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Changes in fish and crustacean assemblages in tidal creeks of Hexham Swamp following the staged opening of Ironbark Creek floodgates

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The collection of all animals in this project was done 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

Changes in fish and crustacean assemblages in tidal creeks of Hexham Swamp following the staged opening of Ironbark Creek floodgates

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Key words

Floodgates, Hunter River, tidal creek, tidal restriction, wetland rehabilitation

The Hunter River has experienced intensive flood mitigation over the past century, with 175 floodgates and hundreds of kilometres of levee banks and drainage canals being constructed. These developments have reduced the extent and biodiversity of the adjacent coastal wetlands such as Hexham Swamp. This Ramsar-listed wetland, located in the Hunter River estuary, was degraded by the installation of floodgates on Ironbark Creek in the early 1970s.

The Hexham Swamp Rehabilitation Project was established to promote the long-term rehabilitation of this important estuarine wetland ecosystem. The project has involved the staged opening of eight floodgates at the downstream end of Ironbark Creek to increase tidal flushing of the swamp and inundate low-lying lands with saline water. This involved three stages: one gate opened (Stage 1), three gates opened (Stage 2), six to eight gates opened (Stage 3).

This report documents the changing assemblage of aquatic fauna (fish and crustaceans) in Hexham Swamp over an 11-year period, between April 2004 and April 2014. It also outlines which of those changes, if any, may be attributed to the staged opening of the Ironbark Creek floodgates. To investigate the changes, seine netting was used to sample fish and crustaceans in tidal creeks above and below the Ironbark Creek floodgates. The catches were compared to nearby natural, un-gated reference creeks and a control creek that remained closed by a floodgate throughout the entire study.

Before the Ironbark Creek floodgates were opened, the following measurable impacts upstream of the floodgates were noted:

- reduced extent of suitable habitat for fish and crustaceans, with the upper site being less suitable than the lower site just upstream of the floodgate. This was reflected by the lower water quality in the upper site, including reduced pH (slightly acidic), dissolved oxygen and salinity
- lower species richness and abundance in the upper site, making the assemblage distinctly different from sites below the floodgates or in the other un-gated reference creeks

- far fewer estuarine–marine dwelling species in the upper reaches of the swamp, including the absence or substantially lower abundance of several commercially important species
- increased abundance of invasive Mosquitofish (Gambusia holbrooki) compared with un gated reference creeks.

After the floodgates were opened, significant recovery was noted in the upper site, which was more degraded than the lower site. Recovery was observed in both water quality and assemblage composition; by the end of the study, the upper site resembled the un-gated reference creeks. Most importantly, this response was not seen in the control creek where the floodgates remained closed. This provides strong evidence for the recovery being related to the opening of the floodgates. The recovery did not occur until all eight floodgates were opened in Stage 3.

Some notable responses in the upper site included:

- increased salinity, dissolved oxygen and pH, which all reached levels seen in un-gated creeks
- doubled species richness, including eight new species in Stage 3 alone. Most of the increase was seen in estuarine–marine dwelling species, but the number of freshwater–estuarine species also rose
- significantly increased abundance of many species, including several commercially important species
    - School prawn (*Metapenaeus macleayi*: 15 times more)
    - Eastern king prawn (*Melicertus plebejus*: absent prior to Stage 3)
    - Yellowfin bream (*Acanthopagrus australis*: 62 times more)
    - Flat-tail mullet (*Liza argentea*: 10 times more)
    - Silver biddy (*Gerres subfasciatus*: 19 times more)
    - School prawn and Eastern king prawn were never sampled in the upper site prior to Stage 1 and Stage 3 opening (respectively), but since then were detected on every sampling occasion
- significantly reduced abundance of Mosquitofish.

On one occasion since the floodgates were opened, fish and crustacean species richness and abundance declined beyond the 75% trigger value that would denote a detrimental impact under the Hexham Swamp Rehabilitation Project’s Operational and Environmental Monitoring Plan. This occurred in the upper Ironbark Creek site in July 2009, immediately following Stage 1 floodgate opening. It was a one-off observation, from which species richness and abundance recovered quickly.

In conclusion, the construction and closure of the Ironbark Creek floodgates significantly decreased the amount of suitable habitat for aquatic life within Hexham Swamp. As a result, the diversity and abundance of fish and crustaceans declined in tidal creeks. Opening the floodgates has reversed much of this impact, and restored connectivity between wetland and estuarine habitats for species across multiple levels in the food chain. It is therefore likely that ecosystem functioning and nutrient supply to the estuary has improved.

Significant improvements in species abundance and diversity were only seen once all eight gates were opened. This suggests that a threshold of increased tidal flushing, habitat improvement or water quality improvement may need to be exceeded before rehabilitation goals relating to fish and prawns can be realised.

The current study provides evidence to support the views of many commercial fishers – that installing the Ironbark Creek floodgates degraded important nursery habitat for species such as the school prawn and Eastern king prawn. The positive response of these and several other commercially important species to floodgate opening suggests that at least some of this impact has been reversed. The degree to which this translates into improved productivity of commercial and recreational fisheries will depend on the contribution of those juveniles occupying the rehabilitated nursery habitats to the adult exploited population. This is the subject of ongoing research by NSW Department of Primary Industries.

