Fish and decapod assemblages in Kooragang Wetlands: the impact of tidal restriction and responses to culvert removal

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Final Fisheries Report for the Kooragang Wetland Rehabilitation Project

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February 2012

NSW Department of Primary Industries –
Fisheries Final Report Series
No. 133
ISSN 1837-2112

NSW GOVERNMENT Primary Industries

NSW GOVERNMENT Catchment Management Authority Hunter–Central Rivers

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February 2011

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Published By: NSW Department of Primary Industries

Postal Address: PO Box 21 Cronulla NSW 2230

Internet: www.industry.nsw.gov.au

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ISSN 1837-2112

Note: Prior to July 2004, this report series was published by NSW Fisheries as the 'NSW Fisheries Final Report Series' with ISSN number 1440-3544. Then, following the formation of the NSW Department of Primary Industries the report series was published as the 'NSW Department of Primary Industries – Fisheries Final Report Series' with ISSN number 1449-9967. The report series is now published by Industry & Investment NSW as the 'Industry & Investment NSW – Fisheries Final Report Series' with ISSN number 1837-2112.

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ACKNOWLEDGEMENTS

We are grateful to all staff of the Department of Primary Industries and all volunteers and staff of the Kooragang Wetland Rehabilitation Project (KWRP) who assisted with field and laboratory work over the 16 years of this study. These individuals include Jack Hannan, Fiona Watford, Vlad Balachov, Darryl Sullings, Brett Louden, Alan Genders, Isabelle Thiebaud, Brooke McCartin, Ben Rampano, Tony Fowler, Gary Reilly, Ben Kearney and Antonia Creese. We are thankful to the leaseholders of Kooragang Island who allowed us access to the wetlands.

This research was initiated in 1993 by a grant from Port Waratah Coal Services Pty. Ltd, sustained by support from the then NSW Fisheries, and subsequently the NSW Department of Primary Industries and NSW Industry and Investment, and finalised by a grant from the KWRP via the Hunter-Central Rivers Catchment Management Authority. We would like to acknowledge the efforts of Craig Copeland, Peggy Svoboda, Rob Henderson and the late Sue Rostas for their involvement at various stages throughout this study.

The collection of all animals in this project was in accordance with the appropriate animal care and ethics research authority (98/11) and a Section 37 research permit in accordance with the NSW Fisheries Management Act 1994.

NON-TECHNICAL SUMMARY

Fish and decapod assemblages in Kooragang Wetlands: the impact of tidal restriction and responses to culvert removal

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NON TECHNICAL SUMMARY:

Background and objectives

At least half of all coastal wetlands within south-eastern Australia have been degraded by human disturbance, and further fragmentation of these important estuarine environments is predicted under climate change. Within New South Wales, there are in excess of 1,000 man-made barriers which restrict tidal flow and the movement of aquatic biota into coastal wetlands. Most significantly, a large proportion of these barriers have been deemed candidates for modification to reinstate tidal flushing, potentially enhancing connectivity and contributing to the rehabilitation of coastal wetlands and aquatic fauna.

In an effort to improve the value of fish and crustacean habitat, tidal flow into creeks on Kooragang Island was increased by culvert removal as part of the Kooragang Wetland Rehabilitation Project (KWRP) in the lower Hunter River. This study provided an opportunity to determine what impact (if any) the presence of tidally restrictive culverts has had on fish and decapod crustaceans (e.g. prawns) in tidal creeks and their upstream ponded waters (marshes), and to subsequently measure how these assemblages respond to culvert removal and the reinstatement of tidal flow.

Methods

The study design involved comparing manipulated locations (creeks and marshes) to control locations (where culverts remained for the entire study) and reference locations (where no culverts were present). The trajectory of response in tidal creeks was monitored over a 16 year period (14 years post-culvert removal) between 1993 and 2009, a timeframe seldom applied to rehabilitation studies, but more applicable to determining whether rehabilitation responses become self-sustaining. Due to a slow but progressive increase in water depth in the manipulated marshes that appeared to bias the efficiency of the sampling gear, marsh data were only analysed for three years at the onset of the study. Two years into the 16 year study, tidally-restrictive culverts were removed from Fish Fry Creek and Crabhole Creek on the south-arm of the Hunter River. Seine and gill netting were carried out in these creeks and two control creeks and two reference creeks. Multiple temporal replicates were taken between late spring and summer in 10 of the 16 years to coincide with the summer recruitment period and spring tides. Additional sampling with fyke nets was performed in two marshes upstream of the removed culverts at Fish Fry Flats and Crabhole Flats, and two control and two reference marshes.

Results and discussion

A comparison between control, reference and manipulated locations provided irrefutable evidence that the presence of culverts at Kooragang Island had significantly changed the fish and decapod assemblage of tidal creeks. One of the more conspicuous ecological changes was the exclusion of many estuarine-marine dwelling species upstream of culverts. This resulted in an overall reduction in species richness (number). In particular species such as glass goby, Port Jackson glassfish, flat-tail mullet, yellow-finned bream, largemouth goby, pink shrimp, school prawn, striped shrimp and Tamar River goby were consistently less abundant in tidally-restricted creeks. The habitat created in these disturbed environments was not detrimental for all estuarine-marine species however. Species such as mangrove goby, blue-spot goby, half-bridled goby, bridled goby and checkered mangrove goby were consistently more abundant in creeks with culverts. This may be due in part to their demersal (bottom-dwelling) nature and preference for muddy substrates and lower velocities, which tend to be more typical of tidally-restricted creeks. Invasive species such as mosquitofish and yellow-fin goby also appear to be favoured in tidally restricted marshes.

Succession is a sequence of directional changes in the composition of a community (or assemblage of species). In this study we were able to observe a clear succession in the fish and decapod assemblage of Fish Fry Creek occurring as two distinct changes over 16 years in response to culvert removal. The first change occurred immediately and persisted for at least six years following culvert removal. Although there was no net increase or decrease in the number of species inhabiting the creek during this time, there was a significant change in assemblage composition. The goby species previously mentioned as dominant in tidally-restricted creeks became less abundant and a reciprocal increases in species such as flat-tail mullet, fantail-mullet and sea mullet was observed.

A secondary significant shift in the assemblage of Fish Fry Creek was observed sometime between years 8-10 (6-8 years after culvert removal). It was only at this time that the assemblage of Fish Fry Creek could be judged to be fully matured and equivalent to that of unrestricted reference creeks. Most importantly, the change was not seen at control creeks and was also not due to a change at reference creeks, providing strong evidence that culvert removal was responsible. Notable changes in Fish Fry Creek at this time involved a reduction in the abundance of the mullet species that were previously noted to be primary colonisers, and an increase in estuarine-marine dwelling species such as pink shrimp, school prawn, yellow-finned bream and glass goby.

In the latter stages of the study, a reduction in species richness and abundance was observed across all creeks. The general nature of the response (including reference locations) suggests it was caused by unknown broader-scale effects on fish and decapod populations across Kooragang Island or in the Hunter estuary. Speculating about the cause of this is outside the scope of this study, but the changes highlight the importance of incorporating suitable reference locations into rehabilitation studies wherever possible. We have shown that rehabilitated wetlands can develop along complex trajectories that may be difficult to predict. Without references it would be impossible to ascertain what unforseen changes can be attributed to the rehabilitation manipulation and what may be due to unpredictable disturbances that can be common in wetlands surrounded by urbanised catchments.

Consistent trajectories of improvement were not observed at all manipulated creeks. In comparison to Fish Fry Creek, Crabhole Creek displayed large inter-annual variability in assemblage composition that could not reasonably be attributed to culvert removal. The reason for differing responses between the manipulated creeks may be due to spatial differences in larval and juvenile supply in the Hunter estuary. However, it is just as likely to be due to the different degree to which culvert removal changed the habitat of the two creeks. Significant channel deepening and widening occurred at Fish Fry Creek, but the changes were less significant at Crabhole Creek. Crabhole Creek ended up being shallower, narrower and had greater mangrove establishment at lower elevations. These physical changes are likely to impact on the relative refuge values of both creeks and on predator-prey dynamics. It may explain why Crabhole Creek maintained an assemblage relatively similar to restricted creeks, whereas Fish Fry Creek underwent a significant shift in assemblage composition driven by increased utilisation by estuarine-marine species such as mullets and yellow-finned bream. The differing responses highlight that caution must be exercised when making generalisations regarding likely ecological outcomes of rehabilitation activities and demonstrates that rehabilitation efforts should have some degree of pre and post-manipulation evaluation to ensure they are meeting stated objectives.

When compared to their associated creeks, responses in Fish Fry Flats and Crabhole Flats (marshes) were less evident (although only monitored over three years). A subtle response observed was that assemblages of manipulated marshes resembled unculverted marshes more than culverted ones as time progressed. For example, culverted control marshes became dominated by invasive mosquitofish, whereas manipulated and reference marshes did not. Significantly more temporal replication is required in culverted, unculverted and manipulated marshes to further resolve uncertainty.

Overview

Tidal restrictions such as culverts appear responsible for modifying the fish and decapod assemblages of creeks and marshes on Kooragang Island, reducing the number of species and in some cases favouring invasive species. Other studies show that the movement of fish and decapods into an out of coastal wetlands can play a major role in the transfer of energy and nutrients between coastal wetlands and an estuary (“trophic cycling”). Based on this we infer from our results that the exclusion of estuarine-marine dwelling species from culverted wetlands may decrease the transfer of wetland-derived production to estuarine and offshore ecosystems. Importantly, however, by showing that fish and decapod assemblages can respond to culvert removal, we support the assertion that wetland rehabilitation such as that applied at Kooragang Island may translate into improved productivity in an estuary.

The Kooragang Wetland Rehabilitation Project has demonstrated the benefits of treating rehabilitation projects as experiments within a rigorous scientific framework that maximises learning potential. This is particularly pertinent for coastal wetland rehabilitation, where responses can be site-specific due to the landscape context. At the very least, this project demonstrates why it is prudent to evaluate responses to wetland remediation so that activities can be adaptively managed in response to unforeseen or negative trajectories. Future coastal wetland rehabilitation projects should acknowledge:

1. The unpredictable nature of ecological assemblages and assume that multiple response trajectories are possible and responses may be location-specific (even within the same wetland).
2. Manipulated sites can show an improvement in the nature of their assemblages.
3. The time taken for manipulated wetlands to reach equivalency with reference states may exceed the usual three year monitoring period of rehabilitation projects.
4. Manipulated sites may never fully replace natural systems in composition or function.

Recommendations

1. Long-term studies (5-10 years) are needed to appreciate responses to rehabilitation strategies such as the replacement of culverts with bridges.

2. Short-term studies should be recognised as giving only a partial indication of response to rehabilitation efforts.
3. Rehabilitated tidal marshes and tidal creeks may contribute to trophic relay and should be further investigated for the Hunter estuary.
4. The presence of distinct groups of fish and/or decapods may indicate stages in the maturation of a rehabilitated wetland and should be further investigated as a potential way of determining whether rehabilitation outcomes are being met.

KEYWORDS

Culverts, Hunter River, Kooragang Wetland Rehabilitation Project, saltmarsh, succession, tidal creek, tidal marsh, tidal restriction, wetland rehabilitation

1. GENERAL INTRODUCTION

1.1.1. Rehabilitation of degraded coastal wetlands

Estuarine wetlands act as important nurseries for fish and invertebrate species during early life stages and contribute considerably to the productivity of many estuarine and offshore fisheries (Morton 1990, Barbier and Strand 1998, Manson *et al.* 2005a). Estuarine habitats afford greater survival to a large number and diversity of juveniles (Boesch and Turner 1984, Kneib 1984, Able *et al.* 1996, Kneib 1997, Paterson and Whitfield 2000, Zedler 2000, Beck *et al.* 2001, Laegdsgaard and Johnson 2001, Levin *et al.* 2001, Manson *et al.* 2005b, Baker and Sheaves 2007), and their role in primary and secondary production, sediment retention and groundwater recharge are thought to have cascading effects which in turn also support greater fisheries production (Wolanski 1995, Furukawa *et al.* 1997, Barbier and Strand 1998, Winter 1999, Alongi and McKinnon 2005, Jordan *et al.* 2009).

Estuarine function becomes compromised once wetland connectivity within the landscape becomes fragmented (Meynecke *et al.* 2007, Meynecke *et al.* 2008, Meynecke 2009). Upsurge in the development of coastal areas over recent decades has disturbed large areas of wetland and placed pressure on biodiversity and sustainability (Vitousek *et al.* 1997, Edgar *et al.* 2000). In particular, the proliferation of extensive flood mitigation schemes, including levee banks, drainage channels, floodgates, dams and weirs has reduced connectivity in coastal habitats in many parts of Europe, North America and south-eastern Australia (NSW Fisheries 1976, Raposa and Roman 2001, Strayer and Findlay 2010 and references within). In turn, restriction of tidal influence and increasing habitat fragmentation has changed the composition of aquatic flora and fauna with concomitant losses of biodiversity (Pressey and Middleton 1982, Herke *et al.* 1992, Pollard and Hannan 1994, Chambers *et al.* 1999, Kroon and Ansell 2006, Valentine-Rose *et al.* 2007, Eberhardt *et al.* 2011).

There are over 950 waterways draining the NSW coast and of these, 150 have a permanent water surface area that exceeds 1ha (Williams *et al.* 1998). Many of these waterways supply adjacent freshwater and estuarine wetlands, and as approximately half of the coastal wetlands in south-eastern Australia have been degraded by human disturbance (DEH 2005), wetland rehabilitation is seen as a valuable tool for addressing biodiversity losses (Warren *et al.* 2002, Callaway 2005, Thayer and Kentula 2005). Within NSW alone, in excess of 4,200 man-made barriers impede tidal flow and the movement of aquatic biota (Williams and Watford 1997). But, a large proportion of these barriers are deemed modifiable to reinstate tidal flushing, enhance connectivity and potentially rehabilitate wetland habitats (Williams and Watford 1997).

When undertaken, wetland rehabilitation is seldom monitored (Zedler and Callaway 2000), or the design of studies is inadequate for determining rehabilitation success (Grayson *et al.* 1999). Those studies that have assessed rehabilitation responses can often produce variable results. For example, in some instances the removal of tidal restrictions has been met with rapid and sustained recovery in wetland assemblages (Able *et al.* 2008, Boys *et al.* 2012), whereas in other instances the response has been less pronounced and possibly related to the extent tidal of restriction prior to rehabilitation (Raposa and Roman 2003, Eberhardt *et al.* 2011). Uncertainty exists when trajectories of recovery have been tracked over longer timeframes, with some wetlands showing a clear path towards rehabilitation targets (Warren *et al.* 2002, Able *et al.* 2008) whilst others appear unlikely to ever achieve functional equivalence (Zedler and Callaway 1999). Such variability in response is hardly surprising given different types of antecedent factors, degradation and management practices associated with rehabilitation studies, and it highlights the need to monitor trajectories of recovery over sufficient spatial and temporal scales to determine if rehabilitation goals are being met (Simenstad *et al.* 2006, Wagner *et al.* 2008). It is therefore of concern when wetland rehabilitation is evaluated over the short-term with little plan to sufficiently determine if rehabilitation goals are being achieved (Zedler and Callaway 2000, Wagner *et al.* 2008).

1.1.2. The Kooragang Wetland Rehabilitation Project (KWRP) and research scope

The natural history and subsequent human-induced degradation of the Kooragang Wetlands in the lower Hunter River of NSW has been well documented (Turner 1997, Streever 1998, Williams 2000). The Kooragang Wetland Rehabilitation Project (KWRP) was established in 1993 to oversee rehabilitation activities within Kooragang Wetlands to improve fish and shorebird habitat (Streever 1998). In 1995 the KWRP removed culverts at the mouths of two creeks to reinstate tidal flow in these creeks as well as tidal ponds further upstream containing salt marsh. Research of short and medium-term responses within a Before-After-Control-Impact framework (BACI) (Underwood 1991) has been conducted looking at nutrient cycling (Howe *et al.* 2009), vegetation (Streever and Genders 1997, Howe *et al.* 2010) and shore bird roosting (Kingsford *et al.* 1998). In this report we describe research undertaken by NSW Department of Primary Industries (formerly NSW Fisheries) on the fish and decapod crustacean assemblages of the wetlands between 1993 and 2009. In doing this we provide a case study in coastal wetland rehabilitation across a long time-frame (14 years post-culvert removal), a temporal extent seldom applied to rehabilitation studies but more applicable to determining whether rehabilitation responses become self-sustaining (Zedler and Callaway 2000).

1.1.3. Objectives

A number of objectives were set in this study:

- To quantify the impact of culverts and subsequent tidal restriction on fish and decapod assemblages in tidal creek and marsh habitats (Chapter 3)
- To evaluate the short-term responses in these assemblages following culvert removal and the reinstatement of tidal flows (Chapter 4).
- To evaluate the long-term responses in these assemblages following culvert removal and the reinstatement of tidal flows (Chapter 4).

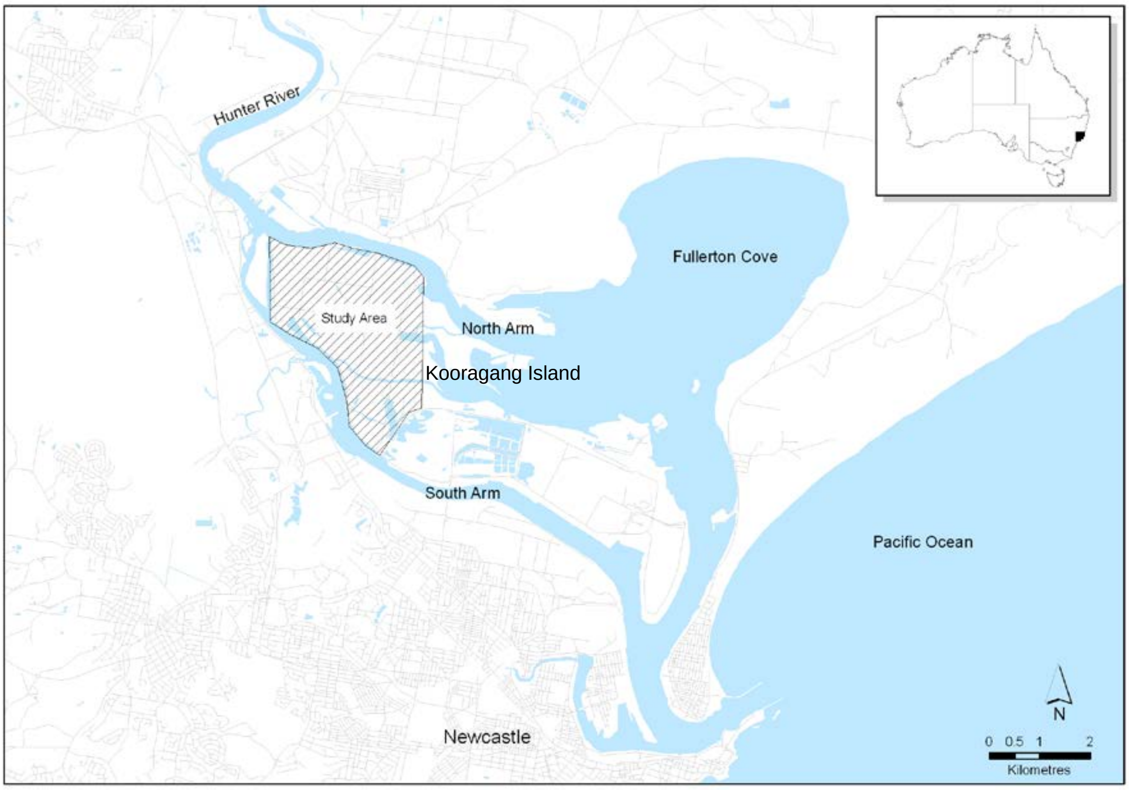

Fig. 1 Location of the study area on Kooragang Island in the lower Hunter River.

2. GENERAL METHODS

2.1.1. Study area

The study area, Kooragang Island (32^{o}51’52”S, 151^{o}42’15”E), is a wetland complex approximately 26 km^2 in area and situated ~ 10km upstream the mouth of the Hunter River, Newcastle, New South Wales (NSW) Australia (Fig. 1). The river’s catchment is the third largest in NSW, draining and area in excess of 22,000 km^2 (Williams *et al.* 2000), and its entrance is a mature barrier estuary (Roy *et al.* 2001). Drill cores show extensive infilling of the river channel and surrounding flood plain since sealevel stabilised about 6,500 years ago (Roy and Crawford 1980). Beginning in the mid 1800s channel depth in the river increased to facilitate ship traffic to the port of Newcastle (Coltheart 1997, Williams *et al.* 2000). Dredging has been continuous since that time, either as maintenance operations to clear the channel after flooding or to progressively enhance capacity for the export of coal. Newcastle is now the world’s largest coal export harbour. The effect of dredging near the mouth on tidal behaviour further upstream has been examined, and an increase of 100mm in the mean tide level has been measured at a gauging station just upstream of the Hexham Bridge (MHL 2003).

The river has been reported as the only barrier estuary in NSW with no seagrass (Williams *et al*. 2000), but other wetlands in the estuarine portion of the catchment typically consist of a mixture of mangrove-lined tidal creeks, transitioning into saltmarsh and freshwater wetlands at higher elevations (Streever 1997, Streever 1998). The dominant mangrove species is grey mangrove (*Avicennia marina*) with some river mangrove (*Aegiceras corniculatum*), and their distribution should be watched with some interest as an increase in distribution of mangrove has been reported for southeast Australia over the past several decades (Saintilan and Williams 1999, Saintilan and Williams 2000).

At higher elevations, tidal creeks transition into saltmarsh consisting of samphire (*Sarcocornia quinqueflora*), salt couch (*Sporobolus virginicus*), seablite (*Suaeda australis*) and streaked arrow grass (*Triglochin striata*). In the lower areas a mixture of native sea rush (*Juncus kraussii*) and exotic spiny rush (*Juncus acutus*) are found. Above the upper limit of tidal influence, localised freshwater run-off often ponds in low-lying areas with species such as common reed (*Phragmites australis*), ribbon grass (*Triglochin procerum*), water couch (*Paspalum distichum*), and river clubrush (*Schoenoplectus validus*) found. The wetlands are typical fringed and interspersed with urban development and upland pasture consisting of exotic grasses including buffalo (*Stenotaphrum secundatum*), couch (*Cynodon dactylon*), and kikuyu (*Pennisetum clandestinum*) (Streever 1998).

There has been a long history of habitat change in the wetlands of the lower Hunter River over the past two centuries (Williams *et al.* 2000). Attracted by the natural resources of the area, settlers colonised the Newcastle region in 1798 (Paterson 1801). The first century following settlement was characterised by the development of logging, cropping, stock grazing that saw a steady clearing of native vegetation (including mangrove forests) and and shipping activities that included dredging (begun in 1859) deposition of spoil from the main river channel. Kooragang Island was mapped in the 1800s as a complex of small islands (Williams *et al.* 1999, Williams *et al.* 2000), but the placement of dredging spoil in the channels separating these islands resulted in a new and larger entity that was given the name Kooragang Island in 1951. The establishment of a steel and coal industry in the late 1800s and the growth of Newcastle to become the largest coal port in the southern hemisphere by the end of the 20th century saw intensive engineering works undertaken at the mouth of the Hunter River and Kooragang Island. Drainage infrastructure (levee banks, culverts, pipelines, the construction of service roads and further land clearing) occurred on the island, and the composite effect of these works was to significantly impact on the hydrology of the wetlands, greatly restricted the tidal flushing within the array of tidal creeks dissecting the island that had not already been filled with dredge spoil. Although the ecological consequences of the altered hydrology and land use were acknowledged early (e.g. Woolls 1867), it was not until after considerable degradation that the Kooragang Nature Reserve was gazetted in 1983 (Williams *et al*. 2000).

2.1.2. Research design

For a rigorous evaluation of the response to rehabilitation at Kooragang Wetland, it was necessary to compare creeks and marshes of two localities where culverts were removed (manipulated), to replicated reference locations (creeks and marshes not under the influence of culverts) and appropriate control locations (creeks and marshes impacted by culverts which were not being removed). By monitoring these locations at replicated times both before and after culvert removal we were able to attribute any changes in the wetlands to the manipulations (Underwood 1991). Such a design is generally based around the hypothesis that the manipulated locations should become similar to the reference locations, but, similar changes must not occur in the control locations if the improvements are to be attributed to culvert removal (Grayson *et al.* 1999).

This experimental design is detailed in Table 1**,** Table 2 and Fig. 2. It is based around the complex hydrology of Kooragang Island and the strategic way in which the KWRP applied culvert upgrades (see Williams *et al.* (1995) for more detail). The design was spatially-balanced so that two manipulated creeks and marshes were compared to two control and two reference creeks and marshes (Table 1). All creeks were sampled for two years prior to culvert removal and in eight years across the 14 years following culvert removal (Table 2) with seine and gill nets. The temporal design of marsh sampling was more restrictive due to changes in the ability to sample marshes throughout the project. It involved sampling the marshes in the year proceeding culvert removal and for two years after (Table 2).

Table 1. The overall spatial design applied during the study. Particular components of this design were used to test different hypotheses in different chapters as specified in respective chapters. Sites are shown in Fig. 2.

Treatment	Habitat	Location*	Sampling method	Sites
Manipulated (culverts removed during study)	Tidal Creek	Fish Fry Creek (Ck 5)	Seine	3
			Gill	2
		Crabhole Creek (Ck 1)	Seine	3
	Marsh	Fish Fry Flats (P5)	Fyke	3
		Crabhole Flats (P1)	Fyke	3
Control (remained restricted by culverts)	Tidal Creek	Dead Mangrove Creek (Ck 3)	Seine	3
			Gill	2
		Wader Creek (Ck 4)	Seine	3
			Gill	2

	Marsh	Swan Pond (SP)	Fyke	3
		Wader Pond (P4)	Fyke	3
Reference (no culverts, remained unrestricted during study)	Tidal Creek	Cobbans Creek (Ck 2)	Seine Gill	3 2
		Mosquito Creek (Ck 6)	Seine Gill	3 2
	Marsh	Cobbans Marsh (NP1)	Fyke	3
		Milhams Pond (NP3)	Fyke	3

* Codes in Parentheses refer to historical names given to locations in a previous interim report (Williams *et al.* 1995)

Table 2. Overall temporal design applied during the study. Particular components of this design were used to test different hypotheses in different chapters as specified in each respective chapter.

Habitat	Before/After (B/A)	Years (Yr)	Months (Mo)
Tidal marsh	x2	(x1 nested in B, x2 nested in A)	(x4, nested in Yr)
	Before (B)	1994/95 (before) (Yr 2)	November
	After (A)	1995/96 (after) (Yr 3)	December
		1996/97 (after) (Yr 4)	January
			February
Tidal creek	x2	(2 Yr nested in before, 8 Yrs nested in after)	(x3, nested in Yr)
	Before (B)	1993/94 (before) (Yr 1)	December
	After (A)	1994/95 (before) (Yr 2)	January
		1995/96 (after) (Yr 3)	February
		1996/97 (after) (Yr 4)	
		1997/98 (after) (Yr 5)	
		1998/99 (after) (Yr 6)	
		2000/01 (after) (Yr 8)	
		2002/03 (after) (Yr 10)	
		2004/05 (after) (Yr 12)	
		2008/09 (after) (Yr 16)	

2.1.3. Fish and decapod sampling

Fish and decapod crustaceans were collected during spring tide in daylight hours from late spring to summer to coincide with the summer recruitment period. Saltmarsh in southeast Australia is only fully inundated during the spring tide. All sites within a sampling month were visited within one week and different gear types were used in marshes and creeks (Table 3). Marshes were sampled using three fine mesh (3mm) fyke nets (5m L x 200mm H x 400mm W), with 2m wings and central septum 3m in length. Fyke nets were set haphazardly approximately one hour before the turn of high tide with a soak time of approximately two hours.

