Chapter 3 Water quality

3.1 Introduction and aims

The ecological and physical condition of aquatic and riparian habitats in sugarcane-producing catchments is one of the main determinants of water quality in streams and floodplain water bodies (Rayment, 2002; Pearson *et al.*, 2003). Chapter 2 presented data on habitat condition of a range of floodplain water bodies, all of which were found to be moderately to highly degraded. The present poor water quality found throughout the Burdekin Floodplain is directly linked to habitat condition (Lukacs, 1995). Furthermore, the modified hydrology of the main study sites and most of the remnant sites also has a major influence on water quality. One of the most important components of the physico-chemical make – up of the water, with respect to the fish communities, is dissolved oxygen (DO) content (Wetzel, 1983; Hogan and Graham, 1994; Matthews, 1998; ANZECC/ARMCANZ , 2000; Pearson *et al.*, 2003). The spatio-temporal distribution of DO in water greatly affects the solubility of inorganic nutrients also (Wetzel, 1983), which aids the growth of exotic weeds such as *Eichhornia crassipes* (Gutierrez *et al.*, 2001). Increases in the cover of *E. crassipes* cause oxygen content to decrease (Willoughby *et al.,* 1993; Scheffer *et al.*, 2003). This chapter will focus on the dynamics of oxygen, *E. crassipes* infestation and nutrient (input) loadings from the modified flow conditions of Burdekin floodplain lagoons.

3.1.1 Oxygen in water

Oxygen is introduced into freshwater through means of a mass balance. Inputs are: inflow + reaeration + photosynthesis (Pearson *et al.*, 2003). Outputs are: loss to atmosphere + respiration + chemical oxygen demand (Pearson *et al.*, 2003). The primary drivers in this mass balance shift depend on the saturation levels. In undersaturated situations, losses to the atmosphere are zero and the main consumption of oxygen is biological and chemical (typical in shallow tropical lagoons). Therefore respiration dominates loses and systems tend to be hypoxic unless the photosynthesis in the water column is significant, which requires 1) light and 2) biomass of submergent autotrophs (Pearson *et al.*, 2003). In natural wetlands of the Australian tropics, inflows (flow rate x concentration of DO in inflowing water) are negligible for most of the year and originate principally from groundwater naturally low in DO (Perna and West, 1998; Pearson *et al.*, 2003). In the current altered hydrological condition in the Burdekin Delta, inflows of Burdekin Falls Dam (BFD) water provide a great increase in oxygen inflow, but at the same time have promoted turbidity, floating/emergent weed infestations (*E. crassipes* and *Brachiaria mutica*) and biomass accumulation, which have increased respiratory consumption and decreased water column photosynthesis (Pearson *et al.*, 2003; McCormick and Laing, 2003).

3.1.2 Oxygen dynamics in tropical freshwater

The consumption of oxygen in a water body is largely determined by the biomass and activity of aerobic respirators (Kaenel *et al.,* 2000). This consumption is often referred to as biological oxygen demand (Wetzel, 1983; Pearson *et al.*, 2003). Oxygen may also be consumed chemically, adding to the overall consumption (for example when ferrous iron is oxidized, however there is no evidence of high concentrations of such chemicals in the waterbodies under investigation here) (B. Butler, pers. com.).

Diel variations in DO follow a pattern in which levels reach a maximum during the late afternoon (the timing of the maximum depends on flow – under high flow conditions the maximum will occur in the middle of the day, under low flow later in the afternoon), followed by a steady decline (in some cases to below lethal thresholds) through the night and early morning (Fig. 3.1). Although there is a recognized stratification effect with depth and temperature (Mackey, 1991), this chapter will deal mainly with the epilimnion, as this is where most local species live. Nocturnal DO sags (the rapid depletion of dissolved oxygen) occur naturally in even the most undisturbed natural wetlands, and this natural “poor water quality” that occurs at certain times and places within complex wetland mosaics probably plays a vital role in the maintenance of biodiversity (Pearson *et al.*, 2003). Once cycling approaches equilibrium the daily rise is about twice as fast as the overnight decline (Pearson *et al.*, 2003).

Figure 3.1 Oxygen cycling (% saturation) in Keelbottom Ck lentic lagoon site showing natural lentic oxygen cycling. (Pearson *et al.*, 2003)

The Australia and New Zealand Environment and Conservation Council (ANZECC/ARMCANZ, 2000) guidelines for tropical lowland fresh waters state that oxygen saturation in healthy sites should range between 85-110% (average basis) at daily maximum (ANZECC/ARMCANZ, 2000). However, the guidelines do not specify a minimum level and it is this level, in systems that tend to have oxygen sags, that may determine the biological limits of the fish, rather than average based guidelines (Pearson *et al.*, 2003). Also turbidity or cloudy conditions can cause associated sags over short periods due to reduction in photosynthetic production, which can't be seen in averages. Throughout most of tropical Queensland, and especially in agriculturally developed regions, this standard is rarely met (Hogan and Graham, 1994; Kennard, 1995; Butler, 2003; Pearson *et al.*, 2003) (Table 3.1). Most water quality data are spot measurements, taken between 0900 and 1630, at which time the oxygen may be significantly above its minimum and could even approach its maximum level. However, the potentially stressful oxygen levels are likely to occur in the early morning when oxygen levels are typically at their lowest (Pearson *et al.*, 2003). Alternatively, in heterotrophic systems, the maximum can occur at the warmest times of day. In open water, autotrophic water bodies minimum values occur just before the sun hits the water surface, between 0630 and 0930 (as shown below in results) (Fig. 3.1). In closed surface situations (such as in the Burdekin floodplain lagoons, and in some closed canopy riparian systems) the minimum depends more on the flow, because the minimum generated by the open waters upstream, can arrive at any time of the day depending on flow (ACTFR, unpublished data). Without recording the diel cycling it is difficult to interpret results meaningfully (Norris and Norris, 1995). It is also vital to capture the temporal variation throughout the year and spatial variation within the site, to be able to interpret the data confidently in relation to the biotic community because conditions fluctuate enormously over even small spatial and temporal scales.

3.1.3 Impacts of habitat condition on oxygen content

Increasingly, the condition of aquatic habitats is being linked to agricultural activities within the catchment (ACTFR, 1994; Hunsaker and Levine, 1995; Lukacs, 1995; Arthington *et al.*, 1997; Nilsson and Svedmark, 2002; Rayment, 2002). Land clearing, soil enrichment, riparian clearing and altered hydrology on tropical floodplains have greatly impacted on instream conditions (Chapter 2), although in the Burdekin the focus of the agricultural and urban development is focused almost entirely on the coastal floodplain. In most large tropical floodplains, the main river channel is spared direct input, as runoff is directed away from the main channel due to the natural morphology of levees, distribution channels and sub-catchments (Hopley, 1970; Pearson *et al.*, 2003). The smaller distribution channels and sub-catchments, with their associated lagoons and wetlands are the main receiving waters for agricultural runoff;

this situation puts these habitats at the highest risk of impact in the floodplain (Gore and Shields, 1995). The remaining wetlands are usually kept or modified to serve as drains, retention basins or irrigation distribution points for agricultural purposes (Gore and Shields, 1995). By retaining preferred habitats that serve similar functions the natural diversity of floodplain habitats is homogenised (Tait, 1994; Tait and Perna, 2001; Pearson *et al.*, 2003).

The loss of riparian vegetation has been found to greatly affect oxygen and thus fish communities in tropical wetlands (Arthington *et al.,* 1997; Bunn *et al.,* 1997; Houston and Duivenvoorden, 2002; Pusey and Arthington, 2003). Without riparian vegetation to reduce solar radiation and to filter inputs, tropical wetlands quickly become eutrophic (Bunn, *et al.* 1998). The typical response by oxygen to this increase in temperature and light availability is a rapid increase in cycling of oxygen that may exceed oxygen inputs (Kaenel *et al.,* 2000; Rayment, 2002; Pearson *et al.*, 2003). The loss of riparian vegetation is usually associated with agricultural activity, which may also increase nutrient inputs to the water by increasing the mobilization of sediments (Rayment, 2002). This input is exacerbated because the riparian zone, which naturally acts as a filter during bank overflow is removed in these developed catchments. Moreover, most agriculture in Australia requires the addition of nutrients to the soil (Rayment, 2002; Pearson *et al.*, 2003).

Table 3.1. Regional data on percent saturation of oxygen in floodplain habitats.

Site	Dissolved Oxygen % Saturation or noted otherwise	Landuse
Normanby River (Kennard, 1995)(NB. Mean from all readings taken in the late dry season)	Lagoon sites 1=34.7% 2=60.3% 3=61.6% 4=90.1% 5=78.6% 6=55.6% River sites 1=81.9% 2=78.1%	National Park and cattle grazing
Lagoon Ck Herbert River Floodplain (Range is presented here) (Pearson, *et al.* 2003)	Event peak=8% Falling hydrograph=6% Stable base flow=10% No flow=19%	Sugar Cane
Johnstone River (Russell and Hales, 1993)	Coastal lowlands Average=68.5%	Sugar and Banana
Tully-Murray Floodplain (Hogan and Graham, 1994)	Tully River=7.92 mg/L Murray River=7.28 mg/L Barrett's Lagoon=5.30 mg/L Selby's Lagoon=2.04 mg/L Racanello's Lagoon=0.20 mg/L	Sugar and Banana
Russell and Mulgrave Rivers (Russell *et al.*, 1996)	Floodplain 80%	Sugar

3.1.4 Nutrients, artificial flows and *Eichhornia crassipes*

The wetlands under investigation in this study are unusual in that agricultural runoff represents only a small proportion of the water they convey, most of the time they carry a relatively constant stream of irrigation water. Conditions in the distribution channels of the Burdekin River delta floodplain are dominated by artificial flows, which are normally turbid, and impacted by weed infestations (especially *E. crassipes*). Altered flows have been identified as the most serious threat to ecological sustainability of rivers and their associated floodplains (Bunn and Arthington, 2002). The altered flows have aided growth and high productivity of *E. crassipes*. The irrigation water maintains constant water levels in the lagoons, which reduces disturbance to *E. crassipes* mats. Irrigation water also supplies a continuous supply of nutrients for growth of the plant (see results below). The main issue with this plant is that it can greatly reduce oxygen content in the water under the mats (Willoughby *et al.*, 1993; Cordo and Center, 2000; Masifwa *et al.*, 2001), and, in non – nutrient limited water, will create a stable state (Scheffer *et al.*, 2003).

Eichhornia crassipes creates anoxic condition under the mats, which favours the release of nitrogen and phosphorus (N and P) from sediments, which in turn favours *E. crassipes* (Masifwa *et al.,* 2001; Julien *et al.*, 2001; Scheffer *et al.,* 2003). Under these conditions, plants contribute both organic material and a physical substrate for microbes, the metabolic activity of which further increases oxygen demand (Kaenel *et al.,* 2000; Pearson *et al.*, 2003; Scheffer *et al.,* 2003). The irrigation water is often turbid enough to cause light limitation in the water column, but contains enough nutrient to support plant growth that favour floating/emergent species (Pearson *et al.*, 2003; Scheffer *et al.,* 2003).

This light limitation in turn reduces oxygen production by submergent macrophytes and phytoplankton (Wood and Armitage, 1997). For example, investigations undertaken in the Herbert River catchment revealed that during small flow events in which turbidity was increased, a substantial decrease in oxygen saturation occurred, due in part to decreases in photosynthetic production (Pearson *et al.*, 2003). In the Burdekin River floodplain, the turbidity of distributary streams is maintained at relatively high levels in the upper reaches. Turbidity may decrease with distance from pumps due to the filtering effects of the weed mats (personal observation; Bunn *et al.*, 1998). Turbidity plus floating weed mats lead to very strong light limitation in the underlying water column, whereby production is restricted and consumption is increased, even though turbidity is decreasing. Isolation of the air/water interface due to prolific growth of *E. crassipes* limits reaeration also (Cordo and Center, 2000; Pearson *et al.*, 2003).

3.1.5 The aims of this component of the study were to:

1) compare oxygen cycling in the ten main study lagoons with special reference to the differences between weeded and non-weeded lagoons and those with natural or modified flows;
2) compare oxygen cycling at two lagoons classified as remnant (lacking supplemental flows) sites with that occurring at the main sites;
3) describe downstream impacts on water quality in lagoons receiving input from modified or weed-infested lagoons upstream; and
4) determine if modified flows have reduced instream water quality and aided *E. crassipes* infestations.

3. 2 Methods

3.2.1 Physico-chemical analysis

On each sampling occasion a Hydrolab® datasonde multimeter was placed in the site to record oxygen, pH, turbidity, salinity and electrical conductivity. During the first two sampling trips, data were recorded manually and collected from the inlet, middle (if not overgrown by weeds) and outlet of each lagoon. Sampling was vertically stratified and data recorded at 50 cm intervals from the surface to the bottom. However, it was subsequently determined that this protocol yielded insufficient understanding of the dynamics of diel oxygen cycling at each site. Thus, Hydrolabs® were subsequently placed at a single location in each lagoon and at a single depth in each site and left in place for 24 hours. The Hydrolabs were set in either one of two ways. The Hydrolab® was attached to a previously established anchor (a star picket) at a depth of 60 cm from the surface or the Hydrolab® was connected to a float and anchor set in the middle of the lagoon at 60 cm depth. The Hydrolabs were always set away from overhanging vegetation and macrophyte beds to minimize microhabitat effects.

3.2.2 Detailed limnological investigations

During the rainy weather, pumping is suspended and for a brief time (few months at most) the distribution channels receive natural runoff from rainfall events. Detailed water quality information was collected during this period of ‘natural’ flow, and after supplemental flows resumed, to determine the variability in water quality due to pumping. Water quality variables examined included chlorophyll *a*, phaeophytin, turbidity, total suspended solids, total nitrogen, ammonia, nitrite, nitrate, total phosphorus and filterable reactive phosphorus (FRP).

Detailed chemical data were collected at the top, middle and bottom at each site under three flow conditions: pumping, falling hydrograph and no flow. For the collection of dissolved inorganic and organic nutrients a 0.45 micron filter was used in conjunction with a 50 mL syringe to filter water before collecting the sample. All samples were cooled and returned to laboratory for analysis.

3.2.3 Data analysis

Due to the highly variable nature of tropical floodplain wetlands, analysis of the detailed chemical data in this chapter is qualitative and based on graphical interpretation of data rather than a quantitative analysis. The decision to adopt this approach is based largely on the findings of a Sugar Research Development Corporation (SRDC) funded project (Pearson *et al.*, 2003) that found that the complexity of the dynamics of water quality coupled with a high degree of internal variability at each site makes accurate data collection and interpretation difficult unless the sampling regime includes substantial spatial and temporal replication. Also water quality data are generally skewed and may be multi-modal (B. Butler, pers. comm.) To highlight this a plot from a storm event in Lagoon Ck is presented (Fig. 3.2) (Pearson *et al.,* 2003). Oxygen cycling data collected by data logging, however, can to some degree be interpreted graphically. The difficulty in interpreting the oxygen cycles is that oxygen cycling in lagoons in the Burdekin are complicated by the addition of supplemental flows.

To examine variations in nitrogen and phosphorus (N and P), calculations based on concentrations at the end of the dry season (Nov. 2002) were made using volume estimates from size and depth of the lagoons to calculate nutrient input loadings. The two remnant sites, Inkerman (Warren's Gully) and Kelly's (Sheep Station) were used as lentic examples and the most upstream sites Fowler's (Warren's Gully) and Payard's (Sheep Station) were used as the supplemented flow sites (now more lotic in nature). Only the upstream sites were used in order to reduce complications added by water extraction and natural dynamics of nutrient cycles within the streams. The calculations for the remnant sites were based on site volume, whereas the pump rates of the previous five months (dry season period) were used in the supplemented sites. The average concentration (from three samples) of N and P from the November sample was used for all calculations. Three calculations were made to examine the impact of water supplementation:

1) For remnants, Length (L) × Width (W) × Average Depth (D) (or pump rate for supplemented sites) (S) = Volume (ML) × 1000 to express nutrients in kilograms.

2) Volumes were then calculated for Payard's and Fowler's L × W × D = V, but using the concentrations from the remnant sites to give a notional estimate of what the input loadings might have been without supplemental flows.

3) Input Load / Volume (no flow) = Concentration (kg) × 1000 = (μg N and P/L), which gives a comparison of the concentrations found at the time of sampling with concentrations that would be needed to equal the amount estimated by supplemented flow at Payard's and Fowler's.

Figure 3.2 Lagoon Creek storm event, February 2002. Total nitrogen , nitrate, suspended particular matter (x40) and turbidity (x40) during the first 8 days of the flow event (Pearson, *et al.,* 2003).

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3.3 Results and discussion

3.3.1 Oxygen dynamics at Warren's Gully sites.

Fowler's Lagoon is located at the upstream end of Warren's Gully and is the first lagoon on the distribution channel. Figure 3.3 shows oxygen saturation levels during a period of high supplementation in March of 2001, occurring immediately after the recommencement of supplementation, and after a period of below-average wet season flows. Figure 3.3 shows natural cycling from a maximum in the late afternoon to a low in the early morning, with a rise after first light (rise from 41% to 64% between 0700 h and 0830 h, and peaking at 92% at 1630 h. A steady decrease to 67% at 2330 h was recorded at which time there is a minor plateau probably caused by higher DO water from closer to the surface cooling and mixing with the deeper water (B. Butler, pers. comm.). Saturation levels fell slightly below the sub-lethal threshold (Pearson *et al.,* 2003) during the early morning, but generally saturation was adequate for most fish species. The water being pumped in, and the clear channels upstream, provide good conditions for oxygen production in the water (i.e. sunlight and temperature). The variability seen in the daylight hours may reflect variability in cloud cover or wind.

Princess Lagoon, the next site downstream of Fowler's, showed a sharp decrease in total oxygen saturation from that recorded at Fowler's Lagoon (Figs. 3.3 and 3.4). Oxygen content was at a minimum (27%) at 0830 h slightly rising to 40% at 1200 h, never exceeding the sub-lethal threshold (Pearson *et al.,* 2003). The great majority of channels and habitats below Fowler's Lagoon were overgrown with *E. crassipes* and emergent grasses. The plot results from the superimposition of two different cycling patterns. A likely temporally displaced version of the upstream cycling (as evidenced by the arrival of a peak during the night at 2130 h) adds to the effects that originate within the upstream channels and sampling site (mainly increased oxygen consumption), reflecting the cumulative impacts of upstream habitat degradation. There was also a notable peak in dissolved oxygen saturation levels at 2200 h (51 %), due to the arrival of water containing the (1630 h) maximum DO concentrations from upstream rather than *in situ* production. In other words, the figure strongly suggests that it took about 5.5 hours for water from upstream to arrive at this site.

Munro's is the next site downstream of Princess Lagoon. Dissolved oxygen levels varied little over time (25-40%) and there was little indication of any diel cycling (Fig. 3.5), indicating consumption is balanced by re-aeration and inflow, therefore levels are largely dependent on the DO concentrations entering the lagoon. The figure shows that the plug of water from upstream had become so mixed that it no longer appeared as a peak. However, mass balance considerations suggest that it will probably have slightly elevated the mean DO within this lagoon by the presence of flow. Saturation levels were lowest at 0900 h (27%) and increased slightly over 11 hours, peaking at 40% at 2030 h; again these levels remain below the sub-lethal threshold (Pearson *et al.,* 2003). Dissolved oxygen levels were further reduced from those recorded in Princess Lagoon, indicating how the weeded reaches between these lagoons actively consumed oxygen and prevented reaeration.

The three sites are presented together to illustrate the gradual stripping of oxygen as water flows through weed-infested reaches (Fig. 3.6). Fowler's cycled between 41% and 92% saturation whereas Princess cycled between 25% and 50% and Munro's cycled between 25% and 40%, with much less pronounced cycling at the two downstream sites. Also the peaks at these two downstream sites were well after sunset, suggesting pulses of oxygenated water from upstream. Note that, without the flow, hypoxia would have been much worse.

Saltwater Lagoon is the most downstream site and receives only tailwater from Warren's Gully. The site had good riparian condition and had been scoured by flooding at the beginning of the study. Diel oxygen cycling is presented for pumped tail water (Fig. 3.7). Cycling was not very pronounced; however, saturation levels were very high, peaking at 1930 h at 93% falling to71% at 0830 h. This site had saturation levels well above the lethal thresholds (Pearson *et al.,* 2003) and exceeded the ANZECC/ARMCANZ minimum trigger values. There was high variability over short time periods through the day which may be attributed to cloud cover or wind.

Inkerman Lagoon is the remnant site for Warren's Gully. This site receives only local flow (no irrigation water), although at the top end there is tailwater from cane fields. Saturation levels are presented for three flow regimes: 1) no flow at June 2001; 2) first flush of the wet season October 2001; and 3) falling hydrograph after wet season flows, December 2000 (Fig. 3.8). This lagoon had high-value remnant habitats with good riparian condition and clear water (see Chapter 2). The DO saturation levels for the no-flow condition on 20-21-June-2001 was high (ranging from minimum of 48% at 0930 to a maximum of 108% at 2200. The low readings during the falling hydrograph and base flow are probably partly related to landuse, which is cattle and high organic input from tailwater, which would increase oxygen consumption. During these flow periods saturation levels cycled between 19% and 55%. Although these levels were low, cycling was observed, so there was sufficient photosynthetic production of DO to prevent hypoxia from becoming critical, but insufficient to achieve full oxygenation. Saturation levels for no flow exceed the lethal thresholds, whereas during base flow and first flush, levels were below the sub-lethal threshold; and during the morning of the first flush, cycle levels dropped below the lethal threshold.

3.3.2 Sheep Station Creek sites

Sheep Station Ck has a larger catchment than Warren's Gully and so receives more local input during times of surface flow. Payard's Lagoon is the first major deep-water lagoon on this distributary stream. Payard's Lagoon was used as a case study that examined the impact of removal of floating and emergent weed, and all the data from this site are presented in Chapter 5.

Figure 3.3 Fowler's Lagoon diel oxygen cycle, supplemented flow 7-8/ March/2000. The dotted line is the ANZECC minimum trigger value, solid bold line is sub-lethal threshold, and grey line is lethal threshold (Pearson *et al*., 2003)

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Figure 3.4 Princess Lagoon diel oxygen cycle, supplemented flow 7-8/March/2000. The dotted line is the ANZECC minimum trigger value, solid bold line is sub-lethal threshold, and grey line is lethal threshold (Pearson *et al*., 2003)

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Time (h)

Figure 3.5 Munro's Lagoon diel oxygen cycle, supplemented flow 6-7/March/2000. The dotted line is the ANZECC minimum trigger value, solid bold line is sub-lethal threshold, and grey line is lethal threshold (Pearson *et al*., 2003)

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Figure 3.6 24 hour oxygen cycles at Fowler's, Princess and Munro's at peak pumping, showing steady decrease in oxygen content from upstream to downstream. The dotted line is the ANZECC minimum trigger value, solid bold line is sub-lethal threshold, and grey line is lethal threshold (Pearson *et al*., 2003)

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Figure 3.7 Saltwater Lagoon diel oxygen cycle, 5-6-March-2000. The dotted line is the ANZECC minimum trigger value, solid bold line is sub-lethal threshold, and grey line is lethal threshold (Pearson *et al*., 2003)

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Figure 3.8 Inkerman Lagoon diel oxygen cycles for three flow conditions. The dotted line is the ANZECC minimum trigger value, solid bold line is sub-lethal threshold, and grey line is lethal threshold (Pearson *et al*., 2003)

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Time (h)

The next site downstream is Dick's Bank Lagoon which is a large deep lagoon with moderate to low *E. crassipes* infestation (Chapter 2). The figure describing oxygen cycling at Dick's Bank reveals higher DO level (Fig. 3.9). This site had DO peaks of 80% at 2330 h indicating flow of oxygenated water from upstream. The saturation then decreased to a low of 44% at 0700 h with a steady increase after sunrise, indicating biologically induced cycling. Saturation levels only dipped below the sub-lethal threshold for three hours in the early morning. The mass balance of oxygen in this site is driven by flow and photosynthesis.

The next lagoon downstream is Gorizia's lagoon. Water quality with respect to DO content at this site was the poorest in Sheep Station Ck (Fig. 3.10). Saturation never exceeded 20%, which is below the acute threshold for many native fish (Pearson *et al*., 2003), until weeds were removed in August 2002. After weeds were removed, oxygen started to show some cycling and saturation levels were from 41% to 49%, a significant increase from weeded conditions (photosynthetic production of oxygen was introduced after weed removal, adding a biological component to the mass balance) (Fig. 3.10). After weed removal saturation never fell below the acute threshold during the 24 hrs of measurement. This site is downstream of lagoon and channel habitats that were highly degraded (Plate 3.1), and floating and emergent weed cover at the site was 100 percent. This demonstrates how oxygen is stripped out of the water by the weed mats through lack of submerged macrophytes, reduced re-aeration and mixing, and increased microbial consumption from continuous organic supply. The oxygen concentrations at this site before weed removal was largely dictated by inflow of DO water from upstream, and without this inflow concentrations may have been even lower.

Jack's Lagoon is the last site on Sheep Station Ck, about 500 m downstream of Gorizia's. No cycling data is available as logging was started after this site fully closed up due to *E. crassipes* growth, creating access problems. The oxygen content rapidly decreased from surface (30 cm) afternoon maximum of 29% to 1 m minimum of 0.3% (Fig. 3.11). Also there was a decrease in surface saturation between the afternoon and night readings: the afternoon maximum was 29% and the evening was 20%, only just exceeding the acute threshold. This site data highlights how the saturation levels of the logged data from the surface water represents more biologically significant results important to the fish that may use the thin surface layer as a refuge from the poor water quality of the hypolimnion.

Kelly's Lagoon was the remnant site for Sheep Station Ck, with black water and little landuse impacts (Chapter 2, section 2.4.1). This site received no direct input of irrigation water but enough surface and groundwater pressure to maintain water levels, but no turbid water entered the lagoon. The diel cycling at Kelly's is presented for no-flow and falling hydrograph

(Fig. 3.12). Oxygen cycling under the no-flow regime was strong, peaking at 148% at 1630 h, reaching a minimum of 68% at 0700 h, rising rapidly after sunrise, dominated by photosynthetic oxygen production. This cycle, however, shows very large variations that may preclude a situation where consumption reduces oxygen over night, to levels below the acute threshold. The falling hydrograph shows the same trend as other sites during falling hydrograph, where cycling is suppressed, and levels are well below lethal thresholds (<20% saturation), with little variation until a short peak of 42% at 1430 h. High organic input from local runoff and cloudy conditions may have further reduced saturation levels during the falling hydrograph.