Introduction

Flood mitigation and degradation of Hexham Swamp

Hundreds of floodgates, levee banks and drainage canals have been constructed along the Hunter River to mitigate occasional flooding (Williams and Watford 1997). Flood mitigation works of this degree have significantly altered the hydrology of the surrounding floodplain, degrading large areas of estuarine wetlands (Boys and Williams 2012a, b). Most notably, tidal flow into floodplain wetlands has been restricted, which typically changes the species assemblage and reduces 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, Boys and Williams 2012b).

Hexham Swamp is the largest wetland in the Hunter River and one of the largest in New South Wales. It covers approximately 2000 ha around the Newcastle area. The ecosystem is internationally recognised under the Ramsar Convention due to its biological diversity and significance for migratory waterbirds.

In the early 1970s, as part of the Hunter Valley Flood Mitigation Scheme, floodgates were installed on Ironbark Creek – the main tributary of Hexham Swamp. This significantly reduced tidal inundation within Ironbark Creek and the creeks and marshes throughout Hexham Swamp (Haines 2011). Consequently, freshwater vegetation (mostly *Phragmites australis*) became dominant and saltmarsh and mangrove areas were replaced by freshwater reeds and pastures (Winning and Saintilan 2009).

The loss of estuarine habitat in Hexham Swamp also significantly reduced the diversity and abundance of estuarine fish (Genders 2001). Estuarine wetlands are important nurseries during early life stages of fish and invertebrate species, and contribute to the productivity of many estuarine and offshore fisheries (Morton 1990, Barbier and Strand 1998, Manson et al. 2005). Reports from commercial fishers indicate that Hexham Swamp once provided a major nursery for commercial species of fish and prawns (Simon Walsh, NSW DPI, unpublished data).

The Hexham Swamp Rehabilitation Project

The Hexham Swamp Rehabilitation Project was initiated for the long-term rehabilitation of the estuarine wetland ecosystem. As part of the project, floodgates located at the downstream end of Ironbark Creek were opened to increase tidal flushing of the swamp and periodically inundate low-lying lands with saline water. An important objective of the project was to re-establish estuarine wetland habitat and encourage recovery of estuarine wetland fauna.

The eight Ironbark Creek floodgates were opened in a staged manner, under the direction of a strict Operations and Environmental Management Plan (OEMP; Haines 2008). The primary reason for this staged approach was to adaptively manage any changes to the wetland, while maximising the opportunity for habitat and fauna recovery and minimising any negative environmental impacts (e.g. unexpected flooding, drops in water quality and associated fish kills, excessive erosion, mosquito outbreaks). The adaptive management framework involved regular monitoring of hydrology, water quality, vegetation, macroinvertebrates, birds, amphibians, fish and crustaceans.

Stage 1 of the Hexham Swamp Rehabilitation Project, which commenced on 19 December 2008, involved the partial opening of one floodgate. The resultant increase in tidal flows into and out of Hexham Swamp was modest, with only a small amount of overbank inundation (Haines 2009). The overall environmental condition within Ironbark Creek and Hexham Swamp appeared largely unaffected (Haines 2009).

Stage 2 of the project, which commenced one year later on 18 December 2009, involved the partial opening of three floodgates. This was equivalent to having two floodgates open completely, and resulted in the overbank inundation of 320 ha of land (BMT WBM 2011). Despite this increased inundation, no evidence was found that Stage 2 sufficiently increased tidal flushing in upstream sites within the swamp to improve fish and crustacean assemblages (Boys et al. 2012b).

Stage 3 commenced on 29 September 2011, when six floodgates were partially opened. The remaining floodgates were opened on 25 July 2013. All eight of the Ironbark Creek gates were then open to tidal flushing.

Scope of this report

This report documents the changing assemblage of fish and crustaceans in Hexham Swamp over an 11-year period (April 2004 to April 2014). It outlines which of those changes, if any, may be attributed to the staged opening of the Ironbark Creek floodgates. Of specific interest is whether estuarine–marine species (in this case, juvenile fish and crustaceans) have increased. The report also examines whether a detrimental threshold has been exceeded, as per the OEMP. Specifically, we sought to determine whether floodgate opening significantly reduced (by more than 75%) the abundance or diversity of aquatic species apart from those species expected to respond negatively, such as freshwater species (Haines 2008, 2009).

Methods

Experimental design

To evaluate the response to rehabilitation at Hexham Swamp, samples from Ironbark Creek (where floodgates were being opened or manipulated) were compared to reference creeks (not under the influence of floodgates) and an appropriate control creek (influenced by a floodgate that remained closed during the study) (Table 1, Figure 1 and Figure 2). This resulted in an unbalanced design consisting of one treatment or manipulated creek (Ironbark Creek), one control creek (Purgatory Creek) and two reference creeks (Cobbans and Mosquito creeks).