Tidal creeks were sampled using three replicate seine net hauls (10m headline x 1.5m drop x 3mm stretch mesh), performed in a 'U'-shape and pursed onto the shore. Hauls were spaced no closer than ~10m apart. A separate pilot study utilizing up to eight seine hauls was performed on various occasions and this verified that three hauls were adequate in capturing the vast majority of species present, with species seldom added with subsequent hauling (R. J. Williams unpublished data). Other studies employing this sampling method have also found that three seine hauls in an area captured 86% of species present (Kroon and Ansell 2006). Seine netting was commenced shortly before high tide and completed shortly after to coincide with maximum depth and minimum velocity. Seine netting was knowingly biased toward the capture of small-bodied species or juveniles of larger commercially important species. To in part overcome this bias, microfilament floating gill nets of various mesh sizes (two panels 10m in length with a 2m drop, each of 25, 50, 75 and 100mm mesh) were also used to target larger fish in tidal creeks (except Crabhole Creek, as it was too shallow to use gill nets). Panels were set haphazardly from alternate banks with the order of mesh sizes randomised, and at 45^{o} angles across the channel into the incoming tide with one end secured to the bank and the other weighted. Gill nets were set approximately one hour before the turn of high tide and allowed to soak for approximately two hours.

Fish and decapods were placed in buckets of estuarine water prior to identification to species level; fork length was recorded for fish and carapace length for decapods. After measurement, all live specimens were returned to the water.

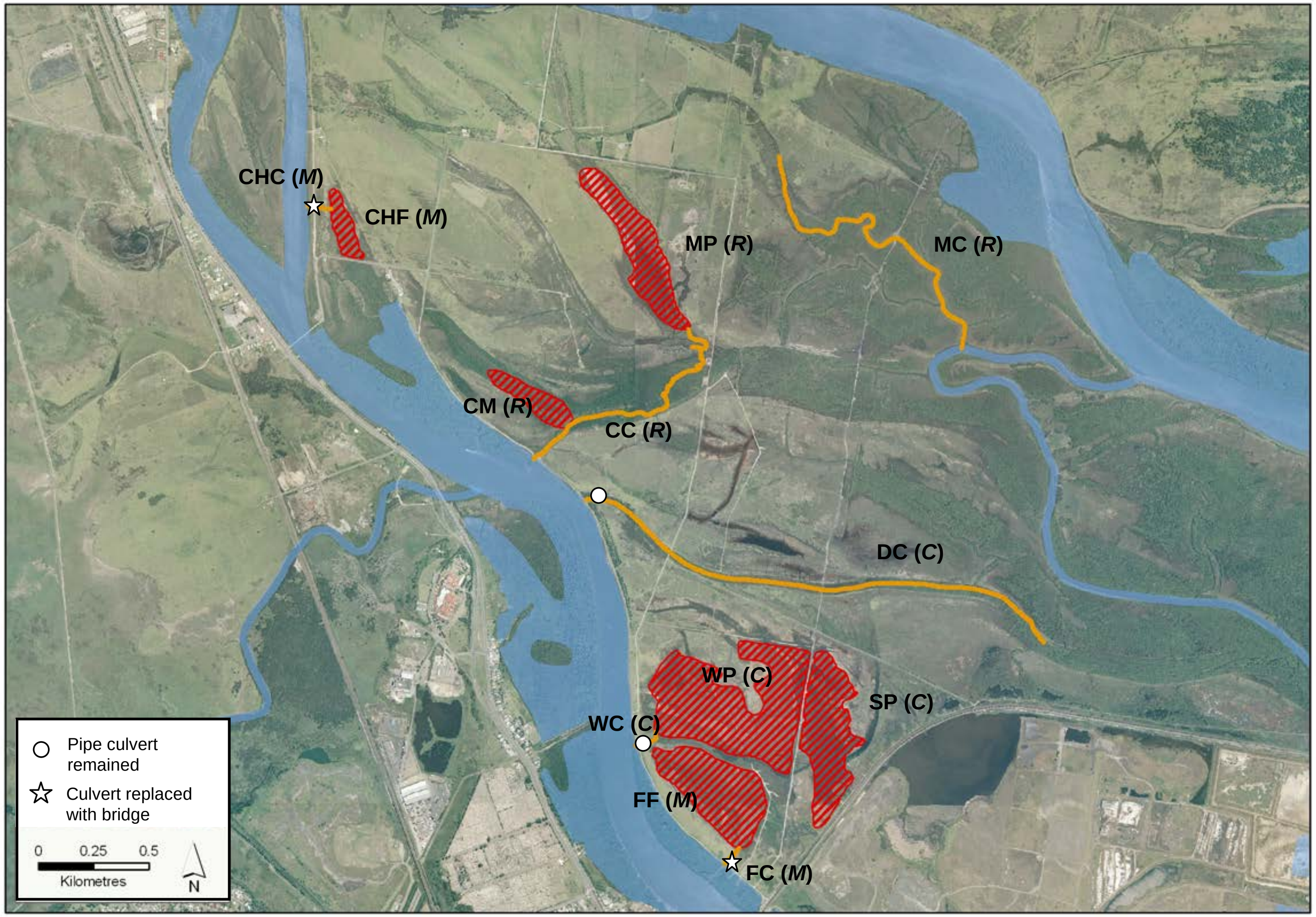

Fig. 2. Control (*C*; tidally-restricted throughout study), Reference (*R*; unrestricted throughout study) and Manipulated (*M*; tidal restriction removed throughout study) marsh and creek locations in Kooragang Island. Refer to Table 1 for the assignment of locations and replicated sites within the BACI design.

CHC (M) = Crabhole Creek
CHF (M) = Crabhole Flats
CM (R) = Cobbans Marsh
CC (R) = Cobbans Creek
MP (R) = Milhams Pond
MC (R) = Mosquito Creek
DC (C) = Dead Mangrove Creek
WC (C) = Wader Creek
WP (C) = Wader Pond
SP (C) = Swan Pond
FFF (M) = Fish Fry Flats
FFC (M) = Fish Fry Creek

3. THE IMPACT OF TIDAL RESTRICTIONS ON MARSH AND CREEK ASSEMBLAGES

3.1. Introduction

The rapid upsurge in the development of coastal areas over recent decades has disturbed large areas of wetland and placed enormous pressure on their biodiversity and sustainability (Vitousek *et al.* 1997, Edgar *et al.* 2000). One common type of disturbance has been the loss of connectivity that results when structures such as levees, floodgates and culverts restrict the tidal amplitude and fragment habitats (NSW Fisheries 1976, Roman *et al.* 1984, Daiber 1986, Pollard and Hannan 1994, Streever 1997, Williams and Watford 1997, Dick and Osunkoya 2000, Kroon *et al.* 2004, Kroon and Ansell 2006). In coastal New South Wales there are over 4,200 anthropogenic structures restricting tidal flow to wetlands, many of which are culverts (Williams and Watford 1997). Whilst most are deemed candidates for removal or manipulation to improve tidal flushing, in most cases there are insufficient data to ascertain the actual severity of restriction and subsequent ecological impact. This makes it impossible to prioritise coastal rehabilitation projects, set appropriate rehabilitation goals and establish benchmarks against which to measure rehabilitation success (Zedler and Callaway 1999).

In this chapter we assessed the impact of tidally-restrictive road culverts on the fish and decapod assemblages of salt marsh and tidal creek habitats of Kooragang Island. It was hypothesised that the assemblages of restricted wetland habitats would differ in species richness as well as relative abundance from those of unrestricted wetlands. Further, it was hypothesised that much of this difference would be due to having fewer species whose life cycles are associated with a need for connectivity between wetlands and the rest of the estuary. The ecological implications of our findings are discussed and the results used to formulate hypotheses relating to fish and decapod responses to culvert removal at Kooragang Island.

3.2. Methods

3.2.1. Experimental design

In this chapter, only a component of the full design outlined in section 2.1.2, page 5 was used to test the hypothesis relating to assemblage differences between tidally-restricted and unrestricted creeks and marshes. This included control and reference sites across three years for marsh sites and four years for creek sites (Table 3).

3.2.2. Statistical analyses

Differences in the composition (species type and abundances) of the combined fish and decapod assemblage between tidally-restricted versus unrestricted marsh and creeks were investigated using the PRIMER v6 suite of non-parametric multivariate analyses (Clarke and Gorley 2006). Marsh data were analysed separately from the creek data and seine net data were analysed separately from the gill net data within the creeks. Seine and fyke net catches were analysed as replicate samples. Gill net catches were combined for different mesh sizes and analysed as two multi-mesh replicates. To reduce the “swamping” of very abundant species (e.g. ambassids, eleotrids, shrimps) on assemblage differences, all data were fourth root transformed (Clarke and Green 1988) before calculating Bray-Curtis dissimilarities between samples (Bray and Curtis 1957).

One-way Analysis of Similarity (ANOSIM: Clarke 1993) was used to determine the degree of assemblage difference between tidally-restricted versus unrestricted marshes and creeks. Assemblage differences were interpreted based on R values, with values approaching 1 indicating greater dissimilarity among groups (Clarke and Gorley 2006). Patterns in the combined fish and decapod assemblage data were also interpreted visually with non-metric multidimensional scaling ordinations (nMDS; Kruskal and Wish 1978). The plots depict the centroids of the multiple replicates at each site and time and therefore each ordination point represents a single site by time record. The distances among centroids were obtained for these ordinations using principal coordinates analyses carried out on the original Bray-Curtis dissimilarity matrices as described by Terlizzi *et al.* (2005).

To examine impact of tidal restriction on the types of species occupying marsh and creek habitats, species were classified according to their tendency to move between different sections of the saltwater to freshwater continuum that exists from the main estuary to ponded freshwater sections beyond the tidal influence. The following categories were used (after Pollard and Hannan 1994, Genders 2001):

- *Estuarine-marine* (E-M) are saltwater species that are primarily estuarine-marine dwelling as adults
- *Freshwater-estuarine* (F-E) are euryhaline species equally well adapted to saline or freshwater habitats as adults
- *Freshwater* (F) species are those typically confined to freshwater.

ANOVA was used to test for differences in the total number of species and for different salinity categories. The SIMPER procedure (Clarke 1993) was used to identify those species that were most important in differentiating tidally-restricted versus unrestricted marshes and creeks. Species were presented if they exceeded an arbitrary threshold value of percent dissimilarity ≥3% (Terlizzi *et al.* 2005). The consistency ratio (Dissimilarity/SD), calculated for all important species, indicated whether a species consistently contributed (values >1) to the average dissimilarity between restricted and unrestricted habitats in the majority of site by time comparisons.

Table 3. Spatial and temporal sampling design to test for differences in fish and decapod assemblages between tidally-restricted and unrestricted marshes and tidal creeks on Kooragang Island. See Fig. 2 and 3 for map of locations.

Habitat	Locations	Replicate samples	Years	Months
Marsh	(x4)	(nested in sites)	(x3)	(x4, nested in years)
	Cobban's Marsh (unrestricted)	Fyke nets (x3)	1994/'95	November
	Milhams Pond (unrestricted)		1995/'96	December
	Wader Pond (tidally-restricted)		1996/97	January
	Swan Pond (tidally-restricted)			February
Tidal creek	(x4)	(nested in sites)	(x4)	(x3, nested in years)
	Cobbans Creek (unrestricted)	Seines (x3)	1993/'94	December
	Mosquito Creek (unrestricted)	Multi-panel Gill nets (25, 50, 75, 100mm) (x2)	1994/'95	January
	Dead Mangrove Creek (restricted)		1995/'96	February
	Wader Creek (restricted)		1996/97	

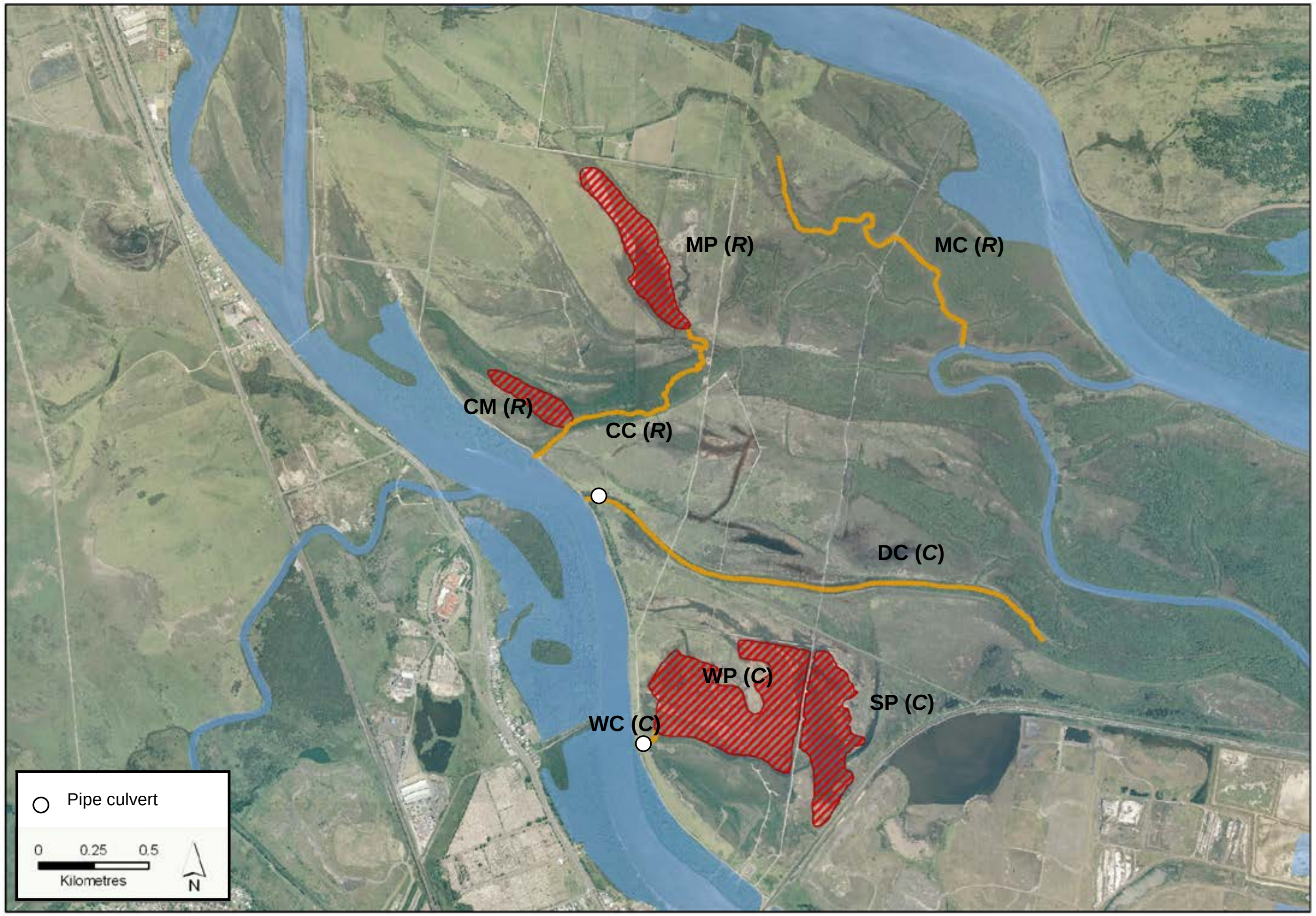

Fig. 3. The subset of locations that were used in the comparison between tidal creek and tidal marsh locations restricted by culverts (controls *C*) and unrestricted (references *R*).

CM (R) = Cobbans Marsh
MP (R) = Milhams Pond

WP (C) = Wader Pond
SP (C) = Swan Pond

CC (R) = Cobbans Creek
MC (R) = Mosquito Creek

DC (C) = Dead Mangrove Creek
WC (C) = Wader Creek

3.3. Results

3.3.1. Marsh assemblages

A total of 13,317 fish and 6,213 decapods (27 and 10 species respectively) were sampled from marsh habitats over three years (Table 4 and Appendix Table A1). At any given marsh at any given sampling time species richness was generally low (mean = 6.15 ± 0.45 species). Fish and decapod assemblages were similar between tidally-restricted and unrestricted marshes (Fig. 4; ANOSIM R=0.295 $p<0.05$), however, there were some notable differences. There were generally fewer species in tidally-restricted (5.04 ± 0.69 species) when compared to unrestricted (7.25 ± 0.49) marshes (ANOVA: $F_{(1,46)}=6.8767$, p=0.01). This difference was due to there being fewer estuarine-marine species in restricted (3.29 ± 0.51) when compared to unrestricted (5.21 ± 0.32) marshes (ANOVA: $F_{(1,46)}=10.1944$, $p<0.01$).

The abundance of individuals also differed, with estuarine-marine species such as glass goby (*Gobiopterus semivestitus*), Swan River goby (*Pseudogobius olorum*), and southern blue-eye (*Pseudomugil signifier*) ,striped shrimp (*Macrobrachium intermedium*) and pink shrimp (*Acetes sibogae australis*) being more abundant in the unrestricted marshes. Importantly, only the glass goby was consistent in this difference across most sites and times, with the other species showing a larger degree of variability between sites and/or times (Table 5: Diss/SD <1 for all species except glass goby). Two species of fish were notable by their high abundances in the restricted marshes. One of these was the introduced mosquitofish (*Gambusia holbrookii*) (Table 5); the other was the silver biddy (*Gerres ovatus*), with 674 individuals caught in the Wader Pond, with none taken at the other restricted marsh location and considerably fewer by an order of magnitude at the unrestricted locations (Appendix Table A1).

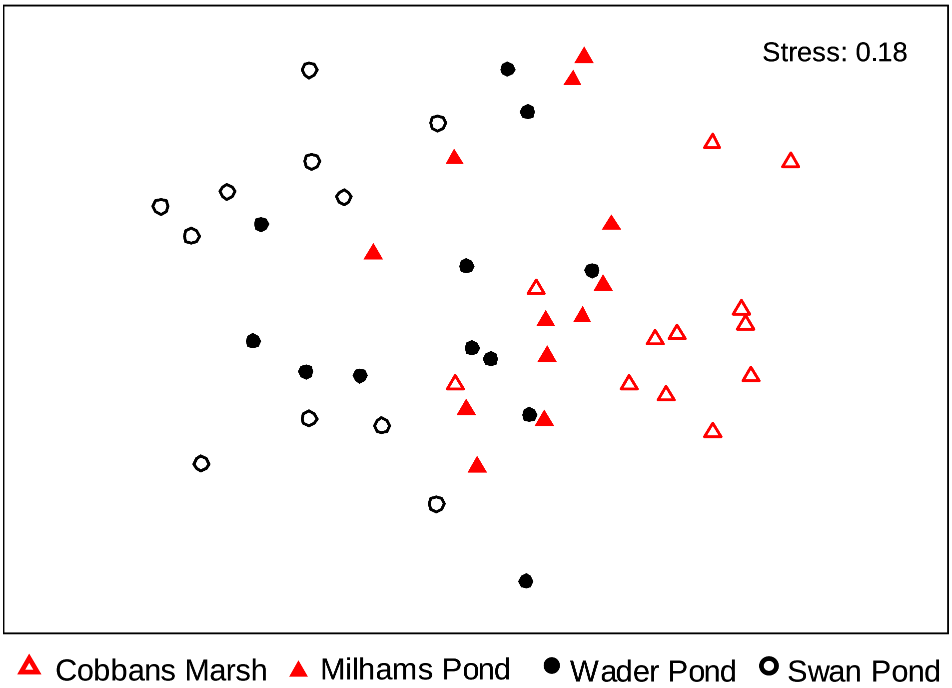

Fig. 4. Two-factor nMDS showing dissimilarity (Bray-Curtis, forth root transformed) between different **MARSH** assemblages on Kooragang Island at each time of sampling (total of 4 months in each of 3 years). Each data point is the centroid of three fyke net replicate samples. Black circles represent those marshes which are tide-restricted and red triangles represent those marshes which are unrestricted.

Table 4. Summary of catch of fyke nets from unrestricted (reference) and restricted (control) MARSH locations on Kooragang Island, 1994/95-1996/97. Abundances for the three most numerous species are shown. See Appendix, Table A1 for all species totals.

Category	Total	Unrestricted marshes no culverts (reference)	Restricted marshes culverted (controls)
NUMBER OF SPECIES (richness)			
Fish	**27**	**22**	**20**
Non C/R species	17	15	15
C/R species	10	7	5
E-M species	15	11	12
F-E species	11	10	7
F species	1	1	1
Decapods	**10**	**9**	**8**
Non C/R species	8	7	7
C/R species	2	2	1
E-M species	9	8	7
F-E species	1	1	1
F species	0	0	0
ABUNDANCE			
Fish	**13,317**	**10,404**	**2913**
Non C/R species	12,548	10,316	2,232
C/R species	769	88	681
Glass goby	8,159	7,990	169
Swan River goby	1,488	728	760
Southern blue-eye	1,184	1,177	7
Decapods	**6,213**	**4,999**	**1,214**
Non C/R species	6,204	4,991	1,213
C/R species	9	8	1
Striped shrimp	5,334	4,246	1,088
Pink shrimp	717	643	74
Shore crab	135	88	47

C/R = species of commercial and/or recreational significance.
E-M = Estuarine-Marine, F-E = Freshwater-Estuarine, F = Freshwater

Table 5. Results of SIMPER analysis showing those species contributing most to the Bray-Curtis dissimilarity (4th root transformed) between tidally-restricted and unrestricted MARSHES on Kooragang Island. Species are ranked in decreasing order of percent contribution with their average abundance in unrestricted and restricted marsh samples shown (untransformed average abundance in parentheses). Only those species most important in discriminating marsh types (Av.Diss ≥3%) are shown. The treatment (restricted or unrestricted) with the largest abundance of each species is shown in bold.

Species	Salinity	Average dissimilarity = 59.73 (79.88)		Av. Diss	Diss/SD	Contrib %
		Unrestricted Av. Abund.	Restricted Av. Abund.			
Glass goby *Gobiopterus semivestitus*	E-M	**2.05 (110.97)**	0.44 (2.34)	17.26	1.08	21.41
Swan River goby *Pseudogobius olorum*	E-M	**1.33 (10.11)**	0.87 (10.56)	11.59	0.95	14.37
Mosquitofish *Gambusia holbrooki*	F-E	0.13 (0.63)	**1.00 (13.86)**	8.88	0.77	11.01
Striped shrimp *Macrobrachium intermedium*	F-E	**0.96 (58.97)**	0.55 (15.11)	8.00	0.89	9.92
Mangrove goby *Mugilogobius paludis*	E-M	**0.53 (2.38)**	0.45 (3.24)	5.36	0.86	6.65
Pink shrimp *Acetes sibogae australis*	E-M	**0.62 (8.93)**	0.12 (1.03)	5.25	0.67	6.52
Shore crab *Paragrapsus laevis*	E-M	**0.36 (1.22)**	0.27 (0.65)	4.56	0.61	5.65
Southern blue-eye *Pseudomugil signifer*	F-E	**0.53 (16.35)**	0.06 (0.10)	4.36	0.60	5.41

Salinity tolerance: Freshwater-Estuarine (F-E), Estuarine-Marine (E-M)

3.3.2. Tidal creek assemblages

A total of 55,951 fish and 26,975 decapods (38 and 15 species respectively) were sampled from creek habitats over four years (Table 6 and Appendix Table A2). More species were caught in seine nets (n=52) than gill nets (n=15), and, the vast majority of individuals (82,551, 99.5% of total catch) were caught in seine nets. Southern herring (*Herklotsichthys castelnaui*) was the only species caught exclusively in gill nets. As expected, gill nets were more effective at catching larger bodied fish (>200mm; Fig. 5), however this size class equated to only a very small proportion of the total catch (20 individuals out of the total 82,926).

The catch of fish was dominated by glass goby (*Gobiopterus semivestitus*, 78% of total abundance), while the decapods were dominated by pink shrimp (*Acetes sibogae australis*, 73%) and striped shrimp (*Macrobrachium intermedium*, 24%). Species of commercial and/or recreational significance did not figure prominently in the catch, with flat-tail mullet (*Liza argentea*, 2%) and school prawn (*Metapenaeus bennettae*, 2%), having third highest abundances in each of the fish and decapod categories, respectively.

Seine net sampling revealed that assemblages were distinct between tidally-restricted and unrestricted creeks (Fig. 6a; ANOSIM R=0.685 $p<0.05$). In comparison, the low abundances, few species, and prevalence of zero catches obtained by gill nets made it impossible to distinguish between assemblages at the different treatments or locations using this gear type

(Fig. 6b; ANOSIM R (*restricted versus unrestricted*)=0.211, R(*among locations*)=0.161, p<0.05). As a result, gill net data was excluded from subsequent SIMPER tests in this chapter.

The unrestricted creeks generally had more estuarine-marine and freshwater-estuarine species (Fig. 7). Glass goby, flat-tail mullet, Port Jackson glassfish and pink shrimp (all estuarine-marine) were more abundant in the unrestricted creeks (Table 7). Although gill net catches were excluded from SIMPER analysis, (Table A2 in the Appendix) shows that fantail mullet (*Paramugil georgii*) was also more abundant in the unrestricted creeks. In comparison, Swan River goby, mangrove goby (estuarine-marine) and striped shrimp (freshwater-estuarine) were more abundant in restricted creeks (Table 7).

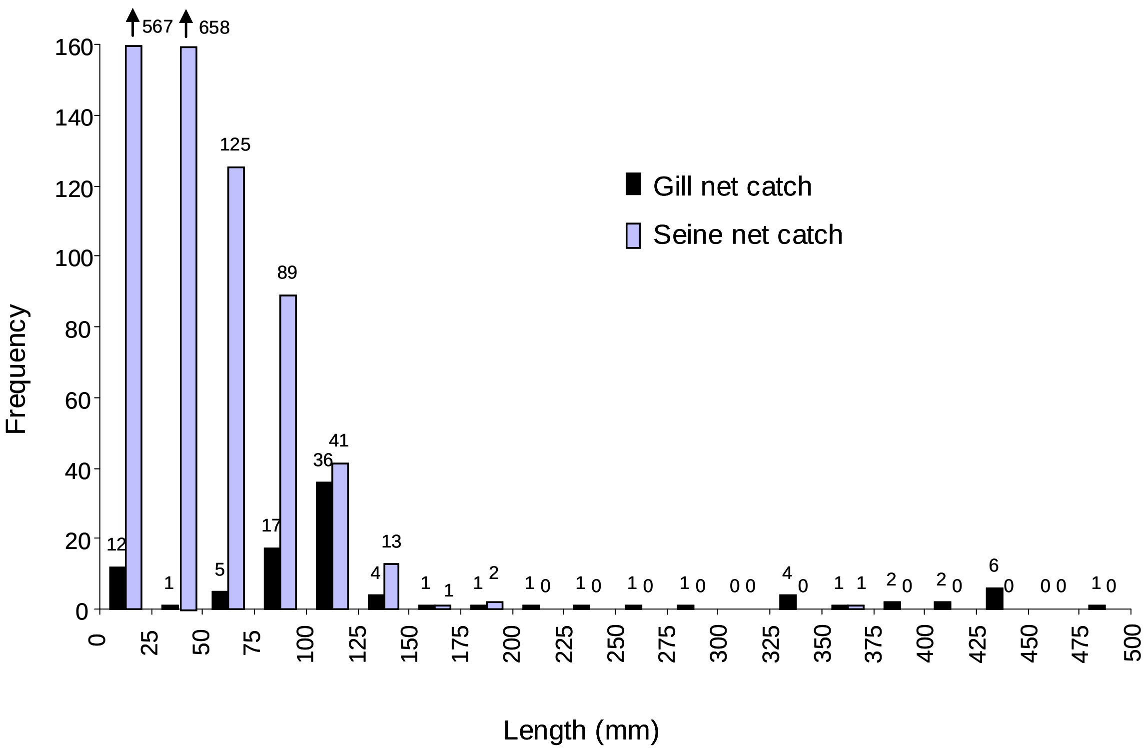

Fig. 5. Length frequency distribution for fish and decapods caught in tidal creeks, showing numbers caught with gill and seine nets.

