		1		44	4	16	5	2	37	8	55	18	4		
Lycidae																																		
Lymnaeidae											1		1	1		13						2	1											1
Macromiidae																															2			
Mesoveliidae														3		2													6					
Nannochoristidae																																		
Naucoridae																				1		3												
Nematoda																																		
Nematomorpha																																		
Nemertea																																		
Nepidae						1		1						4		1	1	2				2							3					
Nuerorthidae																																		
Noteridae																																		
Notonectidae		54				6		7	1		20	8		1				16	23			8				9	1		8		2			
Ochteridae						2	1																											
Odontoceridae																																		
Oligochaeta	2	19	8	19	1	7	14	4	11	8	1	12	16	8		28	1	26	13	3	3	12	13	14	3	2			13	8	2	1	3	5
Orabatidae																																		
Ostracoda	2	8	9	10	2	2	5	7	1	8	1	3	19	12	5	10		13	11	4		1	14	10	2	6	3	2	3	5	18	8		12
Palaemonidae					7	2	6			2										2			3		1		2	2		5	1	1		
Parastacidae								2	1	3			5	8		1																		
Philopotamidae	2			18			20																68	1						50				
Physidae																2																		
Planorbidae					1			1				1	5	18	1	8			7			48	1	4					3		1		1	
Pleidae						2		9	2	1		2	1	19	2	37		8	16	1		1				11	14		121		3	6		27
Polycentropodidae		1				3															1		7								1			
Prosopistomalidae							9																											
Protoneuridae		19				19		24	4	9	2	9	11	36	1	18	3	21	16		8			3		35	27	1	44		2	7		21
Psphenidae		1																					2											
Pyralidae				7			4									1		1					5						1	2	1			
Scirtidae														1				3						6			8		1		3	1		
Sialidae																																		
Simulidae	2			45			14																54							166				
Sisyridae																																		
Sphaeriidae				5																														
Synlestes (Chlorolestidae)																																		
Staphylinidae						2						1		3				1					5			1								
Stratiomyidae														2										3										
Tabanidae		1		9	5		4					1											1						1	4				
Tasimiidae																																		
Temnocephalidea																2		2																
Thiaridae															3	2								44										
Tipulidae	1			1				1						2		1																		
Turbellaria			3	1			7					1	17	22	2	21	1	23	27	94		1	6	1		5	13		5	27				4
Veliidae		3				5		9		1		2		2		1		30	17			8		7		4	9	1	12		12	6		
Zygoptera (undifferentiated)		3				4	1	2				3	1		1	2		1	2		1					6	4		17		1	1		11