Figure 3.9 Dick's Bank Lagoon diel oxygen cycling, supplemented flow 4-5/Apr/2002. The dotted line is the ANZECC minimum trigger value, solid bold line is sub-lethal threshold, and grey line is lethal threshold (Pearson *et al*., 2003)

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Figure 3.10 Gorizia's Lagoon diel oxygen cycling at 3 flow regimes. The dotted line is the ANZECC minimum trigger value, solid bold line is sub-lethal threshold, and grey line is lethal threshold (Pearson *et al*., 2003)

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Time (h)

Figure 3.11 Jack's Lagoon spot measurements to 2m depth 21/August/2000, supplemented flow. The dotted line is the ANZECC minimum trigger value, solid bold line is sub-lethal threshold, and grey line is lethal threshold (Pearson *et al*., 2003)

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Figure 3.12 Kelly's Lagoon diel oxygen cycles at no flow and first flush. The dotted line is the ANZECC minimum trigger value, solid bold line is sub-lethal threshold, and grey line is lethal threshold (Pearson *et al*., 2003)

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Time (h)

3.3.3 Remnant sites oxygen cycles

Castelanelli's Lagoon is a washout lagoon that connects to Sheep Station Ck at times of natural flow (flows directly into Dick's Bank Lagoon, Figure 2.5). This site was sampled as a remnant once on 19-21/August/2002. The diel cycling of DO is presented in Figure 3.13. The data was collected at no flow. This site water levels are maintained by similar means as Kelly's above, and at the time of sampling it was noted that a dirty water line was present at the outlet but there was enough groundwater input to keep the irrigation water out of the site, therefore the site has black water. Slight cycling occurred with DO peaking around 102% at 1700 h each day. After sunset dissolved oxygen levels gradually decreased to an early morning minimum of 82% on both days. This site shows saturation in the ANZECC/ARMCANZ classification for healthy lowland floodplain habitats, with levels only just dropping below 85%. Habitat at the site was of good quality. The diel cycling shows the best representation of natural condition (although tropical wetlands can display a wide variety of "natural" diel oxygen patterns including total oxygen depletion) in all the sites.

The only site where water quality data was logged outside of the two water board areas is Clay Hole (Fig. 3.14). This site is on Pelican Ck in the Barratta Ck system. The site had very high habitat values with 100% continuous riparian cover and, because of the narrow nature of the site; the water was shaded for more than half the day. There was low submerged macrophyte biomass. The main impact here was from upstream condition and landuse. This is seen in the low variability, and almost reverse cycling (maximum of 67% at 1800 h and minimum of 56% at 1600 h) of the oxygen cycle, suggesting high re-aeration and respiration dominate the mass balance. Respiration increases enough during the heat of the day to cause a minimum in the afternoon. Upstream of this site were large paddocks of cane irrigated by the Burdekin River Irrigation Area scheme and there was high tailwater input to the site.

3.3.4 Nutrient, Chlorophyll and Total Suspended Solids

A detailed investigation was carried out at each site under different flow conditions. Results of nutrient, chlorophyll, suspended solids and ammonia analyses are presented for natural base flow (no pumping) and artificial pump flow (Table 3.2). Although many variables were recorded, the focus of this section is on nutrients supplied by modified flows, as they may aid in *E. crassipes* infestations, which leads to low oxygen concentrations, as shown above (Fig. 3.9). This section presents results on total nitrogen and phosphorus.

Figure 3.13 Castelanelli's Lagoon diel cycling over two days, 19-21/August/2002 at no flow. The dotted line is the ANZECC minimum trigger value, solid bold line is sub-lethal threshold, and grey line is lethal threshold (Pearson *et al*., 2003)

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Figure 3.14 Clay Hole diel cycling, 18-19/June/2001, tail water flow. The dotted line is the ANZECC minimum trigger value, solid bold line is sub-lethal threshold, and grey line is lethal (Pearson *et al*., 2003)

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Time (h)

Table 3.2. Average (n=9) of Total N and P, nitrate, Filterable Reactive Phosphorus (FRP), ammonia, Total Suspended Solids and chlorophyll ą for ten sites. Sites are: 1) Fowler's, 2) Munro's, 3) Inkerman, 4) Princess, 5) Saltwater, 6) Payard's, 7) Dick's Bank, 8) Kelly's, 9) Gorizia's and 10) Jack's

Stream	Site	Flow Condition	TSS (mg/L)	Chl a (µg /L)	Total N (µg N/L)	Total P (µg P/L)	FRP (µg P/L)	Ammonia (µg N/L)	Nitrate (µg N/L)
WARREN'S GULLY	1	Falling leg	25.8	41.9	629	75	8.5	2.1	3.9
	2	Falling leg	8.6	9.9	648	534	269	82	7.1
	3	Falling leg	22.3	16.1	628	166	74	82	7.6
	4	Falling leg	11.0	21.4	647	570	272	9.2	9.6
	5	Falling leg	11.6	0.8	490	251	196	14.8	60
	1	High pump	25.3	0.9	305	28.9	11.2	26.4	16.4
	2	High pump	3.3	9.7	312	57	19.0	9.2	5.2
	3	High pump	3.3	5.0	384	27.2	4.1	18.7	7.2
	4	High pump	0.7	0.5	267	30.6	13.1	38.0	6.1
	5	No flow	2.5	7.5	530	36.4	4.9	15.3	5.7
SHEEP STATION	6	Falling leg	18	17	629	228	139	12	6
	7	Falling leg	13.8	37.1	738	366	189	11.7	6.7
	8	Falling leg	16.3	6.5	566	327	212	119	5.0
	9	Falling leg	14.7	3.0	655	364	212	82	7.7
	6	High pump	15	1.3	333	37	18	9	5.4
	7	High Pump	6.0	18.6	862	60	10.9	19.9	5.7
	9	High pump	3.4	5.2	300	23.9	9.3	2.1	8.8
	10	High pump	2.2	2.7	360	31.7	12.0	20.4	7.0

Over the period before the samples were taken water clarity was higher than average in the irrigation distribution streams. This is likely the result of higher volumes of base flow coming from the Bowen River catchment on the coast than the Burdekin Falls Dam. This can be seen in moderate TSS readings during the pumping (Table 3.2, Fig. 3.15a). During the falling hydrograph there were moderate to high levels of chlorophyll in the upper reaches with open water (sites 1, 6 and 7 in Table 3.2). The high levels of chlorophyll were directly linked to high levels of total P in the storm tail water (Table 3.2, Fig. 3.15b). P values were lower at Fowler's but chlorophyll was high, suggesting that the site had high submerged plant productivity and therefore high assimilation capacity. These conditions promoted autotrophic production where oxygen content was driven largely by biological inputs. When the water reached the weeded sections, the conditions changed to heterotrophic, where the phytoplankton and submerged macrophytes were shaded and oxygen production in the water column was restricted. The weeds also restricted re-aeration, further reducing oxygen concentrations. The conditions created by the hypoxia under the weeds are reflected in the nutrient concentrations.

Total P and FRP in the upper reaches of Warren's Gully, and to a lesser extent Sheep Station Ck, were lower during the falling hydrograph than sites downstream with increased weed infestation. Once the water flowed into the weeded reaches the low oxygen environment promoted the release of phosphorus from sediments. The proportion of FRP from the total was high, around 80% in Warren's and 65% in Sheep Station (Table 3.2). This is a very high proportion of reactive P (B. Bulter, pers. comm.) and may be partially due to the low oxygen environment created by weed mats, but are also suggestive of through flows containing high levels of dissolved phosphorus (surface runoff is normally dominated by particulate phosphorus). The proportion of dissolved inorganic nutrients (ammonia and nitrate) was low in relation to total N. Overall, storm tail water was higher in total P and total N, however, loadings and volume need to be considered when calculating these values given the modified nature of the area (Figures 3.15, 3.16, 3.17 and 3.18).

Input loadings of nutrients to the remnant sites during the six months of natural dry-season reduced flow were greatly reduced compared to the lagoons experiencing the elevated modified flow regime (22 to 32 t of N in the no-flow sites and 5,359 to 11,180 t of N in the supplemented sites) (Table 3.3). The same trend was found for total P. This suggests that, during the six months of no-flow, the remnant sites had very low input loadings compared to those with supplemental flows. The concentrations recorded in November 2002, at the end of the dry season, were similar for all sites, which makes it difficult to interpret why weed infestations occurred only in the supplemented sites (pers. obs.). When artificial flows over five months are calculated for Payard's and Fowler's Lagoons, the input loading is up to 300 times that of the remnant, no-flow lagoons. This indicates how modified flows increase input loadings, which may benefit *E. crassipes* infestation. To show how much increase there is with modified flows, Payard's and Fowler's (presently lotic in nature) input loading was calculated on the basis of the nutrient concentrations at the lentic sites and on volumes based on size rather than supply (i.e. if the sites were returned to lentic habitats). Total N was up to 600 times greater with supplemental flows (Tables 3.3 and 3.4).

To show how modified flows contribute to N and P, notional values were calculated to determine the concentration of N and P needed to equal the input loading of the modified flow conditions (Table 3.5). The concentrations in Payard's and Fowler's with no modified flow would need to increase 150 times to have similar input loadings as occurs with modified flows. This demonstrates that under modified flows N and P input loadings are high enough to be not limiting for *E. crassipes*, and may aid in whole lagoon infestations. This is a very simple explanation for the dynamics of *E. crassipes* growth, but it does suggest that the modified flows enhance the *E. crassipes* infestations.

Figure 3.15 Median of a) TSS (mg/L) and b) chlorophyll (µg /L), comparing falling hydrograph and supplemental flows in Sheep Station Ck and Warren's Gully. Flow is: 1) falling hydrograph (25/Feb/2002) and 2) supplemental flow (12/Nov/2002).

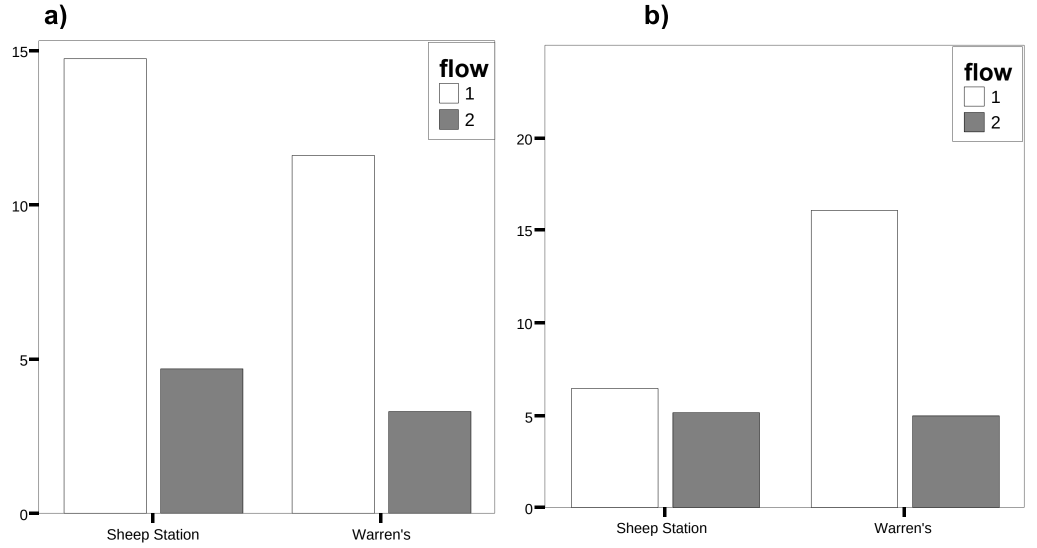


Figure 3.16 Median of a) phosphorus (µg P/L) and b) FRP (µg P/L) , comparing falling hydrograph and supplemental flows in Sheep Station Ck and Warren's Gully. Flow is: 1) falling hydrograph (25/Feb/2002) and 2) supplemental flow (12/Nov/2002).

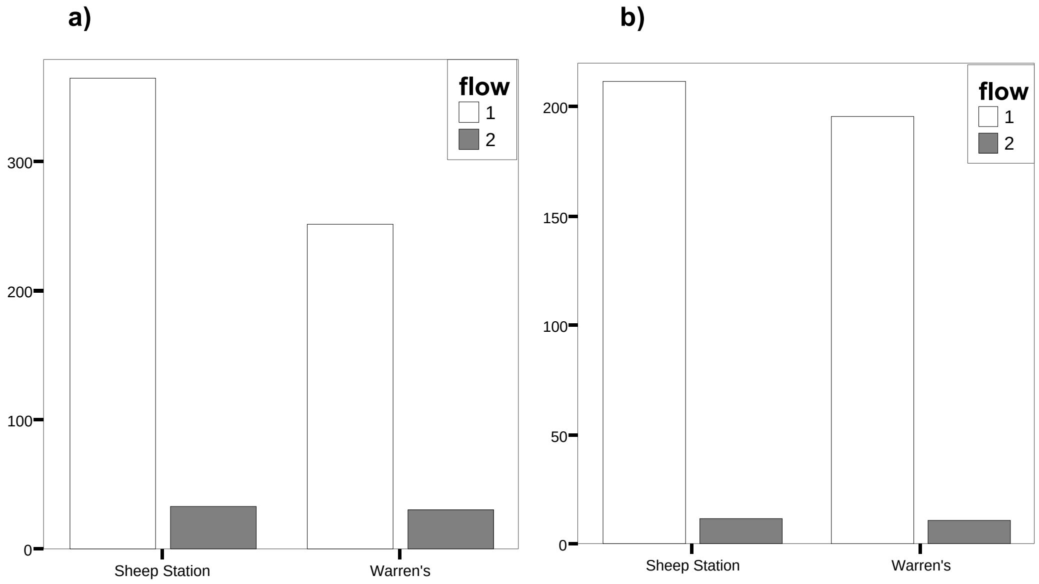


Figure 3.17 Median of a) total nitrogen (µg N/L) and b) nitrate (µg N/L) comparing falling hydrograph and supplemental flows in Sheep Station Ck and Warren's Gully. Flow is: 1) falling hydrograph (25/Feb/2002) and 2) supplemental flow (12/Nov/2002).

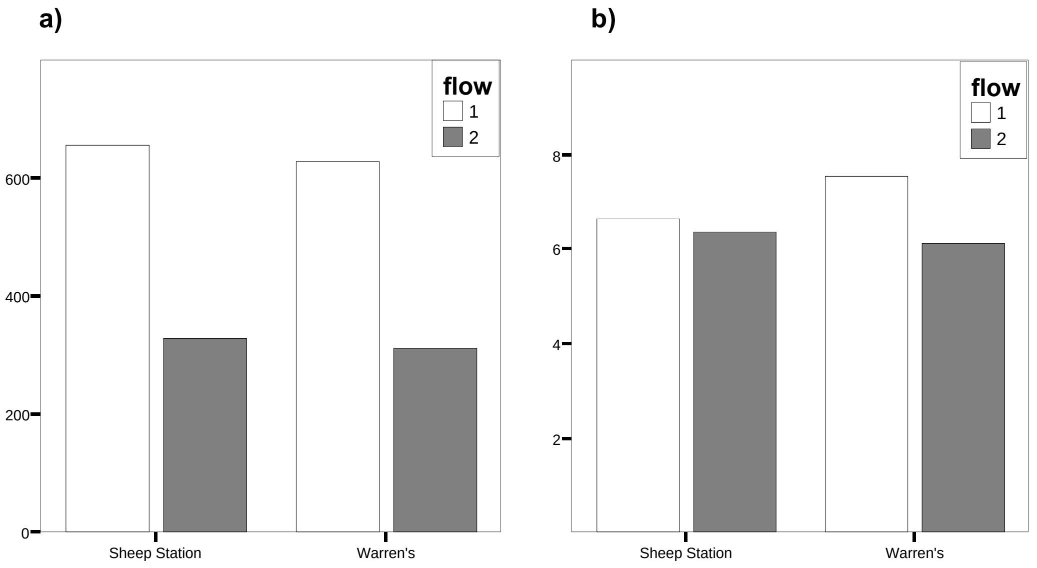


Figure 3.18 Median of total ammonia (µg N/L) comparing falling hydrograph and supplemental flows in Sheep Station Ck. and Warren's Gully. Flow is: 1) falling hydrograph (25/Feb/2002) and 2) supplemental flow (12/Nov/2002).

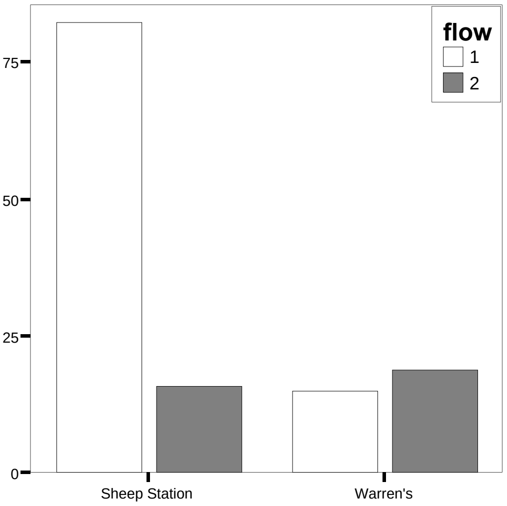


Table 3.3 Estimated total nitrogen and phosphorus input loading in remnant lagoons (no artificial flow) and lagoons with modified flows. Rows in bold and italics are sites with modified flows and loading is based on monthly pump totals from NBWB and SBWB annual reports. Inkerman and Fowler's Lagoons are in Warren's Gully and Kelly's and Payard's Lagoons are in Sheep Station Ck

Flow Condition	Lagoon	Total nitrogen concentration (µg N/L) and phosphorus (µg P/L)		Volume/Pump rate over 6 months (ML)	Total nitrogen and phosphorus input loading (kg)	
Natural	Inkerman	N384.22	P27.18	585	N 225	P 15.9
Natural	Kelly's	N 285.56	P 27.93	112	N 31.98	P 3.1
Supplemented	***Fowler's***	***N 304.86***	***P 28.90***	***17,581***	***N 5359***	***P 508***
Supplemented	***Payard's***	***N 298***	***P 35.83***	***37,518***	***N 11,180***	***P 1344***

Table 3.4 Estimates of total nitrogen and phosphorus input loading in Payard's and Fowler's if no modified flow was present. Based on volume and using total nutrient from the remnant sites as indication of what concentrations would be without flows.

Lagoon	Volume (ML)	Total nitrogen concentration (µg N/L)	Total nitrogen input loading (kg)	Total phosphorus concentrations (µg P/L)	Total phosphorus input loading (kg)
Payard's	228	285.56	65	27.9	6.4
Fowler's	22.4	384.22	8.6	27.2	0.6

Table 3.5 Notional measure of total nitrogen and phosphorus concentration in Payard's and Fowler's, with no flow, that would be needed to equal the input load from pumping.

Lagoon	Volume (ML)	Total nitrogen concentration (µg N/L)	Total nitrogen input loading (kg)	Total phosphorus concentrations (µg P/L)	Total phosphorus input loading (kg)
Payard's	228	49035	11,180	5894	1344

3.4 Discussion

As discussed in Sections 3.1.3 and 3.1.4, there is a dynamic interaction between flow, habitat condition and instream oxygen content. The oxygen is also affected by inputs of nutrients, particulate matter and autotroph biomass (mass balance of re-aeration, photosynthesis and consumption). The results of the oxygen cycles show how the saturation of oxygen is driven mostly by inflow and habitat condition, and how the amounts of nutrients (N and P) are affected by modified flow conditions, which in turn provide better growing conditions for *E. crassipes* (Section 3.3.4).

3.4.1 Flow and oxygen content

Flow is a major factor affecting water quality in the Burdekin floodplain. When there has been a local flow event and there is high organic input from the local catchment oxygen is greatly reduced (Congdon *et al.*, 1994; Pearson *et al.*, 2003). During these flow conditions DO will decrease due to decreased light condition and increased consumption via biological oxygen demand of the increased content of organic matter. The decrease in oxygen will be even greater in streams with degraded riparian areas due to lack of filtration by a riparian buffer, therefore increasing organic inputs (Rayment, 2002; Pusey and Arthington, 2003). There are short-term benefits with increased nutrients where, autotroph biomass will increase, thus increasing respiration rates. However, as biomass increases the balance between photosynthesis and respiration will tend towards respiration and the diel fluctuations may become severe enough to cause fish kills (such as at Kelly's Lagoon Fig. 3.12) (Kaenel, *et al.,* 2000; Pusey and Arthington, 2003; Pearson *et al.*, 2003). However, as seen in the sites above, the superimposition of flow raises the mean concentration levels, thus smoothing the peaks and troughs. Proliferation of submerged macrophytes under these conditions may also have indirect impacts on the fish assemblage by reducing habitat diversity and depressing secondary production (Pusey and Arthington, 2003).

3.4.2 Submerged macrophytes and oxygen cycling

Kelly's Lagoon showed super-saturation of dissolved oxygen, and then fell to levels below 60% saturation (Fig. 3.12). Not long after that data was collected, the landholder reported a fish kill (M. Kelly pers. com). The system presumably crashed due to a combination of high biomass of submerged macrophytes and a series of still, overcast days. In the beginning of the project the site had been scoured by flooding, but over the two years of the project no significant flow events occurred, thus allowing submerged macrophyte biomass to increase to over 80% cover. This became the driver of oxygen cycling in the site. Collapse occurs when consumption exceeds production or during periods of overcast conditions and increased temperature where production decreases but consumption continues (Pearson *et al.*, 2003). It has been found that light availability and temperature most affect oxygen production, and respiration rates in tropical wetlands (Wetzel, 1983; Matthews, 1998; Pearson *et al.*, 2003). The reduced light levels and high respiration rates would have caused the crash in oxygen and thus the fish kill. The other main effect of flow on oxygen is the condition of the habitat that the water flows through, in this case, mainly *E. crassipes* infested reaches.

3.4.3 *Eichhornia crassipes* impacts to downstream oxygen content

During flow, upstream condition greatly affects downstream oxygen content. The three consecutive sites on Warren's Gully highlighted this effect (Fig. 3.6), demonstrating a clear decrease in oxygen saturation from upstream to downstream. Also apparent is the gradual loss of cycling within sites, indicating that the lower – most sites were dominated by *E.crassipes* mats, flow and quality of water from upstream, and were not capable of producing enough oxygen internally to cause an increase in dissolved oxygen concentrations. Lastly, the extent of *E. crassipes* infestation within the sites may cause the flowing water to short circuit, modifying roughness, water depth and current velocity, thus reducing the detention time and greatly inhibiting mixing and re-aeration within the lagoon (Kaenel *et al.,* 2000; ACTFR, unpublished data).

When the water is pumped out of the Burdekin River, it is high in oxygen concentration (e.g., Fowler's Lagoon, Fig. 3.3). However, as this water flows downstream through weed-infested channels, the oxygen is stripped out of the water. *Eichhornia crassipes* is a floating plant that obtains oxygen, carbon dioxide and sunlight from the surface (Scheffer *et al.,* 2003). As this plant grows over a water body, instream production of oxygen is reduced and dissolved oxygen levels are lower than would be found from consumption only, as the plant inhibits re-aeration (Julien *et al.,* 2001; Scheffer *et al.,* 2003; Pearson *et al.*, 2003; ACTFR, unpublished data). Macrophyte decomposition can greatly reduce oxygen levels (Kaenel *et al.,* 2000; Battle and Mihuc, 2000). Therefore, at the downstream sites, the oxygen levels recorded reflected the water quality upstream (a few hours previous to recording), rather than oxygen dynamics within the site itself.

Another factor that was not examined in detail, but is pertinent, is that where the riparian vegetation is disturbed there tends to be extensive growth of invasive semi-emergent grasses on the banks, and these grasses increase the impact of *E. crassipes* by growing out on to the mats. The invasive grass *Brachiaria mutica* (para grass) and the native *Leersia hexandra* (Rice grass) grow on most of the banks in the distributary streams and both grow over *E. crassipes,* binding it together, creating a very solid and stable mat (Scheffer *et al.,* 2003). This mat is very hard to remove or kill with chemicals and establishes a continuous source of *E. crassipes* from the edges. *Brachiaria mutica* on its own has also been found to create poor water quality by growing out over the water and dominating the littoral zone, it accumulates sediment, thus reducing littoral habitats, changing channel morphology and increasing flow velocities (Bunn *et al.*, 1998); and it excludes native plant growth, further reducing habitat diversity (Houston and Duivenvoorden, 2002). By shading out the native submerged and emergent macrophytes there is less production of oxygen, and microhabitats that may have sustained fish populations during low oxygen periods are removed. For example, the littoral area with shallow water and submerged and emergent native macrophyte beds, provides oxygen refuge during times of low water quality (Pearson *et al.*, 2003). *Brachiaria mutica* grass has also been found to contribute very little to food webs and contributes greatly to the thick anoxic ooze layer that provides substrate for microbial oxygen consumption (Bunn *et al.*, 1998; Pusey and Arthington, 2003).

3.4.4 Modified flows, nutrients and proliferation of *E. crassipes*

During irrigation pumping (water from the upper catchment), the volume of water entering the lagoons increases, as does the total amount of nutrients. The concentrations in spot measurements appear to be higher at base flow when the local catchment is the source of input, but the constant flow created by pumping creates a much higher loading because of the high volume of water pumped through (Tables 3.3-3.5). Others have also found an increase in nutrients in downstream reaches in cane farming areas (e.g., Rayment, 2002). It would be of benefit to take more samples to determine if the downstream sections do indeed have higher nutrient concentrations as they may be filtered out by the weed mats before reaching downstream. It has been documented that when nutrients are not limited, floating macrophytes may establish a stable state and out-compete submerged macrophytes as the floating varieties have primacy to light (Scheffer *et al.,* 2003). The increased loading of nutrients coming through the modified flows in the Burdekin distributary streams may tip the competitive advantage in favour of the floating macrophytes. *Eichhornia crassipes* is known to decrease fish abundance and diversity in Lake Victoria, Africa (Willoughby *et al.,* 1993), so increases in nutrients through modified flow may indirectly decrease fish abundance and diversity.