Table 6. Summary of catch of seine nets from each of the unrestricted (reference) and restricted (control) **CREEKS** on Kooragang Island, 1993/94-1996/97. Abundances for the three most numerous species are shown. See Appendix, Table A2 for all species totals.

Category	Total	Unrestricted creeks no culverts (reference)	Restricted creeks culverted (controls)
NUMBER OF SPECIES (richness)			
Fish	**38**	**35**	**23**
Non C/R species	19	18	14
C/R species	19	17	9
E-M species	27	26	16
F-E species	11	9	7
F species	0	0	0
Decapods	**15**	**14**	**13**
Non C/R species	11	10	9
C/R species	4	4	4
E-M species	40	38	27
F-E species	13	11	9
F species	0	0	0
ABUNDANCE			
Fish	**55,951**	**42,192**	**13,759**
Non C/R species	53,778	40,378	13,400
C/R species	2,173	1,814	359
Glass goby	43,045	37,427	5,618
Swan River goby	5,116	852	4,264
Flat-tail mullet*	1,352	1,274	78
Decapods	**26,975**	**21,892**	**5,083**
Non C/R species	26,265	21,290	4,975
C/R species	710	602	108
Pink shrimp	19,588	19,175	413
Striped shrimp	6,527	2,075	4,452
School prawn*	494	481	13

C/R = species of commercial and/or recreational significance.
E-M = Estuarine-Marine, F-E = Freshwater-Estuarine, F = Freshwater
* = C/R species

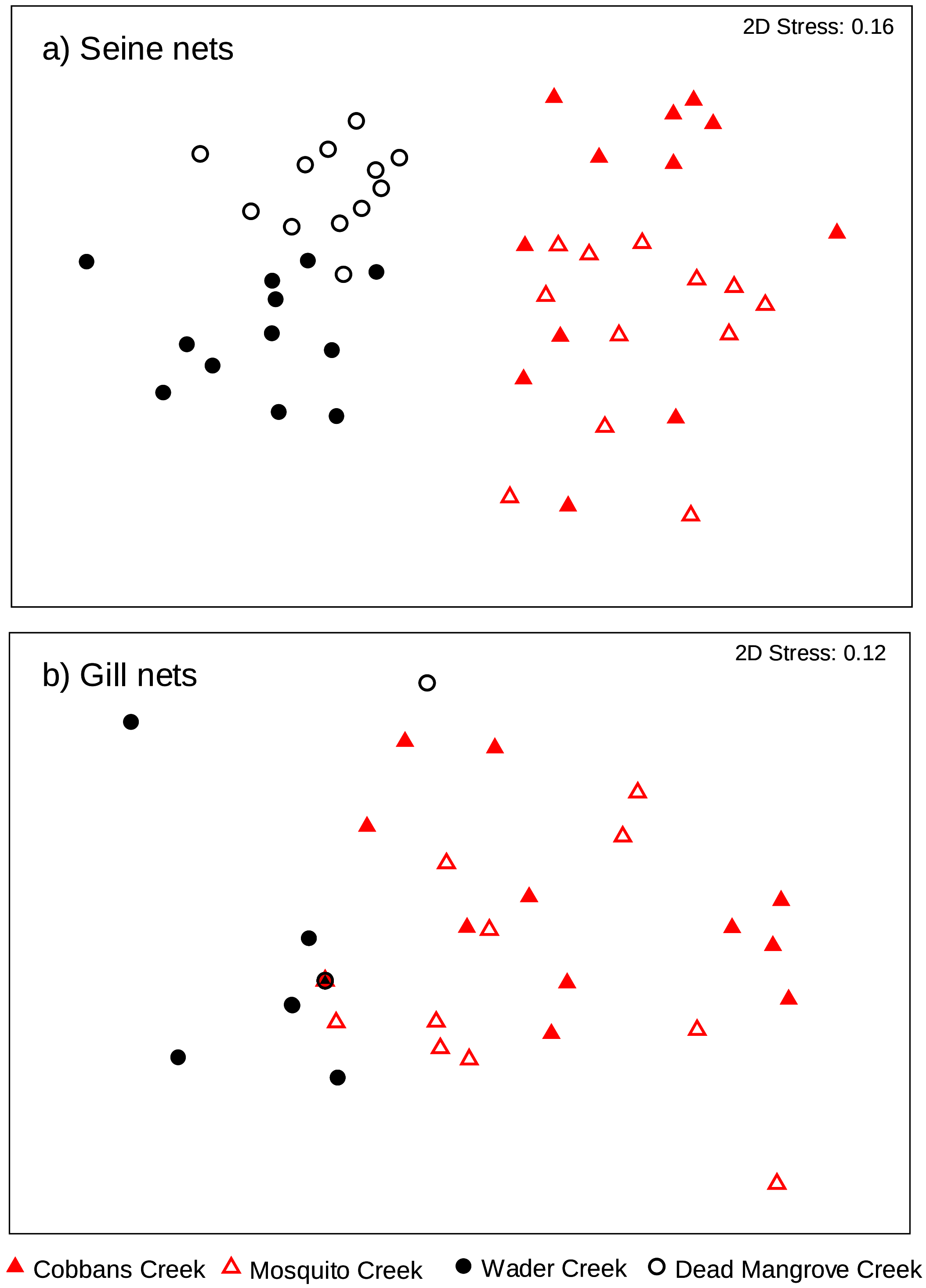

Fig. 6. Two-factor nMDS showing dissimilarity (Bary-Curtis, forth root transformed) between different creek assemblages on Kooragang Island at each time of sampling (total of 3 months in each of 4 years). Each data point is the centroid of a) three seine net replicate samples or b) two gill net replicate samples. Black circles show those creeks which are tide-restricted and red triangles represent those creeks which are unrestricted. Absence of a centroid point/s occurred in some instances due to an absence of fish.

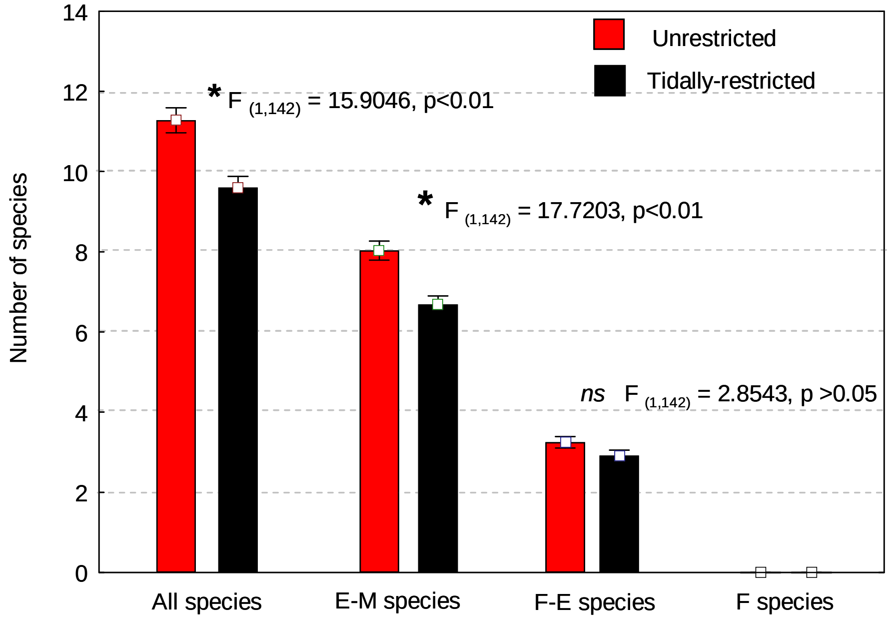

Fig. 7. Species richness of fish and decapod assemblage caught by seine nets within unrestricted (red) and tidally-restricted (black) **CREEKS**. Species split by estuarine-marine (E-M), freshwater-estuarine (F-E) and Freshwater (F). * indicates statistically significant comparison between restricted and unrestricted creeks (ANOVA).

Table 7. Results of SIMPER analysis showing those species contributing most to the Bray-Curtis dissimilarity (4th root transformed) between tidally-restricted and unrestricted **CREEKS** on Kooragang Island. Species are ranked in decreasing order of percent contribution with their average abundance in each unrestricted and restricted creek samples shown (untransformed average abundance in parentheses). Only those species most important in discriminating marsh types (Av.Diss ≥3%) are shown and the treatment (restricted or unrestricted) with the largest abundance of each species is shown in bold.

		Average dissimilarity = 80.62				
Species	Salinity	Unrestricted Av. Abund.	Restricted Av. Abund.	Av. Diss	Diss/SD	Contrib %
Pink shrimp *Acetes sibogae australis*	E-M	**2.67** **(266.32)**	0.66 (5.74)	6.13	1.33	10.27
Glass goby *Gobiopterus semivestitus*	E-M	**3.64** **(519.82)**	2.48 (78.03)	4.66	1.07	7.81
Mangrove goby *Mugilogobius paludis*	E-M	0.21 (0.32)	**1.71** **(16.86)**	4.17	2.02	6.99
Swan River goby *Pseudogobius olorum*	E-M	1.31 (11.83)	**2.55** **(59.22)**	3.76	1.46	6.30
Striped shrimp *Macrobrachium intermedium*	F-E	1.72 (28.82)	**2.21** **(61.83)**	3.25	1.24	5.45
Flat-tail mullet *Liza argentea*	E-M	**1.31** **(15.78)**	0.40 (1.08)	3.25	1.22	5.44
Port Jackson glassfish *Ambassis jacksoniensis*	E-M	**1.12** **(12.60)**	0.00 (0)	3.03	1.12	5.08

Salinity tolerance: Freshwater (F), Freshwater-Estuarine (F-E), Estuarine-Marine (E-M)

3.4. Discussion

There was evidence that the utilisation of Kooragang Wetlands by fish and decapods has been impacted by tidally-restrictive culverts. In this study the composition of fish and decapod assemblages differed between all tidally-restricted and unrestricted creeks. Species richness was significantly reduced in the culverted creeks, with the greatest reduction seen for estuarine-marine dwelling species. This difference was due to a difference in the number of fish species rather than decapod species. In comparison, no effect was seen on the number of freshwater-estuarine species. The differential impact of culverts on these two groups of species likely reflects the fragmenting effect of culverts on wetland habitats. Since estuarine-marine species will be reliant on dispersal between the wetland and the rest of the estuary, they would be disadvantaged by structures such as culverts which restrict their movements (Eberhardt *et al.* 2011). In comparison, it is feasible that freshwater-estuarine species, which are euryhaline and tolerant of larger salinity ranges, may be more resistant to extreme ranges in salinity that can occur in tidally-restricted wetlands. Such extremes have been shown to increase when tidal exchange is reduced because the salinity of upstream habitats fluctuates between periods of elevated freshwater input from local rainfall and hypersaline conditions when evaporation rates are high during summer periods (Weinstein *et al.* 1980).

The differences between the assemblages in the restricted and unrestricted marsh habitats was more subtle than in creeks. Within the tidal marshes the number of estuarine-marine species was about the same in the unrestricted (n=31) and restricted locations (n=28), although a few species were unique to each habitat (Appendix, Tables A1 and A2). This has been observed in other studies (Raposa and Roman 2003, Eberhardt *et al.* 2010) and suggests that marsh habitats upstream of tidally restrictive culverts can still maintain viable fish and decapod assemblages. Despite this, there did appear to be difference in the relative abundance of some species between restricted and unrestricted marshes that may suggest some impact of culverts and a shift in distribution within the wetland (discussed below).

The proportion of commercial-recreational species is much greater in creeks than in marshes, but in neither habitats do the commercial-recreational species form a large proportion of the catch. At the onset of the study it was anticipated that tidal wetland habitats of Kooragang Island might operate as a nursery for species of commercial and/or recreational significance. It appears that they do not do so in any large measure. Instead, the value of these wetlands may reside more in their ability to provide refuge for smaller species and it is therefore these species, with benefits for commercial-recreational species being indirectly related to the role of wetland species in cascading energy transfer into estuarine food webs (discussed below).

The most abundant species captured during this project have been little studied and so their role in estuarine function is not well known. But, their extremely high abundances would suggest that they may be significant in estuarine ecology. The glass goby (*Gobiopterus semivestitus*) and Swan River goby (*Pseudogobius olorum*) are the pre-eminent fish and the striped shrimp (*Macrobrachium intermedium*) and pink shrimp (*Acetes sibogae australis*) are the pre-eminent decapods in this regard. While few shore crabs (*Paragrapsus laevis*) were captured, this species has been strongly implicated in supporting estuarine food chains due to the seasonal discharge of larvae that have been shown to be a major component of the diet of small estuarine fish that in turn may be consumed by larger predators (Mazumder *et al.* 2006).

Two species of introduced fish were found. One, the yellow-fin goby (*Acanthogobious flavimanus*), is thought to have been carried to Australia in the ballast water of bulk carriers trading from the northwest Pacific (Hutchings 1992). This species was not captured in the restricted marshes, but 11individuals were found across the other three habitat types (Tables 4 and 6). The other introduced species, the mosquitofish (*Gambusia holbrooki*), was present in restricted marshes as well as creeks, being one of the most dominant species in the former habitat. This species is a live-bearer, can complete its entire life cycle within landlocked areas and is considered tolerant to habitat disturbance (McDowall 1996). It has also been reported in higher abundances in creeks tidally restricted by floodgates (Kroon and Ansell 2006). In large numbers, the mosquitofish has the ability to alter food webs by feeding heavily on rotifer, crustacean, and insect populations, subsequently increasing phytoplankton populations and impacting on nutrient cycling in ecosystems (Hurlbert *et al.* 1972, Hurlbert and Mulla 1981). They have also been shown to prey heavily on frog eggs and tadpoles (Grubb 1972, Baber *et al.* 2004), including the eggs and tadpoles of the endangered green and golden bell frog (*Litoria aurea*) (Pyke and White 2000), a nationally protected species found to co-inhabit disturbed marshes of Kooragang Wetland (Hamer *et al.* 2002).

Culverts may modify floral and faunal wetland assemblages either through reduced nekton passage rates and/or habitat modification from altered hydrology. Pipe culverts of small diameter, such as the those at control sites and those that were replaced in this study, have been shown to significantly reduce the passage of fish in saltmarsh habitats (Eberhardt *et al.* 2011) and their subsequent removal or enlargement has lead to rapid recolonisation by nekton (Dionee *et al.* 1999, Raposa and Roman 2003, Eberhardt *et al.* 2011). In our study, there were a number of species whose abundance was significantly lower upstream of culverts, suggesting that inward passage for pelagic species was hampered, or, that this part of the wetland was suboptimal for survival. For example, pelagic species such as southern blue-eye, glass goby, striped shrimp and pink shrimp were more abundant in unrestricted marshes, and the pelagic Port Jackson glassfish, glass goby, flat-tail mullet, fan-tail mullet and pink shrimp were more abundant in unrestricted creeks (Table 8). Glass goby and pink shrimp seem to be more abundant in unrestricted habitats regardless of whether they are marsh or creek. Some demersal species, such as mangrove goby and Swan River goby are widely distributed; however, they are rare in the unrestricted creeks perhaps because the velocity associated with tidal flow has modified bottom sediments or some other feature associated with a demersal lifestyle. Another possibility is that they are more accessible to larger predators in the unrestricted creeks. The mosquitofish appears to do particularly well in only in restricted marshes. Such a gradation of favoured and unfavoured habitats has implications in relation to types of manipulation to rehabilitate habitats to favour certain assemblages.

In addition to reducing faunal passage rates, tidally-restrictive culverts greatly alter hydrology, which in turn changes bathymetry, impacts on water quality, and interferes with the cascade of nutrients through the food chain (Roman *et al.* 1984, Streever and Genders 1997, Raposa 2008, Howe *et al.* 2009, Howe *et al.* 2010). These physical habitat changes have been observed at the culverted locations examined during this study (Streever and Genders 1997, Howe *et al.* 2009, Howe *et al.* 2010) and are likely to have contributed (at least in part) to the changes in species abundance observed for some species. When culverts were present, species such as striped shrimp, Swan River goby and mangrove goby were less abundant in marshes and these were paralleled with increased abundance in creeks. This apparent movement from marshes into creeks may have a number of causes. Striped shrimp (more commonly referred to by their genus name *Macrobrachium*) and Swan River goby are euryhaline species that may have an expanded range in tidally-restricted creeks that tend to be more brackish and salinities range between greater extremes (Roman *et al.* 1984).

Differences in fish and decapod abundance between creek and marsh habitats that is dependent of the level of tidal restriction implies a gradation in habitat that is worthy of further investigation of hydrodynamic conditions at the respective locations. A better understanding of hydroperiod, tidal prism, invert and obvert levels as they relate to different types of assemblages would assist in setting rehabilitation targets for other projects.

Table 8. Summary of occurrence (denoted as X) of most abundant fish and decapod species. R = restricted, UR = unrestricted, * = present in limited numbers or absent.

	Tidal marsh abundance (fyke net)			Tidal creek abundance (seine & gill net)		
	UR>R	UR~R	UR<R	UR>R	UR~R	UR<R
FISH						
Southern blue-eye	X				X	
Port Jackson glassfish	*	*	*	X		
Silver biddy	*	*	*			X
Glass goby	X			X		
Mangrove goby		X				X
Swan River goby		X				X
Mosquitofish			X	*	*	*
Flat-tail mullet	*	*	*	X		
DECAPODS						
Striped shrimp	X					X
Pink shrimp	X			X		
Shore crab		X				X

It is worth mentioning that seine netting and gill netting caught substantially different amounts of fish and decapods. Whilst seine net catches were high in abundance of individuals and number of species, gill net samples (incorporating multi-panel meshes: 25, 50, 75 and 100mm) were dominated by zero catches, although allowed a larger size range of fish to be captured (>200mm). These larger individuals accounted for only 0.02% of the total catch. Because of this, the gill net dataset was not sufficiently large enough to detect assemblage-based differences between different creeks. One explanation of this may be that larger-bodied adult species seldom utilise tidal creeks. This has been observed in tidal creeks elsewhere (Reis and Dean 1981, Paterson and Whitfield 2000) and provides support for the hypothesis that shallow tidal creeks may function as important refugia for small fish avoiding larger predatory fish (Rozas and Odum 1988, Ruiz *et al.* 1993, Paterson and Whitfield 2000) and therefore be important fish and decapod nurseries (Beck *et al.* 2001). The apparent rarity of large adults in our study is bound by the assumption that the gill nets were an effective sampling device. There is still some contention surrounding sampling bias and gear avoidance associated with the use of gill nets in estuarine habitats (see Paterson and Whitfield (2000) and references therein). In particular, some studies suggest that gill net avoidance may be higher during the day, with one study reporting low day-time and high night-time catches of large piscivorous fish in creeks (Rountree and Able 1997). This may also reflect a movement of large-bodied piscivorous fish into creeks at night, however, other studies report no such nocturnal changes, with large-bodied fish under-represented in creeks in both day-time and night-time samples (Paterson and Whitfield 2000). It has also been suggested that short set-times for gill nets around the turn-of-tide, as adopted in this study, may not be effective at capturing highly mobile large-bodied fish such as mullet, that may only enter creeks to feed for short periods at the start of the incoming tide (Paterson and Whitfield 2000). Future research that pairs gill net samples taken within the creek with alternative habitats outside of the wetland during day and night may help to clarify this uncertainty.

Studies using stable isotope analysis (Deegan and Garritt 1997) and bioenergetic modelling (Kneib 2003) show that fish and decapods play a major role in the exchange of energy and nutrients between coastal wetlands and the estuary. Even if large adult carnivorous fish do not penetrate small tidal creeks, the free movement between marsh, creek and Hunter River of the small animals found in this study is one mechanism that drives "trophic relay" (Kneib 1986, Deegan and Garritt 1997, Kneib 2003). The restriction of passage into wetlands may therefore have impacts beyond that of the individual or population, also altering food webs and the export of secondary production from the wetland (Eberhardt *et al.* 2011).

The utilisation of Kooragang wetlands by fish and decapods has been impacted by tidally-restrictive culverts. These results lend themselves to the assumption that culvert removal and subsequent improved tidal flushing may lead to changes in fish and decapod assemblage composition in tidal creeks and a significant improvement in the number of species able to utilise these habitats. This hypothesis is further strengthened by the observation of other studies, in particular, that the short and medium-term impact of culvert removal at Kooragang wetlands has been a significant increase in tidal penetration, increased tidal conveyance between Fish Fry Creek and its associated marshes and the expansion high-value fish and decapod habitat such as mangrove and saltmarsh (Streever and Genders 1997, Howe *et al.* 2010). Our assemblage analysis suggests that the response obtained may be less pronounced in the marsh habitats further inland of these creeks, but there is enough evidence to suggest that there may be functional shifts in the assemblage in these habitats, reflecting greater utilisation from estuarine-marine dwelling species. In being able to detect the hypothesised assemblage differences between the reference (unculverted) and control (culverted) locations in this study, we can have more confidence that the experimental design adopted for fish and decapod sampling at Kooragang will have the ability to detect long-term changes that may occur following culvert removal. This assumption will now be explored using a full beyond BACI design in the next chapter.

4. TRAJECTORIES OF CHANGE IN FISH AND DECAPOD ASSEMBLAGES FOLLOWING CULVERT REMOVAL

4.1. Introduction

The fundamental premise of rehabilitation ecology is that the removal of stressors will result in the reinstatement of ecological processes necessary to move degraded systems towards a more natural or unstressed state (Simenstad *et al.* 2006). This progression of ecological recovery through time has been described by many as a pathway or trajectory towards some reference state or functional equivalency (Kentula *et al.* 1992, Hobbs and Norton 1996, Simenstad and Thom 1996). The realisation of different types of trajectories has received some debate (Zedler and Callaway 1999). In coastal wetlands there appears to be a wide variation in responses to manipulations over extended periods, which is hardly surprising given the different types of antecedent factors, degradation and management practices associated with wetland rehabilitation studies (Simenstad *et al.* 2006). Whilst some studies report a rapid and sustained path towards rehabilitation targets (Morgan and Short 2002, Warren *et al.* 2002, Able *et al.* 2008), others demonstrate large temporal variability and an unlikeliness to achieve functional equivalence to a target condition (Zedler and Callaway 1999).

Typically, the assessment of wetland modification relies on short-term comparisons (3-5 years) of the composition of aquatic assemblages in manipulated systems when compared with un-manipulated controls and/or un-impacted references (Grayson *et al.* 1999). Such studies can demonstrate rapid recovery following manipulations such as the removal of tidal restrictions (e.g. Burdick *et al.* 1996, Raposa 2008, Boys *et al.* 2011). But the short-term nature of these studies greatly limits their interpretability, as they fail to monitor over sufficiently-long timeframes to account for successional changes and system maturity (e.g. habitat use, competition and predator/prey interactions) (Zedler and Callaway 1999). As a result, short-term studies run the risk of drawing erroneous conclusions about the achievement of goals (Grayson *et al.* 1999). Given that approximately half of the coastal wetlands in south-eastern Australia have already been degraded by human disturbance (DEH 2005) and that climate change models predict further fragmentation of such estuarine environments (Vinagre *et al.* 2011), there is considerable capacity for well-intentioned but ineffective wetland rehabilitation.

The Kooragang Wetland Rehabilitation Project provided a rare opportunity to monitor over a sufficiently-long time frame to enable both transitional and successional changes in the ecology of a manipulated wetland to be evaluated. By increasing tidal flows to wetlands through the removal of culverts the value of fish and shorebird habitat was to be improved (Streever 1998). To date, research of short and medium-term responses within a before-after-control-impact framework (Underwood 1991) have been conducted at Kooragang Island on nutrient cycling (Howe *et al.* 2009), vegetation change (Streever and Genders 1997, Howe *et al.* 2010) and shore bird roosting (Kingsford *et al.* 1998). In this chapter we compare pre- and post-measures of fish and decapod crustacean assemblages (species richness and abundance) across different habitats (tidal marshes and tidal creeks) at locations where culverts were removed, with un-manipulated tidally-restricted locations as well as unrestricted reference locations. Most notably, at the tidal creeks we examined the trajectory of response over a 16 year period (14 years post-culvert removal), a timeframe seldom applied to rehabilitation studies but more applicable to determining whether rehabilitation responses become self-sustaining (Zedler and Callaway 2000).

4.2. Methods

4.2.1. Research design and sampling methods

The focus in this chapter is on *rehabilitation* rather than *restoration*. Although these terms are all too often used interchangeably and incorrectly in the literature, distinguishing between them is more than just technical semantics as they involve very different trajectories, endpoints and management expectations (Simenstad *et al.* 2006). Here we focus on the impact of remediating tidally-restrictive culverts at Kooragang Wetland on an assemblage of species (or ecosystem element), within the context of a heavily urbanised and disturbed landscape (the lower Hunter estuary). The term rehabilitation is therefore more apt than restoration, which generally refers to attempts to rebuild entire ecosystems to a pre-disturbance state (Aronson and Le Floc'h 1996, Simenstad *et al.* 2006).

For such an evaluation it was necessary to compare creeks and marshes of two localities where culverts were removed (manipulated) to replicated reference locations (creeks and marshes not under the influence of culverts) and appropriate control locations (creeks and marshes impacted by culverts which were not removed). Reference locations were not assumed to be "natural" or "undisturbed", but rather beyond the influence of tidally-restrictive culverts. Therefore we tested hypotheses relating to culvert removal rather than ecosystem restoration. To achieve this, all locations were monitored at replicated times both before and after culvert removal. The hypothesis being that if any responses were to be attributed to culvert removal, manipulated locations would undergo a trajectory of change to become similar to the reference locations, without similar changes occurring in the control locations (Grayson *et al.* 1999). Marshes and creeks were studied over two very different time-scales. Marshes were monitored for up to two years post-culvert removal, whereas creeks were monitored for up to 14 years post-culvert removal. A detailed description of the spatial and temporal design, as well as the methods used to sample fish and decapods is given in the sections 2.1.2 and 2.1.3 (pages 5-7).

4.2.2. Statistical analysis

Analyses of assemblage change within the BACI design were performed on zero-adjusted Bray-Curtis similarity matrices (Clarke *et al.* 2006) derived from fourth-root transformed abundance data. The fourth-root transformation reduced the 'swamping effect' of a few very abundant species on the composition of an assemblage (Clarke and Green 1988) and zero-adjusting with a "dummy value" did not effect the normal functioning of the Bray-Curtis coefficient, but ensured that samples with denuded assemblages could be incorporated into analyses to generate meaningful ordinations which would otherwise 'collapse' (Clarke *et al.* 2006). Fish and decapod data were analysed as the same assemblage, but marsh (fyke) and creek (seine net) data were analysed separately due to the substantially different sampling methodologies and difference in temporal replication between the datasets. Because gill nets targeted only a few large-bodied species (see Chapter 3), species caught in large enough numbers were analysed separately using univariate calculations of means for each location or treatment at each year of sampling, rather than analysing as a multivariate assemblage that would be dominated by samples containing zero fish.

For the multivariate assemblage data, non-metric multidimensional scaling (nMDS; Kruskal and Wish 1978) ordinations were created from the similarity matrices for both the marsh and creek locations. These ordinations were based on the centroids for each manipulated location and centroids of combined control and combined reference locations for each year of sampling. That is, we incorporated different sites and months into a single Location x Time centroid (for manipulated locations) or Treatment x Time centroid (for references and controls). Doing so greatly reduced the number of data points on the ordination and enabled better interpretation of trajectories of assemblage change in manipulated locations against the 'average' response of control and reference locations.

Simple agglomerative hierarchical clustering and similarity profiles (SIMPROF; Clarke *et al.* 2008) were performed on each of the ordinated matrices to identify statistically significant groupings and change in assemblage composition over time. Of particular interest within the context of the BACI design was whether assemblages in manipulated locations became similar to reference locations at any time, whilst controls did not. These assemblage groupings were compared with Similarity Percentages analyses (SIMPER; Clarke 1993) to identify which species contributed most to within group similarity and between group dissimilarity, thus identifying the species driving successional changes across locations. Only those species exceeding an arbitrary threshold value of dis/similarity of 3% were presented (Terlizzi *et al.* 2005). Finally, species richness was averaged across all centroids within each of the assemblage groupings. This was done on the whole assemblage and the following salinity guilds: *Estuarine-marine* (saltwater species that are primarily estuarine-marine dwelling as adults) and *Freshwater-estuarine* (euryhaline species equally well adapted to saline or freshwater habitats as adults) (after Pollard and Hannan (1994)). *Freshwater* species (those dwelling entirely in freshwater) were not analysed as they were too few (1 species) to be of use.

4.3. Results

4.3.1. Catch summary

A total of 1,116 seine, gill and fyke net samples were collected across all twelve creek and marsh locations over the 16 years that the study took place. These samples netted a combined total of 236,269 fish and 102,487 decapods (50 and 21 species respectively) (Table 9 and Appendix Table A4). As would be expected due to the more intensive sampling effort, more fish and decapods species were collected from creek locations by the seine and gill nets over the eight years in which sampling was done (mean per location 30.7+2.0 and 13.8+0.5 for fish and decapods respectively) than were collected by fyke nets in the marsh locations over three years of sampling (16.5+1.2 and 6.0+0.5).

The sections that follow deal respectively with the marsh and creek assemblages. Assemblage groupings are dealt with in the order from which they emerged in respective statistical analyses and are considered firstly in terms of the number of species present (richness), the species that make up the samples (composition) and the number of individuals present (abundance).

4.3.2. Short-term changes in marsh assemblages

Multivariate analyses identified three distinct assemblage groupings in the marshes over the three year sampling period (Fig. 8 and Table 10). Group I included all samples from Crabhole Flats, indicating little change over time at this location. This group also included samples taken in the 4th year from Fish Fry Flats and the reference marshes, indicating that by the 4th year, both manipulated marshes had reference marshes had a similar assemblage. In all three years (including the year prior to culvert removal), the assemblages Fish Fry Flats and the combined reference marshes did not differ (Group III: Fig. 8), suggesting little impact of culvert presence and no response to culvert removal at this location. Group II was exclusively derived from samples taken at the control marshes in the 4th year, implying a unique set of circumstances prevailed at those locations in that year. Fish Fry Flats and the reference marshes did, however, follow a different trajectory to the control marshes in year four (Fig. 8), which may indicate some broader-scale temporal changes in assemblage composition that was not present in culverted controls.

The change at Fish Fry Flats and the reference marshes to become more like Crabhole Flats was due to subtle changes in each of species richness, composition and abundance. In particular, there was a slight increase in the number of estuarine-marine species in Fish Fry Flats and reference marshes that was not observed in control marshes (Fig. 9). Most species increased in abundance in the absence of culverts, with southern blue-eye (F-E) and flat-tail mullet (E-M) being notable additions to the assemblage (Table 11: Group I versus III).

The abundance of glass goby declined in the control marshes while the number of freshwater-estuarine species increased (Table 11, Group I vs. Group III), and there was a marked increase in the abundance of mosquitofish (Table 11, Group II vs. Group III) at the control marshes. Other changes, such as the increase in number of striped shrimp and decrease of pink shrimp occurred across all sites and therefore cannot be attributed to culvert removal (Table 11: Groups 1 vs. III and Groups II vs. III).

Table 9. Catch summary of fish and decapods from all creek (seine and gill nets) and marsh (fyke nets) locations in Kooragang Island during this study. Abundances for the three most numerous species in each habitat are shown. See Appendix, Table A4 for all species totals.

Habitat	**CREEK**					**MARSH**			
Treatment	Total	Control	Reference	Manipulated		Total	Control	Reference	Manipulated
NUMBER OF SPECIES									
Fish	**49**	**32**	**43**	**36**		**31**	**21**	**24**	**28**
Non C/R species	**29**	20	25	21	Non C/R species	**19**	15	15	17
C/R species	**20**	12	18	15	C/R species	**12**	6	9	11
E-M species	**32**	21	31	24	E-M species	**18**	12	11	16
F-E species	**16**	11	11	11	F-E species	**12**	7	10	10
F species	**1**	0	1	1	F species	**1**	2	3	2
Decapods	**21**	**18**	**16**	**17**		**10**	**8**	**9**	**10**
Non C/R species	**17**	14	12	13	Non C/R species	**8**	7	7	8
C/R species	**4**	4	4	4	C/R species	**2**	1	2	2
E-M species	**19**	16	15	16	E-M species	**9**	7	8	9
F-E species	**2**	2	1	1	F-E species	**1**	1	1	1
F species	**0**	0	0	0	F species	**0**	0	0	0
ABUNDANCE									
Fish	**212,744**	**29,038**	**62,341**	**121,365**		**23,525**	**1,671**	**10,314**	**11,540**
Non C/R species	**204,126**	27,645	60,978	115,503	Non C/R species	**22,273**	990	10,226	11,057
C/R species	**8,618**	1,393	1,363	5,862	C/R species	**1,252**	681	88	483
Glass goby	**176,402**	14,427	55,764	106,211	Glass goby	**14,583**	169	7,990	6,424
Swan River goby	**11,795**	8,413	549	2,833	Southern blue-eye	**5,067**	7	1,177	3,883
Flat-tail mullet*	**4,147**	752	352	3,043	Swan River goby	**1,239**	58	728	453
Decapods	**86,720**	**11,860**	**43,223**	**31,637**		**15,767**	**242**	**4,099**	**11,426**
Non C/R species	**84,739**	11,632	41,766	31,341	Non C/R species	**15,746**	241	4,091	11,414
C/R species	**1,981**	228	1,457	296	C/R species	**21**	1	8	12
Pink shrimp	**52,596**	415	39,953	12,228	Striped shrimp	**13,988**	116	3,346	10,526
Striped shrimp	**31,705**	11,032	1,637	19,036	Pink shrimp	**1,453**	74	643	736
School prawn*	**1,475**	77	1,356	42	Shore crab	**233**	47	88	98

The number of samples (and gear type) contributing to each count differed between creek and marsh habitats and therefore comparisons between each treatment within each habitat are the most informative. Refer to Table 1 and Table 2 (pages 5-7) for a full description of this differing sampling effort.* = C/R species

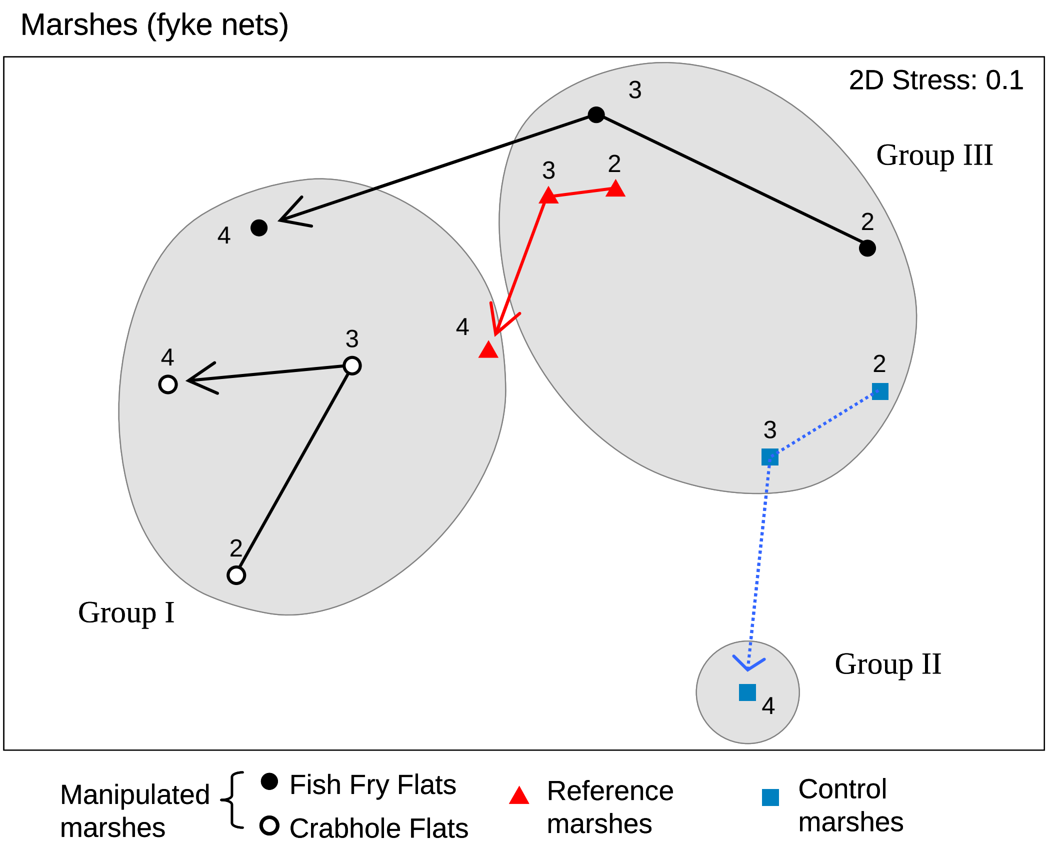

Fig. 8. nMDS of centroids for each manipulated **MARSH** at each year of sampling and the combined control and reference marshes at each year of sampling showing the trajectories in fish and decapod assemblage change. Statistically significant assemblage groupings (SIMPROF, Table 10) are shown. Numbers correspond to years (2 = pre-culvert removal at manipulated marshes and 3 & 4 = two years post-culvert removal at manipulated marshes).

Table 10. Statistically significant groupings (SIMPROF) of **MARSH** assemblages across space and time based upon Bray-Curtis similarity. For referencing, groups are labelled sequentially with Roman numerals as they appear across the ordination in Fig. 8.

		SIMPROF Test			
Group	Contains locations and times*	Split from Group(s)	At similarity	π	Prob.
Fish Fry flats and Crabhole Flats versus combined references and controls *(based on fyke net data and ordination Fig. 8)*					
Group I	(CHF Yr 2-4) (FFF Yr 4) (Rs Yr 4)	II-III	47.45	2.64	0.001
Group II	Cs Yr 4	III	45.87	2.38	0.037
Group III	(FFF Yr 2-3) (Rs Yr 2-3) (Cs Yr 2-3)				

Yr=Year
FFF=Fish Fry Flats, CHF=Crabhole Flats
Rs=combined reference marshes
Cs=combined control marshes

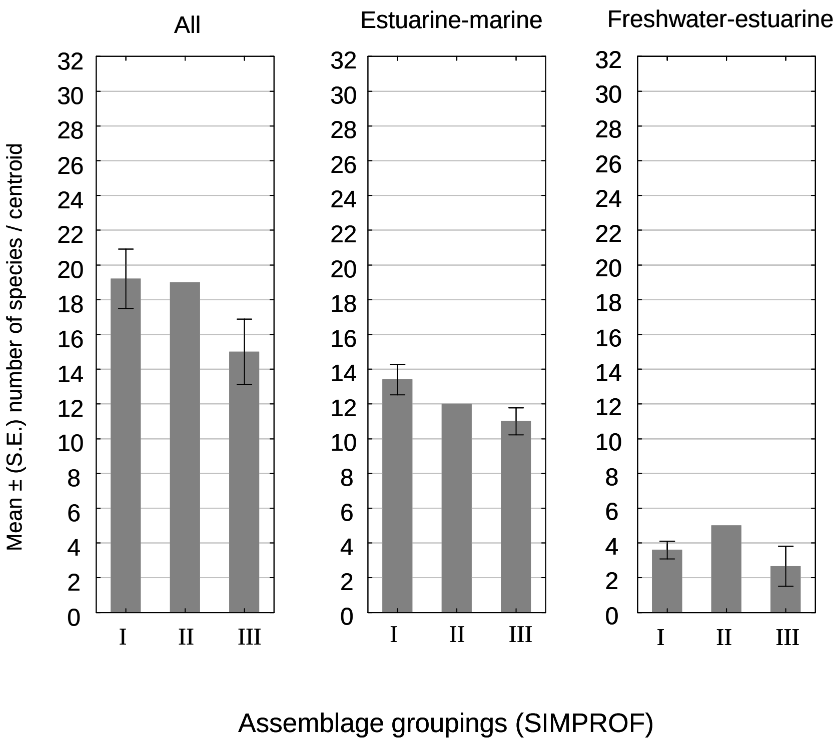

Fig. 9. Mean (± S.E.) number of all species, estuarine-marine species and freshwater-estuarine species for the different **MARSH** assemblage groupings identified by SIMPROF (Table 10).

Table 11. Results of SIMPER analyses showing the species that contributed most to the dissimilarity between statistically significant **MARSH** assemblage groupings identified by SIMPROF (Table 10). Fourth-root transformed average abundance. Group with highest average abundance in bold.

Group I versus Group II		Av. Abun					
Average dissimilarity: 56.69		Group I	Group II	Av. Diss	Sim/SD	Contrib%	Cum.%
Mosquitofish	*Gambusia holbrookii*	0.17	**2.13**	9.36	5.68	16.52	16.52
Striped shrimp	*Macrobrachium intermed.*	**3.09**	1.30	7.93	1.75	13.99	30.51
Glass goby	*Gobiopterus semivestitus*	**2.17**	0.55	7.39	2.35	13.04	43.55
Southern blue-eye	*Pseudomugil signifer*	**1.65**	0.14	7.14	1.12	12.60	56.15
Mangrove goby	*Mugilogobius paludis*	**1.60**	0.73	4.09	1.26	7.22	63.37
Blue-spot goby	*Pseudogobius olorum*	**1.53**	0.75	3.42	1.12	6.02	69.39
Flat-tail mullet	*Liza argentea*	**0.64**	0.05	2.60	0.91	4.59	73.99
Group I versus Group III		Av. Abun					
Average dissimilarity: 60.48		Group I	Group III	Av. Diss	Sim/SD	Contrib%	Cum.%
Striped shrimp	*Macrobrachium intermed.*	**3.09**	0.34	14.01	3.06	23.16	23.16
Southern blue-eye	*Pseudomugil signifer*	**1.65**	0.13	8.17	1.25	13.51	36.67
Glass goby	*Gobiopterus semivestitus*	**2.17**	1.37	6.61	1.45	10.92	47.59
Mangrove goby	*Mugilogobius paludis*	**1.60**	0.35	6.54	1.84	10.81	58.40
Blue-spot goby	*Pseudogobius olorum*	**1.53**	1.15	3.17	1.31	5.24	63.64
Flat-tail mullet	*Liza argentea*	**0.64**	0.03	3.06	1.06	5.05	68.70
Shore crab	*Paragrapsus laevis*	**0.71**	0.41	2.57	1.36	4.25	72.95
Pink shrimp	*Acetes sibogae australis*	0.26	**0.47**	2.08	1.21	3.45	76.39
Group II versus Group III		Av. Abun					
Average dissimilarity: 58.18		Group II	Group III	Av. Diss	Sim/SD	Contrib%	Cum.%
Mosquitofish	*Gambusia holbrookii*	**2.13**	0.15	16.17	6.00	27.79	27.79
Striped shrimp	*Macrobrachium intermed.*	**1.30**	0.34	8.12	2.64	13.95	41.74
Glass goby	*Gobiopterus semivestitus*	0.55	**1.37**	7.40	1.19	12.71	54.45
Pink shrimp	*Acetes sibogae australis*	0.00	**0.47**	3.42	1.10	5.87	60.32
Mangrove goby	*Mugilogobius paludis*	**0.73**	0.35	3.31	2.25	5.69	66.01
Blue-spot goby	*Pseudogobius olorum*	0.75	**1.15**	3.23	1.47	5.56	71.57
Striped gudgeon	*Gobiomorphus australis*	**0.36**	0.01	2.92	6.28	5.01	76.58
Shore crab	*Paragrapsus laevis*	0.40	**0.41**	2.29	0.90	3.93	80.51

Terms: Av. Abund = mean abundance, Av. Sim = mean similarity, Sim/SD = consistency ratio (Av. Sim / standard deviation), Contrib% = percent contribution to total similarity, Cum.% = cumulative percent contribution to total similarity.

4.3.3. Long-term changes in Fish Fry Creek assemblages

Long-term responses (two year pre- and 14 years post-culvert removal) of fish and decapod assemblages to culvert removal were examined in tidal creeks. All locations (references, controls and manipulated) showed change through time (Fig. 10a). When culverted (in year one and two), Fish Fry Creek had an assemblage similar to control (culverted) creeks, but differed significantly from reference creeks (Fig. 10a, Table 12: Group II similarity 40.79, $p = 0.001$). Estuarine-marine (E-M) species typified the difference, as more of these species were found in reference creeks (Fig. 11: Group II versus V) and when present were typically more abundant (Table 13: Group II versus V). In particular, species such as glass goby, Port Jackson glassfish, flat-tail mullet, yellow-finned bream, largemouth goby, pink shrimp, school prawn (all E-M), striped shrimp and Tamar River goby (both F-E) were consistently more abundant in references when compared to the controls and Fish Fry Creek prior to culvert removal (Table 13: Group II versus V). In contrast, the culverted creeks and Fish Fry Creek (before culvert removal) contained greater abundances of mangrove goby, blue-spot goby, half-bridled goby, bridled goby and checkered mangrove goby (all E-M) than the reference creeks (Table 13: Group II versus V).

For the first 10 years of the study, assemblages at the control and reference creeks remained relatively unchanged when compared to the significant shift that occurred in the Fish Fry Creek assemblage immediately following culvert removal (Fig. 10a: after year 2). The change in assemblage at Fish Fry Creek after culvert removal was sustained for at least the next five years (Fig. 10a: Fish Fry Creek years 3-8, Table 12: Group III similarity 37.02, $p = 0.001$), demonstrating a clear and sustained response to culvert removal. The goby species (mangrove, blue-spot, half-bridled, bridled and checkered mangrove) which had characterised the culverted Fish Fry Creek (and control creeks) became immediately less abundant and were replaced by a reciprocal increase in abundance of glass goby, flat-tail mullet, fan-tail mullet, sandy sprat, pink shrimp (all E-M), Tamar River goby and sea mullet (both F-E) (Table 13: II versus III). Whilst these reciprocal changes in species abundance and species replacement significantly changed the composition of the assemblage at Fish Fry Creek, it equated to no net change in the number of estuarine-marine or freshwater-estuarine-marine species (Fig. 11: Group II versus III).

It was not until sometime between years 8-10 (6-8 years after culvert removal) that the assemblage at Fish Fry Creek changed enough to be considered equivalent in composition to the reference creeks (Fig. 10a: Fish Fry Creek years 10-12, Table 12: Group V similarity 32.54, $p = 0.001$). This shift was driven by changing species abundances and a net increase in the number of estuarine-marine species (Table 13 and Fig. 11: Group III versus V). School prawn, Port Jackson glassfish, pink shrimp, yellow-finned bream and large mouth goby (all E-M) became more abundant in Fish Fry Creek during this time whereas the abundance of striped shrimp, glass goby, sandy sprat and the three mullet species fan-tail, flat-tail and sea mullet all fell (Table 13: Group III versus V). Importantly, in the years that Fish Fry Creek changed to become equivalent in assemblage composition to the reference creeks, no such change was seen in the control creeks (Fig. 10a), providing strong evidence that the response was due to culvert removal rather than unexplained environmental variation.

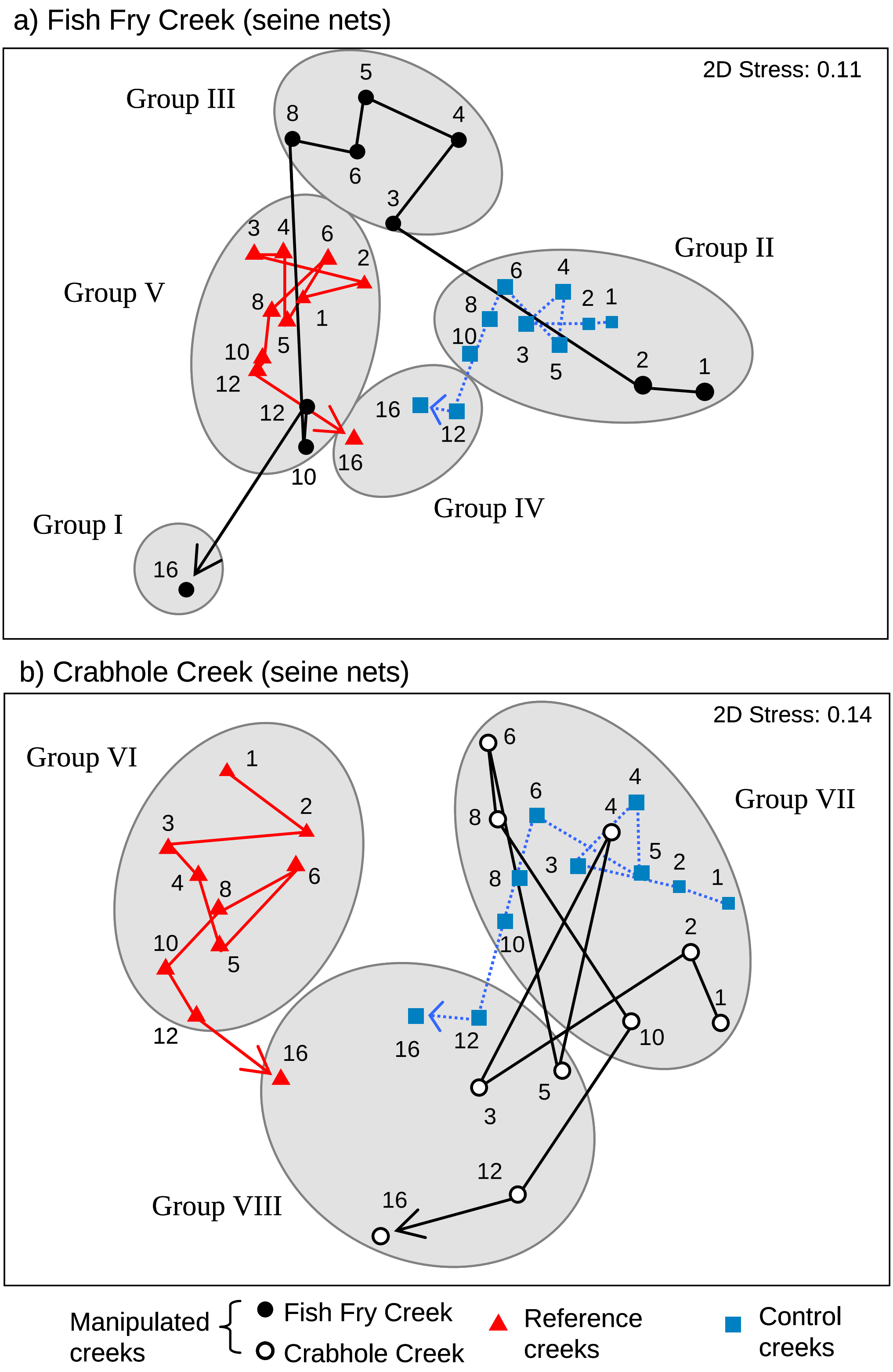

Fig. 10. nMDS of centroids for a) Fish Fry Creek and b) Crabhole Creek and the combined reference and control locations at each year of sampling showing the trajectories in fish and decapod assemblage change (based on seine net data only). Statistically significant assemblage groupings (SIMPROF, Table 12) are shown. Numbers correspond to years (1-2 pre-culvert removal and 3-16 post-culvert removal).

At some point between years 12-16 in the control creeks and later (year 16) in the reference creeks the assemblages of these locations changed significantly. During the same time (year 16), the assemblage at Fish Fry Creek also underwent a significant change, albeit on a somewhat different trajectory from the references (Fig. 10a, Table 12: Group I similarity 50.17, p = 0.001). During this time species richness declined (Fig. 11: Group I versus V and IV versus V) and species such as pink shrimp, striped shrimp, glass goby, flat-tail mullet, sandy sprat, Tamar River goby and large mouth goby all decreased in abundance in both references and Fish Fry Creek, whereas blue-spot goby and school prawn increased in abundance (Table 13: Group I versus V and IV versus V). In controls, similar reductions in abundance were observed for striped shrimp, pink shrimp and glass goby, in addition to those goby species which characterised culverted controls in previous years (i.e. mangrove, blue-spot, half-bridled, bridled and checkered mangrove) (Table 13: Group II versus IV). Opposite responses were seen for Port Jackson glassfish, Tamar River goby and large mouth goby which increased in controls in the latter two years (Table 13: Group II versus IV). The net result of these changes in abundance in reference and control creeks was that their assemblages merged in the latter years so that little discernable difference was noted between them (Fig. 10a, Table 12: Group IV similarity 32.54, p = 0.001).

Whilst the general nature of these changes across both un-manipulated creeks and Fish Fry Creek suggests that factors occurring at scales beyond that of the remediation works were responsible, there did appear to be some interaction between location and background environmental change. This is demonstrated by the different trajectory that Fish Fry Creek took from the controls and references in year 16. This change was driven by the fact that the reductions in species abundance observed in both reference and control creeks occurred to a larger extent in Fish Fry Creek so that at year 16, Fish Fry Creek had lower abundances of these species than the reference (Table 13: Group I versus IV). Furthermore, blue-spot goby and common toadfish increased at Fish Fry Creek but not any of the controls or references (Table 13: Group I versus IV).

The gill nets placed in Fish Fry Creek, the reference and control creeks caught fewer species than the seine nets but targeted larger size ranges: sea mullet (86-442mm), flat-tail mullet (85-305mm), fan-tail mullet (93-134mm) and yellow-finned bream (180-260mm), compared to seine nets that targeted smaller individuals of these species (<125mm, <203mm, <92mm and <185mm, respectively). Of these species, only sea mullet and flat-tail mullet were caught in large numbers. Sea mullet greater than 86mm increased significantly in Fish Fry Creek in the first year following culvert removal (year 3), without any reciprocal increase in the control or reference creeks (Fig. 12). This indicated a clear initial response to culvert removal, but it was not sustained and no further responses were noted in later years. The abundance of flat-tail mullet (>85mm) was far more variable between years (Fig. 12). Increases in abundance observed in reference creeks in the first half of the study were not seen in Fish Fry Creek and the controls. Although flat-tail mullet numbers increased in Fish Fry Creek in year 12 and this was sustained in year 16, a similar increase was observed in the control creeks. As such, this response cannot be attributed to culvert removal.