3.4.5 Conclusion

In general, water quality across the Burdekin floodplain during this study was of moderate to low quality. Oxygen levels rarely exceeded the minimum ANZECC/ARMCANZ (2000) guideline, although these guidelines may need to be reviewed to properly reflect natural conditions in tropical floodplain systems. The altered flow conditions are seen to be the main driver of water quality. Water quality is further eroded by degraded habitats. The floodplain systems have probably always been susceptible to occasional severe oxygen depletion.

However, prior to human development, the severity, extent and duration would have been much less, so that the availability of unaffected refuges would have been much more extensive. Currently conditions are sustained below ecologically viable levels. How this affects fish communities is discussed in Chapter 4.

Chapter 4   The dynamics of fish assemblage structure on the Burdekin Floodplain

4.1 Introduction and aims

There have been several surveys of freshwater fish communities in coastal north Queensland (e.g., Russell and Hales, 1993; Pusey *et al.,* 1995a; Herbert *et al.*, 1995; Kennard, 1995; Russell *et al.*, 1996) but, with the exception of a flora and fauna survey on the Burdekin "left bank" prior to the full expansion of the irrigation area (ACTFR, 1994), none of these studies has specifically examined irrigation areas. This chapter describes the fish communities of the Burdekin floodplain and examines how habitat modification and resulting reduction in water quality have altered them.

The loss of riparian habitat may have many direct and indirect impacts to fish assemblages. Increased insolation may increase temperatures and weed growth, and impair predator avoidance (Pusey and Arthington, 2003). Increased temperatures may also disrupt reproductive strategies and impact on growth rates (Pusey and Arthington, 2003). Aquatic macrophytes, invasive grasses and floating weeds can proliferate under conditions of increased light availability (Russell *et al.*, 1996; Kaenel *et al.,* 2000; Pearson *et al.,* 2003), with major impacts on instream oxygen content (Chapter 3), which may greatly affect fish assemblages, as more sensitive species may become locally extinct. Sub-lethal effects of long-term exposure to low oxygen levels may include a reduced reproductive capacity, foraging efficiency and growth (Tallqvist *et al*, 1999; Taylor and Miller, 2001; Pearson *et al.,* 2003).

Altered flows have increased average velocities and nutrient loadings and have reduced microhabitat diversity in floodplain lagoons (Chapter 3). The increased water velocities may create fish barriers restricting migration capacity and reducing microhabitat diversity by reducing the amount of backwater habitats, which have been identified as important habitat types elsewhere (Gore and Shields, 1995). The irrigation water comes from the upper catchment and is high in sediment and nutrient loads, which themselves can exclude sensitive species (Bruton 1985; Pusey *et al.,* 1993). It was found in Chapter 3 that the nutrient levels were lower during pumping, but the cumulative input of nutrient loads and volumes are increased with the continuous flows from irrigation. This has helped to establish the aquatic and riparian weeds that reduce habitat diversity and quality, which are likely to be reflected by the fish communities. (Bruton 1985; Pusey *et al.,* 1993). It has been documented elsewhere that saltwater-derived species generally increase diversity of freshwater fish assemblages (Pusey and Kennard, 1996) but, in the Burdekin River Delta, most areas are no longer connected to the salt water due to weed infestation, bund walls and irrigation infrastructure (Chapter 2).

The aims of this chapter are to: 1) describe the present condition of fish communities across the floodplain; 2) compare fish assemblages between the 12 “remnant” sites, three highly modified sites and the ten main sites within the Water Board irrigation areas, to determine likely factors causing the absence of certain species in the irrigation areas; and 3) examine spatial variation in fish assemblages in the ten main study sites. It was hypothesized that species richness was low in the irrigation areas due to highly degraded habitats and high weed infestation, which have created barriers between the fresh and saltwater reaches and between refuge sites and degraded sites.

4.2 Methods

4.2.1 Spatial variation in fish assemblage composition

Fish surveys were conducted every three months over 2.5 years at ten main sites, five in the North Burdekin Water Board (NBWB) area (Sheep Station Ck) and five in the South Burdekin Water Board (SBWB) area (Warren’s Gully/Saltwater Ck) (see Chapter 2 for description of the sites). Not all sites were sampled during each survey (Table 4.1). Fifteen additional sites, including 12 remnant sites and three irrigation sites were sampled in June 2001 and August 2002. The remnant sites in the two distributary streams, Inkerman and Kelly’s Lagoon, were only sampled every other survey. For Payard’s Lagoon only survey data for surveys1-3 will be presented in this chapter (Table 4.1).

Fish were collected in the ten main sites mainly using dip nets, gill nets and seine nets (see below). Three other techniques visual – observations, lure-fishing and baited traps – were used opportunistically, and grouped as “other” methods in all analyses. The dip net was 100 cm x 70 cm net with a mesh size of 5 mm. The dip net was placed in the bow of the boat and the boat was driven into a selected habitat type and the net lifted. All the contents were sorted and bagged for later identification.

A set of four gill nets, each 30 m long with a 2 m drop, was used for each survey. Three nets consisted of a single panel of 2.5 cm, 5 cm or 7.5 cm while the fourth consisted of three 10 m panels of 0.75 cm, 1 cm and 1.25 cm. Nets were set one hour before sunset and retrieved one hour after sunset. The nets were set either perpendicular or parallel to the bank depending on the amount of weed infestation: in sites with large amounts of hyacinth the nets were set parallel, whereas in sites with less hyacinth the nets were set perpendicular, as it was found that the nets were more effective when set parallel only in the sites with high hyacinth infestation.

A seine net (20 m x 2 m with 5 mm stretch knotless mesh) was used at the inlet of Payard's Lagoon on every sampling occasion, and once at Castelanelli's Lagoon. These were the only locations where this net could be safely and easily used. Traps were used during two surveys but did not add any species to the total catch and so were no longer used. Catches due to lure fishing, traps and visual census were pooled (as "other methods"); only visual census detected a species not otherwise collected.

The remnant sites were surveyed using a boat-mounted electro-fisher (Smith-Root™ 2.5 GPP pulsed DC current to 6 amps) and dip net only. Up to ten two-minute shots were conducted at each site. Fewer shots were run in smaller sites. The shots were conducted parallel to the bank. After the two minutes of electrofishing, the fish were identified, measured and released.

4.2.2 Statistical analysis

Species presence/absence data at all 25 sites (10 main and 15 remnant), pooled across all sampling techniques, were ordinated using a Semi-Strong Hybrid Multi-Dimensional Scaling (SSHMDS, PATN, Belbin, 1995), based on a species-by-sites association matrix generated by Bray-Curtis dissimilarity measures. The varimax rotation option in PATN was used to rotate axes to illustrate the simplest structure. A 2-axis model is presented. A principal axis correlation analysis was used to identify significant vectors in the ordination for each of the 30 species. Significant species correlations in relation to sites were assessed using 100 Monte Carlo randomized runs, using only correlations with a significance level of $P<0.05$.

Between-site variation in species richness, total abundance, abundance of individual species, diversity (Shannon Weaver), evenness, and the proportion of total abundance contributed by exotic species was compared using univariate techniques in SPSS v.11. Data were log transformed prior to analysis to satisfy concerns about variance heterogeneity except for data expressed as a proportion, which were arcsin transformed. Analysis of variance tests of different model types were used. The ANOVA model is specified below in the results section. Factors examined were: a) ecological condition (1 = highly degraded; 2 = moderately degraded and 3 = remnant); and b) creek position (1 = north (Sheep Station Ck); 2 = south (Warren's Gully); or 3 = other (remnant sites). On occasions, site was used as a third factor, usually nested within one of the other main factors. Chi-squared contingency tests were used to test whether species occurred randomly across treatment groups**.** The treatment groups were 1 = highly degraded (n=6), 2 = moderately degraded (n=7), 3 = remnant lagoon (n=6) and 4 = riverine (n=6).

Table 4.1 Sampling schedule for all 25 sites over the study period.

Site	Survey 1 Jun-2000	Survey 2 Aug-2000	Survey 3 Aug-2000	Survey 4 Dec-2000	Survey 5 Mar-2001	Survey 6 Apr-2001	Survey 7 Jun-2001	Survey 8 Oct-2001	Survey 9 Apr-2002	Survey 10 Aug-2002
Payard's	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**	**X**
Munro's	**X**		**X**	**X**	**X**			**X**	**X**	
Princess	**X**		**X**	**X**	**X**				**X**	
Saltwater	**X**		**X**	**X**	**X**				**X**	
Fowler's	**X**		**X**	**X**	**X**			**X**	**X**	
Gorizia's			**X**	**X**	**X**	**X**		**X**		**X**
Inkerman	**X**			**X**			**X**	**X**	**X**	
Kelly's	**X**			**X**				**X**	**X**	**X**
Dick's Bank	**X**		**X**			**X**				
Jack's	**X**		**X**							
Clay hole							**X**			**X**
Lilliesmere							**X**			
Glady's										**X**
Rita Island										**X**
Alan Rd.										**X**
Horseshoe										**X**
Hutchinson										**X**
Bowen R.										**X**
The Rocks										**X**
Castanelli's										**X**
Clare Weir										**X**
Swan's										**X**
Warren's										**X**
W. Barratta										**X**
Woodhouse										**X**

4.3 Results

4.3.1 Fish assemblage structure and species richness.

Thirty species from 22 families were recorded from 25 sites over two and a half years (Tables 4.2 and 4.3). A total of 25,524 fish were collected, of which 76% were collected by dip nets (76%), 10% by electrofishing, 8% by seine-netting and 5% by gill nets. Very few fish were recorded by other methods (Table 4.4). Although some taxa, such as *Megalops cyprinoides*, *Nematalosa erebi* and plotosid catfish, were caught mostly in gill nets, a combination of electrofishing and dip netting was able to effectively sample all sites.

Table 4.2 Fish families and species recorded during this project, across the Burdekin floodplain. The asterisk (*) indicates introduced species.

- **Anguillidae**
  - *Anguilla reinhardtii* Steindachner
- **Clupeidae**
  - *Nematalosa erebi* (Gunther)
- **Ariidae**
  - *Arius graeffei* Kner and Steindachner
- **Plotosidae**
  - *Neosilurus ater* (Perugia)
  - *Neosilurus hyrtlii* Steindachner
  - *Porochilus rendahli* (Whitley)
- **Hemiramphidae**
  - *Arramphus sclerolepis* Gunther
- **Belonidae**
  - *Strongylura krefftii* (Gunther)
- **Atherinidae**
  - *Craterocephalus stercusmuscarum* (Gunther)
- **Melanotaeniidae**
  - *Melanotaenia splendida splendida* (Peters)
- **Synbranchidae**
  - *Ophisternon bengalense* McClelland
- **Ambassidae**
  - *Ambassis agrammus* Gunther
- **Centropomidae**
  - *Lates calcarifer* Bloch
- **Terapontidae**
  - *Leiopotherapon unicolor* (Gunther)
  - *Hephaestus fuliginosus* (Macleay)
  - *Scortum parviceps* (Macleay)
  - *Amniataba percoides* (Gunther)
- **Apogonidae**
  - *Glossamia aprion* (Richardson)
- **Toxotidae**
  - *Toxotes chatareus* (Hamilton)
- **Gobiidae**
  - *Redigobius bikolanus* (Herre)
- **Eleotrinae**
  - *Giurus margaritacea* (Valenciennes)
  - *Oxyeleotris lineolatus* (Steindachner)
  - *Mogurnda adspersa* (Castelnau)
  - *Hypseleotris* spp.
- **Megalopidae**
  - *Megalops cyprinoides* (Broussonet)
- **Lutjanidae**
  - *Lutjanus argentimaculatus* (Forsskal)
- **Gerreidae**
  - *Gerres filamentosus* Cuvier
- **Scatophagidae**
  - *Scatophagus argus* (L.)
- ***Poeciliidae**
  - *Gambusia holbrooki* (Giriard)
- ***Belontidae**
  - *Trichogaster trichopterus* (Pallas)

Table 4.3 Habitat condition, collecting methods used and species list for each of the 25 study sites. Conditions scores are: 1=Highly degraded, 2=Moderately degraded, 3=Remnant lagoons and 4=Riverine sites. Sites are: 1. Gorizia's, 2. Jack's, 3. Lilliesmere, 4. Munro's, 5. Payard's (includes only surveys 1-3 and not post-weed harvest surveys), 6. Princess, 7. Dick's Bank, 8. Fowler's, 9. Hutchinson, 10. Saltwater, 11. Warren's Gully, 12. Glady's, 13. Horseshoe, 14. Woodhouse, 15. Castelanelli's, 16. Inkerman, 17. Kelly's, 18. Swan's, 19. Rita Island, 20. W. Barratta, 21. Burdekin (The Rocks), 22. Clay Hole, 23. Clare Weir, 24. Barrata at Alan Rd., 25. Bowen Junction. Sites are ordered by condition and species are ordered to indicate the associated gradient in assemblage composition. * indicates exotic species

SITE	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**	**21**	**22**	**23**	**24**	**25**
Condition	1	1	1	1	1	1	2	2	2	2	2	2	2	2	3	3	3	3	3	3	4	4	4	4	4
Method																									
Dipnet	X	X	X	X	X	X	X	X	X			X	X		X	X	X	X			X	X	X		X
Gill nets	X	X	X	X	X	X	X	X			X	X				X	X								
Electrofisher			X	X	X	X			X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Seine nets					X										X				X						
Visual															X				X			X			
Ophisternon bengalense	X			X																					
Mogurnda adspersa					X		X																		
*Trichogaster Trichopterus**	X	X			X		X										X								
Porochilus rendahli	X		X	X			X	X		X			X		X	X	X	X							
Leiopotherapon unicolor			X		X		X	X		X	X		X	X			X	X							X
Neosilurus hyrtlii				X		X	X	X		X			X				X	X							
*Gambusia holbrooki**	X	X	X	X	X	X	X	X	X	X		X			X	X	X	X	X			X			X
Megalops cyprinoides	X	X	X	X	X	X	X	X		X	X		X			X	X		X			X		X	
Hypseleotris spp.	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X		X	X			X
Anguilla reinhardtii	X		X	X		X		X	X	X		X	X	X	X		X	X			X	X	X	X	X
Neosilurus ater	X			X	X	X	X	X		X			X	X	X	X	X			X		X		X	X
Melanotaeniaspl. splendida				X		X	X	X	X	X	X	X	X	X		X			X	X	X	X	X		X
Giurus margaretacea	X			X		X				X										X		X		X	
Craterocephalus stercusmuscarum			X		X		X	X	X	X		X	X	X	X	X	X	X	X			X	X		X
Ambassis agrammus	X				X	X	X	X		X			X		X	X	X	X	X		X	X			X
Amniataba percoides							X				X				X				X		X	X		X	X
Glossamia aprion							X	X	X		X	X	X	X	X	X	X	X	X	X	X	X	X		X
Nematalosa erebi							X	X	X	X		X		X	X		X	X		X	X	X	X	X	X
Oxyeleotris lineolatus							X	X	X		X	X		X	X		X	X	X	X	X	X	X	X	X
Lates calcarifer				X						X				X		X			X	X	X	X		X	X
Redigobius bikolanus														X	X	X			X						
Toxotes chatareus									X						X				X		X	X	X		X
Strongylura krefftii									X												X		X		X
Hephaestus fuliginosus											X												X	X	X
Arramphus sclerolepis									X														X		X
Scortum parviceps											X														
Lutjanus argentimaculatus																			X						
Scatophagus argus																				X					
Arius graeffei																				X		X	X		X
Gerres filamentosus																					X				
Total Species Richness	**10**	**4**	**7**	**11**	**9**	**9**	**16**	**14**	**11**	**14**	**9**	**8**	**11**	**11**	**13**	**11**	**14**	**11**	**13**	**9**	**12**	**16**	**11**	**9**	**18**

Table 4.4 Total number of fishes collected by each sampling technique. "Other" is the sum of records from visual observations, lures and baited traps. * represents introduced species. This table includes all surveys at Payard's lagoon (i.e. before and after weed harvesting).

	Technique						
Species	**Dip net**	**Electro-fishing**	**Gill**	**Seine**	**Other**	**Total Numbers**	**Percentage of total catch**
Ambassis agrammus	436	46	1	1501	85	2069	8.0
Amniataba percoides	2	16		3		21	<0.1
Anguilla reinhardtii	2	61	4			67	0.3
Arius graeffei		16				16	0.1
Arramphus sclerolepis		23				23	0.1
Craterocephalus stercusmuscarum	761	384		300	1	1446	6.0
*Gambusia holbrooki**	3353	4		81		3438	13.
Gerres filamentosus		4				4	<0.1
Giurus margaretacea		11	5			16	0.1
Glossamia aprion	216	123	2	8	1	350	1.0
Hephaestus fuliginosus		11				11	<0.1
Hypseleotris spp.	14316	141	3	34	4	14498	57.0
Lates calcarifer		93	79		3	175	0.7
Leiopotherapon unicolor	5	21	6	15		47	0.2
Lutjanus argentimaculatus				1		1	<0.1
Megalops cyprinoides		38	934	1		973	4.0
Melanotaenia spl. splendida	63	385	1	19		468	2.0
Mogurnda adspersa	36			7		43	0.2
Nematalosa erebi	8	852	99	184		1143	4.0
Neosilurus ater	7	69	101	1		178	0.7
Neosilurus hyrtlii	3	53	22			78	0.3
Ophisternon bengalense	1	1				2	<0.1
Oxyeleotris lineolatus	3	90	3	1		97	0.4
Porochilus rendahli	51	60	27	7		145	0.6
Redigobius bikolanus	4			12		16	0.1
Scatophagus argus		1				1	<0.1
Scortum parviceps		2				2	<0.1
Strongylura krefftii		25				25	0.1
Toxotes chatareus	1	57		1	5	69	0.3
*Trichogaster trichopterus**	107					107	0.4
Total for technique	**19375**	**2587**	**1287**	**2176**	**99**	**25524**	
Species recorded per technique	**19**	**26**	**14**	**17**	**6**		
Species unique to technique	**1**	**7**	**0**	**1**	**0**		
Percent of catch per technique	**76**	**10**	**5**	**9**	**0.004**		

The most abundant taxon caught was *Hypseleotris* spp., accounting for approximately 57% of the total (Table 4.4). Other abundant species were *Gambusia holbrooki* (13%), *Ambassis agrammus* (8%), *Craterocephalus stercusmuscarum* (6%) and *Nematalosa erebi* (4%). Dip net samples accounted for the majority of the catch and *Hypseleotris* spp. accounted for 74% of fish in the dip net samples (Table 4.4). Gill net samples were dominated by *Megalops cyprinoides* (73%) while plotosid catfishes (*Neosilurus ater, N. hyrtlii* and *Porochilus rendahli*) accounted for 10% and *Nematalosa erebi* approximately 8%. Seine net samples were dominated by *Ambassis agrammus* (69%), *C. stercusmuscarum* (14%) and *N. erebi* (8%).

The ten main sites (sampled every quarter) contained only 21 species from 14 families (Table 4.5). *Hypseleotris* spp. was the most numerically dominant taxon, comprising 69% of the catch. Species absent from these sites, but present in remnant sites or the main channel of the river, were either amphidromous and could not access the lagoons from downstream (e.g., *A. graeffei, Scatophagus argus* and *Lutjanus argentimaculatus*), or were riverine species that preferred the main channel (*Gerres filamentosis, Hephaestus fuliginosus, Scortum parviceps*). Other species historically recorded in floodplain lagoons of the area but absent in this study included: *Arrhamphus sclerolepis, Arius graeffei, Strongylura kreffti and Toxotes chatareus* (A. Darwen, J. Tait, A. Stennet, pers. comm.; MacLeay, 1883).

Hypseleotris spp. was the most widely distributed taxon, being recorded in 22 of the 25 sites surveyed. Other species that were widely distributed were *Gambusia holbrooki* (18 sites), *Anguilla reinhardtii* (18 sites) and *Megalops cyprinoides* (16 sites). The most widely distributed species were also apparently the most tolerant to low oxygen levels (Table 4.6). Five species, *Scortum parviceps*, *Lutjanus argentimaculatus*, *Scatophagus argus* and *Gerres filamentosus,* were recorded at a single site only and these species were either riverine or amphidromous (Table 4.3). Two uncommon species were restricted to specific habitats: *Gerres filamentosus* was only sampled in the Burdekin River and *Ophisternon bengalense* was only recorded from two highly disturbed lagoons (Table 4.3).

SSHMDS ordination and principal axis correlation analysis of the species' presence/absence show strong grouping along axis one and weak grouping on axis 2 (2-axis stress of 0.19) (Fig. 4.1). Condition 1 sites are arrayed positively whereas riverine sites (condition 4) were arrayed negatively on Axis 1, indicating a gradient of site condition. Condition 2 and 3 sites mix in the middle due to lack of strong separation in the condition classification of these two groups. Taxa for which significant correlations between ordination scores and presence/absence were detected included *M. cyprinoides, Hypseleotris* spp., and *G. holbrooki* (associated with degraded sites), and *N. erebi, T. chatareus, L. calcarifer* and *G. aprion* (associated with sites with good habitat conditions) (Table 4.3). Thus, habitat condition is a good predictor of fish assemblage composition. Axis 2 (45% of the variation) indicates a separation of river sites from lagoon sites. The species that correlate with this gradient such as *A. sclerolepis* and *A. graeffei,* are all characteristic river species, whereas *O. bengalense, G. holbrooki* and *P. rendahli* are characteristic of lagoon habitats (Table 4.3). Fish assemblages thus appear to be habitat-specific.

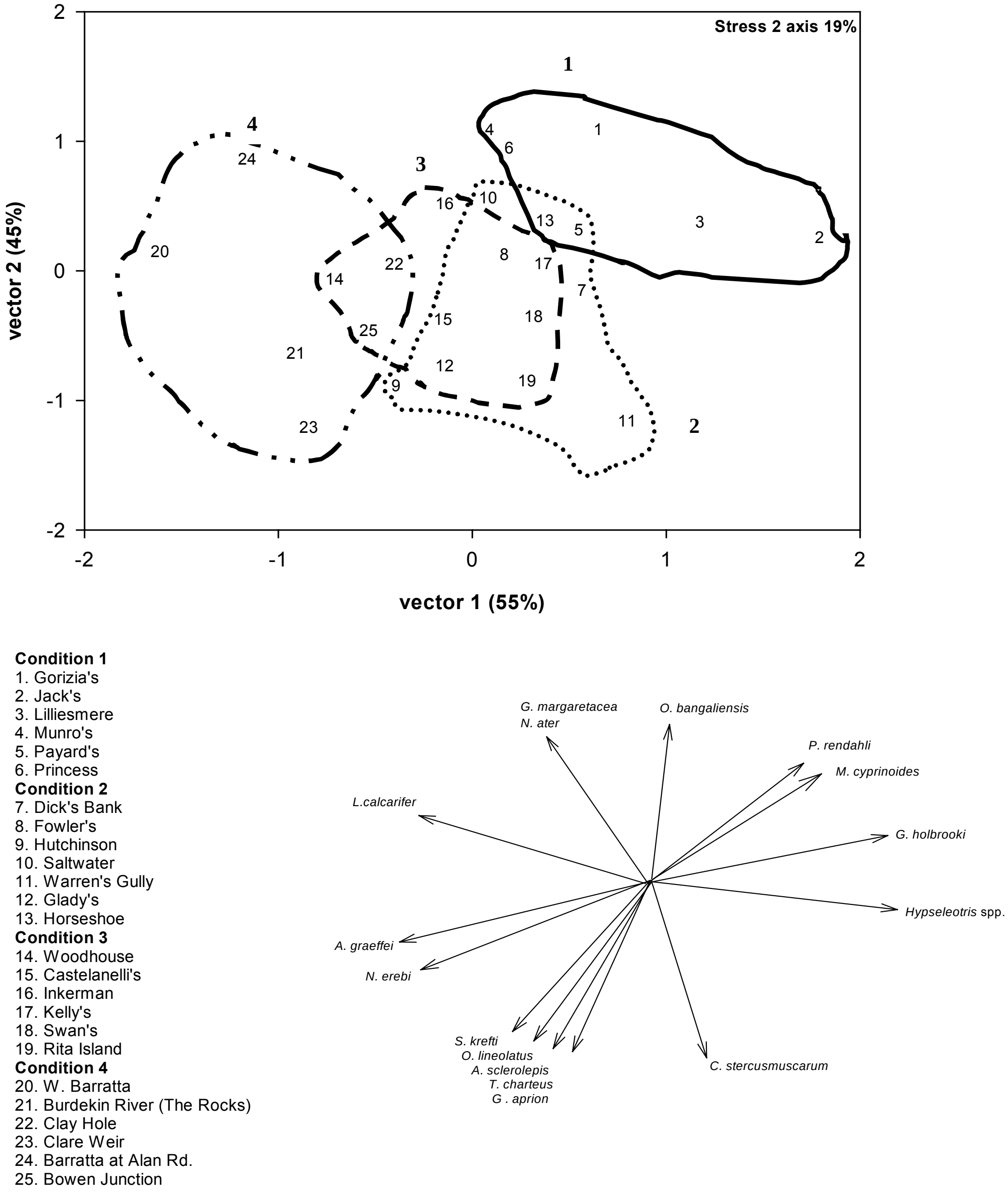
Table 4.5 Total abundance of species caught by each technique in the 10 main study sites.

Species	Dip net	Electro	Gill net	seine net	Other	Total	% of catch
Ambassis agrammus	210	3		666		**879**	**5.43**
Amniataba percoides	2					**2**	**0.01**
Anguilla reinhardtii	2		4			**6**	**0.04**
Craterocephalus stercusmuscarum	376	6		8		**390**	**2.41**
Gambusia holbrooki	2234			11		**2245**	**13.86**
Glossamia aprion	136	1	1		1	**139**	**0.86**
Hypseleotris spp.	11176	15	3	4		**11198**	**69.14**
Lates calcarifer			63		1	**64**	**0.40**
Leiopotherapon unicolor	2		3	9		**14**	**0.09**
Megalops cyprinoides		20	851			**871**	**5.38**
Melanotaenia spl. splendida	33	23	1	2		**59**	**0.36**
Mogurnda adsperssa	8			3		**11**	**0.07**
Nematalosa erebi	2	4	46			**52**	**0.32**
Neosilurus ater	6		78	1		**85**	**0.52**
Neosilurus hyrtlii	1		13			**14**	**0.09**
Giurus margaretacea		7	4			**11**	**0.07**
Ophisternon bengalense		1				**1**	**0.01**
Oxyeleotris lineolatus	2	1	3			**6**	**0.04**
Porochilus rendahli	18		17			**35**	**0.22**
Trichogaster trichopterus	106					**106**	**0.65**
Redigobius bikolanus	2					**2**	**0.01**
Total	**14322**	**81**	**1087**	**704**	**2**	**16196**	
% catch by method	**88.43**	**0.50**	**6.71**	**4.35**	**0.01**		

Table 4.6 Species composition, site condition and dissolved oxygen concentration . Sites include the 10 main study sites plus two remnant sites and 1 degraded site in Plantation Creek (Lilliesmere) showing mean, maximum, minimum and S.E. of percent saturation of oxygen at each site. Sites are arrayed in ascending order of minimum dissolved oxygen content. Lowest to highest minimum content of oxygen. NB. oxygen content collected at Lilliesmere was unusually high for that site because of recent weed removal and high pump rates. Conditions are: 1=Highly degraded, 2=Moderately degraded, 3=Remnant lagoons and 4=Riverine sites. Sites are: 1. Jack's, 2. Princess, 3. Gorizia's, 4. Payard's (survey's 1-3 only, does not included post weed harvest survey's), 5. Saltwater, 6. Munro's, 7. Dicks Bank, 8. Inkerman, 9. Kelly's, 10. Lilliesmere, 11. Fowlers, 12. Castelanelli's, 13. Clay hole.

SITE	1	2	3	4	5	6	7	8	9	10	11	12	13
Condition	1	1	1	1	2	1	2	3	3	1	2	3	4
Average DO % Saturation	9.6	17.0	15.4	36.6	34.8	23.9	50.8	36.2	58.7	45.5	70.1	92.9	60.2
Maximum DO % Saturation	30.4	78.3	92.8	89.8	92.9	44.7	80.0	107.5	148.6	75.0	99.7	104.0	65.8
Minimum DO % Saturation	0.2	0.2	0.4	0.6	2.3	3.4	4.2	4.7	5.1	37.1	41.5	84.4	55.9
S.E. DO % Saturation	1.6	1.3	1.1	3.0	2.4	0.9	1.6	1.6	3.2	0.9	1.0	0.8	0.1
Species Richness	4	9	10	9	14	11	15	11	14	7	14	11	15
Gambusia holbrooki	X	X	X	X	X	X	X	X	X	X	X	X	X
Megalops cyprinoides	X	X	X	X	X	X	X	X	X	X	X		X
Hypseleotris spp.	X	X	X	X	X	X	X	X	X	X	X	X	X
Anguilla reinhardtii		X	X		X	X			X	X	X	X	X
Neosilurus ater		X	X	X	X	X	X	X	X		X	X	X
Trichogaster trichopterus	X		X				X		X				
Neosilurus hyrtlii		X			X	X	X		X		X		
Porochilus rendahli			X		X	X	X	X	X	X	X	X	
Giurus margaretacea		X	X		X	X							X
Melanotaenia spl. splendida		X		X	X	X	X	X			X		X
Ambassis agrammus		X	X	X	X		X	X	X		X	X	X
Ophisternon bengalense			X			X							
Craterocephalus stercusmuscarum				X	X		X	X	X	X	X	X	X
Mogurnda adsperssa				X			X						
Lates calcarifer					X	X		X					X
Leiopotherapon unicolor				X	X		X		X	X	X		
Glossamia aprion							X	X	X		X	X	X
Redigobius bikolanus								X				X	
Nematalosa erebi					X		X		X		X	X	X
Oxyeleotris lineolatus							X		X		X	X	X
Toxotes chatareus													X
Arius graeffei													X

Figure 4.1 Semi-Strong Hybrid Multidimensional Scaling ordination plot of species by sites matrix using total species richness at all 25 sites and including all sampling methods. Inset reflects correlation between species abundances and position in ordination space.



Condition 1
1. Gorizia's
2. Jack's
3. Lilliesmere
4. Munro's
5. Payard's
6. Princess

Condition 2
7. Dick's Bank
8. Fowler's
9. Hutchinson
10. Saltwater
11. Warren's Gully
12. Glady's
13. Horseshoe

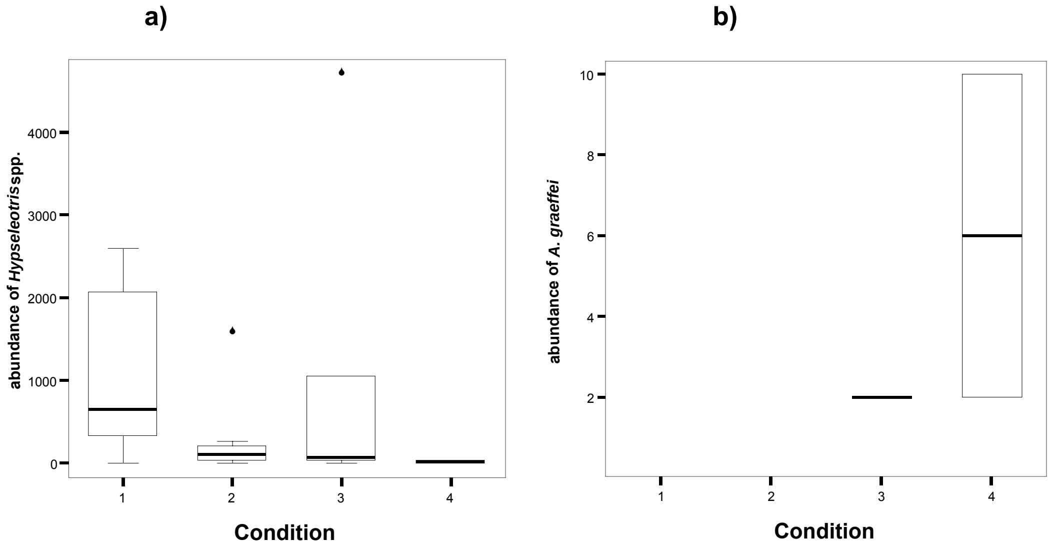
Condition 3
14. Woodhouse
15. Castelanelli's
16. Inkerman
17. Kelly's
18. Swan's
19. Rita Island

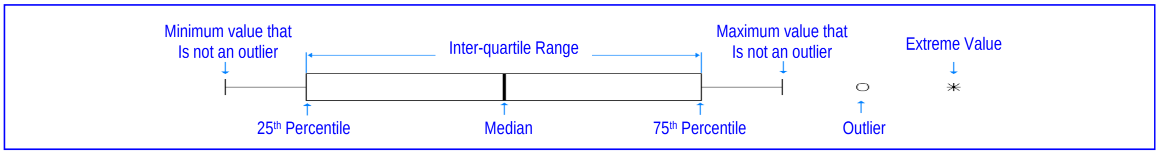
Condition 4
20. W. Barratta
21. Burdekin River (The Rocks)
22. Clay Hole
23. Clare Weir
24. Barratta at Alan Rd.
25. Bowen Junction

4.3.2 Species richness and site condition across all sites

Ordination suggests that certain species show preference for particular habitat types. One species, *A. graeffei* was sampled in river sites only (Fig. 4.2 b). The most common taxon, *Hypseleotris* spp., was recorded across all habitat condition types but occurred significantly more frequently in the degraded sites (Fig 4.2 a). To further examine this effect a χ^2 test was run using four conditions (Table 4.7). The results show significant preference by some species for good condition lagoon and riverine sites (*Redigobius bikolanus, Oxyeleotris lineolatus, Lates calcarifer, Nematalosa erebi*) whereas *Giurus magaritacea,* and *Hypseleotris* spp. appear to prefer impacted lagoon habitats (Table 4.7).

Figure 4.2 a) Abundance of *Hypseleotris* spp. for all sites and all methods, across condition; b) abundance of *A. graeffei* for all sites and all methods, across condition. The boxplot symbols are explained below the figure. Conditions are: 1= highly degraded, 2= moderately degraded, 3= remnant lagoon and 4= riverine. The * in A are extremes.





Explanation of Boxplots

Site condition also had an effect on the abundance and number of exotic species. Two species of exotic fish were recorded (*Gambusia holbrooki* and *Trichogaster trichopterus*), both of which have high tolerance for poor water quality (Cech *et al*., 1985; Herbert and Wells, 2001). Analysis of this data comparing differences between site conditions reveals that the exotic species richness is significantly higher in condition 1 sites ($F_{2,24}$=3.566, p=0.046). The number of exotic species and proportion of exotics to total species richness was significantly higher in Sheep Station Ck (Fig. 4.3, Table 4.8), whereas abundance of species was more evenly distributed (evenness) in Warren's Gully and other streams. Degraded sites (especially in Sheep Station Ck) were dominated by a few very tolerant species, with exotic species being prominent in this group of species. A two-way ANOVA (main factors being stream and site, the interaction was stream by condition) on the diversity of exotics found that only stream was significant ($F_{2,25}$ 5.859, p=0.012).

Table 4.7 Number of sites within each condition category in which selected fish species were recorded, with χ^2 results testing the significance of the pattern. Only those species for which significant differences in incidence across the gradient of habitat condition were detected are shown. Comparison of observed versus expected values by χ^2 test. Condition: 1 = Highly degraded, 2 = Moderately degraded, 3 = Remnant lagoon and 4 = Riverine habitat.

	Condition				df	χ^2	P
	1	2	3	4			
Number of sites:	6	8	6	5			
Arius				4	3	15.08	0.002
Redigobius			4		3	15.08	0.002
Oxyeleotris		5	5	5	3	15.18	0.002
Lates	1	1	3	5	3	8.23	0.041
Nematalosa	1	5	4	5	3	9.40	0.024
Glossamia	1	6	6	5	3	12.48	0.006
Giurus	4	1		3	3	8.04	0.045
Hypseleotris spp.	6	7	6	3	3	10.80	0.013

Figure 4.3 Proportion of exotics to the total species richness for all sites and all methods, across three streams (all sites, all methods). Streams are 1= Sheep Station Ck., 2= Warren's Gully and 3= Others. See Figure 4.2 for explanation of boxplot symbols.

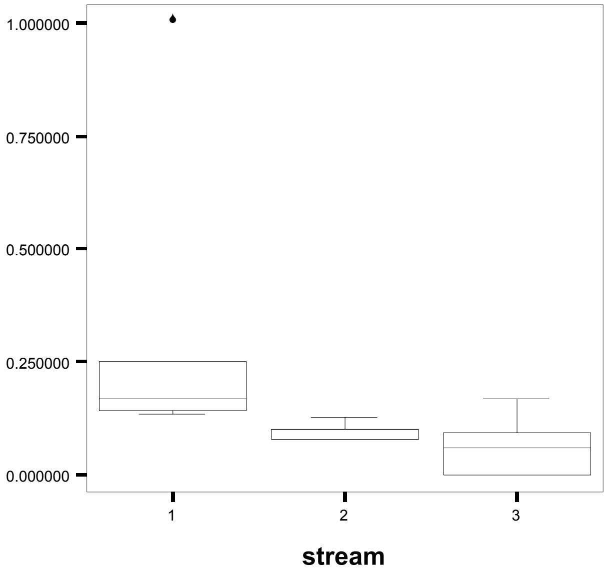


Table 4.8 Comparison of F values by one-way ANOVA for assemblage descriptors. Data transformed to $\log_{10}$, except proportion of exotics and evenness, which were arcsin transformed.

	Stream			$F_{2,24}$	**P**
	Sheep Station Ck Mean (S.E)	Warren's Gully Mean (S.E)	Other Mean (S.E)		
N	5	5	15		
Total species richness	12 (2)	12 (1)	11 (1)	0.073	0.93
Number natives	10 (2)	11 (1)	11 (1)	0.577	0.57
Number exotics	2 (0)	1 (0)	1 (0)	12.709	0.00***
Evenness	0.459 (0.049)	0.413 (0.083)	0.707 (0.036)	9.566	0.001***
Proportion exotics to total species richness	0.335 (0.167)	0.096 (0.009)	0.054 (0.015)	4.170	0.029*

4.3.3 Spatial variation of fish species richness and abundance in dip nets in the 10 main sites.

Dip net samples accounted for the majority of fish sampled and the following analyses are restricted to this sampling methodology and to only those sites included in the main groups of sites (i.e. ten sites sampled quarterly). Two-way ANOVA was used to examine spatial variation by site, stream and condition with interactions between these variables. Total abundance (Log_{10} transformed) was modeled by condition, stream and the interaction between condition and stream (Table 4.9). Abundances were significantly higher in the sites of good condition (despite the proliferation of exotics in the poor sites) (Fig. 4.4). Site nested within condition by stream was not significant; therefore, condition is the main factor determining total abundance of species in dip net samples. Spatial variation in species richness was tested for site, condition, stream, condition by stream and site nested within condition by stream. Only when site was nested within stream was there a significant effect (Table 4.10). Although significant, this nested effect is driven by one site, Fowler's Lagoon (Mean richness 5.80± 1.30) (Fig. 4.5). Overall the dip net samples reveal no spatial or treatment differences in species richness.

Total native species abundance (Log_{10} transformed) was examined for effects of condition, stream, condition by stream, and site within condition by stream variations. In the condition, stream and condition by stream model, condition was found to be significant, with more native species in good sites (Table 4.11). When the interaction was changed to site within condition by stream, both condition and the interaction were found to be significant (Table 4.12). Once again there were more native fish in the good sites (Fig. 4.6) and the site nested within the interaction indicates that there are significant differences between site effects and one of the four interaction components. In this case Jack's Lagoon in the north had high abundance of one taxon (*Hypseleotris* spp.), even though this site was ranked as being of very poor condition (Fig. 4.7).

The proportion of total abundance contributed by exotics within the 10 main sites, was analyzed by condition, stream, condition by stream and site nested within condition by stream. The condition by stream model shows that numerically there were more exotics in the north and more in poor condition sites (Table 4.13; Fig. 4.8), but no significant interaction. The model of site nested in condition by stream is very significant (Table 4.14). There were many more exotic fishes in Sheep Station Ck lagoons in poor condition. Other than condition, the proportion of exotic abundances in sites and streams was variable as some sites of poor condition were dominated by exotic fishes whereas others were not.

Table 4.9 F values and related statistics for a two-factor ANOVA comparing species abundance (Log_{10} transformed) in dip net samples in 10 main study sites. The treatments were condition, stream and condition by position.

Factor	d.f.	F	P
condition	1,38	4.224	0.047
stream	1,38	1.531	0.224
condition*stream	1,38	2.266	0.140

Table 4.10 F values and related statistics for a two-factor nested ANOVA comparing species richness in dip net samples from 10 main sites. The treatments used were stream and site nested within stream.

Factor	d.f.	F	P
Stream	1,32	0.241	0.627
Site(Stream)	6,32	3.436	0.010

Table 4.11 F values and related statistics for a two-factor ANOVA comparing native species abundance (Log_{10} transformed) of dip net samples in 10 main sites and stream. The treatments were condition, stream and the interaction condition by stream.

Factor	d.f.	F	P
condition	1,38	6.807	0.013
stream	1,38	0.000	0.999
condition*stream	1,38	3.128	0.085

Table 4.12 F values and related statistics for a two-factor ANOVA comparing native species abundance (Log_{10} transformed) in dip net samples from 10 main sites. The treatments were condition, stream and the interaction site within condition by stream.

Factor	d.f.	F	P
condition	1,32	8.620	0.006
stream	1,32	1.856	0.183
site (condition*stream)	7,32	2.958	0.17

Table 4.13 F values and related statistics for a two-factor ANOVA comparing proportion of total abundance contributed by exotics (Arcsine transformed) of dip net samples in 10 main sites. The treatments were condition, stream and the interaction condition by stream.

Factor	d.f.	F value	Significance
condition	1,38	6.763	0.013
stream	1,38	9.559	0.004
condition*stream	1,38	1.165	0.287

Table 4.14 F values and related statistics for a two-factor ANOVA comparing proportion of total abundance contributed by exotics (Arcsine transformed) of dip net samples in 10 main sites. The treatments were condition, stream and the interaction site within condition by stream.

Factor	d.f.	F value	Significance
condition	0,32		
stream	1,32	29.017	0.000
site (condition*stream)	7,32	4.252	0.002

Figure 4.4 Total species abundance in dip net samples of 10 main sites, in two conditions. The conditions are 1= degraded and 2= remnant. The extreme not shown is Kelly's lagoon (2753 *Hypseleotris* spp.) and Saltwater lagoon (941 *Hypseleotris* spp.). See Figure 4.2 for explanation of boxplot symbols.

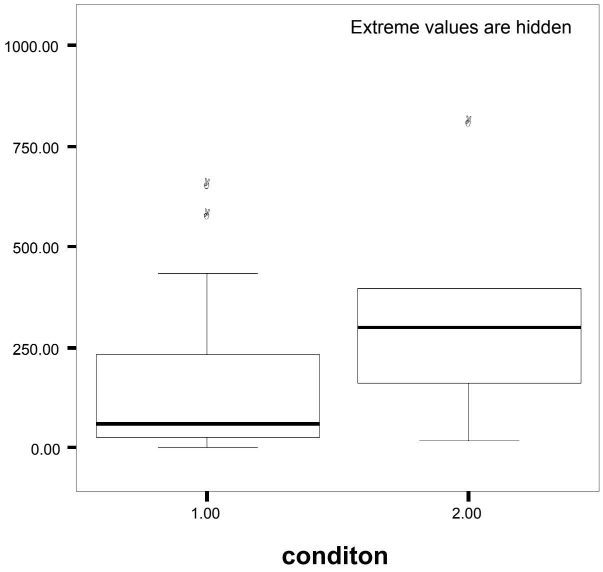


Figure 4.5 Total species richness in dip net samples of 10 main sites, across sites. Sites are 1= Jack's, 2= Gorizia's, 3= Dick's Bank, 4= Payard's, 5= Kelly's, 6= Saltwater, 7= Munro's, 8= Princess, 9= Fowler's and 10= Inkerman. Streams are 1= Sheep Station and 2= Warren's gully. See Figure 4.2 for explanation of boxplot symbols.

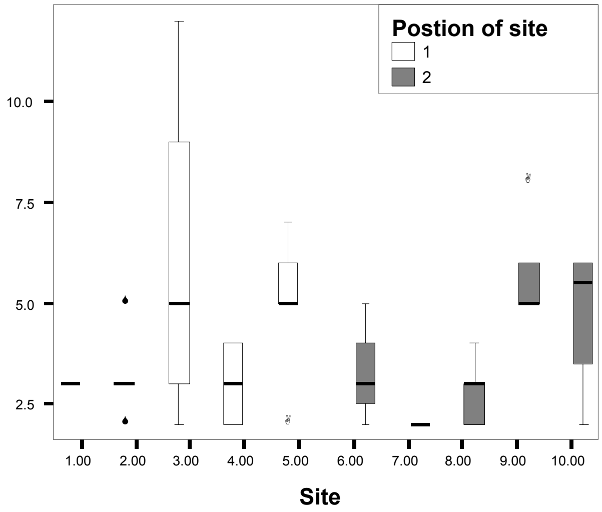


Figure 4.6 Total abundance of native species in dip net samples of 10 main sites, by condition. Condition 1= degraded and 2= remnant. The outliers are extremely high catches of *Hypseleotris* spp. An extreme in condition 2 is not shown, the value was 2758, 99% *Hypseleotris* spp. See Figure 4.2 for explanation of boxplot symbols.

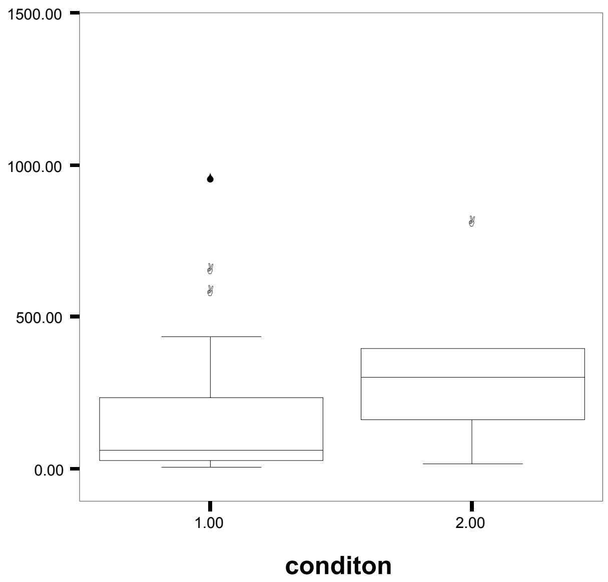


Figure 4.7 Total native species abundance in dip net samples of 10 main sites, by site. Sites are 1= Jack's, 2= Gorizia's, 3= Dick's Bank, 4= Payard's, 5= Kelly's, 6= Saltwater, 7= Munro's, 8= Princess, 9= Fowler's and 10= Inkerman. Streams are 1= Sheep Station and 2= Warren's Gully. An extreme in site 5 is not shown, the value was 2758, 99% *Hypseleotris* spp. See Figure 4.2 for explanation of boxplot symbols.

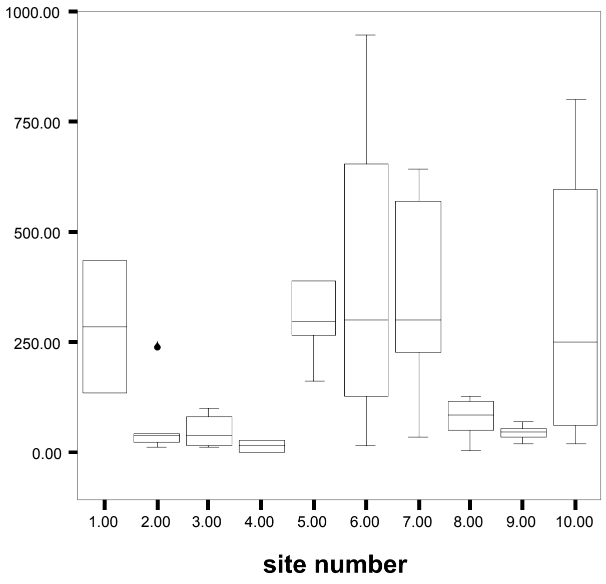
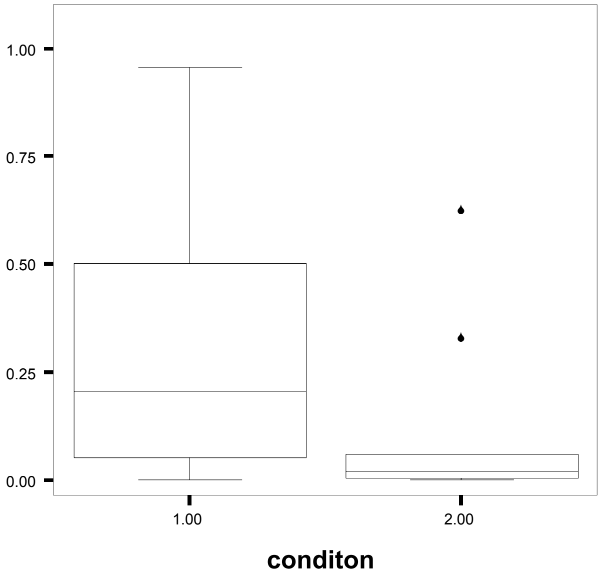


Figure 4.8 Proportion of exotics to total species abundance in dip net samples of 10 main sites, by condition. Condition is 1= degraded and 2= remnant. See Figure 4.2 for explanation of boxplot symbols.



4.3.4 Spatial variation of fish species abundance and richness in gill nets at 9 main sites.

Fish abundance and species richness in gill net samples were analyzed using two-way ANOVA. The models were the same as for dip net analyses. Fish abundances showed significant effect of stream and stream by condition interaction (Table 4.15), but when site was nested within the interaction term, the stream effect was no longer significant (Table 4.16). This suggests that local factors at the site level influence abundance levels irrespective of stream position on the delta or condition. (Fig. 4.9). Species richness showed similar results. Only the interaction effect was significant (Table 4.17). There was no significant difference between condition or stream, and the interaction was largely the result of high species richness in Saltwater Creek (Fig. 4.10). When site was nested in the interaction (stream by stream) the same result-no significance of site or stream only interaction – occurred (Table 4.18). This re – emphasizes the extent of between-site variation irrespective of condition or stream.

Table 4.15 F values and related statistics for a two-factor ANOVA comparing total abundance (Log10 transformed) of gill net samples in 9 main sites. The treatments used were condition, stream and the interaction condition by stream.

Factor	d.f.	F value	Significance
condition	1,30	0.001	0.982
stream	1,30	4.296	0.047
condition*stream	1,30	4.333	0.046

Table 4.16 F values and related statistics for a two-factor ANOVA comparing total abundance (Log10 transformed) of gill net samples in 9 main sites. The treatments used were condition, stream and the interaction site within condition by stream.

Factor	d.f.	F value	Significance
condition	1,25	0.040	0.844
stream	1,25	1.528	0.228
Site (condition*stream)	1,25	3.202	0.018

Table 4.17 F values and related statistics for a two-factor ANOVA comparing species richness (Log10 transformed) of gill net samples in 9 main sites. The treatments used were condition, stream and the interaction condition by stream.

Factor	d.f.	F value	Significance
condition	1,30	0.281	0.600
stream	1,30	1.124	0.297
condition*stream	1,30	4.591	0.040

Table 4.18 F values and related statistics for a two-factor ANOVA comparing species richness (Log10 transformed) of gill net samples in 9 main sites. The treatments used were condition, stream and the interaction site within condition by stream.

Factor	d.f.	F value	Significance
condition	1,25	0.141	0.711
stream	1,25	0.264	0.620
Site (condition*stream)	1,25	3.338	0.015

Figure 4.9 Total species abundance in gill nets for 9 main study sites, by condition. Condition is 1= degraded and 2= remnant. See Figure 4.2 for explanation of boxplot symbols.