Table 12. Statistically significant groupings (SIMPROF) of **CREEK** assemblages across space and time based upon Bray-Curtis similarity. For referencing, groups are labelled sequentially with Roman numerals as they appear across the ordinations in Fig. 10.

		SIMPROF Test			
Group	Contains locations and times*	Split from Group(s)	At similarity	π	Prob.
Fish Fry Creek versus combined references and controls *(based on seine net data and ordination Fig. 10a)*					
Group I	FFC Yr 16	II-V	50.87	4.34	0.001
Group II	(FFC Yr 1-2) (Cs Yr 1-10)	III-V	40.79	3.86	0.001
Group III	FFC Yr 3-8	IV-V	37.02	2.32	0.001
Group IV	(Cs Yr 12-16) (Rs Yr 16)	V	32.54	2.07	0.001
Group V	(FFC Yr 10-12) (Rs Yr 2-12)				
Crabhole Creek versus combined references and controls *(based on seine net data and ordination Fig. 10b)*					
Group VI	Rs Yr 2-12	VII-VIII	38.96	3.08	0.001
Group VII	(CHC Yr 1-2, 4, 6-10) (Cs Yr 1-10)	VIII	34.45	2.60	0.001
Group VIII	(CHC Yr 3, 5, 12-16) (Cs Yr 12-16) (Rs Yr 16)				

Yr=Year
FFC=Fish Fry Creek, CHC=Crabhole Creek
Rs=combined reference creeks
Cs=combined control creeks

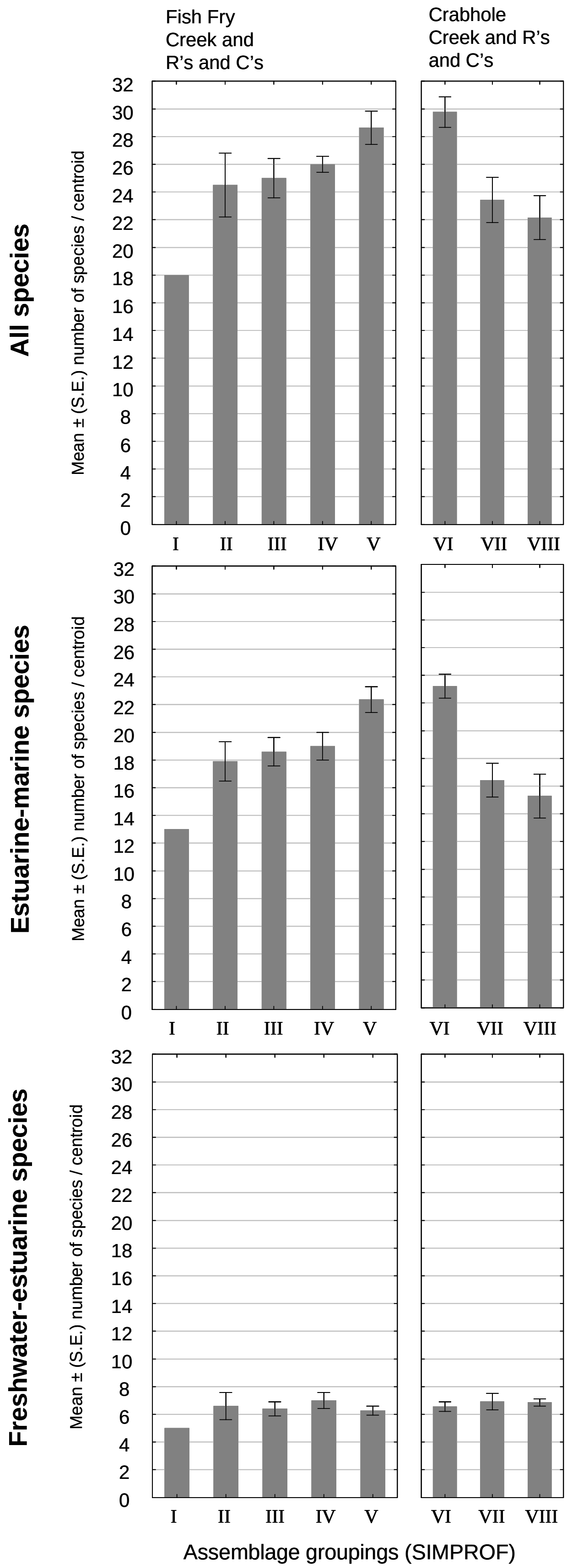

Fig. 11. Mean (± S.E.) number of all species, estuarine-marine species and freshwater-estuarine species for the different CREEK assemblage groupings identified by SIMPROF (Table 12).

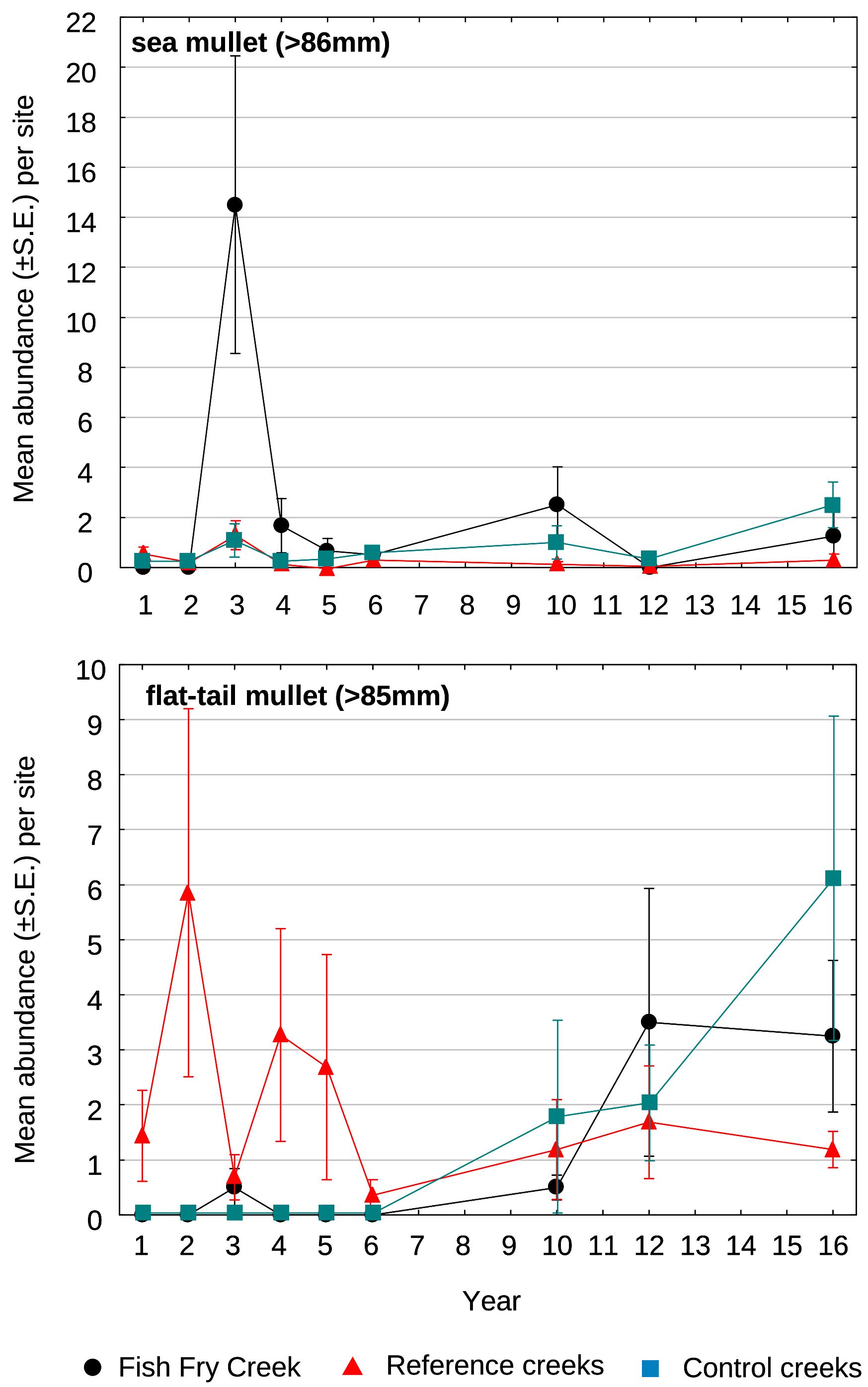

Fig. 12. Mean abundance of sea mullet and flat-tail mullet caught in multi-panel gill nets at Fish Fry Creek and the combined control and combined reference creeks. Year 1-2 was before culvert removal at Fish Fry Creek and years 3-16 are post-culvert removal.

4.3.4. Long-term changes in Crabhole Creek assemblages

As at Fish Fry Creek, Crabhole Creek prior to culvert removal was similar in assemblage to the control creeks and different to the reference creeks (Fig. 10b, Table 12: Group VI defined from VII-VII at similarity 38.96, p = 0.001). More estuarine-marine (E-M) species were found in reference creeks (Fig. 11: Group VI versus VII) and they were generally more abundant (Table 13: Group VI versus VII). Like Fish Fry Creek, large changes in the assemblage at Crabhole Creek were observed at different times after culvert removal (Fig. 10b). However, unlike Fish Fry Creek, a consistent trajectory towards reference condition was not observed in any of the years that would suggest a sustained response to culvert removal. At various times (years 3 and 5), the assemblage at Crabhole Creek changed from that of the controls, however, this was never maintained and the assemblages between Crabhole Creek and the control creeks resumed significant similarity in alternative years (Fig. 10b). In years 12 and 16, the assemblage at Crabhole Creek did become relatively similar to that of reference locations in year 16 (Fig. 10b,: Table 12: Group XIII similarity 34.45, p=0.001), however, this shift also occurred at control creeks in these years and is likely due to broader-scale assemblage changes occurring across Kooragang Island beyond the influence of the manipulations.

Table 13. Results of SIMPER analyses showing the species that contributed most to the dissimilarity between statistically significant **CREEK** assemblage groupings identified by SIMPROF (Table 12). Fourth-root transformed average abundance.

Fish Fry Creek versus combined references and controls

Group I versus Group II	**Average dissimilarity: 72.01**	Av. Abun					
		Group IV	Group V	Av. Diss	Sim/SD	Contrib%	Cum.%
Blue-spot goby	*Pseudogobius olorum*	0.42	**2.36**	8.32	6.53	11.55	11.55
Striped shrimp	*Macrobrachium intermed.*	0.13	**2.00**	8.06	4.53	11.20	22.74
Glass goby	*Gobiopterus semivestitus*	0.48	**2.38**	7.98	4.08	11.08	33.82
Port Jacks. glassfish	*Ambassis jacksoniensis*	**1.70**	0.12	6.81	4.27	9.45	43.27
Half-bridled goby	*Arenigobius frenatus*	0.00	**1.28**	5.69	1.88	7.90	51.18
Mangrove goby	*Mugilogobius paludis*	0.22	**1.52**	5.56	3.48	7.73	58.90
Bridled goby	*Arenigobius bifrenatus*	0.11	**1.06**	3.96	2.50	5.50	64.41
Check. mang. goby	*Mugilogobius stigmaticus*	0.00	**0.73**	3.03	1.85	4.21	68.61
Eastern king prawn	*Penaeus plebejus*	0.11	**0.60**	2.31	0.90	3.21	71.83

Group I versus Group III	**Average dissimilarity: 69.01**	Av. Abun					
		Group IV	Group VI	Av. Diss	Sim/SD	Contrib%	Cum.%
Glass goby	*Gobiopterus semivestitus*	0.48	**5.36**	16.27	11.16	23.58	23.58
Striped shrimp	*Macrobrachium intermed.*	0.13	**3.61**	11.73	4.41	16.99	40.57
Pink shrimp	*Acetes sibogae australis*	0.39	**1.63**	4.00	1.31	5.80	46.37
Port Jacks. glassfish	*Ambassis jacksoniensis*	**1.7**	0.54	3.96	2.17	5.74	52.11
Sandy sprat	*Hyperlophus vittatus*	0.00	**1.18**	3.77	1.63	5.47	57.57
Flat-tail mullet	*Liza argentea*	0.77	**1.81**	3.47	3.41	5.03	62.60
Blue-spot goby	*Pseudogobius olorum*	0.42	**1.29**	3.02	2.82	4.38	66.99
Fan-tail mullet	*Paramugil georgii*	0.00	**0.79**	2.59	1.38	3.75	70.74
Sea mullet	*Mugil cephalus*	0.15	**0.72**	2.23	0.88	3.23	73.96
Tamar River goby	*Afurcagobius tamarensis*	0.31	**0.94**	2.17	1.18	3.14	77.10

Group I versus Group IV **Average dissimilarity: 54.08**		Av. Abun Group IV	Group VII	Av. Diss	Sim/SD	Contrib%	Cum.%
Glass goby	*Gobiopterus semivestitus*	0.48	**2.02**	8.34	4.07	15.42	15.42
Port Jacks. glassfish	*Ambassis jacksoniensis*	**1.70**	0.65	5.72	3.65	10.57	26.00
Blue-spot goby	*Pseudogobius olorum*	0.42	**1.36**	5.11	3.30	9.45	35.44
Striped shrimp	*Macrobrachium intermed.*	0.13	**1.03**	4.84	2.44	8.94	44.38
School prawn	*Metapenaeus macleayi*	0.45	**0.85**	3.44	1.17	6.37	50.75
Flat-tail mullet	*Liza argentea*	**0.77**	0.29	2.68	1.32	4.95	55.70
Common toadfish	*Tetractenos hamiltoni*	**0.49**	0.04	2.42	4.55	4.47	60.17
Largemouth goby	*Redigobius macrostoma*	0.00	**0.42**	2.29	7.16	4.24	64.41
Bridled goby	*Arenigobius bifrenatus*	0.11	**0.51**	2.17	3.03	4.01	68.41
Half-bridled goby	*Arenigobius frenatus*	0.00	**0.36**	2.03	1.48	3.75	72.16
Tamar River goby	*Afurcagobius tamarensis*	0.31	**0.68**	1.90	1.20	3.51	75.68

Group I versus Group V **Average dissimilarity: 56.59**		Av. Abun Group IV	Group VIII	Av. Diss	Sim/SD	Contrib%	Cum.%
Glass goby	*Gobiopterus semivestitus*	0.48	**3.13**	10.43	4.13	18.43	18.43
Pink shrimp	*Acetes sibogae australis*	0.39	**2.5**	8.50	3.06	15.01	33.45
Striped shrimp	*Macrobrachium intermed.*	0.13	**1.37**	4.88	3.16	8.62	42.07
Blue-spot goby	*Pseudogobius olorum*	0.42	**1.20**	3.13	2.10	5.53	47.61
Largemouth goby	*Redigobius macrostoma*	0.00	**0.79**	3.11	2.32	5.50	53.11
Tamar River goby	*Afurcagobius tamarensis*	0.31	**0.94**	2.68	1.61	4.73	57.84
Bridled goby	*Arenigobius bifrenatus*	0.11	**0.65**	2.11	2.76	3.74	61.58
Flat-tail mullet	*Liza argentea*	0.77	**1.03**	1.86	2.22	3.30	64.87
Port Jacks. glassfish	*Ambassis jacksoniensis*	**1.70**	1.23	1.83	1.33	3.24	68.11
Common toadfish	*Tetractenos hamiltoni*	**0.49**	0.19	1.80	3.04	3.18	71.29

Group II versus Group III **Average dissimilarity: 49.04**		Av. Abun Group V	Group VI	Av. Diss	Sim/SD	Contrib%	Cum.%
Glass goby	*Gobiopterus semivestitus*	2.38	**5.36**	7.57	3.06	15.44	15.44
Striped shrimp	*Macrobrachium intermed.*	2.00	**3.61**	4.11	2.01	8.37	23.81
Flat-tail mullet	*Liza argentea*	0.39	**1.81**	3.66	3.20	7.46	31.27
Pink shrimp	*Acetes sibogae australis*	0.54	**1.63**	2.94	1.42	6.00	37.27
Sandy sprat	*Hyperlophus vittatus*	0.01	**1.18**	2.89	1.74	5.89	43.17
Blue-spot goby	*Pseudogobius olorum*	**2.36**	1.29	2.70	3.71	5.50	48.67
Half-bridled goby	*Arenigobius frenatus*	**1.28**	0.39	2.38	1.43	4.86	53.53
Mangrove goby	*Mugilogobius paludis*	**1.52**	0.63	2.28	2.26	4.64	58.17
Fan-tail mullet	*Paramugil georgii*	0.06	**0.79**	1.90	1.56	3.88	62.05
Bridled goby	*Arenigobius bifrenatus*	**1.06**	0.38	1.80	1.62	3.66	65.71
Tamar River goby	*Afurcagobius tamarensis*	0.63	**0.94**	1.74	1.47	3.55	69.26
Sea mullet	*Mugil cephalus*	0.17	**0.72**	1.68	1.02	3.43	72.69
Check. mang. goby	*Mugilogobius stigmaticus*	**0.73**	0.14	1.48	1.47	3.01	75.70

Group II versus Group IV **Average dissimilarity: 43.39**		Av. Abun Group V	Group VII	Av. Diss	Sim/SD	Contrib%	Cum.%
Mangrove goby	*Mugilogobius paludis*	**1.52**	0.41	3.95	2.95	9.11	9.11
Blue-spot goby	*Pseudogobius olorum*	**2.36**	1.36	3.53	3.32	8.14	17.25
Striped shrimp	*Macrobrachium intermed.*	**2.00**	1.03	3.49	2.00	8.04	25.29
Half-bridled goby	*Arenigobius frenatus*	**1.28**	0.36	3.38	1.48	7.78	33.08
School prawn	*Metapenaeus macleayi*	0.17	**0.85**	2.68	1.16	6.17	39.24
Check. mang. goby	*Mugilogobius stigmaticus*	**0.73**	0.02	2.47	1.83	5.69	44.94
Eastern king prawn	*Penaeus plebejus*	**0.60**	0.02	2.20	1.07	5.08	50.01

Port Jacks. glassfish	*Ambassis jacksoniensis*	0.12	**0.65**	2.15	3.26	4.96	54.97
Glass goby	*Gobiopterus semivestitus*	**2.38**	2.02	2.05	1.54	4.72	59.69
Bridled goby	*Arenigobius bifrenatus*	**1.06**	0.51	2.03	1.71	4.68	64.37
Tamar River goby	*Afurcagobius tamarensis*	0.63	**0.68**	1.67	1.48	3.85	68.22
Pink shrimp	*Acetes sibogae australis*	**0.54**	0.53	1.34	1.39	3.10	71.32
Largemouth goby	*Redigobius macrostoma*	0.37	**0.42**	1.30	1.91	3.00	74.32

Group II versus Group V
Average dissimilarity: 47.51

		Av. Abun					
		Group V	Group VIII	Av. Diss	Sim/SD	Contrib%	Cum.%
Pink shrimp	*Acetes sibogae australis*	0.54	**2.5**	5.76	2.50	12.13	12.13
Mangrove goby	*Mugilogobius paludis*	**1.52**	0.19	3.88	3.47	8.16	20.29
Blue-spot goby	*Pseudogobius olorum*	**2.36**	1.20	3.38	2.63	7.12	27.42
Port Jacks. glassfish	*Ambassis jacksoniensis*	0.12	**1.23**	3.32	2.39	6.98	34.40
Glass goby	*Gobiopterus semivestitus*	2.38	**3.13**	2.84	1.33	5.97	40.37
Half-bridled goby	*Arenigobius frenatus*	**1.28**	0.37	2.76	1.49	5.81	46.18
Striped shrimp	*Macrobrachium intermed.*	**2.00**	1.37	2.05	1.32	4.32	50.50
Flat-tail mullet	*Liza argentea*	0.39	**1.03**	2.00	1.48	4.20	54.70
Tamar River goby	*Afurcagobius tamarensis*	0.63	**0.94**	1.82	1.35	3.83	58.54
Check. mang. goby	*Mugilogobius stigmaticus*	**0.73**	0.13	1.73	1.51	3.65	62.18
School prawn	*Metapenaeus macleayi*	0.17	**0.71**	1.72	1.34	3.62	65.80
Largemouth goby	*Redigobius macrostoma*	0.37	**0.79**	1.69	1.57	3.56	69.36
Yellow-finned bream	*Acanthopagrus australis*	0.11	**0.66**	1.62	1.71	3.40	72.76
Bridled goby	*Arenigobius bifrenatus*	**1.06**	0.65	1.45	1.46	3.06	75.83

Group III versus Group IV
Average dissimilarity: 51.65

		Av. Abun					
		Group VI	Group VII	Av. Diss	Sim/SD	Contrib%	Cum.%
Glass goby	*Gobiopterus semivestitus*	**5.36**	2.02	9.55	5.66	18.49	18.49
Striped shrimp	*Macrobrachium intermed.*	**3.61**	1.03	7.50	3.23	14.52	33.01
Flat-tail mullet	*Liza argentea*	**1.81**	0.29	4.44	3.47	8.61	41.61
Pink shrimp	*Acetes sibogae australis*	**1.63**	0.53	3.33	1.45	6.44	48.05
Sandy sprat	*Hyperlophus vittatus*	**1.18**	0.00	3.28	1.73	6.36	54.41
School prawn	*Metapenaeus macleayi*	0.14	**0.85**	2.23	1.16	4.31	58.72
Fan-tail mullet	*Paramugil georgii*	**0.79**	0.02	2.21	1.51	4.28	63.00
Sea mullet	*Mugil cephalus*	**0.72**	0.28	1.82	1.05	3.52	66.52
Tamar River goby	*Afurcagobius tamarensis*	**0.94**	0.68	1.70	1.67	3.30	69.82

Group III versus Group V
Average dissimilarity: 40.05

		Av. Abun					
		Group VI	Group VIII	Av. Diss	Sim/SD	Contrib%	Cum.%
Striped shrimp	*Macrobrachium intermed.*	**3.61**	1.37	5.53	2.52	13.80	13.80
Glass goby	*Gobiopterus semivestitus*	**5.36**	3.13	5.44	2.00	13.57	27.37
Pink shrimp	*Acetes sibogae australis*	1.63	**2.50**	2.95	1.41	7.36	34.73
Sandy sprat	*Hyperlophus vittatus*	**1.18**	0.38	2.14	1.36	5.34	40.07
Flat-tail mullet	*Liza argentea*	**1.81**	1.03	2.03	1.51	5.07	45.14
Port Jacks. glassfish	*Ambassis jacksoniensis*	0.54	**1.23**	1.91	1.50	4.76	49.90
Fan-tail mullet	*Paramugil georgii*	**0.79**	0.07	1.81	1.55	4.52	54.42
Tamar River goby	*Afurcagobius tamarensis*	0.94	0.94	1.57	1.44	3.92	58.34
Sea mullet	*Mugil cephalus*	**0.72**	0.35	1.52	1.13	3.80	62.13
School prawn	*Metapenaeus macleayi*	0.14	**0.71**	1.48	1.38	3.70	65.83
Yellow-finned bream	*Acanthopagrus australis*	0.54	**0.66**	1.16	1.55	3.10	66.93
Largemouth goby	*Redigobius macrostoma*	0.50	**0.79**	1.15	1.42	3.00	69.93

Group IV versus Group V
Average dissimilarity: 38.80

		Av. Abun					
		Group VII	Group VIII	Av. Diss	Sim/SD	Contrib%	Cum.%

Pink shrimp	*Acetes sibogae australis*	0.53	**2.50**	6.71	2.7	17.28	17.28
Glass goby	*Gobiopterus semivestitus*	2.02	**3.13**	3.63	1.51	9.37	26.65
Flat-tail mullet	*Liza argentea*	0.29	**1.03**	2.62	1.73	6.75	33.39
School prawn	*Metapenaeus macleayi*	**0.85**	0.71	2.39	1.37	6.17	39.56
Port Jacks. glassfish	*Ambassis jacksoniensis*	0.65	**1.23**	2.13	1.65	5.50	45.07
Tamar River goby	*Afurcagobius tamarensis*	0.68	**0.94**	1.78	1.59	4.59	49.65
Striped shrimp	*Macrobrachium intermed.*	1.03	**1.37**	1.76	1.58	4.52	54.18
Largemouth goby	*Redigobius macrostoma*	0.42	**0.79**	1.51	2.17	3.88	58.06
Sandy sprat	*Hyperlophus vittatus*	0.00	**0.38**	1.27	1.03	3.26	61.32
Blue-spot goby	*Pseudogobius olorum*	**1.36**	1.20	1.25	1.51	3.21	64.53
Yellow-finned bream	*Acanthopagrus australis*	0.32	**0.66**	1.24	1.29	3.21	67.74

Crabhole Creek versus combined references and controls

Group VI versus Group VII
Average dissimilarity: 45.19

		Av. Abun					
		Group IX	Group X	Av. Diss	Sim/SD	Contrib%	Cum.%
Pink shrimp	*Acetes sibogae australis*	**2.61**	0.40	6.42	2.68	14.20	14.20
Mangrove goby	*Mugilogobius paludis*	0.18	**1.50**	3.80	2.95	8.41	22.61
Port Jacks. glassfish	*Ambassis jacksoniensis*	**1.17**	0.10	3.17	2.37	7.01	29.62
Blue-spot goby	*Pseudogobius olorum*	1.14	**2.07**	2.81	1.98	6.21	35.83
Glass goby	*Gobiopterus semivestitus*	**3.32**	2.60	2.77	1.41	6.14	41.97
School prawn	*Metapenaeus macleayi*	**0.86**	0.13	2.11	1.71	4.67	46.64
Striped shrimp	*Macrobrachium intermed.*	1.51	**2.11**	1.93	1.36	4.26	50.90
Largemouth goby	*Redigobius macrostoma*	**0.92**	0.36	1.81	1.68	4.02	54.92
Tamar River goby	*Afurcagobius tamarensis*	**1.02**	0.73	1.81	1.36	4.00	58.92
Half-bridled goby	*Arenigobius frenatus*	0.33	**0.91**	1.74	1.80	3.85	62.76
Flat-tail mullet	*Liza argentea*	**1.04**	0.70	1.73	1.46	3.82	66.58
Yellow-finned bream	*Acanthopagrus australis*	**0.71**	0.20	1.57	1.70	3.47	70.05
Check. mang. goby	*Mugilogobius stigmaticus*	0.16	**0.62**	1.51	1.35	3.35	73.40

Group VI versus Group VIII
Average dissimilarity: 42.75

		Av. Abun					
		Group IX	Group XI	Av. Diss	Sim/SD	Contrib%	Cum.%
Pink shrimp	*Acetes sibogae australis*	**2.61**	0.42	7.59	2.67	17.77	17.77
Glass goby	*Gobiopterus semivestitus*	**3.32**	2.25	3.67	1.63	8.58	26.35
Port Jacks. glassfish	*Ambassis jacksoniensis*	**1.17**	0.45	2.64	1.60	6.18	32.53
Largemouth goby	*Redigobius macrostoma*	**0.92**	0.22	2.44	2.50	5.70	38.23
Tamar River goby	*Afurcagobius tamarensis*	**1.02**	0.47	2.42	1.79	5.67	43.89
School prawn	*Metapenaeus macleayi*	**0.86**	0.48	2.31	1.56	5.39	49.29
Flat-tail mullet	*Liza argentea*	**1.04**	0.64	2.15	1.52	5.04	54.32
Striped shrimp	*Macrobrachium intermed.*	**1.51**	1.27	1.88	1.51	4.40	58.72
Yellow-finned bream	*Acanthopagrus australis*	**0.71**	0.27	1.54	1.55	3.59	62.32
Blue-spot goby	*Pseudogobius olorum*	**1.14**	1.00	1.49	1.28	3.49	65.80
Mangrove goby	*Mugilogobius paludis*	0.18	**0.58**	1.44	1.77	3.37	69.17

Group VII versus Group VIII
Average dissimilarity: 41.83

		Av. Abun					
		Group X	Group XI	Av. Diss	Sim/SD	Contrib%	Cum.%
Blue-spot goby	*Pseudogobius olorum*	**2.07**	1.00	4.08	1.94	9.75	9.75
Striped shrimp	*Macrobrachium intermed.*	**2.11**	1.27	3.45	1.72	8.25	18.01
Mangrove goby	*Mugilogobius paludis*	**1.50**	0.58	3.44	2.11	8.22	26.23
Half-bridled goby	*Arenigobius frenatus*	**0.91**	0.24	2.64	1.79	6.31	32.54
Bridled goby	*Arenigobius bifrenatus*	**0.93**	0.37	2.20	1.37	5.27	37.81
Flat-tail mullet	*Liza argentea*	**0.70**	0.64	2.14	1.42	5.12	42.93
Check. mang. goby	*Mugilogobius stigmaticus*	**0.62**	0.10	2.08	1.35	4.97	47.90
Glass goby	*Gobiopterus semivestitus*	**2.60**	2.25	2.06	1.40	4.92	52.82

Tamar River goby	*Afurcagobius tamarensis*	**0.73**	0.47	1.89	1.55	4.51	57.33
Port Jacks. glassfish	*Ambassis jacksoniensis*	0.10	**0.45**	1.67	1.31	3.99	61.32
School prawn	*Metapenaeus macleayi*	0.13	**0.48**	1.64	0.86	3.93	65.24
Pink shrimp	*Acetes sibogae australis*	0.40	**0.42**	1.51	1.26	3.61	68.86
Southern blue-eye	*Pseudomugil signifer*	**0.51**	0.32	1.42	0.94	3.39	72.25

4.4. Discussion

4.4.1. Changes in marshes following culvert removal

No substantial change was observed in the assemblage at Crabhole Flats during the three year observation period. There was, however, a measurable change in the assemblage at Fish Fry Flats after the removal of the culvert. But many of these changes were also observed in the reference marshes, and therefore are likely to have been caused by influences beyond that of the manipulation.