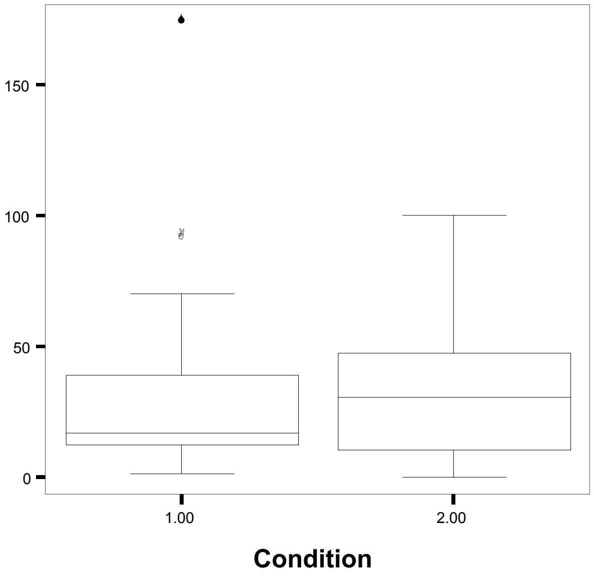
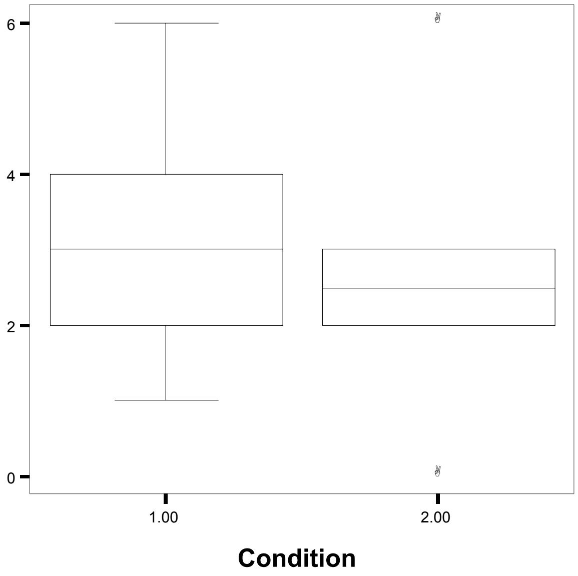


Figure 4.10 Species richness in gill net samples for 9 main sites, by condition. Condition is 1= degraded and 2= remnant. See Figure 4.2 for explanation of boxplot symbols.



4.4 Discussion

4.4.1 Fish assemblage structure and habitat condition.

Species richness in floodplain lagoons of the Burdekin River delta (30 species) is low compared with other systems in the region (40 species in the Black/Alice River - Beumer, 1980; 66 species from 11 areas Bloomfield to Cardwell - Pusey and Kennard, 1996; 36 species from Mulgrave and South Johnstone Rivers - Pusey *et al*., 1995a). Floodplain lagoons of the Burdekin delta contained very few amphidromous species, in contrast to other aquatic systems of northern Queensland (Beumer, 1980; Kennard, 1995; Pusey and Kennard, 1996; Lokkers *et al.*, 2000). The lack of saltwater-derived species is due largely to the loss of connection to the saltwater reaches, either by physical barriers (bunding, drop boards and sand dams) or barriers created by weed infestation (Tait and Perna, 2001). There are an estimated 43 species of fish that occur or once occurred, on the floodplain and most of the missing species would be excluded by the above impacts (Tait and Perna, 2001). When species richness is examined in the context of catchment size (>100,000 km^2) the Burdekin can be seen as low in fish species richness. Catchment size and species richness have been shown to be correlated (Pusey *et al*., 1995a). The low species richness is attributed to the river's low gradient, sand-dominated substrate and low predictability of flooding (Pusey *et al*., 1995a). The Burdekin also has one of the largest floodplains in Australia (Chapter 2) and a wide range of habitats within this floodplain that should accommodate more species. The degraded nature of habitats across the floodplain is likely to account for the low species richness recorded.

Habitat condition is one of the most important factors in fish assemblage structure across the sites sampled. Species richness is higher in the better condition sites and lower in poor condition sites. The relationship between habitat structure and fish assemblage structure is well documented, especially that of species richness (Schlosser, 1987; Bishop and Forbes, 1991; Pusey *et al*., 1993). Habitat cover, either large woody debris, macrophyte beds or overhanging vegetation, has been shown to be important to fish for refuge from predation, foraging and spawning (Pusey *et al.,* 1993; Pusey and Arthington, 2003). Macrophytes especially have been found to provide habitat and are well correlated with fish distribution in many Queensland rivers (Perna 1996; Pusey and Arthington, 2003). Macrophytes provide spawning substrate (Pusey *et al*., 2001), and refuge from predators (Webb, 2003) and high water velocities (Losee and Wetzel, 1993; Pusey and Arthington, 2003). Woody debris was found to be significantly correlated with species richness in the Normanby River and necessary for predator avoidance (Kennard, 1995). The Burdekin, however, is one of the most modified rivers in north Queensland and lacks many of the habitat characteristics found in the studies mentioned above.

Within the Burdekin floodplain, riparian areas have been cleared and light penetration has increased. This has led to an increase in macrophyte cover in many degraded sites. Macrophyte cover can become too high under these circumstances, and cause massive fluctuations in diel oxygen cycles, leading to fish kills (Chapter 3; Pusey and Arthington, 2003; Pearson *et al.,* 2003). However, not all sites experienced an increase in macrophyte cover as the water being pumped into the lagoons is high in sediment and light attenuation is high (ACTFR unpublished data), thus restricting the growth of native macrophytes in deeper water. More importantly, the proliferation of exotic floating weeds (*E. crassipes*) and emergent pasture grasses has greatly reduced submerged native macrophyte cover, by shading out the native emergent plants (Chapter 3).

In Chapter 2 the extent of infestation of exotic floating weeds and emergent pasture grasses was documented. These weed mats have created anoxic conditions in large reaches of the main two streams, which exclude a large number of species as well as inhibit migration between sites by increasing channel flow velocities and reducing oxygen concentrations (Bunn *et al*., 1998; Pusey and Arthington, 2003). The data show that, in the most degraded sites, only those species capable of aquatic surface respiration (ASR), or species with very high tolerance for low oxygen are present. Facultative air-breathers (*M. cyprinoides*) are present in sites that still have some open water. The weed mats have created migration barriers, which are indicated in the low species richness of the remnant sites that have good habitat and water quality, compared to those sites with connectivity to marine or other recruitment sources. The low richness is attributed to a lack of connection to a recruitment source, such as the river and saltwater reaches. Floodplain lagoons are likened to islands in that the fish species are often a selected sample of potential recruits from the source population in the parent stream (Kennard, 1995). Fish diversity is reliant on a matrix of habitat patches across the floodplain that may be interconnected at times of high water, allowing for migration and interaction (Townsend, 1989; Kennard, 1995). Within the Burdekin floodplain much of the diversity of floodplain habitats has been lost and the unpredictable nature of flooding and magnitude of flooding, coupled with highly degraded migration pathways, has led to lowered fish diversity across all habitats.

Although species distributions were shown to be correlated with habitat condition, some species may have been more abundant in particular sites due to biotic factors rather than condition alone. For example, *Hypseleotris* spp. was most abundant in degraded lagoon sites, and abundances of this species have been strongly correlated with predation and habitat cover (Kennard, 1995). In the remnant lagoons and riverine sites the abundance and richness of predators is higher, which may be the reason for the reduced abundance of the smaller species in these habitats. Large amounts of habitat cover in the highly degraded sites (despite the fact that it is dominated by an exotic weed), allows species such as *Hypeleotris spp., G. holbrooki*, and *Trichogaster trichopterus* to avoid predation (Webb, 2003); also, the abundance and diversity of predators is lower in these sites. Abundances of these small, tolerant species are much higher in degraded sites that lack predation pressure. In some of the remnant sites, such as Clay Hole, the same species were present as in the distributary streams but the abundance of these species was very low (such as *Hypseleotris* spp. in Fig. 4.2a).

Conversely, the species that appear to be associated with good sites or river sites may not be habitat-specific, but rather excluded from the poor sites. The fork-tailed catfish (*A. graeffei*) was once considered a pest by anglers in most lagoons on the floodplain (A. Darwen and A. Stennet, pers comm.). Throughout this study no individuals of this species were recorded in lagoon habitats, probably because poor water quality and migration barriers exclude the species. Alternatively, the sampling techniques were not suitable, but other research suggests this species is prone to capture in gill nets (Halliday, *et al.* 2001), so individuals should have been recorded in the main sampling program, had they been present.

4.4.2 Fish assemblages and water quality.

As was shown in Chapter 3 and Table 4.6, oxygen levels are generally low across the floodplain. High abundances of *Hypseleotris* spp. in conditions approaching hypoxia suggest that this taxon utilizes ASR or is capable of tolerating prolonged exposure to low oxygen levels (ACTFR, unpublished data). The other abundant and widely distributed species utilize ASR (*G. holbrooki*) or facultative air breathing (*M. cyprinoides* and *T. trichogaster*). *M. cyprinoides* gulp air into their air bladder, which is highly vascularised, and the oxygen diffuses into the blood stream from the air bladder (Geiger *et al.*, 2000). Also *M. cyprinoides* from the degraded sites had much more vascularised gas bladders than individuals collected from good quality sites (N. Flint pers. comm.). It was also noted that *M. cyprinoides* in the degraded sites almost always died in gill nets as they were restricted from the surface and so died of hypoxia (pers. obs.) . The Belontidae, to which *T. trichopterus* belongs, utilize the labyrinthine organ to obtain atmospheric oxygen in a similar manner (Munshi, 1976). The plotosid catfishes may be able to slow metabolism and utilize shallow microhabitats to avoid hypoxia (personal observation). The anguilid eels utilize their skin for respiration (Dehadrai and Tripathi, 1976). Although the average oxygen levels appear to be low in the sites, there are many microhabitats that fish will use to avoid hypoxia and this is apparent in the lack of a strong separation of low and high tolerant fish across the sites, and the presence of intolerant species such as *N. erebi* in sites with average saturation below 40%, which is at or below this species' tolerance limit. As well as excluding species, low oxygen levels can differentially affect different size classes (Hogan and Graham, 1994).

High levels of suspended solids have been found to reduce native species richness (Pusey *et al.*, 1993). Turbid waters may exclude visual predators such as *A. sclerolepis* and *T. chatareus*. Both of these species were rare or absent from samples in the turbid distributary streams. High turbidity indirectly affects fish assemblages by reducing light penetration and the growth of macrophytes, which are important to spawning, foraging and predator avoidance (Pusey *et al.*, 1993). The reduction in instream growth may affect overall primary production, with cascading effects on the rest of the food chain (Wood and Armitage, 1997). Turbidity appears to affect *A. percoides* indirectly. This species is dependent on macrophytes for foraging and predator avoidance (B. Pusey pers. comm.), and was very rare in distributary streams of high turbidity and reduced macrophyte cover. The high turbidity of the distributary streams also appears to directly benefit the exotic floating weeds (Chapter 3). Turbidity reduces native macrophyte competition and supplies continuous input of nutrients for rapid growth of floating weed mats. These weed mats create anoxic conditions and thereby present a chemical barrier to fish migration, contributing to low fish diversity across the floodplain.

4.4.3 Connectivity and fish assemblage structure

Total species richness and abundance was less in the ten main sites compared with sites across the floodplain (even though the main sites were repeat sampled), but there was an increase in the abundance of exotic species at these sites. Eight of the ten main sites were highly impacted with low habitat values and input of irrigation water. The two remnant sites had high habitat values and had no irrigation flows. The species richness and abundance was higher in these remnant sites but still lower than in sites outside of the distribution systems. The main reason is that for species richness to be maintained or increased there must be connection to a recruitment source, but in Sheep Station Creek and Warren's Gully migration pathways have all but disappeared except during times of major flooding (the last bank overflow event was in 1991). There has been recent documentation of migration by amphidromous species in Sheep Station Ck, such as *Chanos chanos,* recorded in Dick's Bank Lagoon (ACTFR, 1994). This record was made in 1993, just under two years after a major flow event in which the Burdekin overflowed into Sheep Station Ck, removing the weed mats. The result was, therefore, a recruitment of these amphidromous species; however, by 2000 when the present project began, the weed mats had re-established and closed off the migration pathways and most of these species again became locally extinct.

The most economically important species in the area, *Lates calcarifer*, is extinct in Sheep Station Ck, due largely to the presence of weed infestations throughout the lower reaches. This species was found in Warren's Gully sites, but its presence in Saltwater Creek and Munro's Lagoon is due to stocking (Burrows, 2003; A. Stennet, pers. comm.). Only the fish recorded at Inkerman could have been wild stock as no stocking occurs at this lagoon. It is suspected that juvenile *L. calcarifer* migrating from the saltwater reaches cannot survive the poor water quality in the lower reaches sufficiently long enough to colonize the open water lagoons upstream, hence their absence in Sheep Station Ck (which was not stocked previous to this research). However *Megalops cyprinoides* were found and are a diadromous species which due to surface respiration may be able to overcome the chemical barriers it is also suspected that this species may be introduced to the creek as larvae through the pumps within the river. It also must be noted that many *L. calcarifer* were recorded in the Burdekin River and Barratta Ck sites, but this may be the result of stocking (Table 4.19).

Table 4.19 Stocking data from Burdekin Fish Stocking Association for *L. calcarifer* stocking locations and rates.

Location	Date	Number	Size (mm)
Groper Ck Landing	26-Mar-95	3000	50
Allen Rd, Barrattas	22-Apr-99	2000	?
Allen Rd, Barrattas	26-Jan-00	2000	?
Collinsons Lagoon	26_Jan-00	8500	?
Barramundi Ck	8-Nov-00	7000	30
Launders Ck	8-Nov-00	6125	30
Saltwater Ck	8-Nov-00	7500	30
Barrattas Ck	8-Nov-00	9375	30
Collinsons Lagoon	17-Mar-01	12	30
Hutchinsons Lagoon	6-Apr-01	500	25
Iyah Ck	6-Apr-01	410	25
Millaroo/Burdekin River	6-Apr-01	3636	25
Collinsons Lagoon	25-Oct-01	2400	30
"3 places in Burdekin River"	31-Oct-01	50000	25
Horseshoe Lagoon	29-Mar-02	8000	?
TOTAL		**110,458**	

Many purely freshwater species have restricted distributions on the floodplain as well. For example *A. percoides* was found only in Dick's Bank Lagoon and Castelanelli's Lagoon. Data suggest that this species recruited back into Dick's Bank only after a flow event allowed migration from Castelanelli's just upstream of Dick's Bank (Fig. 2.5). Castelanelli's Lagoon is a remnant site in the Sheep Station catchment that receives no irrigation water and has high habitat values. Both *A. percoides* and *N. erebi* were abundant in this site. It is possible that this site acts as both refuge and source for these intolerant species within this catchment.

4.4.3 Irrigation impacts on fish assemblages in Sheep Station and Warren's Gully Creeks.

Two exotic species, *G. holbrooki* and *T. trichopterus* (Three-spot gourami), were found in Sheep Station Ck The gourami was restricted to Sheep Station Ck and was most likely the result of direct release of aquarium fish. *G. holbrooki* were located throughout the floodplain. Mats of exotic weeds appear to aid exotic fish abundance and distribution (Pusey and Arthington, 1993; Webb, 2003). The complex cover created by the weed mats is effectively used by exotic fish for predator avoidance (Webb, 2003). Additionally, the weed mats create anoxic conditions, which exclude many native fish, but may favour tolerant exotic fish.

Clearly, the most significant impact within these distributary streams is the presence of weed mats, which create migration barriers and proliferate from the modified conditions associated with supplemental flows (Chapter 3). Migration barriers in lotic systems are well documented as reducing fish diversity (Cotterell, 1998), especially in tropical northern Australia, where a large component of the diversity is derived from the saltwater reaches (Beumer, 1980). To restore fish diversity, removal of physical or chemical barriers is required. This is the subject of Chapter 5.

Chapter 5 Effects of weed removal on fish communities and water quality at Payard's Lagoon

5.1 Introduction and aims

Fish communities and habitats across the Burdekin floodplain have been highly modified and degraded. Loss of connectivity, weed infestation, altered hydrology and poor water quality have greatly reduced fish diversity and abundance. The previous chapters have described the present condition of habitats, water quality and fish communities across a variety of sites throughout the floodplain and showed that weed infestation and habitat condition are the most important drivers of fish community structure. During this project, the Burdekin Shire Council and the Department of Primary Industries conducted mechanical removal of mats of *Eichhornia crassipes*. This provided the opportunity to examine the impact, to both the fish community and water quality, of large-scale weed control in tropical floodplain lagoons. The focus of this chapter is on the effects of restoration work at Payard's Lagoon, which was first cleared in August 2000.

Eichhornia crassipes has been labeled the worst aquatic weed on the planet (Willoughby *et al.*, 1993; Julien *et al.*, 2001), especially in the tropics where nutrients are said to be rarely limited (Scheffer *et al.*, 2003). Infestations of this plant hinder boat traffic, fishing activities, hydroelectric power generation, and access to water by local communities (Julien *et al.* 2001; Masifwa, *et al.*, 2001; Scheffer *et al.*, 2003). *E. crassipes* infestations also increase the risk of flood damage and can increase water loss through evapo-transpiration by 3.5 times that of evaporation (Ogutu-Ohwayo *et al.*, 1997). Many recent studies have also found that *E. crassipes* increases the abundance of major human and animal disease vectors by providing protective habitat from predators (Ogutu, *et al.* 1997; Masifwa *et al.* 2001; Julien, *et al.* 2001). *E. crassipes* has been found to create "dark anoxic conditions under thick floating-plant cover, leaving little opportunity for animal or plant life" (Scheffer *et al.* 2003). However, Masifwa *et al.*, (2001) showed that macroinvertebrate richness and abundance increase under the outer edges of *E. crassipes*, and that these habitats provide protection from predation of fish species thought to be extinct; nevertheless, this benefit is not thought to outweigh the negative aspects of infestation (Masifwa *et al.*, 2001).

In the Burdekin distributary streams, *E. crassipes* is the major driver of poor fish habitat and low fish diversity (Chapter 2, 3 and 4). Due to the high variability in flood magnitude in the distributary streams and the development of the irrigation areas, which maintain water levels, *E.*

crassipes has become the dominant macrophyte in most of the lagoons across the delta. Due to the ideal conditions of sunny warm conditions with continuous nutrient input from dam water (and to a lesser degree irrigation runoff), *E. crassipes* can cover whole lagoons, creating anoxic conditions and eliminating most fish species (Chapter 4). Recent initiatives to rehabilitate the distribution streams in the Burdekin region have identified the removal of *E. crassipes* as being of high priority.

5.1.1 Aims

The main aim of this chapter is to identify changes in fish assemblages and habitat condition resulting from weed removal, including analysis of the medium-term recovery of fish assemblages and water quality. Given the impacts of *E. crassipes* described above, it was hypothesized that, after weed removal:

- both species richness and abundance of the fish assemblage would increase;
- abundances of small-bodied fish species that were present before the weed removal would show a decrease in the medium term due to the re-invasion of predators; and
- oxygen content and production would increase, and would be stable over the medium term.

5.2 Methods

5.2.1 The weed removal project

A mechanical weed harvester was used to clear *E. crassipes* weed infestations in August 2000. The weed removal took 20 days to complete and 90% of the infestation was removed. Previous removal of weeds occurred as a result of flooding in 1991 (A. Darwen, pers. com.). Prior to weed removal, only one fish sample had been conducted, which was restricted to the open water channel at the inlet. During this time, researchers from CSIRO were conducting work on runoff in sugar cane areas and water quality monitoring was being carried out at the inlet and outlet of Payard’s. That project aimed to examine the cost and effectiveness of the weed removal process and monitor any environmental changes.

5.2.2 Fish surveys

Surveys were conducted at three-month intervals. An added survey was conducted on the August 2, 2000, while weed removal was being conducted. One survey was conducted prior to weed removal; second and third surveys were conducted during and after initial weed removal respectively. These three surveys are grouped in the analyses as “pre-weed removal” as no flow events occurred which would have allowed recruitment. Six surveys were conducted after the first flow following weed removal and are referred to as “post-weed removal”. For all fish sampled, measurements of total length (TL) or length at caudal fork (LCF), were taken. If fewer than 20 individuals were caught, all were measured. If more than 20 were caught, a sub-sample of 20 were measured. Dip net, gill net and seine net sampling techniques were used as in Chapter 4. This was the only site with seine net access. The seine net (20 m x 2 m with a 5 mm stretch knotless mesh) was pulled out from the bank in the top channel of the lagoon. A wide sweep was made into the middle of the channel and back to the bank. All fish were sorted and put on ice for identification and measurement.

5.2.3 Water quality

Physico-chemical water quality was measured as per Chapter 3. The first three samples were taken manually (spot measurements). After these surveys all data were logged as per Chapter 3. In conjunction with this survey work the same parameters were logged, hourly, at the inlet and outlet by CSIRO. The Hydrolab® was placed in a stainless steel tube at the inlet and outlet of the lagoon at a depth of 60 cm. Logging began on August 1, 2000, and continued until August 30, 2001, enabling description of water quality before and after weed removal.

5.2.4 Statistical analysis

Temporal variation in fish assemblages were examined by SSHMDS ordination in PATN using Log_{10} transformed species’ abundances to normalize the data, as described in Chapter 4. To compare Payard’s Lagoon with three other lagoons on the creek to examine impacts of weed removal, another SSHMDS was run using Log_{10} transformed abundance data from dip and gill net samples, as seine nets were only used at Payard’s Lagoon. Temporal variations in fish assemblages were analyzed using Independent T-Test in SPSS on pre- and post weed clearing samples. Assemblage descriptors used were total abundance (log_{10} transformed), species richness, evenness (arcsin transformed), proportion of exotics to total abundance (arcsin transformed) and proportion of *Gambusia* to total species abundance (arcsin transformed). Data were analyzed separately by sampling technique or pooled from all methods, as appropriate.

5.3 Results

5.3.1 Fish assemblage structure in Payard's Lagoon

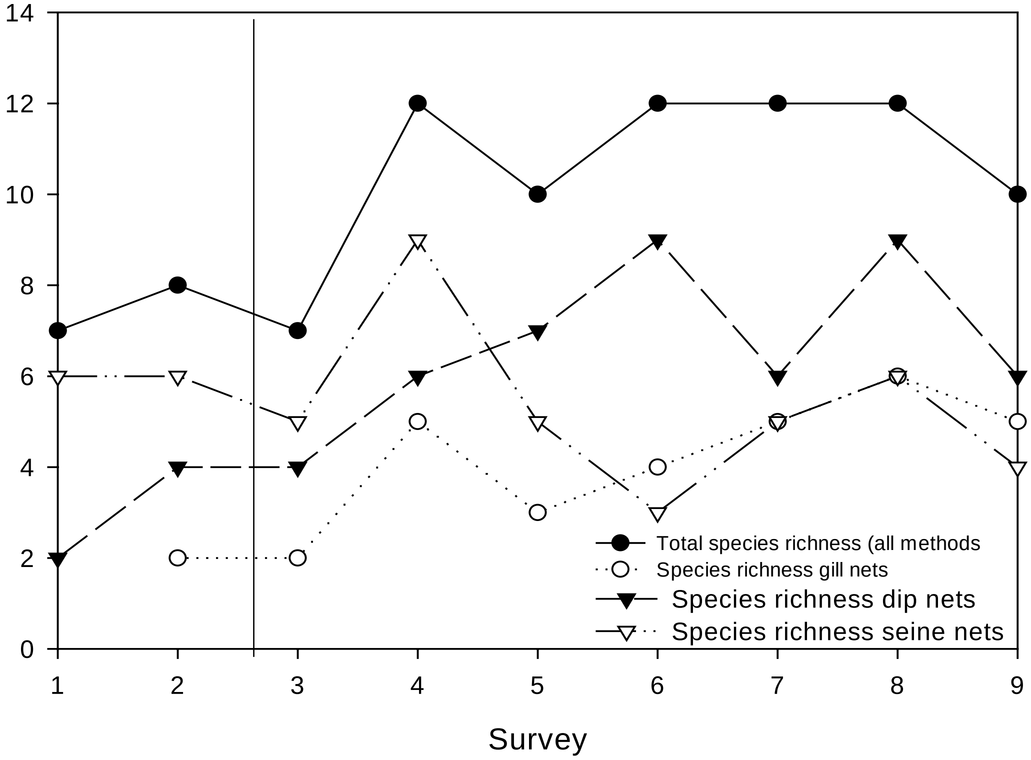
The fish assemblage structure of the lagoon showed significant temporal changes in abundance and richness. As with the other sites, *Hypseleotris* spp. (39% of total number of fish collected) was the most abundant taxon followed by *A. agrammus* (27%) and *G. holbrooki* (19%) (Table 5.1). These three taxa were the only taxa recorded on every occasion. Two species were recorded only once – *Trichogaster trichopterus* (survey 3 pre-clearing) and *T. chatareus* (on the last survey, post-clearing). Six species were recorded only after *E. crassipes* was removed (Table 5.1). Dip nets caught the most fish – 67% - with seine nets catching 31% and gill nets only 2% of the total. Gill nets caught three species that were not recorded in the other methods. The total abundance increased rapidly after weed removal with a subsequent gradual decrease (Table 5.1). Species richness increased overall, fluctuating between 10 and 12 species, although new species were recorded during each survey after weed removal (Fig. 5.1 and Table 5.1). The seine net samples were the only method to show a decrease in richness (Fig. 5.1).

The 2-axis ordination (14% stress) of species abundance data shows clear separation between weeded and non-weeded samples on both axes (Fig. 5.2). Axis 1 (58% of the variation) represents a temporal gradient from weeded (positive) to cleared (negative) assemblage structure. There appears to be a continuous transition from right to left on this axis, which may represent the increase in species richness recorded after weed removal (see Table 5.1). Also, significant movement of sites from left to right on this axis occurs after survey 4, when new species started to recruit into the lagoon (Table 5.1 and Fig. 5.1). Axis 2 (42% of the variation) represents a habitat gradient, showing the gradual shift from channel to lagoon habitat. In the three pre-clearing samples, total abundance and species richness were dominated by seine samples in the inlet channel (Table 5.1). After weed removal the fish in the channel recruited into the lagoon. Surveys 4 and 5 show significant increases in dip net catches and species richness, which may represent this shift from channel to lagoon and rapid reproduction in the newly available habitat.

Table 5.1 Abundances of fishes recorded by each method in Payard's Lagoon before and after weed removal (Treatment 1 = pre-weed removal/recruitment and 2 = post-weed removal/recruitment).