Not all of the changes in species abundances observed at Fish Fry Flats/reference sites were also observed at control marshes. There was a slight increase in the number estuarine-marine species in the manipulated marsh, whereas in the control marshes there was a slight decrease in the number of estuarine-marine species and increase in freshwater-estuarine species. As a consequence of this, two years after manipulation, culverted control sites had slightly more freshwater-estuarine species and less estuarine-marine species than all other marshes. This supports the notion presented in the previous chapter that whilst culverts may not change the fish and decapod assemblage of marshes in its entirety, they appear to impact different functional groupings within the assemblage, relating to a shift in guilds of fish based around their ability to tolerate salinity extremes and need to access estuarine-marine environments. This differential impact on nekton assemblages has been observed in culverted salt marsh in the New England region of the United States of America (Raposa and Roman 2001, 2003).

A species of particular interest in our study due to its documented ability to withstand disturbance and negatively impact on native fish, frogs and ecosystems is the introduced mosquitofish (see discussion of previous chapter, Grubb 1972, Hurlbert *et al.* 1972, Hurlbert and Mulla 1981, Baber *et al.* 2004). This species increased significantly in abundance in the control marshes between years three and four and was largely responsible for driving the differential trajectories seen between the control sites and Fish Fry Flats/reference marshes. The low abundance, although consistent presence of yellowfin goby, indicates this introduced species has also adapted to its new home after presumably been introduced to the Port of Newcastle in ballast water (Hutchings 1992).

The changes at the marshes were subtle when compared to the tidal creeks. This is hardly surprising given the assemblages in control marshes did not differ significantly from reference sites in the first two of the three years studied (Chapter 3). Other studies have reported similar findings, with moderately restricted marshes containing viable nekton assemblages equivalent to unrestricted marshes (Raposa and Roman 2003, Eberhardt *et al.* 2011). This may suggest that the marshes of Kooragang have not been as significantly impacted by tidal restriction as tidal creeks. If so, this response may relate to the fact that marsh assemblages are driven by water level at spring tide under natural circumstances, and that while culverts may reduce tidal height and prism they do not eliminate it. Hence, marsh assemblages can establish at lower elevations when culverts or other tidal barriers are constructed. Removal of culverts will see a change in tidal dynamics that allows assemblages to re-establish at former and higher elevations. Rise in sea level due to climate change is expected to further exacerbate this where connectivity with suitable higher elevation substrata exists.

It has been well-documented that increasing the access of estuarine-marine dwelling species to marsh surfaces can have numerous ecosystem benefits (Zedler (2000) and references within). Nekton migrating onto marsh surfaces during high tides have access to rich foraging areas and tend to have fuller guts and more varied diets than those fish restricted to creeks (Kneib 1997, West and Zedler 2000). The shallower, vegetated habitat of marshes can also provide refuge from larger predatory fish (Boesch and Turner 1984). As discussed in the previous chapter, besides benefiting individuals and populations, increased access of species to productive feeding and refuge areas on marsh surfaces, combined with their subsequent ability to return to the estuary may be an important process which cycles energy and nutrients from the wetlands into the open estuary.

An important caveat with any of the marsh results presented here, is the short study period (three years total/two years post culvert removal). It is possible that more significant shifts including successional changes occurred beyond this sampling frame as increased tidal penetration into Fish Fry Flats was noted in subsequent years (Howe *et al.* 2010). A progressive increase in water depth after year four was so great as to assume fyke nets could no longer be relied upon to effectively and reliably sample the habitats. As well, it became difficult to find habitat in which to set the fyke nets that was unobstructed by pneumatophores (there was an explosion in extent of *Avicennia marina* that prior to manipulation had formed a small encircling ring around the marshes). Because this sampling bias could not be resolved, fyke net data at all locations beyond year four was disregarded for the purpose of this report. At some future time the utility of the data from the later years can be more closely inspected to determine the degree of bias and the time at which bias became unacceptable.

4.4.2. Successional changes in tidal creeks following culvert removal

A comparison between control (culverted), reference (unculverted) and manipulated creeks at Kooragang Island suggests that the installation of culverts many decades ago significantly changed the fish and decapod assemblage of tidal creeks. One of the more conspicuous ecological changes is likely to have been the exclusion of many estuarine-marine species upstream of culverts, which resulted in a reduction in species richness and abundance. Numerous species were absent or consistently less abundant in tidally-restricted creeks, including: glass goby, Port Jackson glassfish, flat-tail mullet, yellow-finned bream, largemouth goby, pink shrimp, school prawn (all estuarine-marine dwelling), striped shrimp and Tamar River goby (both freshwater-estuarine dwelling).

It is important to note that not all estuarine-marine dwelling species appeared disadvantaged by the presence of tidally-restrictive culverts. This has been noted in studies elsewhere (Raposa and Roman 2003, Eberhardt *et al.* 2010), where restricted wetlands that maintain some degree of tidal connectivity can support viable assemblages of estuarine-marine species. In fact, the habitat and inter-specific conditions created in these disturbed environments may be of direct advantage to certain species of fish (Raposa 2008). At Kooragang Island, mangrove goby, blue-spot goby, half-bridled goby, bridled goby and checkered mangrove goby were consistently more abundant in creeks with culverts. This may be in part due to their demersal (bottom-dwelling) nature (Cole and Shapiro 1995) and preference for muddy substrates (Allen *et al.* 1989), the latter being typical of tidally-restricted creeks (Raposa 2008).

Tidal creeks are also thought to function as important refugia for small fish avoiding larger predatory fish (Rozas and Odum 1988, Ruiz *et al.* 1993, Paterson and Whitfield 2000). Since yellow-finned bream (a piscivore) was more abundant in the absence of culverts, it may be possible that restricted creeks provide more suitable refuges than unrestricted creeks for small-bodied species such as gobies. However, this assumption is questionable considering the small size of a large proportion of bream sampled during this study. In addition, no significant increase in predatory fish species paralleled the observed decrease in mangrove goby, blue-spot goby, half-bridled goby, bridled goby and checkered mangrove goby following culvert removal, and it is unlikely that increased predation based on a reduction in the refuge value of the creek was driving this initial response.

Succession is a sequence of directional changes in the composition of a community (or assemblage of species) towards a stable condition (Shugart 2001). In this study we were able to observe clear succession in the fish and decapod assemblage of Fish Fry Creek over 16 years in response to culvert removal. The first change occurred immediately and persisted for at least six years following culvert removal. Although there was no net increase or decrease in the mean number of species inhabiting the creek, there were significant changes in assemblage composition. The goby species previously mentioned as dominant in tidally-restricted creeks became less abundant and a reciprocal increase in species such as flat-tail mullet, fantail-mullet and sea mullet was observed.

Mullet are highly mobile and have been shown to quickly colonise habitats once passage is improved (Kroon and Ansell 2006, Boys *et al.* 2011). They move immediately into tidal creeks once floodgates are opened and frequently use newly constructed fishways and widened culverts in coastal streams (Boys *et al.* 2011). Because mullet feed predominately on benthic organic matter and other detritus (Blaber 1977) they are important primary consumers in estuarine foodwebs. Studies using stable isotope analysis (Deegan and Garritt 1997) and bioenergetic modelling (Kneib 2003) show that fish can play a major role in the exchange of energy and nutrients between coastal wetlands and the estuary. By feeding in tidal creeks and subsequently returning to estuarine and offshore environments where they interact with predators and prey, species such as mullet may play an important role in the export of energy and nutrients out of wetlands (Kneib 1986, Deegan and Garritt 1997) This concept was defined as ‘trophic relay by Kneib (2003). Given that rehabilitated wetlands such as Kooragang can act as important carbon sinks (Howe *et al.* 2009), improved passage rates of nekton may help translate these energy stores into increased ecosystem and fisheries productivity.

Changes in the assemblage was potentially driven by a change in habitat. Culvert removal at Fish Fry Creek allowed significant increases in tidal exchange that resulted in increased velocities and extensive widening and deepening of the channel through erosion (Howe *et al.* 2010). Although it was not quantified in this study, these changes in velocity and geomorphology would have changed benthic habitats and made the sites sampled less desirable for the previously mentioned demersal goby species. The increasing tidal range and subsequent migration of creek habitats upstream into higher elevations (Howe *et al.* 2010), may have lead to a movement of these species further up the creek, away from the sampling sites that were randomly occupied within a fixed zone throughout the study. To resolve this uncertainty, it is recommended that in future studies fish and decapod sampling be conducted at extended distances along culverted, reference and manipulated creeks, to better account for spatial differences in species distributions.

A second shift in the assemblage of Fish Fry Creek was observed after year 8 (6 years after culvert removal). At this time the assemblage of Fish Fry Creek appeared to have matured as it was equivalent to that of reference creeks. Most importantly, this change was not seen at control creeks, providing strong evidence that culvert removal was responsible. Notable changes in Fish Fry Creek at this time involved a reduction in the abundance of the mullet species (fan-tail, flat-tail and sea mullet), that were previously noted to be colonisers immediately after culvert removal. The reduction of these benthic detrital feeding species may reflect changes in foraging conditions that occurred as the tidal creek matured into a deeper, wider channel (Howe *et al.* 2010) where greater tidal exchange flushes sediments and organic matter from the wetlands. Other species to significantly decrease in abundance at this time were sandy sprat, glass goby and striped shrimp.

Whilst there were notable declines in abundance of some species, many others increased and overall there was a net gain in the number of estuarine-marine species moving into Fish Fry Creek during this secondary maturation stage. School prawn, Port Jackson glassfish, pink shrimp, yellow-finned bream and large mouth goby increased in abundance. Other studies at the Macleay and Clarence River estuaries (Kroon and Ansell 2006, Boys *et al.* 2011) have observed pink shrimp in greater abundance in reference creeks relative to creeks with floodgates. This species is an important food source for carnivorous and omnivorous species co-habiting manipulated creeks, such as yellow-finned bream and large mouth goby (Xiao and Greenwood 1993). Thus an increase in prey may be driving reciprocal increases in other species.

Our findings are consistent with those of other studies from the Clarence, Macleay and Hunter Rivers that have shown that of prawn species inhabiting estuaries, school prawns appear to respond the greatest to rehabilitation in wetlands located in lower regions of estuaries (Kroon and Ansell 2006, Boys *et al.* 2011 and C.A. Boys unpublished data from Hexham Swamp). The data from Kooragang show significantly fewer king prawns at our sites compared to school prawns. Juvenile king prawn are historically known to inhabit wetlands in this section of estuary and commercial fisherman and fisheries officer reports show that Hexham Swamp (a wetland immediately adjacent to Kooragang Island on the South Arm of the Hunter River was a significant king prawn nursery (R. Hyde and C. Copeland, personal communication). It may be that king prawns are not occupying rehabilitated wetlands to the same extent that they did prior to the onset of tidal restriction and wetland degradation. No evidence contrary to this has been obtained in this study, or from three years of monitoring in Hexham Swamp following the opening of Iron Bark Creek floodgate and increased tidal flushing (C.A. Boys unpublished data). Further work, supplementing day-time seine netting with night-time prawn trawls may provide a more definitive picture about prawn responses to wetland rehabilitation.

A third shift in was observed across all manipulated, control and reference creeks sometime after year 10 for control sites and after year 12 for reference and manipulated sites. Typically, this change manifested in a reduction in species richness and abundance, and the general nature of it across all creeks strongly suggests it was caused by unknown broader-scale effects across the whole of Kooragang Island, in the Hunter estuary, or at even larger scales. Although reductions were largest at Fish Fry Creek, this may be contributed to by the collapse of the bridge across the creek at some stage between years 12 and 16, which may have significantly impacted on the passage of fish and decapods.

Speculating about the cause of broader-scale effects on fish and decapod populations across Kooragang Island or in the Hunter estuary is outside the scope of this study, but the observed changes highlight the importance of incorporating suitable reference locations into rehabilitation studies. We have shown that rehabilitated wetlands can develop along complex trajectories that may be difficult to predict. Without references it would be impossible to ascertain what unforseen changes can be attributed to the rehabilitation intervention and what may be purely due to apparently unpredictable disturbances that can be common in wetlands due to natural circumstances such as El Niño/La Niña-Southern Oscillation (ENSO) or due to changes in land use brought about by urbanised catchments.

When conducting rehabilitation projects in urbanised areas it can often be difficult to find locations that are sufficiently unaltered to be considered reference sites. This can make it all but impossible to gauge how manipulated locations are progressing towards rehabilitation goals (Grayson *et al.* 1999). But, in the absence of references, groupings of species such as those identified in this study as driving successional changes may provide useful indicator groups. That is, it may be possible to determine where on a rehabilitation response trajectory a system may be and when it is likely to reach maturation. For instance the dominance of benthic species and absence of mobile species such as yellow-finned bream and mullet may indicate a level of fragmentation between the wetland and estuary and hence a compromised trophic function. A subsequent change in assemblage composition reflecting a dominance of estuarine-marine, detrital feeders such as mullet may suggest that connectivity has been re-established, but that the system is early in its maturation and yet to reach a steady state. The variable nature of response seen between the two manipulated creeks within the same wetland (discussed below), demonstrates that the generality of these successional groupings require further consideration and testing within the context of other rehabilitation studies before their reliability can ascertained.

4.4.3. Variability in rehabilitation trajectories

Unlike Fish Fry Creek, the response to culvert removal at Crabhole Creek did not proceed as expected. Significant changes in the assemblage, reflecting a reduction in the abundance and number of species were detected in the first and third year following culvert removal, but were not observed in alternative years. Additionally, with the exception with the latter years (12 and 16) when assemblages declined across all Kooragang locations, the changes at Crabhole Creek were not observed at any of the reference or control creeks. This finding suggests that ecological responses to wetland rehabilitation are to some degree location-specific. This has been noted in other studies that have looked at compositional responses of fish and decapod assemblages to wetland remediation (Raposa 2008, Boys *et al.* 2012). Similarly, high inter-annual variability and weak directional response to rehabilitation are commonly reported for some indicators of ecosystem functioning but not others (Bishel-Machung *et al.* 1996, Simenstad and Thom 1996, Minello and Webb Jr 1997, Zedler and Callaway 2000). At the very least, this inconsistency shows that caution must be exercised when generalising ecological responses across different rehabilitation projects and demonstrates that it is imperative for all rehabilitation efforts to have some degree of pre- and post-manipulation evaluation. This is particularly relevant for wetland rehabilitation in disturbed urbanised environments, where a multitude of other stressors may be constraining responses (Grayson *et al.* 1999).

In trying to understand why some locations respond differently to rehabilitation than others, consideration is often given to either pre-settlement processes (such as the ability of individuals to colonise new habitats (Bell *et al.* 1988, Ford *et al.* 2010), or post-settlement processes such as competitive and predator-prey interactions between species (e.g. Raposa and Roman 2003). Crabhole Creek and Fish Fry Creek are both located on the South Arm of the Hunter River, with Fish Fry Creek located approximately 10km from the entrance to the Hunter River and Crabhole Flats situated a further four kilometres upstream. The closer proximity of Fish Fry Creek to the estuary mouth may be sufficient to create the difference in colonisation rates between the two locations and therefore different trajectories of rehabilitation response. Colonisation rates tend to be faster the closer a habitat is to a good supply of post-settlement individuals, although most research concerns artificial reefs (Bohnsack and Sutherland 1985, Matthews 1985, Alevizon and Gorham 1989, Hueckel *et al.* 1989, Golani and Diamant 1999). Within estuaries, it has been shown that fish densities in some habitats are higher the closer they are to the estuary mouth, possibly reflecting a greater supply of larvae and juvenile entering from offshore habitats (Bell *et al.* 1988, Hannan and Williams 1998).

It is also likely that the different responses between Fish Fry Creek and Crabhole Creek were mediated by the type and speed of habitat change, which differed substantially between creeks. As mentioned, culvert removal at Fish Fry Creek lead to significant increases in tidal exchange and resulted in increased velocities and widening and deepening of the channel through erosion (Howe *et al.* 2010). This change in geomorphology appears to have been substantial enough to contribute to the collapse of the bridge after the “Pasha Bulker” storm in June of 2007 (between years 12 and 16). Although mangrove colonisation proliferated in Fish Fry Flats and Fish Fry Creek after culvert removal (Howe *et al.* 2010), this occurred further up the creek than the sampling sites. In comparison, Crabhole Creek underwent little if any change in water velocity or creek depth. A slight increase in mangrove density at Crabhole Creek made sampling with seine nets more difficult, the mangrove pnuematophores providing greater refuge for small-bodied fish and decapods from larger predatory fish (Sasekumar *et al.* 1992, Primavera 1997). This may explain why predatory fish such as yellow-finned bream did not utilise Crabhole Creek after culvert removal as much as they did in the deeper and more open habitats afforded by the reference creeks and Fish Fry Creek following remediation. Conversely, it may explain why Crabhole Creek with its absence of these predators and more refuge, had a higher abundance of the small-bodied goby species that were typically found to be more abundant in restricted creeks.

5. CONCLUSIONS AND RECOMMENDATIONS

5.1. Responses to culvert removal at Kooragang Island

This study has provided irrefutable evidence that the presence of culverts at Kooragang Island has reduced the richness and changed the composition of fish and decapod assemblages of tidal creeks. These changes are driven by a reduction in the number and abundance of estuarine-marine dwelling species. Whilst the impact of culverts on marsh assemblages are less discernable than those of tidal creeks, there is evidence that certain estuarine-marine species have less access to restricted marshes, whilst other species appear to change their distribution to occupy tidally restrictive creeks. A number of demersal goby species were in particular found to be more abundant in culverted creeks.

Most importantly, it has been demonstrated that culvert removal can lead to clear successional changes and a trajectory of improvement in fish and decapod assemblages in tidal creeks. Whilst the assemblages of manipulated creeks that have had culverts removed can become equivalent to unculverted reference creeks, this may take many years and is not experienced at all locations. Whilst various mechanisms are discussed as possibly being responsible for successional changes and variability of responses between Fish Fry Creek and Crabhole Creek, many of these warrant further investigation.

Short-term responses in Fish Fry Flats and Crabhole Flats (marshes) were less evident than in their associated tidal creeks. A subtle response in manipulated marshes that may be evident, however, is that they track more closely in condition to unculverted marshes through time, whereas culverted marshes can move in a different assemblage direction. In this instance, culverted controls became dominated by invasive mosquitofish, whereas manipulated and reference marshes did not. Significantly more temporal replication is required in culverted, unculverted and manipulated marshes to resolve this uncertainty.

5.2. Recommendations for further research at Kooragang Island

Sampling at fixed sites throughout the study was unable to resolve uncertainties regarding possible shifts in nekton-habitat use within the wetland, as channels and habitats migrated as a result of culvert removal. As such, species that were found to decline in rehabilitated creeks, such as several demersal goby species, may have done so purely as a result of movement further into the wetlands. That is, they moved beyond the sampling frame of the fixed sampling sites. If this was indeed the case, responses to rehabilitation could have been underestimated. To resolve this uncertainty, it is recommended that all locations be re-sampled at greater spatial intensity. That is, sampling sites should be located at various points longitudinally from the creek mouths all the way to the creek-marsh interface. This should also resolve some of the mechanisms responsible for variable responses between the two manipulated creeks. Since the collapsed bridge at Fish Fry Creek has been replaced and passage restored since the sixteenth year of sampling, any additional sampling will also resolve whether the divergence of Fish Fry Creek from reference state in the final year was an artefact of the bridge collapse or a real product of continuing succession in this manipulated wetland.

Given the significant hydrological change that occurred in manipulated marshes over the past 16 years, it is likely that there have also been significant changes in the assemblage of marshes that were not detected by the limited dataset collected with three years of sampling. It would be beneficial to resolve the difficulties that were faced with fyke sampling in this study and re-sample manipulated, control and reference marshes at this point in time so that comparisons can be made after the marshes have matured over a sufficiently long time post culvert removal..

The successional changes observed in this study have raised interesting questions regarding the potential role of wetland rehabilitation in estuarine productivity. In particular, the role that improved nekton passage plays in the export of energy and nutrients from wetlands to the estuary and marine environment, and the contribution this makes to fisheries productivity. As well as having a set of long-established rehabilitated, control and reference wetland habitats at Kooragang, similar rehabilitation projects are underway in adjacent wetlands like Hexham Swamp and Tomago Wetlands. The strategic way that these works are being applied (within a BACI framework) and the dedication that has been shown to date into monitoring long-term responses means that the lower Hunter River provides a unique opportunity to test the relative contribution these different works are having on the productivity of this estuary. Answering these questions would involve stable isotope analyses to establish trophic links, carbon sources and energy movements through the estuary, linking these back to rehabilitated, impacted and more natural creeks and marshes.

5.3 General recommendations for coastal wetland rehabilitation studies

The Kooragang Wetland Project has demonstrated benefits that can arise by treating rehabilitation projects as experiments within a rigorous scientific framework that maximises the potential to learn from the results. This seems particularly pertinent for coastal wetland rehabilitation, where responses can be site-specific and have a strong landscape context. At the very least, it demonstrates why it is prudent to continuously evaluate responses so that activities can be adaptively managed if unforeseen or negative trajectories eventuate. Based on the results of our study, we concur with Zedler and Callaway (2000) and Choi (2004) that when developing wetland rehabilitation projects it is important to acknowledge:

1. The unpredictable nature of ecological assemblages and assume that multiple response trajectories are possible and responses may be location-specific (even within the same wetland).
2. Manipulated sites can show an improvement in the nature of their assemblages.
3. The time taken for manipulated wetlands to reach equivalency with reference states may exceed the usual 3-5 year monitoring period of rehabilitation projects.
4. Manipulated sites may never fully replace natural systems in composition or function.

We have demonstrated the importance of adopting rigorous experimental design incorporating appropriate controls and references. This is particularly relevant for long-term studies so that the true response to wetland manipulation can be ascertained from background variation occurring at scales beyond the treatment (which can be significant as illustrated at Kooragang over the last decade). In acknowledging these points, we recommend that where possible the rehabilitation of coastal wetlands or ecological communities of significance should be monitored for at least five to ten years after works have ceased to ensure that any responses (as rapid as they may be) are sustained and in-line with long-term objectives.

5.4. Summary of recommendations

1. Long-term studies (5-10 years) are needed to appreciate responses to rehabilitation strategies such as the replacement of culverts with bridges.
2. Short-term studies should be recognised as giving only a partial indication of response to rehabilitation efforts.
3. Rehabilitated tidal marshes and tidal creeks may contribute to trophic relay and should be further investigated for the Hunter estuary.

4. The presence of distinct groups of fish and/or decapods may indicate stages in the maturation of a rehabilitated wetland and should be further investigated as a potential way of determining whether rehabilitation outcomes are being met.

6. REFERENCES

Able, K. W., Witting, D. A., McBride, R. S., Rountree, R. A. and Smith, K. J. (1996). Fishes of polyhaline estuarine shores in Great Bay-Little Egg Harbor, New Jersey: a case study of seasonal and habitat influences. In: K. F. Nordstrom and C. T. Roman (eds.). *Estuarine shores: evolution, environments, and human alterations*. John Wiley and Sons, West Sussex, England, 335–353 pp.

Able, K. W., Grothues, T. M., Hagan, S. M., Kimball, M. E., Nemerson, D. M. and Taghon, G. L. (2008). Long-term response of fishes and other fauna to restoration of former salt hay farms: multiple measures of restoration success. *Reviews in Fish Biology and Fisheries* **18**: 65-97.

Alevizon, W. S. and Gorham, J. C. (1989). Effects of artificial reef deployment on nearby resident fishes. *Bulletin of Marine Science* **44**: 646-661.

Allen, G. R., Midgley, S. H. and Allen, M. (1989) *Freshwater fishes of Australia*. T.F.H Publications, Sydney. 240 pp.

Alongi, D. M. and McKinnon, A. D. (2005). The cycling and fate of terrestrially-derived sediments and nutrients in the coastal zone of the Great Barrier Reef shelf. *Marine Pollution Bulletin* **51**: 239-252.

Aronson, J. and Le Floc'h, E. (1996). Vital landscape attributes: missing tools for restoration ecology. *Restoration Ecology* **4**: 377-387.

Baber, M. J., Babbitt, K. J. and Douglas, M. E. (2004). Influence of habitat complexity on predator–prey interactions between the fish (*Gambusia holbrooki*) and tadpoles of *Hyla squirella* and *Gastrophryne carolinensis*. *Copeia* **2004**: 173-177.

Baker, R. and Sheaves, M. (2007). Shallow-water refuge paradigm: conflicting evidence from tethering experiments in a tropical estuary. *Marine Ecology Progress Series* **349**: 13-22.

Barbier, E. B. and Strand, I. (1998). Valuing mangrove-fishery linkages. A case study of Campeche, Mexico. *Environmental Resource Economics* **12**: 151-166.

Beck, M. W., Heck Jr, K. L., Able, K. W., Childers, D. L., Eggleston, D. B., Gillanders, B. M., Halpern, B., Hays, C. G., Hoshino, K. and Minello, T. J. (2001). The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates. *BioScience* **51**: 633-641.

Bell, J. D., Steffe, A. S. and Westoby, M. (1988). Location of seagrass beds in estuaries: effects on associated fish and decapods. *Journal of Experimental Marine Biology and Ecology* **122**: 127-146.

Bishel-Machung, L., Brooks, R. P., Yates, S. S. and Hoover, K. L. (1996). Soil properties of reference wetlands and wetland creation projects in Pennsylvania. *Wetlands* **16**: 532-541.