Treatment	1	1	1	2	2	2	2	2	2	Total	Percent
Date	09-06-00	02-08-00	24-08-00	11-12-00	05-03-01	22-06-01	22-10-01	03-04-02	27-08-02	caught	of total
Dip net total abundance	23	108	257	1277	1757	495	336	102	48		
(species richness)	(2)	(4)	(4)	(6)	(7)	(9)	(6)	(8)	(6)	4403	67
Gill total abundance		7	3	24	21	36	7	14	14		
(species richness)		(2)	(2)	(5)	(3)	(4)	(5)	(6)	(5)	126	2
Seine total abundance	150	554	401	411	195	24	119	125	39		
(species richness)	(6)	(6)	(5)	(9)	(5)	(3)	(5)	(6)	(4)	2018	31
All methods total											
abundance	173	669	661	1712	1973	555	462	241	101	6547	100
(species richness)	(7)	(8)	(8)	(12)	(10)	(12)	(12)	(12)	(10)		
A. agrammus	134	534	281	483	207	17	87	44	4	1791	27.4
G. holbrooki	23	93	292	341	170	142	170	28	2	1261	19.3
Hypseleotris spp.	1	25	7	773	1352	311	78	25	22	2594	39.6
C. stercusmuscarum	8		75	57	197	38	186	47	32	640	9.8
L. unicolor	5	4		1	2	1	1	7		21	0.3
Neosilurus ater	1	6	1	1	2	11		1	1	24	0.4
M. adsperssa	1	4	2	9	7	4	10			37	0.6
M. cyprinoides		1	2	16	9	11	2	2	3	46	0.7
M. spl. splendida		2		7			3	6		18	0.3
T. trichopterus			1							1	0.0
G. margaretacea				1						1	0.0
N. hyrtlii				4		2	1	3	1	11	0.2
P. rendahli				19	2	4	6	12		43	0.7
Nematalosa erebi					25	13	6	58	22	124	1.9
G. aprion						1	2	10	16	29	0.4
T. chatareus									1	1	0.0

Figure 5.1 Species richness for all methods and for each method separately over nine surveys in Payard's Lagoon. The solid vertical line represents weed removal.



Comparison of samples before and after reconnection of lagoons and waterways is presented in Table 5.2. Total species richness increased significantly after weed removal (Fig. 5.1). Both species richness and abundance in gill nets significantly increased after weed removal (Fig. 5.3). Species richness in dip nets significantly increased but abundance did not, which is likely the result of highly variable abundances in dip nets (Fig. 5.4). The proportion of *Gambusia* in the dip nets decreased significantly after weed removal (Fig. 5.5). The evenness of dip net samples significantly increased after weed removal, showing a decrease in dominance of one or two species (*Hypseleotris* spp. and *Gambusia*).

Increases in species richness at Payard's after weed removal were significant. A comparison of temporal changes in species richness at Payard's and Kelly's Lagoons (the remnant control for Sheep Station Ck) was made to validate that changes observed in Payard's were related to weed removal and not repeat sampling (Fig. 5.6). Species richness increased significantly after weed removal in Payard's, whereas no significant increases were recorded in Kelly's (Fig. 5.6). This suggests that weed removal allowed for significant recruitment of fish back into Payard's after weed removal and a wet season flow. Further analysis was conducted using a SSHMDS ordination comparing species abundances at 4 sites along Sheep Station Ck

Figure 5.2 Semi-Strong Hybrid Multidimensional Scaling ordination plot of species by sites matrix using total species abundance, $\log_{10}$ transformed, for nine surveys using all methods (gill, dip and seine nets) at Payard's lagoon. Inset reflects correlation between species abundances and position in ordination space

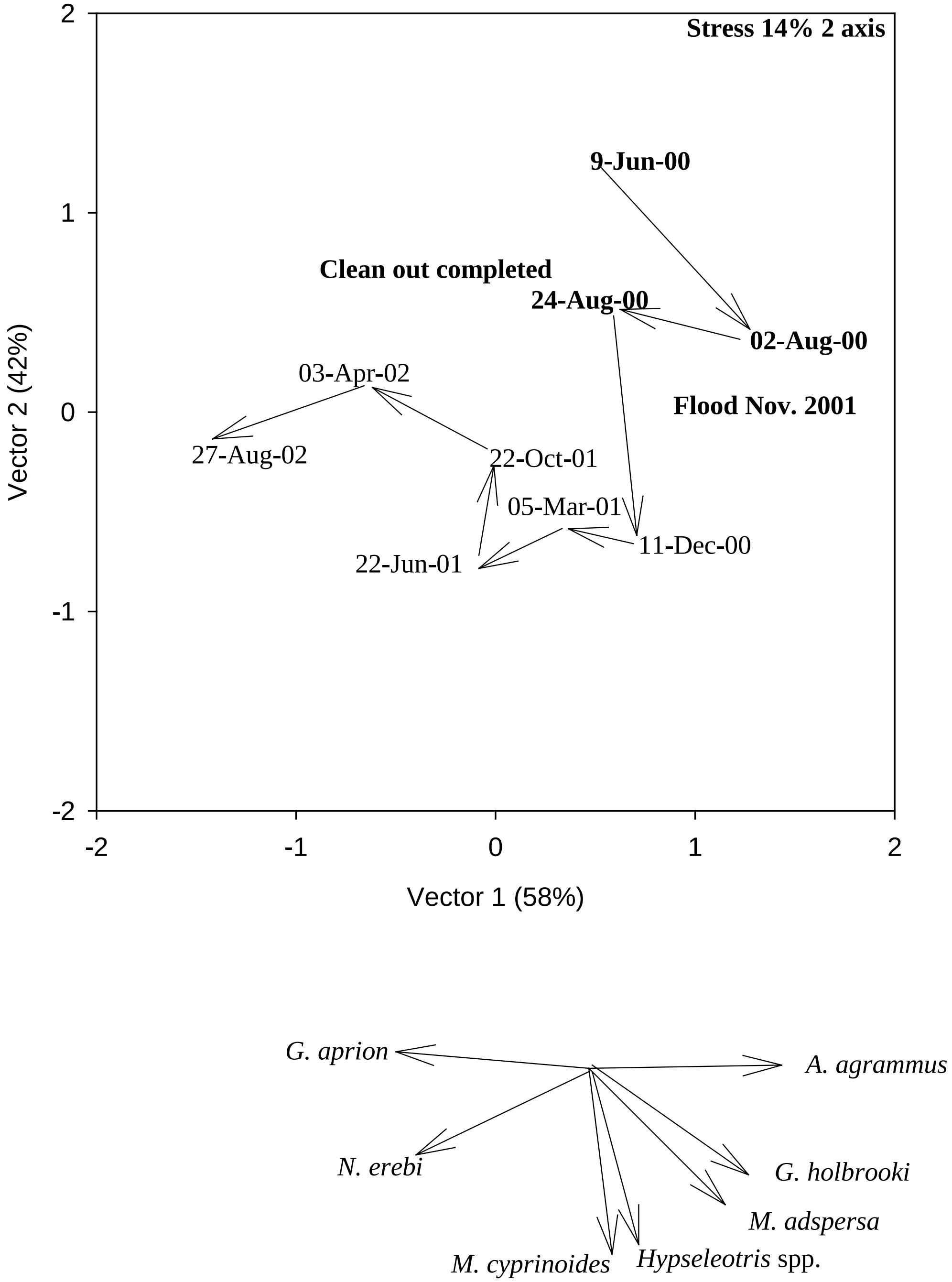


Table 5.2 Results of independent t-test on summary data for Payard's Lagoon comparing assemblage level descriptors prior to (treatment 1) and after (treatment 2) treatment. Showing only those assemblage descriptors with significant differences between treatments.

Parameter	Treatment Mean (SE)		t value	p
	1	2		
N	3	6		
Total species richness	7.33 (0.33)	11.33 0.42)	-6.845	0.000
Gill net species richness	2 (0.00)	4.67 (1.03)	-4.567	0.004
Gill net total abundance	5.00 (2.00)	19.33 (4.13)	-2.829	0.030
Dip net species richness	3.33 (0.67)	7.17 (0.601)	-4.147	0.004
Proportion of *Gambusia* in dip nets	0.89 (0.063)	0.229 (0.67)	6.718	0.008
Dip net evenness	0.26 (0.103)	0.65 (0.079)	-2.587	0.036

Figure 5.3 Total species richness and abundance for gill net samples over eight surveys in Payard's Lagoon. The solid vertical line represents weed removal.

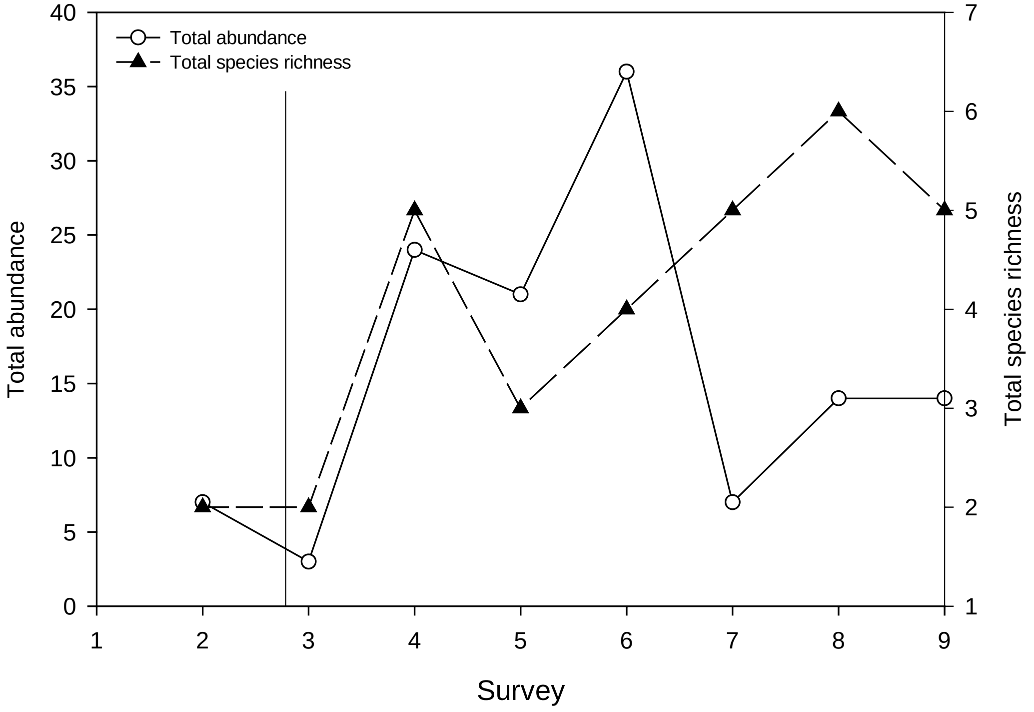
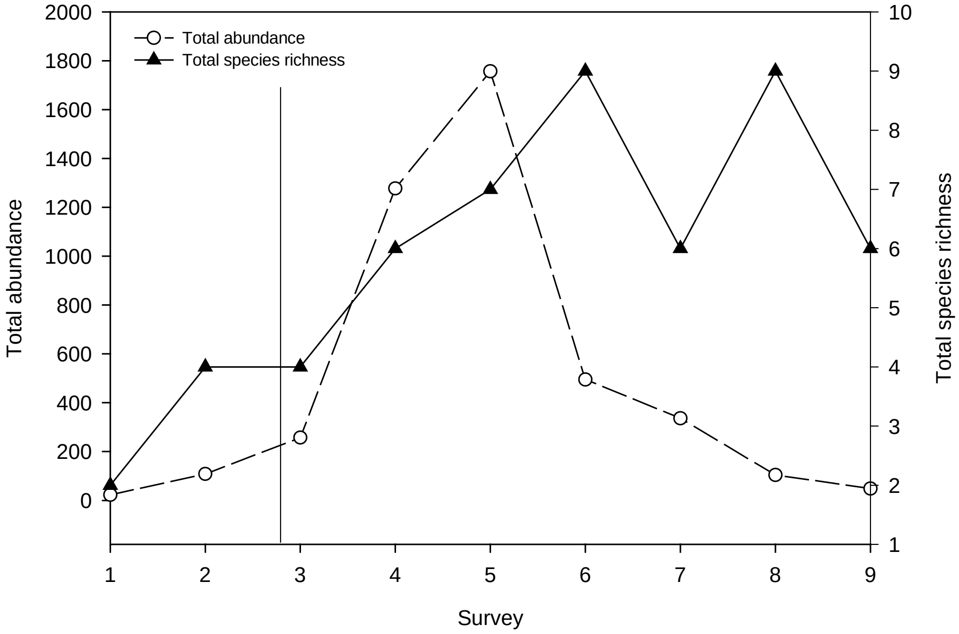


Figure 5.4 Total species richness and abundance for dip net samples over nine surveys in Payard's Lagoon. The solid vertical line represents weed removal.



The ordination plot shows all Payard's post-clearing sites grouping together in the lower right corner (Fig. 5.7). Kelly's survey 8 and Dick's Bank survey 7 both group there as well. Axis 1 accounts for 43% of the variance and reflects a gradient in condition, with sites of poor condition grouping to the left and sites of good condition grouping to the right (Fig. 5.7). The species that most greatly influence this pattern are *G. aprion* and *N. erebi* (correlated positively on Axis 1) and *N. ater, G. holbrooki* and *Hypseleotris* spp. (correlated negatively with Axis 1). Axis 2 (57%) reflects a temporal gradient from pre-clearing (positive) and post-clearing (negative). The pre-clearing surveys at Payard's (survey 1 & 2) and Gorizia's (survey 6) group in the very upper limits of this axis (Fig. 5.7). These surveys were characterized by low diversity (2-4 species) and low abundances. After weed removal (or in the case of sites that did not have weeds removed, repeat samples) species numbers increased and moved down Axis 2 on the plot. Kelly's, Dick's Bank and Gorizia's Lagoons group together with a few exceptions, showing that weed removal at Payard's had a substantial effect on fish abundances that were significantly different from sites with no weed removal.

Figure 5.5 Proportion of *Gambusia* to the total fish abundance in dip net samples over nine surveys in Payard's Lagoon

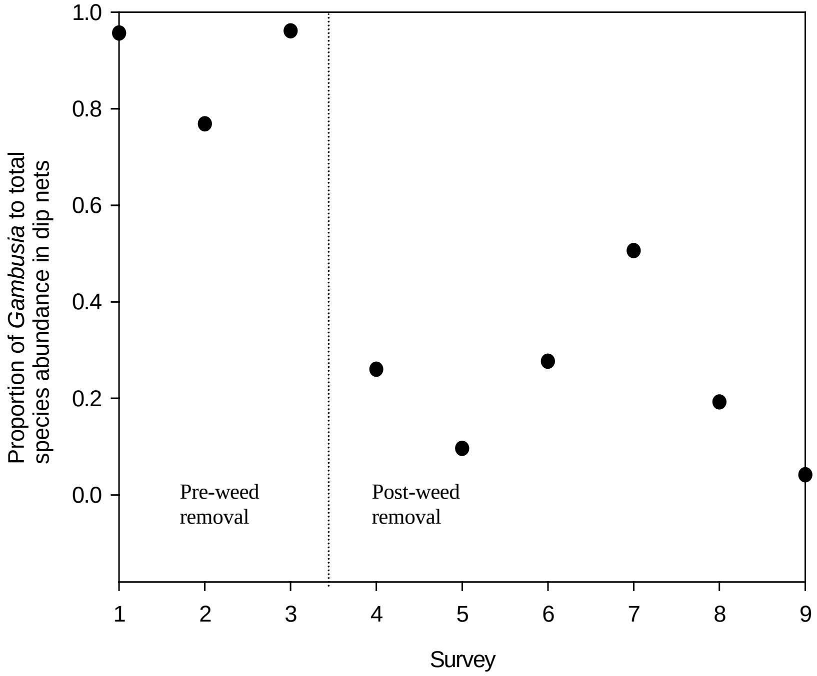


Figure 5.6 Comparison of total species richness before and after weed removal in Payard's Lagoon and Kelly's Lagoon (Sheep Station remnant site). NB. Data from dip and gill nets only, as seine nets were not used at Kelly's.

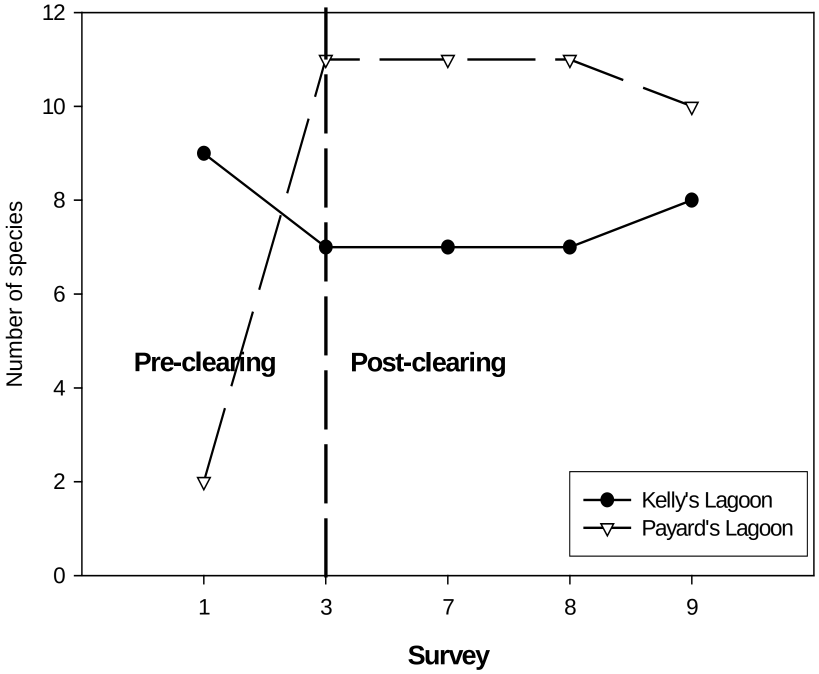
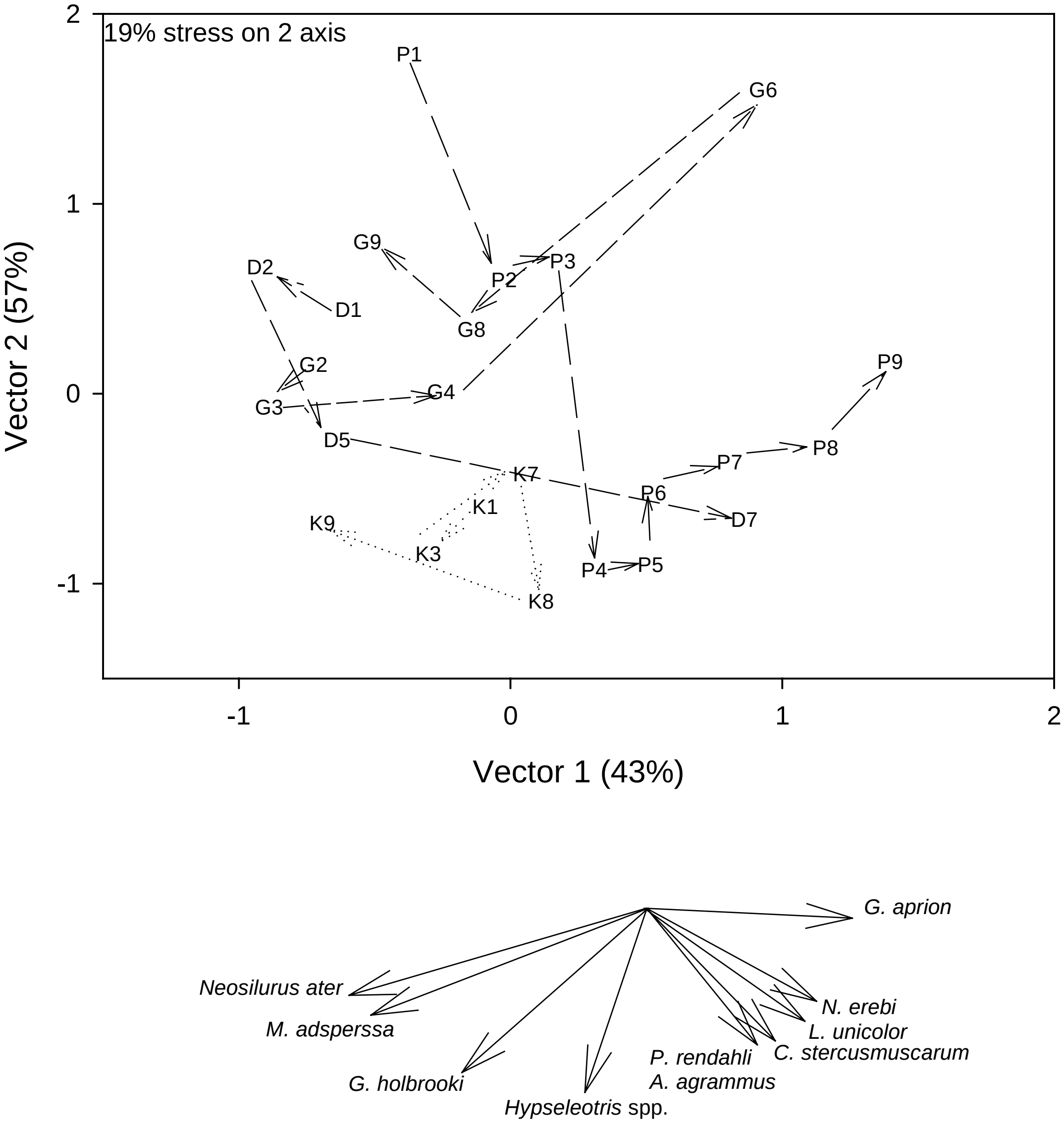


Figure 5.7 Semi-Strong Hybrid Multidimensional Scaling ordination plot of species by sites matrix using total species abundance, log10 transformed from 4 lagoons along Sheep Station Ck. Inset reflects correlation between species abundances and position in ordination space. **Data is for gill and dip net samples from Dick's Bank Lagoon (D), Gorizia's Lagoon (G), Kelly's Lagoon (K) and Payard's Lagoon (P). Survey number is shown with site identification.**



This is also illustrated by Dick's Bank survey 7, which groups with the post-clearing Payard's samples (Fig. 5.7). This sample was taken after the channels and lagoons upstream of the site were cleared and a flow event connected Dick's Bank with Castelanelli's Lagoon, a significant refuge and remnant.

5.3.2 Temporal variation in assemblage structure by techniques

Each of the sampling techniques was examined for temporal variations in catch. Dip nets caught the most fish with a significant increase in species richness after weed removal. The total abundance for 13 species was tested for differences before and after weed removal (Table 5.3). Three species – *Craterocephalus stercusmuscarum*, *Hypseleotris* spp. and *Porochilus rendahli* – significantly increased after weed removal (Table 5.3). The *Hypseleotris* spp. had a very high S.E. due to the very high catches in surveys 4 and 5. Neither *C. stercusmuscarum* nor *P. rendahli* were recorded pre-clearing.

Eleven species were recorded in seine nets and their abundances were tested for temporal variation. There was a significant decrease in *Ambassis agrammus* after weed removal (Table 5.4), whereas *Nematalosa erebi* significantly increased with none recorded before weed removal (Table 5.4). No significant differences in abundances were recorded for nine species caught in gill nets.

Total abundance for all methods and all species was also analyzed. Out of 16 species, five showed significant differences in abundance after weed removal. Three species – *N. erebi, P. rendahli* and *Neosilurus hyrtlii* – significantly increased after weed removal, with none recorded pre-clearing (Table 5.5). Both *Hypseleotris* spp. and *Megalops cyprinoides* significantly increased after weed removal (Table 5.5). Once again *Hypseleotris* spp. had a very high S.E. due to the high catch during surveys 4 and 5 (Table 5.5). This shows boom and bust characteristics where over time community diversity and abundance could be expected to stabilize if weeds are keep out.

5.3.3 Contribution of *Gambusia* to assemblage structure

The introduced species *Gambusia holbrooki* contributed significantly to the total abundance of dip net samples over the study. Differences in proportion of *G. holbrooki* were compared before and after weed removal. The proportion of *G. holbrooki* to total abundance of dip net samples was significantly lower after weed removal ($t_{3,6}=6.717$; $p<0.001$), with a mean (and S.E.) before weed removal of 0.90 (0.06), and after removal of weed 0.23 (0.07). There was a dramatic drop in the proportion of total abundance by *G. holbrooki* directly after weed removal and the trend downward appeared to continue (Fig. 5.5). There was a slight increase in survey 7, but there were no clear links to recruitment (Fig 5.5 and 5.8).

5.3.4 Recruitment of species, invasion and direct recruitment

Evidence of recruitment was found in length-frequency data. For this section only those species that were found to have significantly different mean abundances before and after weed removal are presented (Table 5.5), except *G. aprion*, which is included as it may have important effects on assemblage structure (Webb, 2003). The most noticeable recovery after weed removal was that of *N. erebi*. This species had not been seen in Payard's for over seven years (A. Darwen, pers. comm.). *N. erebi* re-invaded after the 2000/01 wet season. They were recorded first on the March 5, 2001. A wide range of size classes were recorded (Fig. 5.9), all of which would have been by invasion rather than direct recruitment, as only larger size classes were caught and the species had not been present long enough to spawn. High numbers of small - size - class fish of this species were recorded after October of 2001, over a year after weed removal. The mean size of fish caught after October 2001 dropped from just above 100 mm LCF to 50 mm LCF or smaller (Fig. 5.10). In the March 2002 survey a large number of 50-60 mm individuals were recorded suggesting strong summer recruitment (Fig. 5.9 and 5.10). The wet season was below average rainfall with little overflow from other areas but the post-wet season sample on April 3, 2002 showed major recruitment of fish of <50mm and a high frequency of smaller size classes (Fig. 5.9 and 5.10). This indicates that after the initial re-invasion, *N. erebi* found suitable conditions to breed locally.

Table 5.3 Results of independent t-test, showing only species with significantly different means between treatments, on total species abundance in dip nets. Treatments were 1=pre-clean out and 2= post clean.

Species	Treatment Mean (SE)		t value	p
	1	2		
N	3	6		
Craterocephalus stercusmuscarum	0 (0)	57 (24.27)	-6.450	0.000
Hypseleotris spp.	9.33 (6.56)	423.17 (218.11)	-2.702	0.031
Porochilus rendahli	0 (0)	6.4 (1.91)	-2.773	0.028

Table 5.4 Results of independent t-test, showing only species with significantly different means between treatments, on total species abundance (Log10 transformed) in seine nets over nine samples. Treatments were 1=pre-clean out and 2= post clean.

Species	Treatment Mean (SE)		t value	p
	1	2		
N	3	6		
A. agrammus	314.65 (116.35)	109.80 (66.63)	2.590[b]	0.039
N. erebi	0 (0)	19.25 (8.61)	-2.835 [b]	0.036

Table 5.5 Results of independent t-test, showing only species with significantly different means between treatments, on total species abundance for all methods. Treatments were 1=pre-clean out and 2= post clean.

Species	Treatment Mean (SE)		t value	p
	1	2		
N	3	6		
Hypseleotris spp.	11 (7.21)	427 (219)	-2.665	0.032
Megalops cyprinoides	1.5 (0.500)	7.17 (2.36)	-2.528	0.039
Nematalosa erebi	0 (0)	24.8 (8.95)	-2.963	0.021
Porochilus rendahli	0 (0)	8.6 (3.09)	-2.657	0.033
Neosilurus hyrtlii	0 (0)	2.2 (0.58)	-2.642	0.012[b]

[b] =Non-equal variance test.

Figure 5.8 Length frequencies (TL mm) for *Gambusia holbrooki* over nine surveys in Payard's Lagoon

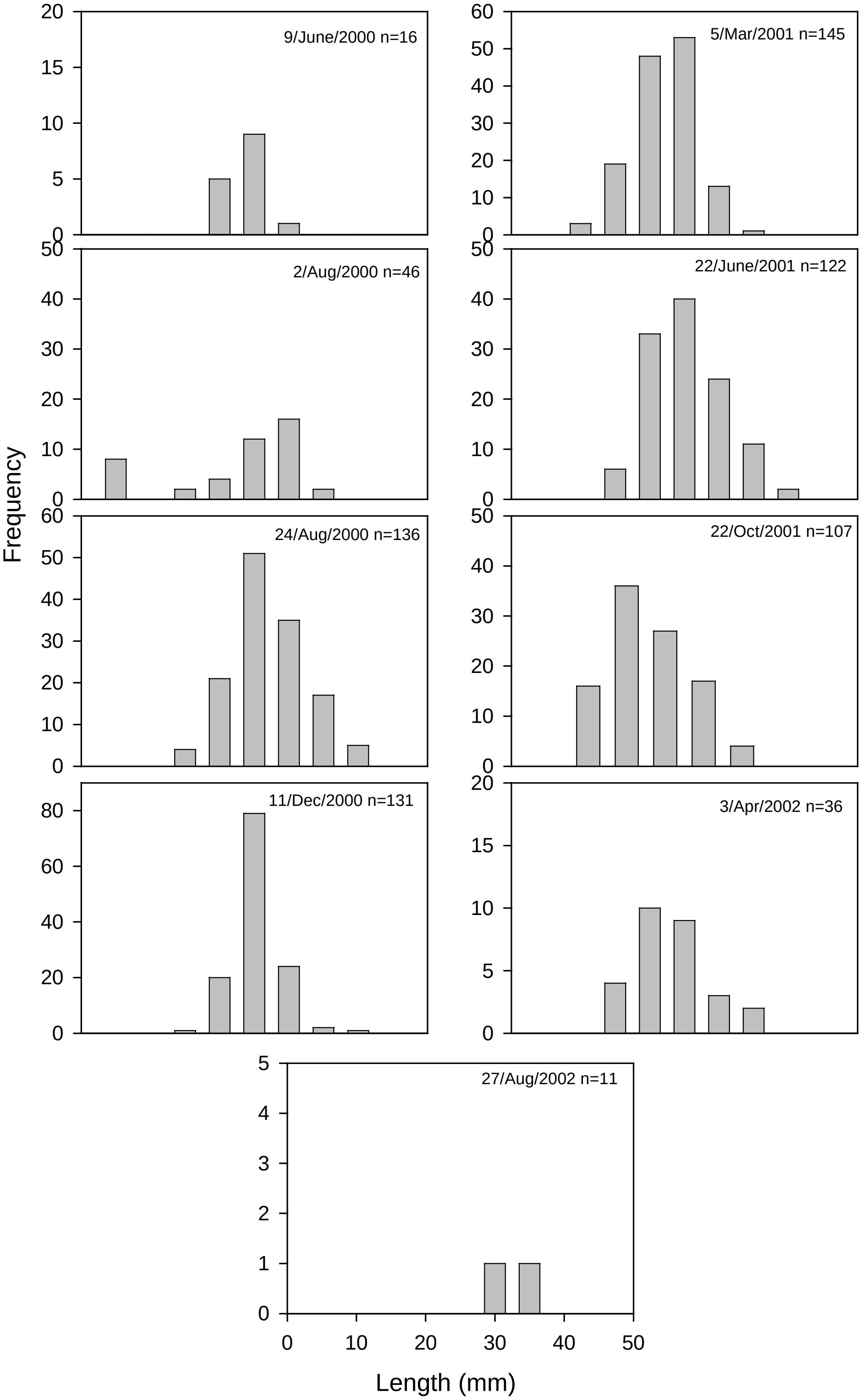


Figure 5.9 Length frequency (LCF mm) for *Nematalosa erebi* over five surveys in Payard's Lagoon.

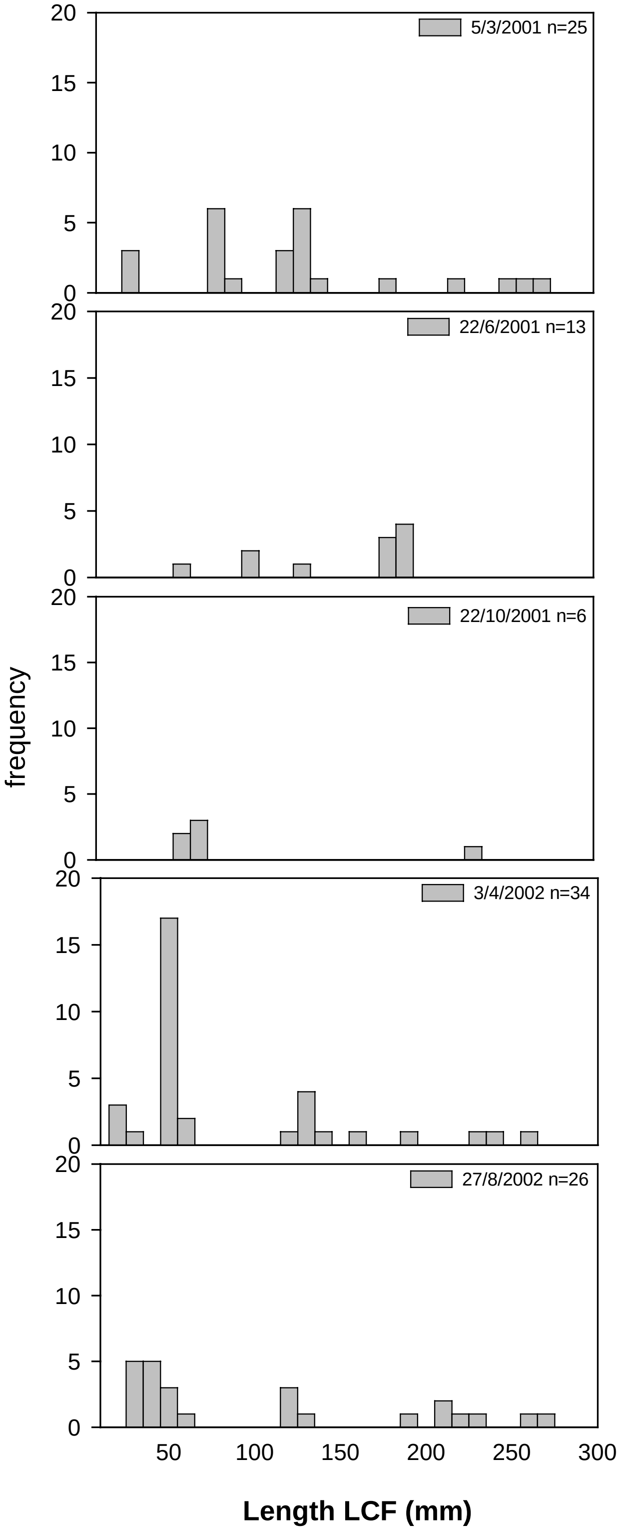
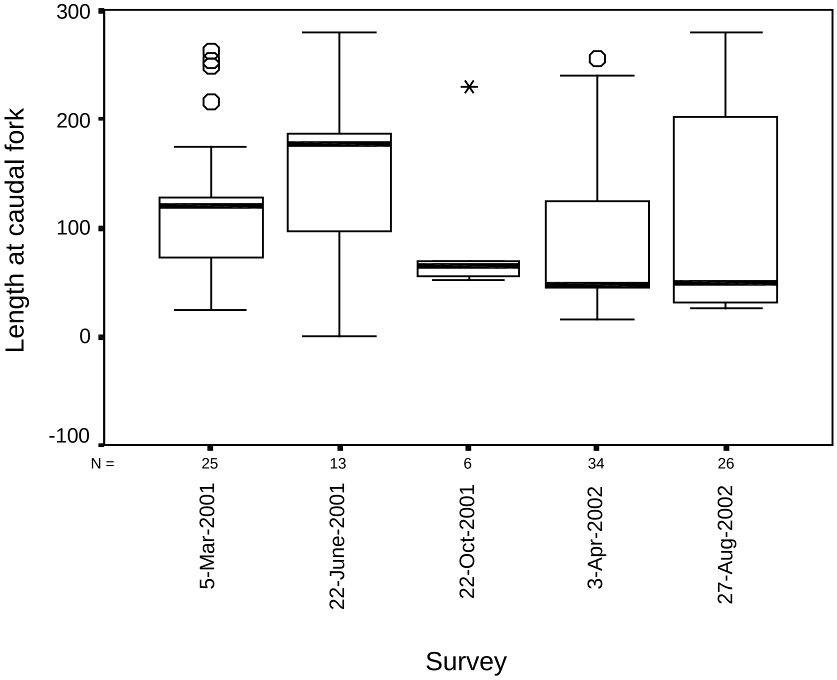


Figure 5.10 Boxplot graph of *N. erebi* lengths (LCF mm) over five surveys. See Figure 4.2 for explanation of boxplot symbols. Only five surveys presented as *N. erebi* only invaded after weed removal and subsequent flooding.



Megalops cyprinoides showed little evidence of recruitment, although there is some evidence of an increase in abundance of fish in the 210-220 mm range after the 2000/01 wet season (Fig. 5.11). The main point of interest for this species was the growth found between March 5, 2001 and June 22, 2001: it appears that *M. cyprinoides* had a high growth rate in the site after weeds were removed (Fig. 5.11). *Hypseleotris* spp. showed evidence of local recruitment after wet season flows (Fig. 5.12). There appeared to be small size classes present throughout all seasons (Fig. 5.12), but it is difficult to draw conclusions, as there was large variation in catch over the study period. *Glossamia aprion* first appeared in the 6th survey (June 22, 2001) and very few were recorded until the last two surveys (Table 5.1). There was very clear recruitment between April and August 2002 (Fig. 5.13). This species appeared to become well established and to be actively recruiting locally. This species may (have) influence (on) the assemblage structure, especially of the smaller - bodied species.

Figure 5.11 Length frequency (LCF mm) for *Megalops cyprinoides*, over eight surveys at Payard's Lagoon. **Only eight surveys presented as no open water was present for gill net samples during the first survey.**

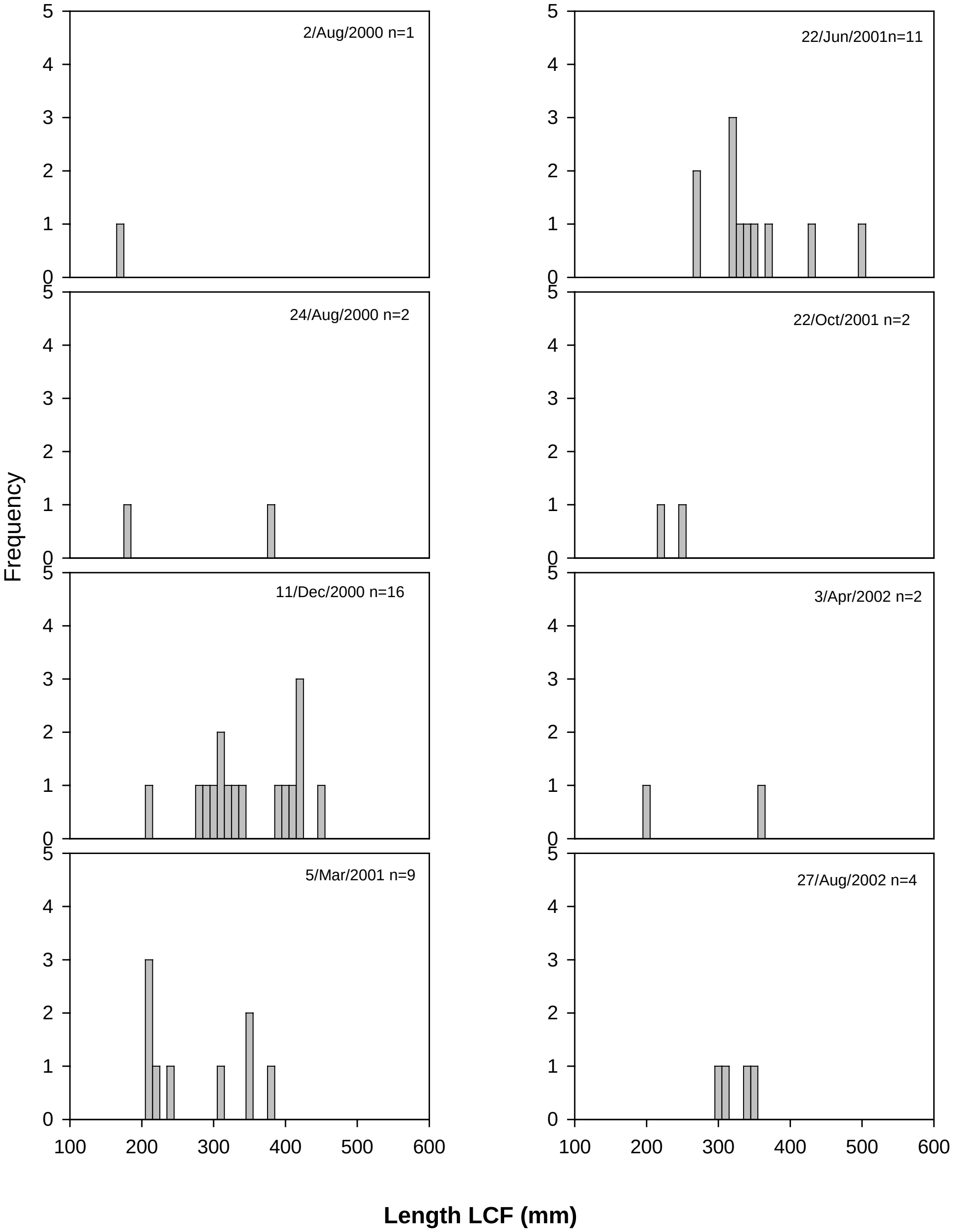


Figure 5.12 Length frequency (TL mm) for *Hypseleotris* spp. over nine surveys at Payard's Lagoon. NB 9-June-2000, 2-Aug.-2000 and 24-Aug.-2000 are pre-weed clearing surveys, and not all Y axis are equal due to large variation in abundance.

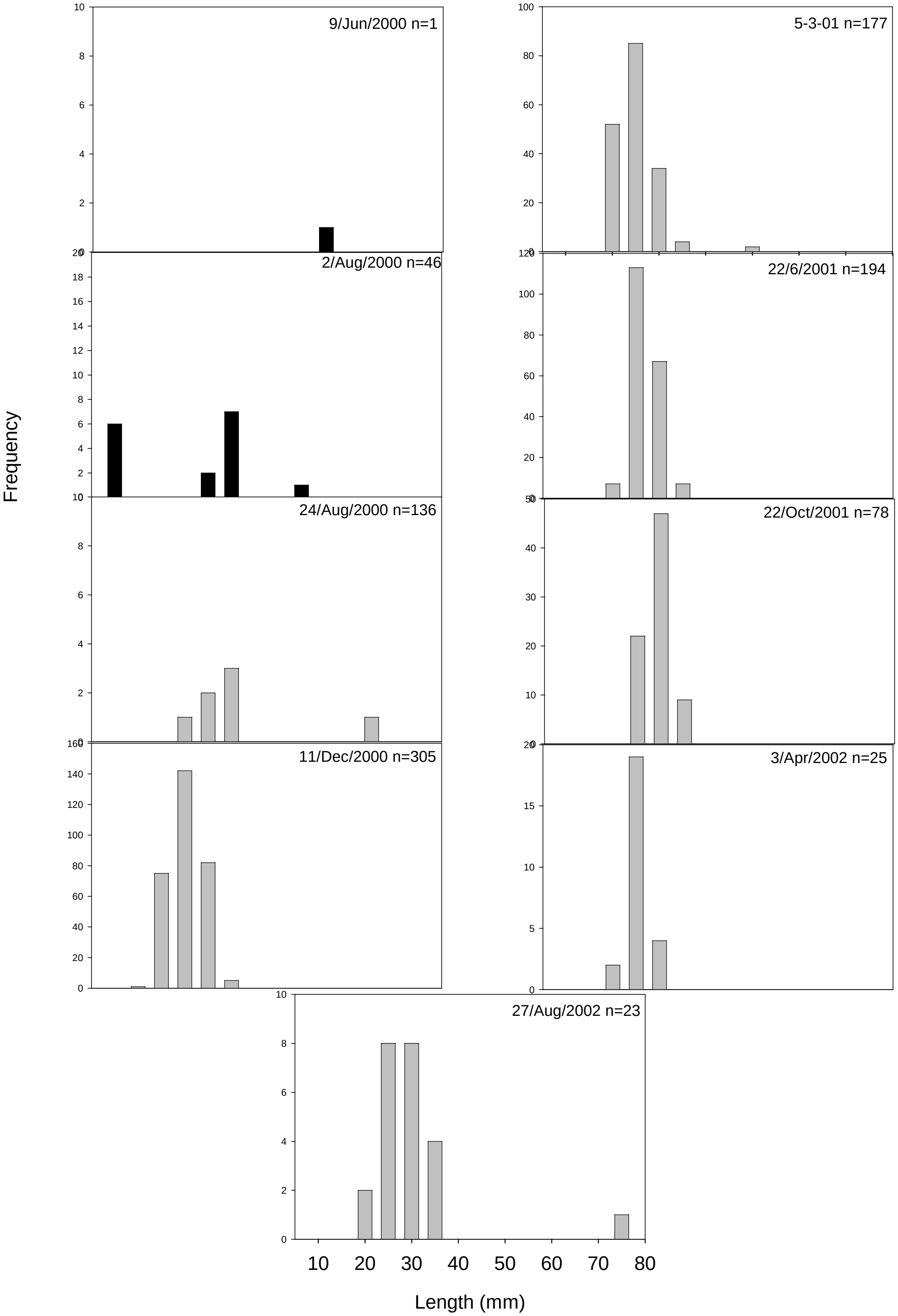
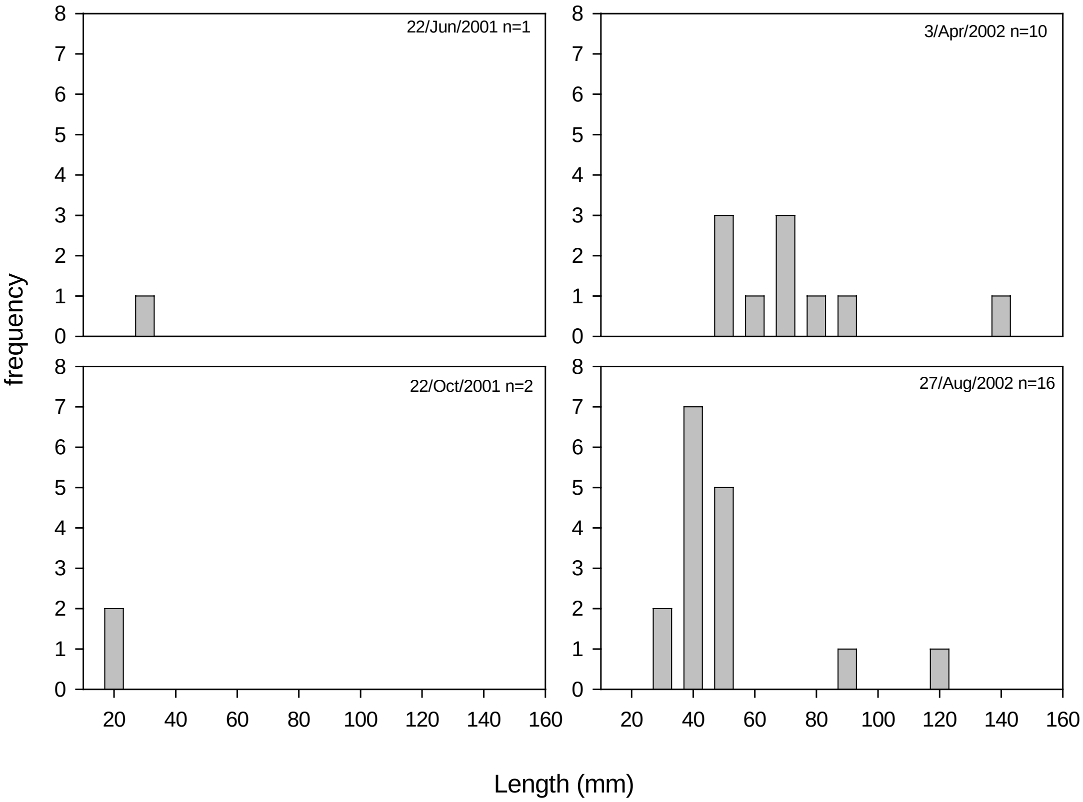


Figure 5.13 Length frequency (TL mm) for *Glossamia aprion* over four surveys in Payard's Lagoon. **Only four surveys are presented as this species invaded only after survey five.**



5.3.5 Temporal variation in oxygen content and the impact of *E. crassipes*

The oxygen content at the lagoon outlet before weed removal was very low (average below 30% saturation), and showed very little cycling (Fig. 5.14). After weed removal, a "slump" in oxygen occurred during the afternoon (Fig. 5.14), probably as a result of flow of overnight, introducing low oxygen levels from upstream, and not due to cycling within the lagoon. The main change was that median levels were significantly higher within the lagoon after the weeds were removed. There were still periods when oxygen concentrations fell below 25% in the afternoon.

The source of the slump can be seen in the inlet data (Fig. 5.15). The logged data from the inlet shows a deep slump in the early morning hours and then as the sun rises and photosynthesis occurs the more natural cycling occurred (Fig. 5.15). The slump in the early morning is probably due to a body of water that slowly flows through the lagoon to be recorded in the outlet in the afternoon. The data from the inlet shows a different effect of weed removal where the post-clearing oxygen levels are lower than pre-clearing (Fig. 5.15). This reflects the fact that upstream factors rather than weed infestation are influencing oxygen content within the inlet, which dominates the oxygen cycling in the outlet. The same two months were logged one year after the weeds were removed to examine long-term effects.

The oxygen content in the inlet was lower one year later (Fig. 5.16b). This again shows how the cycling and content of oxygen in the inlet are driven by upstream factors. The same negative afternoon cycle at the outlet was still evident but much less pronounced (Fig. 5.16a). The overall content was much higher in 2001 showing that instream oxygen production was well established. The median at the outlet only fell below 50% saturation during seven hours in the late afternoon, whereas in 2000 only two hours were recorded above 50% saturation (Fig.5.16a). This data shows that production of oxygen begins almost as soon as floating weeds are removed. Suggesting that removal of weeds reduces oxygen consumption and increases atmospheric exchange.

Figure 5.14 Boxplot for percent saturation of oxygen, each box displaying one month of logged data at each hour of the day. Data was logged at the outlet of Payard's Lagoon. See Figure 4.2 for explanation of boxplot symbols. Data are for August and September 2000, status is 1=pre-clearing and 2= post-clearing. Boxes are described in methods.

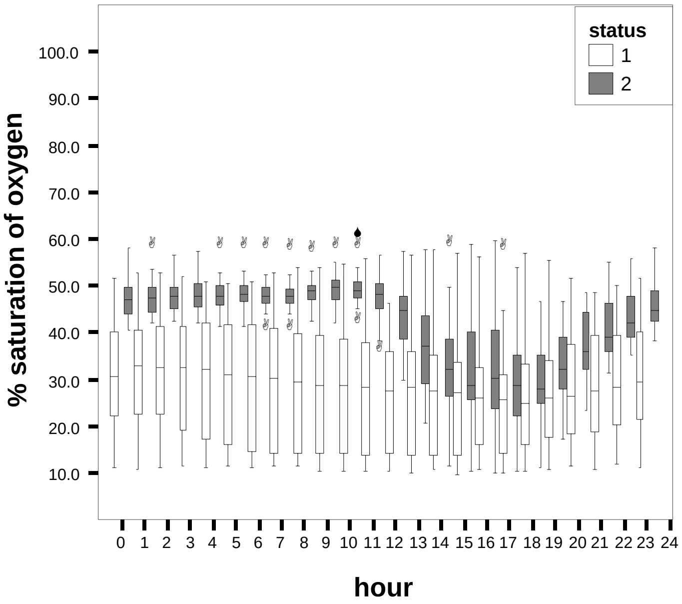


Figure 5.15 Boxplot for percent saturation of oxygen, each box displaying one month of logged data at each hour of the day. Data was logged at the inlet of Payard's Lagoon See Figure 4.2 for explanation of boxplot symbols. Data are for August and September 2000, status is 1=pre-clearing and 2= post-clearing. Boxes are described in methods.