Blaber, S. (1977). The feeding ecology and relative abundance of mullet (Mugilidae) in Natal and Pondoland estuaries. *Biological Journal of the Linnean Society* **9**: 259-275.

Boesch, D. F. and Turner, R. E. (1984). Dependence of fisheries species on salt marshes: The role of food and refuge. *Estuaries* **7**: 460-468.

Bohnsack, J. A. and Sutherland, D. L. (1985). Artificial reef research: a review with recommendations for future priorities. *Bulletin of Marine Science* **37**: 11-39.

Boys, C. A., Glasby, T., Kroon, F. J., Baumgartner, L. J., Wilkinson, K., Reilly, G. and Fowler, T. (2011). *Case studies in restoring connectivity of coastal aquatic habitats: floodgates, box culvert and rock-ramp fishway*. Fisheries Final Report Series No. 130. NSW Department of Primary Industries, Cronulla, 70 pp.

Boys, C. A., Kroon, F. J., Glasby, T. and Wilkinson, K. (2012). Improved fish and crustacean passage in tidal creeks following floodgate remediation. *Journal of Applied Ecology* **49**: 223-233.

Bray, J. R. and Curtis, J. T. (1957). An ordination of the upland forest communities of Southern Wisconsin. *Ecological Monographs* **27**: 325-349.

Burdick, D. M., Dionne, M., Boumans, R. M. and Short, F. T. (1996). Ecological responses to tidal restorations of two northern New England salt marshes. *Wetlands Ecology and Management* **4**: 129-144.

Callaway, J. C. (2005). The challenge of restoring functioning salt marsh ecosystems. *Journal of Coastal Research* **40**: 24-36.

Chambers, R. M., Meyerson, L. A. and Saltonstall, K. (1999). Expansion of Phragmites australis into tidal wetlands of North America. *Aquatic Botany* **64**: 261-273.

Choi, Y. D. (2004). Theories for ecological restoration in changing environment: toward 'futuristic' restoration. *Ecological Restoration* **19**: 75-81.

Clarke, K. R. and Green, R. H. (1988). Statistical design and analysis for a 'biological effects' study. *Marine Ecology Progress Series* **46**: 213-226.

Clarke, K. R. (1993). Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology* **18**: 117-143.

Clarke, K. R. and Gorley, R. N. (2006) *PRIMER v6: User manual/tutorial*. PRIMER-E, Plymouth, UK. 91 pp.

Clarke, K. R., Somerfield, P. J. and Chapman, M. G. (2006). On resemblance measures for ecological studies, including taxonomic dissimilarities and a zero-adjusted Bray-Curtis coefficient for denuded assemblages. *Journal of Experimental Marine Biology and Ecology* **330**: 55-80.

Clarke, K. R., Somerfield, P. J. and Gorley, R. N. (2008). Testing of null hypotheses in exploratory community analyses: similarity profiles and biota-environment linkage. *Journal of Experimental Marine Biology and Ecology* **366**: 56-69.

Cole, K. S. and Shapiro, D. Y. (1995). Social facilitation and sensory mediation of adult sex change in a cryptic, benthic marine goby. *Journal of Experimental Marine Biology and Ecology* **186**: 65-75.

Coltheart, L. (1997) *Between Wind & Water, A History of the Ports and Coastal Waterways of New South Wales*. Hale & Iremonger Pty. Ltd. 208 pp.

Daiber, F. C. (1986) *Conservation of Tidal Marshes*. Van Nostrand Reinhold Company, New York.

Deegan, L. A. and Garritt, R. H. (1997). Evidence for spatial variability in estuarine food webs. *Marine Ecology Progress Series* **147**: 31-47.

DEH (2005). *Background paper to the wildlife conservation plan for migratory shorebirds*. Department of the Environment and Heritage, Canberra.

Dick, T. M. and Osunkoya, O. O. (2000). Influence of tidal restriction floodgates on decomposition of mangrove litter. *Aquatic Botany* **68**: 273-280.

Dionee, M., Short, F. and Burdick, D. (1999). Fish utilization of restored, created and reference salt-marsh habitat in the Gulf of Maine. *American Fisheries Society Symposium* **22**: 384-404.

Eberhardt, A. L., Burdick, D. M. and Dionne, M. (2010). The Effects of Road Culverts on Nekton in New England Salt Marshes: Implications for Tidal Restoration. *Restoration Ecology*: DOI: 10.1111/j.1526-100X.2010.00721.x.

Eberhardt, A. L., Burdick, D. M. and Dionne, M. (2011). The effects of road culverts on nekton in New England salt marshes: Implications for tidal restoration. *Restoration Ecology* **19**: 776-785.

Edgar, G. J., Barrett, N. S., Graddon, D. J. and Last, P. R. (2000). The conservation significance of estuaries: a classification of Tasmanian estuaries using ecological, physical and demographic attributes as a case study. *Biological Conservation* **92**: 383-397.

Ford, J. R., Williams, R. J., Fowler, A. M., Cox, D. R. and Suthers, I. M. (2010). Identifying critical estuarine seagrass habitat for settlement of coastally spawned fish. *Marine Ecology Progress Series* **408**: 181-193.

Furukawa, K., Wolanski, E. and Mueller, H. (1997). Currents and sediment transport in mangrove forests. *Estuarine, Coastal and Shelf Science* **44**: 301-310.

Genders, A. J. (2001). Distribution and abundance of larval fishes and invertebrates along the Hunter River estuary, New South Wales, Australia, with specific reference to the effects of floodgates. University of Newcastle, Newcastle. 217 pp.

Golani, D. and Diamant, A. (1999). Fish colonization of an artificial reef in the Gulf of Elat, northern Red Sea. *Environmental Biology of Fishes* **54**: 275-282.

Grayson, J. E., Chapman, M. G. and Underwood, A. J. (1999). The assessment of restoration of habitat in urban wetlands. *Landscape and Urban Planning* **43**: 227-236.

Grubb, J. C. (1972). Differential predation by *Gambusia affinis* on the eggs of seven species of anuran amphibians. *American Midland Naturalist* **88**: 102-108.

Hamer, A. J., Lane, S. J. and Mahony, M. J. (2002). Management of freshwater wetlands for the endangered green and golden bell frog (*Litoria aurea*): roles of habitat determinants and space. *Biological Conservation* **106**: 413-424.

Hannan, J. and Williams, R. J. (1998). Recruitment of juvenile marine fishes to seagrass habitat in a temperate Australian estuary. *Estuaries* **21**: 29-51.

Herke, W. H., Knudsen, E. E., Knudsen, P. A. and Rogers, B. D. (1992). Effects of semi-impoundment of Louisiana marsh on fish and crustacean nursery use and export. *North American Journal of Fisheries Management* **12**: 151–160.

Hobbs, R. J. and Norton, D. A. (1996). Towards a conceptual framework for restoration ecology. *Restoration Ecology* **4**: 93-110.

Howe, A. J., Rodrı´guez, J. R. and Saco, P. M. (2009). Surface evolution and carbon sequestration in disturbed and undisturbed wetland soils of the Hunter estuary, southeast Australia. *Estuarine, Coastal and Shelf Science* **84**: 75-83.

Howe, A. J., Rodrı´guez, J. F., Spencer, J., MacFarlane, G. R. and Saintilan, N. (2010). Response of estuarine wetlands to reinstatement of tidal flows. *Marine and Freshwater Research* **61**: 702-713.

Hueckel, G. J., Buckley, R. M. and Benson, B. L. (1989). Mitigating rocky habitat loss using artificial reefs. *Bulletin of Marine Science* **44**: 913-922.

Hurlbert, S. H., Zedler, J. and Fairbanks, D. (1972). Ecosystem alteration by mosquitofish (*Gambusia affinis*) predation. *Science* **175**: 639.

Hurlbert, S. H. and Mulla, M. S. (1981). Impacts of mosquitofish (*Gambusia affinis*) predation on plankton communities. *Hydrobiologia* **83**: 125-151.

Hutchings, P. (1992). Ballast water introductions of exotic marine organisms into Australia:: Current status and management options. *Marine Pollution Bulletin* **25**: 196-199.

Jordan, S. J., Smith, L. M. and Nestlerode, J. A. (2009). Cumulative effects of coastal habitat alterations on fishery resources: toward prediction at regional scales. *Ecology and Society* **14**: 16.

Kentula, M. W., Brooks, R. P., Gwin, S. E., Holland, C. C., Sherman, A. D. and Sifneos, J. C. (1992) *An Approach to Improving Decision Making in Wetland Restoration and Creation*. Island Press, Corvallis.

Kingsford, R. T., Ferster Levy, R., Geering, D., Davis, S. T. and Davis, J. S. E. (1998). *Rehabilitating Estuarine Habitat on Kooragang Island for Waterbirds, Including Migratory Wading Birds (May 1994 –May 1997)*. NSW National Parks and Wildlife Service, Hurstville.

Kneib, R. T. (1984). Patterns of invertebrate distribution and abundance in the intertidal salt marsh: causes and questions. *Estuaries* **7**: 392-412.

Kneib, R. T. (1986). The role of *Fundulus heteroclitus* in salt marsh trophic dynamics. *American Zoologist* **26**: 259.

Kneib, R. T. (1997). The role of tidal marshes in the ecology of estuarine nekton. *Oceanography and Marine Biology: an Annual Review* **35**: 163-220.

Kneib, R. T. (2003). Bioenergetic and landscape considerations for scaling expectations of nekton production from intertidal marshes. *Marine Ecology Progress Series* **264**.

Kroon, F. J., Bruce, A. M., Housefield, G. P. and Creese, R. G. (2004). *Coastal floodplain management in eastern Australia: barriers to fish and invertebrate recruitment in acid* *sulphate soil catchments*. Fisheries Final Report Series No. 67. NSW Department of Primary Industries, Cronulla, 212 pp.

Kroon, F. J. and Ansell, D. H. (2006). A comparison of species assemblages between drainage systems with and without floodgates: implications for coastal floodplain management. *Canadian Journal of Fisheries and Aquatic Sciences* **63**: 2400-2417.

Kruskal, J. B. and Wish, M. (1978) *Multidimensional Scaling*. Sage Publications, London. 93 pp.

Laegdsgaard, P. and Johnson, C. (2001). Why do juvenile fish utilise mangrove habitats? *Journal of Experimental Marine Biology and Ecology* **257**: 229-253.

Levin, L. A., Boesch, D. F., Covich, A., Dahm, C., Erséus, C., Ewel, K. C., Kneib, R. T., Moldenke, A., Palmer, M. A. and Snelgrove, P. (2001). The function of marine critical transition zones and the importance of sediment biodiversity. *Ecosystems* **4**: 430-451.

Manson, F. J., Loneragan, N. R., Harch, B. D., Skilleter, G. A. and Williams, L. (2005a). A broad-scale analysis of links between coastal fisheries production and mangrove extent: a case-study for northeastern Australia. *Fisheries Research* **74**: 69-85.

Manson, F. J., Loneragan, N. R., Skilleter, G. A., Phinn, S. R. and Gibson, R. N. (2005b). An evaluation of the evidence for linkages between mangroves and fisheries: a synthesis of the literature and identification of research directions. *Oceanography and Marine Biology: An Annual Review* **43**: 483-513.

Matthews, K. R. (1985). Species similarity and movement of fishes on natural and artificial reefs in Monterey Bay, California. *Bulletin of Marine Science* **37**: 252-270.

Mazumder, D., Saintilan, N. and Williams, R. J. (2006). Trophic relationships between itinerant fish and crab larvae in a temperate Australian saltmarsh. *Marine and Freshwater Research* **57**: 193-199.

McDowall, R. M. (ed.) (1996) *Freshwater fishes of south-eastern Australia*. Reed Books, Sydney.

Meynecke, J.-O., Lee, S. Y., Duke, N. C. and Warnken, J. (2007). The relationship between fish catch and estuarine habitats in Queensland, Australia. *Bulletin of Marine Science* **80**: 773-793.

Meynecke, J.-O., Lee, S. Y. and Duke, N. C. (2008). Linking spatial metrics and fish catch reveals the importance of coastal wetland connectivity to inshore fisheries in Queensland, Australia. *Biological Conservation* **141**: 981-996.

Meynecke, J.-O. (2009). Coastal habitat connectivity - implications for declared fish habitat networks in Queensland, Australia. *Pacific Conservation Biology* **15**: 96-101.

MHL (2003). *Hunter Estuary Process Study*. Manly Hydraulics Laboratory Report No. 1095.

Minello, T. and Webb Jr, J. (1997). Use of natural and created *Spartina alterniflora* salt marshes by fishery species and other aquatic fauna in Galveston Bay, Texas, USA. *Marine Ecology Progress Series* **151**: 165-179.

Morgan, P. A. and Short, F. T. (2002). Using functional trajectories to track constructed salt marsh development in the Great Bay estuary, Maine/New Hampshire, USA. . *Restoration Ecology* **10**: 461–473.

Morton, R. M. (1990). Community structure, density and standing crop of fishes in a subtropical Australian mangrove area. *Marine Biology* **105**: 385-394.

NSW Fisheries (1976). *Impact of dam construction and flood mitigation works on the aquatic environment.* NSW State Pollution Control Commission's Technical Advisory Committee. New South Wales State Fisheries, Sydney.

Paterson, A. W. and Whitfield, A. K. (2000). Do shallow-water habitats function as refugia for juvenile fishes? *Estuarine, Coastal and Shelf Science* **51**: 359–364.

Paterson, W. (1801). *Journal at Hunter's River*. Historical Records of New South Wales, Sydney.

Pollard, D. A. and Hannan, J. C. (1994). The ecological effects of structural flood mitigation works on fish habitats and fish communities in the lower Clarence river system of south-eastern Australia. *Estuaries* **17**: 427-461.

Pressey, R. L. and Middleton, M. J. (1982). Impacts of flood mitigation works on coastal wetlands in New South Wales. *Wetlands* **2**: 27-44.

Primavera, J. (1997). Fish predation on mangrove-associated penaeids: The role of structures and substrate. *Journal of Experimental Marine Biology and Ecology* **215**: 205-216.

Pyke, G. H. and White, A. W. (2000). Factors influencing predation on eggs and tadpoles of the endangered Green and Golden Bell Frog *Litoria aurea* by the introduced Plague Minnow Gambusia holbrooki. *Australian Zoologist* **31**: 496-505.

Raposa, K. B. and Roman, C. T. (2001). Seasonal habitat-use patterns of nekton in a tide-restricted and unrestricted New England salt marsh. *Wetlands* **21**: 451-461.

Raposa, K. B. and Roman, C. T. (2003). Using gradients in tidal restriction to evaluate nekton community responses to salt marsh restoration. *Estuaries and Coasts* **26**: 198-205.

Raposa, K. B. (2008). Early ecological responses to hydrologic restoration of a tidal pond and salt marsh complex in Narragansett Bay, Rhode Island. *Journal of Coastal Research* **55**: 180-192.

Reis, R. R. and Dean, J. M. (1981). Temporal variation in the utilization of an intertidal creek by the Bay Anchovy (*Anchoa mitchilli*). *Estuaries* **4**: 16-23.

Roman, C. T., Niering, W. A. and Warren, R. S. (1984). Salt marsh vegetation change in response to tidal restriction. *Environmental Management* **8**: 141-149.

Rountree, R. A. and Able, K. W. (1997). Nocturnal fish use of a New Jersey marsh creek and adjacent bay shoal habitats. *Estuarine, Coastal and Shelf Science* **44**: 703-711.

Roy, P. S. and Crawford, E. A. (1980). Quarternary geology of Newcastle Bight inner continental shelf, New South Wales, Australia. *New South Wales Geological Survey Records* **19**: 145-188.

Roy, P. S., Williams, R. J., Jones, A. R., Yassini, I., Gibbs, P. J., Coates, B., West, R. J., Scanes, P. R., Hudson, J. P. and Nichol, S. (2001). Structure and function of south-east Australian estuaries. *Estuarine, Coastal and Shelf Science* **53**: 351-384.

Rozas, L. P. and Odum, W. E. (1988). Food, predation risk, and microhabitat selection in a marsh fish assemblage. *Ecology* **69**: 1341-1351.

Ruiz, G. M., Hines, A. H. and Posey, M. H. (1993). Shallow water as a refuge habitat for fish and crustaceans in non-vegetated estuaries: an example from Chesapeake Bay. *Marine Ecology Progress Series* **99**: 1-16.

Saintilan, N. and Williams, R. J. (1999). Mangrove transgression into saltmarsh environments in south-east Australia. *Global Ecology and Biogeography* **8**: 117-124.

Saintilan, N. and Williams, R. J. (2000). Short note: The decline of saltmarsh in southeast Australia: results of recent surveys. *Wetlands (Australia)* **18**: 49-54.

Sasekumar, A., Chong, V., Leh, M. and D'Cruz, R. (1992). Mangroves as a habitat for fish and prawns. *Hydrobiologia* **247**: 195-207.

Shugart, H. H. (2001). Phenomenon of succession. In: S. A. Levin (ed.). *Encyclopedia of Biodiversity,Vol. 5*. Academic Press, San Diego, 541-552 pp.

Simenstad, C., Reed, D. and Ford, M. (2006). When is restoration not? Incorporating landscape-scale processes to restore self-sustaining ecosystems in coastal wetland restoration. *Ecological Engineering* **26**: 27-39.

Simenstad, C. A. and Thom, R. M. (1996). Functional equivalency trajectories of the restored Gog-Le-Hi-Te estuarine wetland. *Ecological Applications* **6**: 38-56.

Strayer, D. and Findlay, S. (2010). Ecology of freshwater shore zones. *Aquatic Sciences - Research Across Boundaries* **72**: 127-163.

Streever, W. J. (1997). Trends in Australian wetland rehabilitation. *Wetlands Ecology and Management* **5**: 5-18.

Streever, W. J. and Genders, A. J. (1997). Effect of improved tidal flushing and competitive interactions at the boundary between salt marsh and pasture. *Estuaries and Coasts* **20**: 807-818.

Streever, W. J. (1998). Kooragang Wetland Rehabilitation Project: opportunities and constraints in an urban wetland rehabilitation project. *Urban Ecosystems* **2**: 205-218.

Terlizzi, A., Bernedetti-Cecchi, L., Bevilacqua, S., Fraschetti, S., Guidetti, P. and Anderson, M. J. (2005). Multivariate and univariate assymetrical analyses in environmental impact assessment: a case study of Mediterranean subtidal sessile assemblages. *Marine Ecology Progress Series* **289**: 27-42.

Thayer, G. W. and Kentula, M. E. (2005). Coastal restoration: where have we been, where are we now, and where should we be going? *Journal of Coastal Research* **40**: 1-5.

Turner, J. (1997). *An environmental history of Ash Island. Report to Kooragang Wetland Rehabilitation Project*. Hunter History Consultants, Wallsend.

Underwood, A. J. (1991). Beyond BACI: Experimental designs for detecting human environmental impacts on temporal variations in natural populations. *Australian Journal of Marine and Freshwater Research* **42**: 569-587.

Valentine-Rose, L., Cherry, J. A., Jacob Culp, J., Perez, K. E., Pollock, J. B., Arrington, D. A. and Layman, C. A. (2007). Floral and faunal differences between fragmented and unfragmented Bahamian tidal creeks. *Wetlands* **27**: 702-718.

Vinagre, C., Salgado, J., Cabral, H. and Costa, M. (2011). Food web structure and habitat connectivity in fish estuarine nurseries—Impact of river flow. *Estuaries and Coasts* **34**: 663-674.

Vitousek, P. M., Mooney, H. A., Lubchenco, J. and Melillo, J. M. (1997). Human Domination of Earth's Ecosystems. *Science* **277**: 494-499.

Wagner, K. I., Gallagher, S. K., Hayes, M., Lawrence, B. A. and Zedler, J. B. (2008). Wetland restoration in the new millennium: Do research efforts match opportunities? *Restoration Ecology*.

Warren, R. S., Fell, P. E., Rozsa, R., Brawley, A. H., Orsted, A. C., Olson, E. T., Swamy, V. and Niering, W. A. (2002). Salt marsh restoration in Connecticut: 20 years of science and management. *Restoration Ecology* **10**: 497-513.

Weinstein, M. P., Weiss, S. L. and Walter, M. F. (1980). Multiple determinants of community structure in shallow marsh habitats, Cape Fear River estuary, North Carolina, USA. *Marine Biology* **58**: 227-243.

West, J. M. and Zedler, J. B. (2000). Marsh-creek connectivity: fish use of a tidal salt marsh in southern California. *Estuaries and Coasts* **23**: 699-710.

Williams, K. (2000). *Assessment of floodgated watercourses and drains for management improvements - Clarence river coastal floodplain*. Clarence River County Council, Grafton.

Williams, R. J., Hannan, J. and Balashov, V. (1995). *Kooragang Wetland Rehabilitation Project: fish, decapod crustaceans and their habitats, first interim report, summer 1993/94*. NSW Fisheries, Cronulla, 106 pp.

Williams, R. J. and Watford, F. A. (1997). Identification of structures restricting tidal flow in New South Wales, Australia. *Wetlands Ecology and Management* **5**: 87-97.

Williams, R. J., Watford, F. A., Taylor, M. A. and Button, M. L. (1998). New South Wales Coastal Aquatic Estate. *Wetlands (Australia)* **18**: 25-48.

Williams, R. J., Watford, F. and Balashov, V. (1999). *Kooragang Wetland Rehabilitation Project: change in wetland habitats over time*. NSW Fisheries, Cronulla.

Williams, R. J., Watford, F. A. and Balashov, V. (2000). *Kooragang Wetland Rehabilitation Project: History of changes in estuarine wetlands of the lower Hunter River*. Fisheries Final Report Series No. 22. NSW Fisheries, Cronulla, 82 pp.

Winter, T. C. (1999). Relation of streams, lakes, and wetlands to groundwater flow systems. *Hydrogeology Journal* **7**: 28-45.

Wolanski, E. (1995). Sediment transport in mangrove swamps. *Hydrobiologia* **295**: 51-58.

Woolls, W. (1867). *A Contribution to the Flora of Australia*. Sydney.

Xiao, Y. and Greenwood, J. G. (1993). The biology of *Acetes* (Crustacea; Sergestidae). *Oceanography and Marine Biology: An Annual Review* **31**: 259-444.

Zedler, J. B. (ed.) (2000) *Handbook for restoring tidal wetlands*. CRC Press LLC, Boca Raton.

Zedler, J. B. and Callaway, J. C. (2000). Evaluating the progress of engineered tidal wetlands. *Ecological Engineering* **15**: 211-225.

Zedler, J. G. and Callaway, J. C. (1999). Tracking wetland restoration: Do mitigation sites follow desired trajectories? *Restoration Ecology* **7**: 69-73.

APPENDICES

Table A1. Abundance of fish and decapods sampled in restricted (Wader Pond and Swan Pond) and unrestricted (Cobbans Marsh and Milhams Pond) **MARSHES** on Kooragang Island using fyke nets during the months of November, December, January and February in each of the years 1994/95, 1995/96 and 1996/97.

Family	Scientific Name	Common Name	Cobbans Marsh	Milhams Pond	Swan Pond	Wader Pond	Total	Length range (mm)	Salinity
FISH									
Atherinidae	*Pseudomugil signifer*	Southern blue-eye	3	1174		7	**1184**	11 - 34	F-E
Chandidae	*Ambassis jacksoniensis*	Port Jackson glassfish			17		**17**	29 - 36	E-M
Eleotridae	*Gobiomorphus australis*	Striped gudgeon		43	12	11	**66**	21 - 41	F
	Hypseleotris compressa	Empire gudgeon	1		4		**5**	23 - 31	F-E
	Philypnodon grandiceps	Flathead gudgeon	1	2	1		**4**	19 - 21	F-E
	Philypnodon macrostoma	Dwarf flathead gudgeon		1			**1**	31	F-E
Elopidae	*Elops hawaiensis**	Giant herring		1			**1**	38 -	F-E
Engraulidae	*Engraulis australis**	Australian anchovy	3				**3**	28 - 31	E-M
Gerreidae	*Gerres subfasciatus**	Silver biddy	11	52		674	**737**	11 - 37	E-M
Gobiidae	*Acanthogobius flavimanus*	Yellowfin goby	2				**2**	36 - 44	F-E
	Afurcagobius tamarensis	Tamar River goby	29		2	1	**32**	17 - 44	F-E
	Acentrogobius bifrenatus	Bridled goby	86	11	2	8	**107**	15 - 123	E-M
	Acentrogobius frenatus	Half bridled goby	2			2	**4**	22 - 57	E-M
	Gobiopterus semivestitus	Glass goby	7252	738	74	95	**8159**	15 - 32	E-M
	Mugilogobius paludis	Mangrove goby	108	63	29	204	**404**	16 - 52	E-M
	Mugilogobius stigmaticus	Checkered mangrove goby	25			2	**27**	11 - 49	E-M
	Pseudogobius olorum	Swan River goby	313	415	200	560	**1488**	12 - 46	F-E
	Redigobius macrostoma	Largemouth goby	2		1	1	**4**	15 - 19	E-M
Mugilidae	*Liza argentea**	Flat-tail mullet		15		3	**18**	25 - 38	E-M
	*Mugil cephalus**	Sea mullet			1	1	**2**	43 - 45	F-E
Platycephalidae	*Platycephalus fuscus**	Dusky flathead	1				**1**	9	E-M
Poeciliidae	*Gambusia holbrookii*	Mosquitofish		45	597	401	**1043**	13 - 45	F-E
Sillaginidae	*Sillago ciliata**	Sand whiting				1	**1**	39	E-M
Sparidae	*Acanthopagrus australis**	Yellow-finned bream	3	1			**4**	10 - 23	E-M
	*Rhabdosargus sarba**	Tarwhine			1		**1**	74	E-M
Terapontidae	*Terapon jarbua**	Crescent perch		1			**1**	14	F-E
Tetraodontidae	*Tetractenos hamiltoni*	Common toadfish				1	**1**	56	E-M
		TOTAL ABUNDANCE OF FISH	**7,842**	**2,562**	**941**	**1,972**	**13,317**		
		TOTAL NUMBER OF FISH SPECIES	**16**	**14**	**13**	**16**	**27**		
DECAPOD CRUSTACEA									
Alpheidae	*Alpheus spp.*	Pistol shrimp	3				**3**	5	E-M
Grapsidae	*Helograpsus haswellianus*	Honey shore crab				1	**1**	9	E-M
Grapsidae	*Paragrapsus laevis*	Shore crab		88		47	**135**	3 - 24	E-M
Grapsidae	*Sesarma erythrodactyla*	Red-fingered crab	2	1		1	**4**	7 - 11	E-M
Ocypodidae	*Australoplax tridentata*	Clown-faced crab		3	1		**4**	5	E-M
Ocypodidae	*Macrophthalmus setosus*	Blue-clawed sentinel crab		5		1	**6**	5 - 8	E-M
Palaemonidae	*Macrobrachium intermedium*	Striped shrimp	3229	1017	180	908	**5334**	2 - 8	F-E
Penaeidae	*Melicertus plebejus**	Eastern king prawn		6	1		**7**	5 - 10	E-M
Penaeidae	*Metapenaeus macleayi**	School prawn	2				**2**	12 - 21	E-M
Sergestidae	*Acetes sibogae australis*	Pink shrimp	634	9	68	6	**717**	2 - 8	E-M
		TOTAL ABUNDANCE OF DECAPOD	**3,870**	**1,129**	**250**	**964**	**6,213**		
		TOTAL NUMBER OF DECAPOD SPECIES	**5**	**7**	**4**	**6**	**10**		