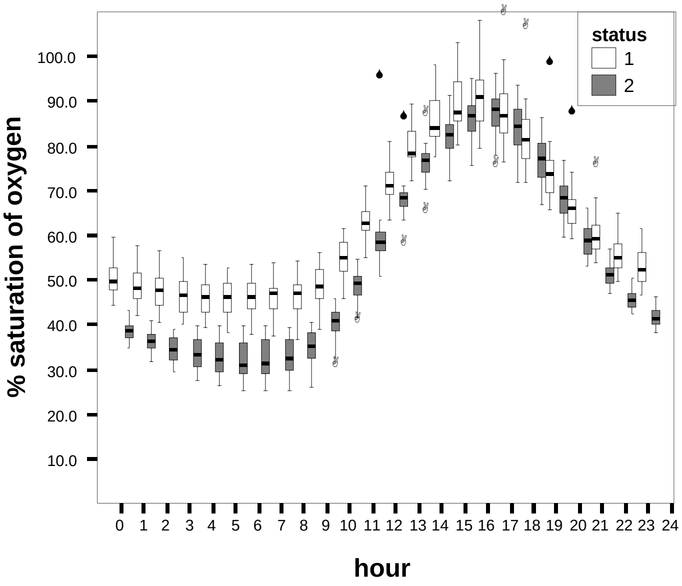
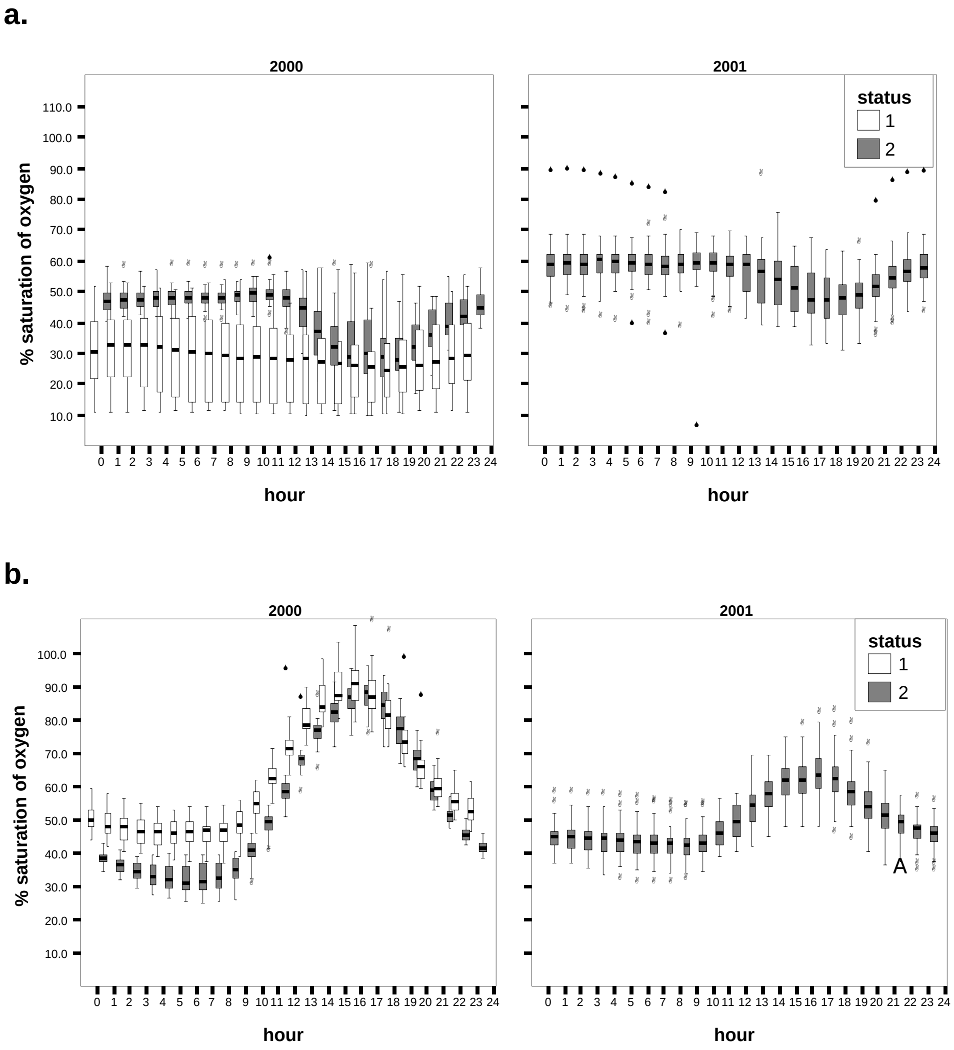


Figure 5.16 Boxplot for percent saturation of oxygen comparing oxygen saturation before and after weed removal and one year after weed removal. The boxes for the year 2000 each display one month of logged data, for the year 2001 the boxes display data for two months. Data was logged at the inlet and outlet of Payard's Lagoon. See Figure 4.2 for explanation of boxplot symbols. (a) outlet and (b) inlet. Data are for August and September 2000 and 2001, status is 1=pre-clearing and 2= post-clearing.



5.4 Discussion

5.4.1 Temporal changes in oxygen content resulting from weed removal

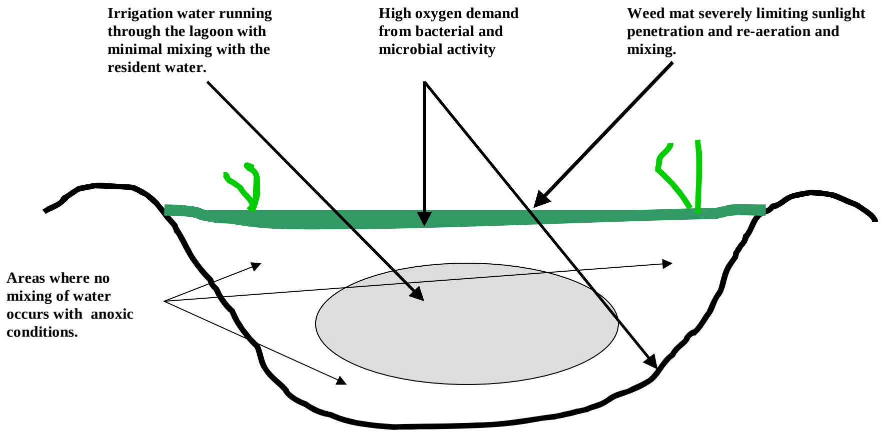
The infestation of *Eichhornia crassipes* resulted in very poor water quality. This has been well documented in other tropical regions (Wijeyaratne and Perera, 2000; Masifwa *et al.*, 2001; Scheffer *et al.*, 2003). The lack of connection between the water and atmosphere severely inhibits re-aeration (Pearson *et al.*, 2003, Scheffer *et al.*, 2003). This is also compounded by the high amount of organic material contributed by the weed mats, which increase biological oxygen demand (Battle and Mihuc, 2000; Pearson *et al.*, 2003) (see Figure 5.17). *Eichhornia crassipes*, like other floating weeds, has primacy to light over submerged macrophytes and when nutrients are not limited, as in Sheep Station Ck, can establish a stable state where anoxic conditions under the mats dominate to the exclusion of most aerobic life forms (Scheffer *et al.*, 2003).

Therefore, *E. crassipes* mats probably block all avenues for reaeration of the water in the lagoon, and severely limit autotrophic production. The process is described in Chapter 2 and here is presented as a conceptual model (Fig. 5.17). Due to the tight packing of the plant on the surface of the water there is no interface for air and water to mix. There is no opportunity for wind-generated or flow-driven mixing. The flows coming into Payard's Lagoon essentially flow through as if in a pipeline, short circuiting and not mixing with cooler anoxic resident water (Fig. 5.17, ACTFR, unpublished data). The residence time for the incoming irrigation water is low and oxygen is partly stripped from this water due to respiration processes under the mat (Fig. 5.16 a and b). Once these weeds are removed the processes for autotrophic oxygen production (largely by phytoplankton and increasing growth of native macrophytes) and re-aeration begin almost immediately (Fig. 5.14), which in turn allows for recolonization by native fish species.

Before weeds were removed there was very little aerobic biological activity under the mats. The fish and macrophytes that were present before weed removal were restricted to the refuge at the inlet (where the *E. crassipes* could not establish due to higher flow velocities (Oguto-Ohwayo, *et al.* 1997)), or were present in low diversity and abundance. Willoughby *et al.*(1993) describe a similar situation in Lake Victoria. The lagoon was dominated by a few small-bodied species that used air-surface respiration (ASR) or other physiological methods to cope with low oxygen levels. Only the small-bodied *G. holbrooki* was found in the *E. crassipes*, above the root mass and surviving largely due to its ability to use ASR.

Figure 5.17 Processes by which *Eichhornia crassipes* blocks re-aeration pathways, causes flows to short circuit and increases biological oxygen demand through supply of organic materials, in Payard’s Lagoon.

Cross-section of weeded lagoon showing how flow and oxygen are affected



5.4.2 Recovery of native macrophytes after weed removal

While Payard’s Lagoon was infested with floating weed mats, submerged and emergent native macrophytes were largely absent from the lagoon. Only in the open water was there any significant growth of these plants. In lentic tropical floodplain lagoons a large component of the oxygen mass balance comes from autotrophic inputs, but under the fully weeded conditions at Payard’s this input was largely missing (Pearson *et al.*, 2003; Butler, 2003). Once the weeds were removed, native plants re-established very quickly. Lilies (*Nymphaea* spp.), for example, appeared by August 24, 2000, just weeks after the *E. crassipes* was removed. By June 2003, beds of *Cerataphyllum demersum, Hydrilla verticillata, Utricularia gibba* and others were distributed through the lagoon. These species are very important to fish such as *M. splendida* (for reproduction), *A. agrammus* and *G. aprion* (for foraging) and *A. percoides* (for food and shelter) (Pusey *et al.*, 1993; Perna, 1996; Pusey and Arthington, 2003). The establishment of native macrophytes was, therefore, probably crucial to the re-invasion of fish species.

5.4.3 Major changes in fish assemblages

The major changes in fish assemblages documented over this experiment were increased fish abundance and diversity and a decrease in exotic fish abundance. Most notable was the re-invasion of *N. erebi*, which is a highly oxygen - sensitive species that had not been seen in the lagoon for almost a decade (A. Darwen, pers. comm.), and larger piscivorous fish species. Not only did these species reinvade but actively reproduced within the site, so conditions were clearly above sub-lethal limits for these species.

a) Changes in fish communities in response to water quality changes

Recent research at ACTFR shows that most native fish will start to show physiological stress at oxygen levels below 50% saturation (Pearson *et al.*, 2003). These stresses lower the ability of fish to forage, grow and reproduce (Pusey and Arthington, 2003; Pearson *et al.*, 2003). While Payard's Lagoon was covered by weeds oxygen concentrations within the lagoon only reached 50% saturation during 3 hours in the day over a month of sampling and only as maximum values well above the 75th percentile of the data (Fig. 5.14). The mean oxygen levels were 30% or lower and every hour of the day minima were recorded at or below 20%, which is the lethal threshold for most fish species, excluding those that utilize aquatic surface respiration (ASR) such as *G. holbrooki*, facultative air breathers such as *M. cyprinoides* and *T. trichopterus* or that can slow their metabolism enough to wait out the low oxygen conditions (e.g., *Hypseleotris* spp. and *Neosilurus* spp.) (Dehadrai and Tripathi, 1976). After the weeds were removed the mean oxygen saturation rose to just under 50% for twelve hours of the day and dropped slightly in the afternoon. Minimum levels only went below 20% for five hours of the day (Fig. 5.14). However, once the weeds were removed and the water/air interface was restored, fish were able to move into the surface layers where oxygen levels are higher than the rest of the water column (Matthews, 1998; Pearson *et al.*, 2003). These two effects of increased oxygen concentrations and the re-introduction of the water/air interface facilitated the re-invasion of fish species, including oxygen sensitive species, into Payard's Lagoon.

b) Fish species invasions after weed removal

With the increase in oxygen, after weed removal, fish that are intolerant of poor conditions were able to reinvade the lagoon. During the weeded conditions the diversity and abundance of fish was highest in the seine samples at the inlet of the lagoon. Once the *E. crassipes* was removed and habitat quality improved, the inlet fish assemblage invaded the lagoon (Section 5.3.1). This assemblage was the first source of fish to recruit into the lagoon and this recruitment happened over two months. Following a flood, the recruitment source became larger (although this recruitment was of short duration as the flood was only moderate) by connecting Payard's Lagoon to upstream areas and off-channel lagoons that had remnant fish communities. After the flood event of 2000/2001 oxygen sensitive species, such as *N. erebi*, invaded via overflow from Pink Lily Lagoon, upstream (A. Darwen pers. com.). Another notable result was the lack of larger piscivorous fish before the weeds were removed, but when the first mats were removed, *M. cyprinoides* re-invaded from the channels. Over the two years of sampling and six post-weed harvest surveys, new species were recorded on each survey, suggesting continued invasion from other parts of the catchment (Section 5.3.1). The flooding clearly demonstrated invasion through the system, but as only a few species invaded the lagoon, it is likely that loss of connectivity in the catchment restricted the extent of the re-invasions. Because of extensive weed infestations throughout the sub-catchment, there was a limited suite of species available for dispersal into the lagoon. Only those species that were present in refuge habitats in the channel and in lagoons (e.g., Pink Lily and Castinalli's) were available for invasion. The only other source was the pump stations, which transfer fish from the river into distribution channels. Once a dispersal pathway is open to the saltwater reaches, it is expected that species diversity will increase further by allowing access to amphidromous species such as *Lates calcarifer, Chanos chanos, Scatophagus argus* etc. Weed removal from the top to the bottom of the catchment would facilitate this rehabilitation.

c) Fish recruitment and predation

Once the new species invaded Payard's Lagoon, repeat sampling documented recruitment within the site. Most notable was *N. erebi*, one of the most oxygen - sensitive species in the region. High numbers of small-size-class fish of this species were recorded starting in October of 2001, over a year after weed removal. This is a most important short- to medium-term result with regard to rehabilitation of lagoons and biodiversity.

Another aspect of removal of floating weeds and recruitment of native plant species is that the exotic fish *G. holbrooki* was found to use the *E. crassipes* very effectively to avoid predation, whereas, in the less compacted and complex habitat of native submerged and emergent macrophytes, there was increased predation on this species (Webb, 2003), evidenced by a major drop in proportion of *G. holbrooki* in the dip net samples (Fig. 5.5). I also observed *M. cyprinoides* shoaling the *G. holbrooki* in open water and aggressively feeding on them. The increase in *G. aprion* (by the last sample *G. aprion* was caught in all nine dip net samples) after weed removal may also have had an impact on abundances of *G. holbrooki* as this fish has been shown to be a major predator of *G. holbrooki* (Webb, 2003). The re-introduction of predators is known to be a major control on fish assemblage structure. Kennard 1995 found that in lagoons on the Normanby River, Cape York Peninsula, large predators such as *L. calcarifer* and habitats such as woody debris were found to control small - bodied fish species through the dry season. In this study, sites that were not cleared of weed and thus had no associated increase in piscivorous fish, did not show the same decrease in abundance of small bodied fish or proportion of exotic species abundances (Section 4.3.3) In summary, weed removal was shown to increase water quality and fish diversity, and decrease abundance of exotic fish.

5.4.4 Long-term effects of weed removal

The results of this work record medium-term effects. Fish assemblages increased in diversity and water quality improved over one year but no information is available on the long-term effects of weed removal. One notable situation is the rapid increase in submerged macrophyte growth. Due to lack of scouring the biomass of submerged macrophytes in Kelly's Lagoon (the remnant site for Sheep Station Ck) became so high that the diel oxygen cycle was very large and slight changes in respiration (most likely cloudy hot calm days) caused the oxygen to fall well below acute levels. This resulted in the local extinction of *N. erebi* again from this site.

Another impact may be the long-term maintenance of the weeds with chemical sprays. Chemical control is seen as the most economically viable method for long-term control of weeds; however, no information is available on the long-term effects of these chemicals on food webs and biodiversity . Recent work in the Northern Territory found that herbicides used on para grass (*U. mutica*) had little effect on the macroinvertebrate community (Douglas and O'Connor, 2003) (this may be misleading as spray was applied just prior to onset of wet season inundation). This study, however, did not present data on the effects of spraying on microinvertebrates and larval fish, which could have a great effect on the trophic ecology of these systems. Finally, it is clear that without establishing connectivity through the whole system, only marginal improvements will be seen in fish diversity, and lagoons will be islands in a sea of poor habitats.

Chapter 6 Summary and Conclusions

6.1 The Burdekin River floodplain, a highly modified landscape

Floodplains are among the most diverse habitats on the planet today (Gopal and Junk, 2000). They contain habitats that fluctuate from aquatic to terrestrial depending on water levels, which create a mosaic of habitat types and conditions (Junk *et al.*, 1989; Junk, 1997; Pearson *et al.*, 2003). This diversity of habitat types and conditions is largely responsible for the diversity of biological life forms that inhabit these areas. However, when floodplains are modified for agriculture this diversity of habitats is largely lost (EPA, 1999). On the Burdekin River floodplain, habitat diversity and quality have been massively degraded as land has been cleared and levelled for sugar cane production, natural distribution channels have been modified for irrigation distribution and aquifer recharge, and flows have been greatly modified. This reduction in habitat diversity, quantity and quality is seen in both the water quality and assemblage structure of the fish communities throughout the Burdekin River floodplain.

Habitats on the Burdekin River floodplain were found to be most impacted in the delta area managed by the North and South Burdekin Water Boards. Riparian vegetation was largely cleared or trees had been water – logged and killed by increased water levels from irrigation water. Instream habitats were impacted by the increased flow frequency and turbid water from the upper catchment that is used for irrigation. However, there were sites within the delta that had high habitat values and these sites most likely play a vital role as refuges within the sub-catchments. The modification of the floodplain for agriculture has had many effects on fish communities that include, but are not limited to: decreased habitat diversity, weed infestation, increased nutrient input loads and migration barriers (ACTFR, 1994; Arthington *et al.*, 1997; Pusey and Arthington, 2003*)*.

Burdekin River floodplain habitats include shallow ephemeral sedge and *Melaleuca* swamps, saline super-tidal swamps, deep black-water lagoons, backwaters and overflow channels. Agricultural activities have reduced these habitats to about 30% of their original extent (G. Lukacs, pers. comm.), with only those habitats that serve some function for the cropping activities generally being retained. On the Burdekin River floodplain many of the shallow ephemeral swamps have been lost, and only the deep water lagoons and associated channels remain. Shallow swamps, especially super-tidal varieties, are very important for economically important species such as *Lates calcarifer* (Barramundi), *Scilla serratta* (mud crab), and a host of prawn species (Davis, 1988). The remaining deep-water lagoons and channels have had much of their riparian vegetation cleared, which has lead to degradation of water quality, increased infestation of exotic emergent grass and floating weed, and consequent loss of instream and riparian habitat values.

Weed infestation, especially by Para grass (*Brachiaria mutica*), is very common through out the floodplain. This grass grows into the channels and lagoons, and can increase flow velocities, reduce oxygen content and has also been found to have no input into the aquatic food web (Bunn *et al.*, 1997, 1998). The floating weed Water hyacinth (*Eichhornia crassipes*) has also benefited from the increased light, sustained water levels from supplemental flows and the grasses. *Brachiaria mutica* and the native Rice grass (*Leersia hexandra*) bind the *E. crassipes*, creating stable platforms that are difficult to flush. These grasses, in combination with supplemental flows, most probably have aided in the full cover of lagoons by *E. crassipes* within the irrigation Water Board areas, as full cover was not documented in any other part of the floodplain.

The loss of native riparian vegetation and the proliferation of exotic emergent and floating weeds have greatly impacted instream water quality. Increased insolation increases temperature and submerged macrophyte growth. Increased temperatures can impact fish foraging behaviour, survivorship, and egg and larval development (Pusey and Arthington, 2003). Increases in macrophyte production cause greater cycling of dissolved oxygen, often resulting in hypoxia overnight.

Increased light availability, resulting from reduced riparian cover, facilitates the growth of invasive weeds and may contribute to the extensive infestations of *E. crassipes*. Section 3.3 showed how *E. crassipes* infestations prevent oxygen from entering the water and contribute significantly to the processes for consumption of oxygen in the water column. Therefore these exotic plants reduce habitat diversity and quality, and have negative impacts on the water quality, which excludes sensitive native fish species from establishing. The poor quality habitat created by these invasive weeds may aid the proliferation of exotic fish also.

The mosquito fish (*Gambusia holbrooki*) was found in very high abundances in the most degraded sites (section 4.3.2 and 4.3.4). This species is very effective at utilizing dense habitat (characteristic of *E. crassipes*) to avoid predators (Webb, 2003) and is capable of tolerating very low oxygen concentrations typical of *E. crassipes* infested lagoons (Cech *et al.*, 1985). Throughout the study, sites that were heavily infested with this weed were dominated by this species and a single native fish (the *Hypseleotris* spp.), which also appears to be very tolerant of low oxygen concentrations (ACTFR, unpublished data). The modification of the flows in the distributary streams was shown to benefit the proliferation of these weeds, which indirectly may benefit the exotic fish species.

The water used for irrigation comes from the turbid upper catchment of the river and is high in sediments. These high sediment flows supply continuous nutrient inputs which prevent the weeds from being nutrient – limited (Section 3.3.4), and they are therefore capable of out-competing native submerged macrophytes (there may also be an effect of light limitation by the water itself on submergent macrophytes). Once the weed mats establish and cover a lagoon, or reach in the channels, they effectively create a chemical barrier excluding oxygen sensitive fish species. In sites that had irrigation water, weed infestations were usually the worst. These sites had low fish diversity, generally comprising the species most tolerant to low oxygen levels (Table 4.6). This is exemplified by the occurrence of an average of eight species of fish in the highly degraded sites within the Water Board areas and the average of 12 species in the remnant sites outside of the Water Board areas.

One of the greatest threats to fish diversity in this region are barriers to migration. The weed infestations indirectly have created barriers by reducing oxygen concentrations to levels that exclude many native fish species. In conjunction with this chemical barrier, fish also have a maze of bund walls at the saltwater interface, drop board structures, high flow culverts and weirs that prevent free movement up and down the catchments. The infrastructure used for managing irrigation water has created these barriers and has also been a benefit to weed infestation, creating a complex relationship between flow modification, weed infestation and infrastructure that combine to prevent upstream and downstream migration. Native fish communities of the tropical east coast of Australia include a large percentage of marine – derived species (Pusey and Kennard, 1996), but in the Burdekin wetlands a large proportion of these species are excluded by barriers.

Thus, the remaining wetlands of the Burdekin River floodplain, especially in the delta, are in a moderately to highly degraded condition.

6.2 Restoring fish habitat and water quality – a case study

Chapter 5 presented the results of a large-scale weed removal project on Payard’s Lagoon. This project showed that fish habitat restoration in the distributary streams can be successful in the short to medium term. Before weeds were removed from this lagoon, oxygen content was well below levels that are required to sustain fish communities, and the fish present were mostly restricted to the open water in the inlet channel. Once the weeds were removed both water quality and fish community composition immediately improved. There was an immediate effect following weed removal, and a longer term pattern of progressive improvement over the subsequent year. Clearly, the main culprit in degradation of fish habitat quality in this system was *E. crassipes*, mats of which block the water/air interface needed for re-aeration, as well as blocking light needed by submergent macrophytes to produce oxygen in the water by biological means.

Within two weeks of weed removal native submerged and emergent vegetation was establishing. This greatly contributes to oxygen concentrations in the water column that allow for fish recolonisation. The plants not only provide the oxygen but vital habitats for spawning, foraging and predator avoidance (Perna, 1996; Pusey and Arthington, 2003). The regrowth of native submerged macrophytes in conjunction with the increased oxygen concentration allowed the fish in the inlet to invade the downstream lagoon. A few new species were recorded immediately after weed removal, but it was not until flooding that fish from other parts of the catchment gained access to Payard's Lagoon. After the flood six new species of fish invaded the lagoon, including oxygen – sensitive species such as *Nematalosa erebi* and *Glossamia aprion*. By the end of the study, these species were actively recruiting within the lagoon, further evidence that conditions were well above viability thresholds. The number of invaders, however, was low.

Although the lagoon now provides suitable habitat for a host of local fish species, connectivity to the lagoon is likely preventing these species from invading. During this project Payard's Lagoon was the pilot trial for weed removal works in Burdekin Shire. Works continued downstream and the fish followed the open water. One very important site was Dick's Bank Lagoon, which eventually recorded the most species (16). This is likely the result of a direct high-water connection to the high-value remnant, Castelanelli's Lagoon (Fig. 2.6). This site was sampled once and had many of the species missing from the rest of the catchment (e.g., *Oxyeleotris lineolata*, *Amniataba percoids* and *Toxotes chatareus*). This highlights the importance of connectivity throughout the catchment, which needs to be addressed in any management planning.

6.3 Conclusions and recommendations

The major findings of this thesis are:

1. Habitats are moderately to highly degraded within the distribution streams controlled by the Water Boards.

2. High value habitats still occur within these distributary streams and act as refuges and recruitment sources.
3. Habitats across the floodplain, outside of the Water Board areas, were somewhat less degraded.
4. Water quality in the Water Board areas is largely driven by a combination of poor habitat condition and supplemental flows.
5. Native fish species showed a preference for good condition sites with the least impacted habitats and better water quality.
6. Exotic fish species appeared to prefer degraded habitats with poor water quality.
7. Weed removal improves lagoon water quality in the short to medium term.
8. Weed removal allows native fish to re-colonise and recruit into the lagoon.
9. Downstream barriers are preventing colonisation of the wetlands by marine species that would normally be an important component of the community.

Any management planning should take into account the highly modified nature of flows in the Burdekin River floodplain. These flows have been shown to aid weed growth and thus decrease habitat quality, however, they help to flush out the poor water quality in sites that are cleared of weed and they increase minimum oxygen concentrations in weeded section by inflow from open water sections upstream. This may be important in the long-term as submerged macrophyte biomass may increase, thus driving very large fluctuations in the diel oxygen cycles. Future weed removal projects should aim to remove weed from upstream and proceed to downstream. This approach will firstly prevent battling the downstream flow of weed mats. Secondly, tropical floodplain fishes in the Normanby River have been found to not discriminate in lagoon selection during flooding (Kennard, 1995). If downstream lagoons are cleared before upstream, sensitive fish may recruit during flooding and then be trapped in a lagoon that has low oxygen inflows from weeded upstream sections, thus leading to fish kills. Therefore it is vital to open up not just lagoons but all possible migration pathways within the Burdekin distributary streams, for restoring fish habitats and fish communities.

Clearly, development on the Burdekin River floodplain has had devastating effects on water quality, instream and riparian habitats, and fish communities that are of conservation, recreation and commercial value. While attempts at rehabilitation have commenced, with very encouraging results, it will require an enormous effort at a landscape scale to properly restore some of the important conservation and fishery values to the remnants of once extensive, productive and unique wetlands.