Family Name	Scientific Name	Common Name	Dead Mangrove Creek	Wader Creek	Cobbans Creek	Mosquito Creek	Total	Length range (mm)	Salinity
FISH									
Anguillidae	*Anguilla australis**	Short-finned eel			2	1	**3**	680 -	F-E
Atherinidae	*Pseudomugil signifer*	Southern blue-eye	364		105	37	**506**	10 - 34	F-E
Chandidae	*Ambassis jacksoniensis*	Port Jackson glassfish			318	591 (2)	**909**	9 - 73	E-M
	Ambassis marianus	Ramsey's glassfish				45 (44)	**45**	68 - 90	E-M
Clupeidae	*Herklotsichthys castelnaui**	Southern herring			1 (1)	2 (2)	**3**	70 - 70	E-M
Clupeidae	*Hyperlophus vittatus**	Sandy sprat	1		62	6	**69**	18 - 41	E-M
Dasyatididae	*Dasyatis fluviorum*	Estuary stingray			1		**1**	-	E-M
Eleotridae	*Philypnodon grandiceps*	Flathead gudgeon	6		2	3	**11**	24 - 31	E-M
Engraulidae	*Elops hawaiensis**	Giant herring			3		**3**	30 - 34	F-E
Gerreidae	*Gerres subfasciatus**	Silver biddy	138	26	5	6	**175**	6 - 41	E-M
Girellidae	*Girella tricuspidata**	Blackfish				1	**1**	13 -	E-M
Gobiidae	*Acanthogobius flavimanus*	Yellowfin goby	4	2		3	**9**	30 - 71	F-E
	Afurcagobius tamarensis	Tamar River goby	276	17	182	158	**633**	17 - 52	F-E
	Acentrogobius bifrenatus	Bridled goby	212	609	12	183	**1,016**	12 - 128	E-M
	Acentrogobius frenatus	Half bridled goby	49	391	16	10	**466**	15 - 105	E-M
	Favonigobius exquisites	Exquisite sand goby	2	2	47	20	**71**	21 - 50	E-M
	Gobiopterus semivestitus	Glass goby	4,019	1,599	33,836	3,591	**43,045**	14 - 33	E-M
	Mugilogobius paludis	Mangrove goby	894	320	17	6	**1,237**	17 - 52	E-M
	Mugilogobius stigmaticus	Checkered mangrove goby	230	69	4	12	**315**	16 - 65	E-M
	Pseudogobius olorum	Swan River goby	2,830	1,434	217	635	**5,116**	10 - 42	F-E
	Redigobius macrostoma	Largemouth goby	31	5	78	235	**349**	12 - 39	E-M
Hemiramphidae	*Hyporhamphus regulatus*	River garfish				2	**2**	21 -	F-E
Mugilidae	*Liza argentea**	Flat-tail mullet	76	2	510 (67)	764 (71)	**1,352**	8 - 167	E-M
	*Mugil cephalus**	Sea mullet	53 (10)	45	67 (22)	93 (6)	**258**	32 - 478	F-E
Mugilidae	*Myxus elongatus**	Sand mullet	1				**1**	25 -	F-E
	*Paramugil georgii**	Fantail mullet	6		108 (105)	15 (10)	**129**	12 - 160	E-M
Paralichthyidae	*Pseudorhombus arsius**	Large-tooth flounder				1	**1**	60 -	E-M
	*Pseudorhombus jenynsii**	Small-tooth flounder			2 (1)		**2**	74 -	E-M
	*Pseudorhombus spp.**	Flounder spp.			1		**1**	20 -	E-M
Platycephalidae	*Platycephalus fuscus**	Dusky flathead			4 (2)	1 (1)	**5**	38 -	E-M
Poeciliidae	*Gambusia holbrookii*	Mosquitofish	33	1			**34**	15 - 39	F-E
Pomatomidae	*Pomatomus saltatrix*	Tailor			13 (4)	5 (1)	**18**	63 - 108	E-M
Scorpaenidae	*Centropogon australis*	Fortescue			9		**9**	8 - 63	E-M
	Notesthes robusta	Bullrout			1 (1)		**1**	255 -	F-E
Sillaginidae	*Sillago ciliata**	Sand whiting	1				**1**	32 -	E-M
Sparidae	*Acanthopagrus australis**	Yellow-finned bream	7	1	70	67 (1)	**145**	9 - 180	E-M
	*Rhabdosargus sarba**	Tarwhine	2		1	1	**4**	10 - 11	E-M
Tetraodontidae	*Tetractenos hamiltoni*	Common toadfish	1		4		**5**	15 - 70	E-M
		TOTAL ABUNDANCE OF FISH	**9,236**	**4,523**	**35,698**	**6,494**	**55,951**		
		TOTAL NUMBER OF FISH SPECIES	**23**	**15**	**30**	**28**	**38**		

Table A2. Abundance of fish (this page) and decapods (next page) sampled in restricted (Dead Mangrove and Wader Creek) and unrestricted (Cobbans and Mosquito Creek) **CREEKS** on Kooragang Island using seine and gill nets during the months of December, January and February in each of the years 1993/94, 1994/95, 1995/96 and 1996/97. The number of each species caught by gill nets is shown in parentheses. * = commercially important species. Salinity tolerance: Freshwater (F), Freshwater-Estuarine (F-E), Estuarine-Marine (E-M).

(continued over page)

Table A2. (continued from previous page)

Family Name	Scientific Name	Common Name	Dead Mangrove Creek	Wader Creek	Cobbans Creek	Mosquito Creek	Total	Length range (mm)	Salinity
DECAPOD CRUSTACEA									
Alpheidae	*Alpheus spp.*	Pistol shrimp	3	1	6	21	31	2 - 7	E-M
Grapsidae	*Paragrapsus laevis*	Shore crab	36	8	1		45	3 - 22	E-M
	Sesarma erythrodactyla	Red-fingered crab	13	9 (4)	1	1	24	3 - 18	E-M
Hymenosomatidae	*Halicarcinus ovatus*	Oval Spider Crab	2			1	3	4 - 10	E-M
Ocypodidae	*Australoplax tridentata*	Clown-faced crab	7	11	1	1	20	3 - 8	E-M
	Heloecius cordiformis	Semaphore crab	2	6 (4)		1	9	4 - 17	E-M
	Macrophthalmus setosus	Blue-clawed sentinel crab	1	11 (5)			12	10 - 21	E-M
Palaemonidae	*Macrobrachium cf novaehollandiae*	Long-armed prawn			2	2	4	17 - 25	E-M
	Macrobrachium intermedium	Striped shrimp	3,156	1,296	1,602	473	6,527	2 - 9	F-E
Penaeidae	*Melicertus plebejus**	Eastern king prawn	45	45	28	74	192	1 - 22	E-M
	*Metapenaeus bennettae**	Greasyback prawn		1	2	10	13	5 - 17	E-M
	*Metapenaeus macleayi**	School prawn	2 (1)	11	64	417 (1)	494	2 - 23	F-E
Portunidae	*Scylla serrata**	Mud crab		4 (4)	6 (4)	1	11	49 - 53	E-M
Sergestidae	*Acetes sibogae australis*	Pink shrimp	303	110	9,679	9,496	19,588	1 - 10	E-M
Xanthidae	*Xanthid spp.*	Stone crab				2	2	10 - 16	E-M
		TOTAL ABUNDANCE OF DECAPOD	**3,570**	**1,513**	**11,392**	**10,500**	**26,975**		
		TOTAL NUMBER OF DECAPOD SPECIES	**11**	**12**	**11**	**13**	**15**		

Family	Scientific name	Commmon name	DC	WC	WP	SP	CC	CM	MP	MC	FC	FF	CHC	CHF	Total	Salinity
		Number of samples	150	150	36	36	150	36	36	150	150	36	150	36	1,116	
FISH																
Anguillidae	*Anguilla australis**	Short-fined eel	1												**1**	F-E
	Anguilla reinhardtii	Long-finned eel	2				2			1					**5**	F-E
Atherinidae	*Pseudomugil signifer*	Southern blue-eye	73	7	7		154	3	1,174	18	3	2	1,772	3,881	**7,094**	F-E
Cepolidae	*Cepola australis*	Band fish					1								**1**	E-M
Chandidae	*Ambassis jacksoniensis*	Port Jackson glassfish	55	732		17	1,219			124	1,327	6	49	1	**3,530**	E-M
	Ambassis marianus	Ramsey's glassfish		3			13			62	1				**79**	E-M
Clupeidae	*Herklotsichthys castelnaui**	Southern herring					3			6	2				**11**	E-M
	*Hyperlophus vittatus**	Sandy sprat	1				18			91	1,446	115	2	4	**1,677**	E-M
Dasyatididae	*Dasyatis fluviorum*	Estuary stingray					1								**1**	E-M
	Dasyatis spp.	Stingray								1					**1**	E-M
Eleotridae	*Gobiomorphus australis*	Striped gudgeon			11	12			43				63	1	**130**	F
	Hypseleotris compressa	Empirefish				4	1	1			2			2	**10**	F-E
	Philypnodon grandiceps	Flathead gudgeon	37	4		1	12	1	2	16	1		49	1	**124**	F-E
	Philypnodon macrostoma	Dwarf flathead gudgeon	1						1						**2**	F-E
Elopidae	*Elops hawaiensis**	Giant herring							1		8	1			**10**	F-E
Engraulidae	*Engraulis australis**	Australian anchovy					6	3							**9**	E-M
Gerreidae	*Gerres subfasciatus**	Silver biddy	2	272	674		2	11	52	54	146	1	12	27	**1,253**	E-M
Girellidae	*Girella tricuspidata**	Blackfish	2	4			2			3	3		2	1	**17**	E-M
Gobiidae	*Acanthogobius flavimanus*	Yellowfin goby	9	1			1	2		12	18	1	2		**46**	F-E
	Acentrogobius bifrenatus	Bridled goby	274	199	8	2	78	86	11	445	114	61	144	3	**1,425**	E-M
	Acentrogobius frenatus	Half-bridled goby	244	75	2		27	2		158	569	13	181		**1,271**	E-M
	Afurcagobius tamarensis	Tamar River goby	527	29	1	2	1,219	29		426	492	6	188		**2,919**	F-E
	Cryptocentroides gobiodes	Oyster goby		2						3	8				**13**	E-M
	Favonigobius exquisites	Exquisite sand goby	2	28			52			26	42		2		**152**	E-M
	Gobiopterus semivestitus	Glass goby	9,955	4,472	95	74	46,926	7,252	738	8,838	97,382	5,238	8,829	1,186	**190,985**	E-M
	Mugilogobius paludis	Mangrove goby	1,329	559	24	29	7	18	63	23	136	18	833	135	**3,174**	E-M
	Mugilogobius stigmaticus	Checkered mangrove goby	327	85	2		6	25		22	23	19	79	17	**605**	E-M
	Pseudogobius olorum	Swan River goby	3,817	4,596	56	2	368	313	415	181	1,459	141	1,374	312	**13,034**	F-E
	Redigobius macrostoma	Largemouth goby	142	11	1	1	16	2		491	123		41	1	**829**	E-M
	Taenioides purpurascens	Eel Goby								1					**1**	E-M
Hemiramphidae	*Hyporhamphus regulatus*	River garfish								2					**2**	F-E
Mugilidae	*Liza argentea**	Flat-tail mullet	573	179	3		113		15	239	2,948	27	95	277	**4,469**	E-M
	*Mugil cephalus**	Sea mullet	65	169	1	1	78			278	395	1	158	12	**1,158**	F-E
	*Myxus elongatus**	Sand mullet	7							2	4				**13**	F-E
	*Paramugil georgii**	Fantail mullet	39	3			115			75	461		35	7	**735**	E-M
Paralichthyidae	*Pseudorhombus arsius**	Large-tooth flounder					2			3					**5**	E-M
	*Pseudorhombus jenynsii**	Small-tooth flounder					3			1					**4**	E-M
	*Pseudorhombus spp.**	Flounder spp.					1								**1**	E-M
Platycephalidae	*Platycephalus fuscus**	Dusky flathead	1	1			8	1		8	5	1			**25**	E-M
Poeciliidae	*Gambusia holbrookii*	Mosquitofish	33	11	41	597			45		2	1	115	7	**852**	F-E
Pomatomidae	*Pomatomus saltatrix**	Tailor					2			15	1				**18**	E-M
Scorpaenidae	*Centropogon australis*	Fortescue					16				8				**24**	E-M
	Notesthes robusta	Bullrout					1								**1**	F-E
Sillaginidae	*Sillago ciliata**	Sand whiting	1	2	1		1				24	7	2		**38**	E-M
Sparidae	*Acanthopagrus australis**	Yellow-finned bream	37	22			17	3	1	213	27	2	51		**373**	E-M
	*Rhabdosargus sarba**	Tarwhine	5	7		1	1			3	33				**50**	E-M
Syngnathidae	*Urocampus carinirostris*	Hairy pipefish												1	**1**	F-E
Terapontidae	*Terapon jarbua**	Crescent perch							1				2		**3**	F-E
Tetraodontidae	*Tetractenos glaber*	Smooth toadfish	1								6				**7**	E-M
	Tetractenos hamiltoni	Common toadfish	1	2	1		8				66	2		1	**81**	E-M
		TOTAL ABUNDANCE OF FISH	**17,563**	**11,475**	**928**	**743**	**50,500**	**7,752**	**2,562**	**11,841**	**107,285**	**5,663**	**14,080**	**5,877**	**236,269**	
		TOTAL NUMBER OF FISH SPECIES	**30**	**26**	**16**	**13**	**37**	**16**	**14**	**33**	**34**	**20**	**24**	**20**	**50**	

Table A3. Abundance and taxa richness of fish (this page) and decapods (next page) sampled at all locations in Kooragang Island during this study. The number of samples contributing to each count at each location is given. Refer to Table 1 and Fig. 2 (Chapter 1) for the assignment of locations within the BACI design.

DC (C) = Dead Mangrove Creek
WC (C) = Wader Creek
WP (C) = Wader Pond
SP (C) = Swan Pond
CC (R) = Cobbans Creek
CM (R) = Cobbans Marsh
MP (R) = Milhams Pond
MC (R) = Mosquito Creek
FFC (M) = Fish Fry Creek
FFF (M) = Fish Fry Flats
CHC (M) = Crabhole Creek
CHF (M) = Crabhole Flats

* = commercially important species.
Salinity tolerance: Freshwater (F), Freshwater-Estuarine (F-E), Estuarine-Marine (E-M)

Table A3. (continued from previous page)

Family	Scientific name	Commmon name	DC	WC	WP	SP	CC	CM	MP	MC	FC	FF	CHC	CHF	Total	Salinity
		Number of samples	150	150	36	36	150	36	36	150	150	36	150	36	1,116	
DECAPOD CRUSTACEA																
Alpheidae	*Alpheus spp.*	Pistol shrimp	11	14			56	3		78	5	3	4		**174**	E-M
Grapsidae	*Grapsidae type 1*	Marsh crab								1	1				**2**	E-M
	Helograpsus haswellianus	Honey shore crab	3		1								1	1	**6**	E-M
	Paragrapsus laevis	Shore crab	44	1	47		2		88	1	18	12	6	86	**305**	E-M
	Sesarma erythrodactyla	Red-fingered crab	17	23	1		3	2	1	3	1		11	24	**86**	E-M
Hymenosomatidae	*Amarinus lacustris*	False Spider Crab		1											**1**	F-E
	Halicarcinus ovatus	Oval Spider Crab	2							2			1		**5**	E-M
Mysidae	*Mysidae*	Opossum shrimp					5						2		**7**	E-M
Ocypodidae	*Australoplax tridentata*	Clown-faced crab	13	25		1	2		3	6	8	13	11		**82**	E-M
	Heloecius cordiformis	Semaphore crab	2	6						4	3				**15**	E-M
	Macrophthalmus latifrons	Southern sentinel crab	1												**1**	E-M
	Macrophthalmus setosus	Blue-clawed sentinel crab	2	11	1				5		2	13			**34**	E-M
Palaemonidae	*Macrobrachium cf novaehollandiae*	Long-armed prawn	8				7			4	3				**22**	E-M
	Macrobrachium intermedium	Striped shrimp	9,219	1,813	98	18	298	3,229	117	1,339	18,664	9,847	372	679	**45,693**	F-E
Penaeidae	*Melicertus plebejus**	Eastern king prawn	58	73		1	61		6	12	237	4	3	4	**459**	E-M
	*Metapenaeus bennettae**	Greasyback prawn	1	11			6			11	3		4		**36**	E-M
	*Metapenaeus macleayi**	School prawn	22	55			214	2		1,142	23		19	4	**1,481**	E-M
Portunidae	*Portunus pelagicus*	Blue swimmer crab		1											**1**	E-M
	*Scylla serrata**	Mud crab		8			9			2	6		1		**26**	E-M
Sergestidae	*Acetes sibogae australis*	Pink shrimp	118	297	6	68	37,834	634	9	2,119	12,175	732	53	4	**54,049**	E-M
Xanthidae	*Xanthid spp.*	Stone crab								2					**2**	E-M
		TOTAL ABUNDANCE OF DECAPOD	**9,521**	**2,339**	**154**	**88**	**38,497**	**3,870**	**229**	**4,726**	**31,149**	**10,624**	**488**	**802**	**102,487**	
		TOTAL NUMBER OF DECAPOD SPECIES	**15**	**14**	**6**	**4**	**12**	**5**	**7**	**15**	**14**	**7**	**13**	**7**	**21**	

		Habitat type	CREEK						MARSH							
		Treatment	Control		Reference		Manipulated		Control		Reference		Manipulated			
		Location	DC	WC	CC	MC	FC	CHC	WP	SP	CM	MP	FF	CHF	Total	
		Number of samples	150	150	150	150	150	150	36	36	36	36	36	36	1,116	
Family	Scientific name	Commmon name														Salinity
FISH																
Anguillidae	*Anguilla australis**	Short-fined eel	1												**1**	F-E
	Anguilla reinhardtii	Long-finned eel	2		2	1									**5**	F-E
Atherinidae	*Pseudomugil signifer*	Southern blue-eye	73	7	154	18	3	1,772	7		3	1,174	2	3,881	**7,094**	F-E
Cepolidae	*Cepola australis*	Band fish			1										**1**	E-M
Chandidae	*Ambassis jacksoniensis*	Port Jackson glassfish	55	732	1,219	124	1,327	49		17			6	1	**3,530**	E-M
	Ambassis marianus	Ramsey's glassfish		3	13	62	1								**79**	E-M
Clupeidae	*Herklotsichthys castelnaui**	Southern herring			3	6	2								**11**	E-M
	*Hyperlophus vittatus**	Sandy sprat	1		18	91	1,446	2					115	4	**1,677**	E-M
Dasyatididae	*Dasyatis fluviorum*	Estuary stingray			1										**1**	E-M
	Dasyatis spp.	Stingray				1									**1**	E-M
Eleotridae	*Gobiomorphus australis*	Striped gudgeon						63	11	12		43		1	**130**	F
	Hypseleotris compressa	Empirefish			1		2			4	1			2	**10**	F-E
	Philypnodon grandiceps	Flathead gudgeon	37	4	12	16	1	49		1	1	2		1	**124**	F-E
	Philypnodon macrostoma	Dwarf flathead gudgeon	1									1			**2**	F-E
Elopidae	*Elops hawaiensis**	Giant herring					8					1	1		**10**	F-E
Engraulidae	*Engraulis australis**	Australian anchovy			6						3				**9**	E-M
Gerreidae	*Gerres subfasciatus**	Silver biddy	2	272	2	54	146	12	674		11	52	1	27	**1,253**	E-M
Girellidae	*Girella tricuspidata**	Blackfish	2	4	2	3	3	2						1	**17**	E-M
Gobiidae	*Acanthogobius flavimanus*	Yellowfin goby	9	1	1	12	18	2			2		1		**46**	F-E
	Acentrogobius bifrenatus	Bridled goby	274	199	78	445	114	144	8	2	86	11	61	3	**1,425**	E-M
	Acentrogobius frenatus	Half-bridled goby	244	75	27	158	569	181	2		2		13		**1,271**	E-M
	Afurcagobius tamarensis	Tamar River goby	527	29	1,219	426	492	188	1	2	29		6		**2,919**	F-E
	Cryptocentroides gobiodes	Oyster goby		2		3	8								**13**	E-M
	Favonigobius exquisites	Exquisite sand goby	2	28	52	26	42	2							**152**	E-M
	Gobiopterus semivestitus	Glass goby	9,955	4,472	46,926	8,838	97,382	8,829	95	74	7,252	738	5,238	1,186	**190,985**	E-M
	Mugilogobius paludis	Mangrove goby	1,329	559	7	23	136	833	24	29	18	63	18	135	**3,174**	E-M
	Mugilogobius stigmaticus	Checkered mangrove goby	327	85	6	22	23	79	2		25		19	17	**605**	E-M
	Pseudogobius olorum	Swan River goby	3,817	4,596	368	181	1,459	1,374	56	2	313	415	141	312	**13,034**	F-E
	Redigobius macrostoma	Largemouth goby	142	11	16	491	123	41	1	1	2			1	**829**	E-M
	Taenioides purpurascens	Eel goby				1									**1**	E-M
Hemiramphidae	*Hyporhamphus regulatus*	River garfish				2									**2**	F-E
Mugilidae	*Liza argentea**	Flat-tail mullet	573	179	113	239	2,948	95	3			15	27	277	**4,469**	E-M
	*Mugil cephalus**	Sea mullet	65	169	78	278	395	158	1	1			1	12	**1,158**	F-E
	*Myxus elongatus**	Sand mullet	7			2	4								**13**	F-E
	*Paramugil georgii**	Fantail mullet	39	3	115	75	461	35						7	**735**	E-M
Paralichthyidae	*Pseudorhombus arsius**	Large-tooth flounder			2	3									**5**	E-M
	*Pseudorhombus jenynsii**	Small-tooth flounder			3	1									**4**	E-M
	Pseudorhombus spp.*	Flounder spp.			1										**1**	E-M
Platycephalidae	*Platycephalus fuscus**	Dusky flathead	1	1	8	8	5				1		1		**25**	E-M
Poeciliidae	*Gambusia holbrookii*	Mosquitofish	33	11			2	115	41	597		45	1	7	**852**	F-E
Pomatomidae	*Pomatomus saltatrix**	Tailor			2	15	1								**18**	E-M
Scorpaenidae	*Centropogon australis*	Fortescue			16		8								**24**	E-M
	Notesthes robusta	Bullrout			1										**1**	F-E
Sillaginidae	*Sillago ciliata**	Sand whiting	1	2	1		24	2	1				7		**38**	E-M
Sparidae	*Acanthopagrus australis**	Yellow-finned bream	37	22	17	213	27	51			3	1	2		**373**	E-M
	*Rhabdosargus sarba**	Tarwhine	5	7	1	3	33			1					**50**	E-M
Syngnathidae	*Urocampus carinirostris*	Hairy pipefish												1	**1**	F-E
Terapontidae	*Terapon jarbua**	Crescent perch						2				1			**3**	F-E
Tetraodontidae	*Tetractenos glaber*	Smooth toadfish	1				6								**7**	E-M
	Tetractenos hamiltoni	Common toadfish	1	2	8		66		1				2	1	**81**	E-M
TOTAL ABUNDANCE OF FISH			**17,563**	**11,475**	**50,500**	**11,841**	**107,285**	**14,080**	**928**	**743**	**7,752**	**2,562**	**5,663**	**5,877**	**236,269**	
TOTAL NUMBER OF FISH SPECIES			**30**	**26**	**37**	**33**	**34**	**24**	**16**	**13**	**16**	**14**	**20**	**20**	**50**	

Table A4. Abundance and taxa richness of fish (this page) and decapods (next page) sampled at all locations in Kooragang Island during this study. The number of samples contributing to each count at each location is given.

DC = Dead Mangrove Creek
WC = Wader Creek
CC = Cobbans Creek
MC = Mosquito Creek
FFC = Fish Fry Creek
CHC = Crabhole Creek
WP = Wader Pond
SP = Swan Pond
CM = Cobbans Marsh
MP = Milhams Pond
FF = Fish Fry Flats
CHF = Crabhole Flats

* = commercially important species. Salinity tolerance: Freshwater (F), Freshwater-Estuarine (F-E), Estuarine-Marine (E-M)

		Habitat type	CREEK						MARSH							
		Treatment	Control		Reference		Manipulated		Control		Reference		Manipulated			
		Location	DC	WC	CC	MC	FC	CHC	WP	SP	CM	MP	FF	CHF	Total	
		Number of samples	150	150	150	150	150	150	36	36	36	36	36	36	1,116	
Family	Scientific name	Commmon name														Salinity
DECAPOD CRUSTACEA																
Alpheidae	*Alpheus* spp.	Pistol shrimp	11	14	56	78	5	4			3		3		**174**	E-M
Grapsidae	*Grapsidae* type 1	Marsh crab				1	1								**2**	E-M
	Helograpsus haswellianus	Honey shore crab	3					1	1					1	**6**	E-M
	Paragrapsus laevis	Shore crab	44	1	2	1	18	6	47			88	12	86	**305**	E-M
	Sesarma erythrodactyla	Red-fingered crab	17	23	3	3	1	11	1		2	1		24	**86**	E-M
Hymenosomatidae	*Amarinus lacustris*	False Spider Crab		1											**1**	F-E
	Halicarcinus ovatus	Oval Spider Crab	2			2		1							**5**	E-M
Mysidae	*Mysid* spp.	Opossum shrimp			5			2							**7**	E-M
Ocypodidae	*Australoplax tridentata*	Clown-faced crab	13	25	2	6	8	11		1		3	13		**82**	E-M
	Heloecius cordiformis	Semaphore crab	2	6		4	3								**15**	E-M
	Macrophthalmus latifrons	Southern sentinel crab	1												**1**	E-M
	Macrophthalmus setosus	Blue-clawed sentinel crab	2	11			2		1			5	13		**34**	E-M
Palaemonidae	*Macrobrachium cf novaehollandiae*	Long-armed prawn	8		7	4	3								**22**	E-M
	Macrobrachium intermedium	Striped shrimp	9,219	1,813	298	1,339	18,664	372	98	18	3,229	117	9,847	679	**45,693**	F-E
Penaeidae	*Melicertus plebejus**	Eastern king prawn	58	73	61	12	237	3		1		6	4	4	**459**	E-M
	*Metapenaeus bennettae**	Greasyback prawn	1	11	6	11	3	4							**36**	E-M
	*Metapenaeus macleayi**	School prawn	22	55	214	1,142	23	19			2			4	**1,481**	E-M
Portunidae	*Portunus pelagicus*	Blue swimmer crab		1											**1**	E-M
	*Scylla serrata**	Mud crab		8	9	2	6	1							**26**	E-M
Sergestidae	*Acetes sibogae australis*	Pink shrimp	118	297	37,834	2,119	12,175	53	6	68	634	9	732	4	**54,049**	E-M
Xanthidae	*Xanthid* spp.	Stone crab				2									**2**	E-M
TOTAL ABUNDANCE OF DECAPOD			**9,521**	**2,339**	**38,497**	**4,726**	**31,149**	**488**	**154**	**88**	**3,870**	**229**	**10,624**	**802**	**102,487**	
TOTAL NUMBER OF DECAPOD SPECIES			**15**	**14**	**12**	**15**	**14**	**13**	**6**	**4**	**5**	**7**	**7**	**7**	**21**	

Table A4. (continued from previous page).

DC = Dead Mangrove Creek
WC = Wader Creek
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Salinity tolerance: Freshwater (F), Freshwater-Estuarine (F-E), Estuarine-Marine (E-M)

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