In each of the floodgated creeks, sampling was undertaken at a site immediately below the floodgate (referred to as 'below' sites) and at a site immediately above the floodgate ('lower' sites) (Figure 3). At Ironbark Creek, a site was also sampled in the upper reaches, known as Fisheries Creek ('upper' sites). Purgatory Creek, which is situated off the Hunter River upstream of Hexham Bridge, is too short for an upper site to be sampled, so only 'below' and 'lower' sites were sampled. The reference creeks are within the Kooragang Wetlands on Ash Island, and in each creek both an upper and lower site were sampled. Cobbans Creek is located off the south arm of the Hunter River, with its confluence with the Hunter River directly opposite that of Ironbark Creek (Figure 2). Mosquito Creek is located off the north arm of the Hunter River.

Fish and crustaceans were sampled at two replicate locations within each site (Table 1 and Figure 1) at various times before and after floodgate opening, during Stage 1, 2 and 3 (Figure 4). All 18 sites were sampled within a 1 to 2-week period during April, July, October and December.

Table 1 Spatial experimental design applied on all sampling occasions

Treatment	Creek	Site[a]	Replicate no. (location)[b]	Collated samples per location
Manipulated (floodgates opened during study)	Ironbark	Below floodgate	x 2 (IBB1, IBB2)	4 seines
		Lower Ck above floodgate	x 2 (IBL1, IBL2)	4 seines
		Upper Ck above floodgate	x 2 (IBU1, IBU2)	4 seines
Control (floodgate remained closed)	Purgatory	Below floodgate	x 2 (PCB1, PCB2)	4 seines
		Lower Ck above floodgate	x 2 (PCL1, PCL2)	4 seines
Reference (no floodgates, remained tidally unrestricted during study)	Cobbans	Lower Ck	x 2 (C2L1, C2L2)	4 seines
		Upper Ck	x 2 (C2U1, C2U2)	4 seines
	Mosquito	Lower Ck	x 2 (C6L1, C6L2)	4 seines
		Upper Ck	x 2 (C6U1, C6U2)	4 seines

a Refer to Figure 1 for schematic of spatial experimental design.
b Location codes in parentheses relate to Figure 1 and Figure 3.

Figure 1 Spatial experimental design showing two replicate locations (each comprised of four collated seines) per site within each creek

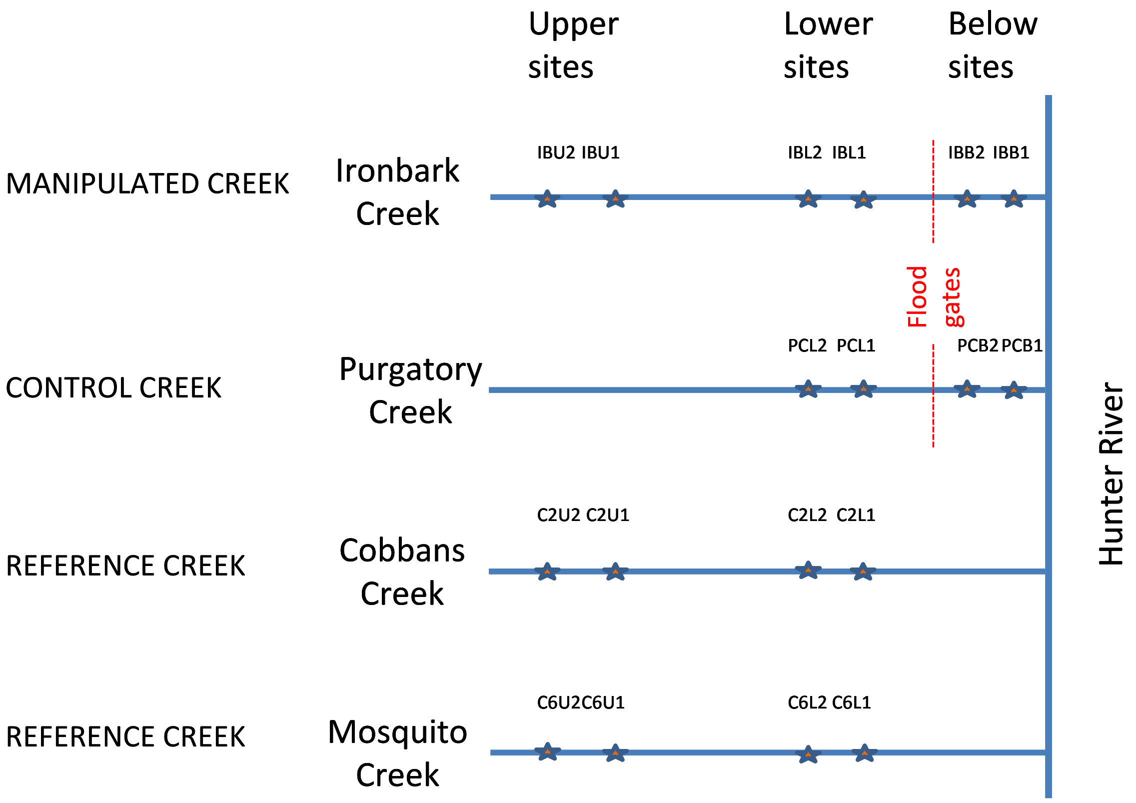
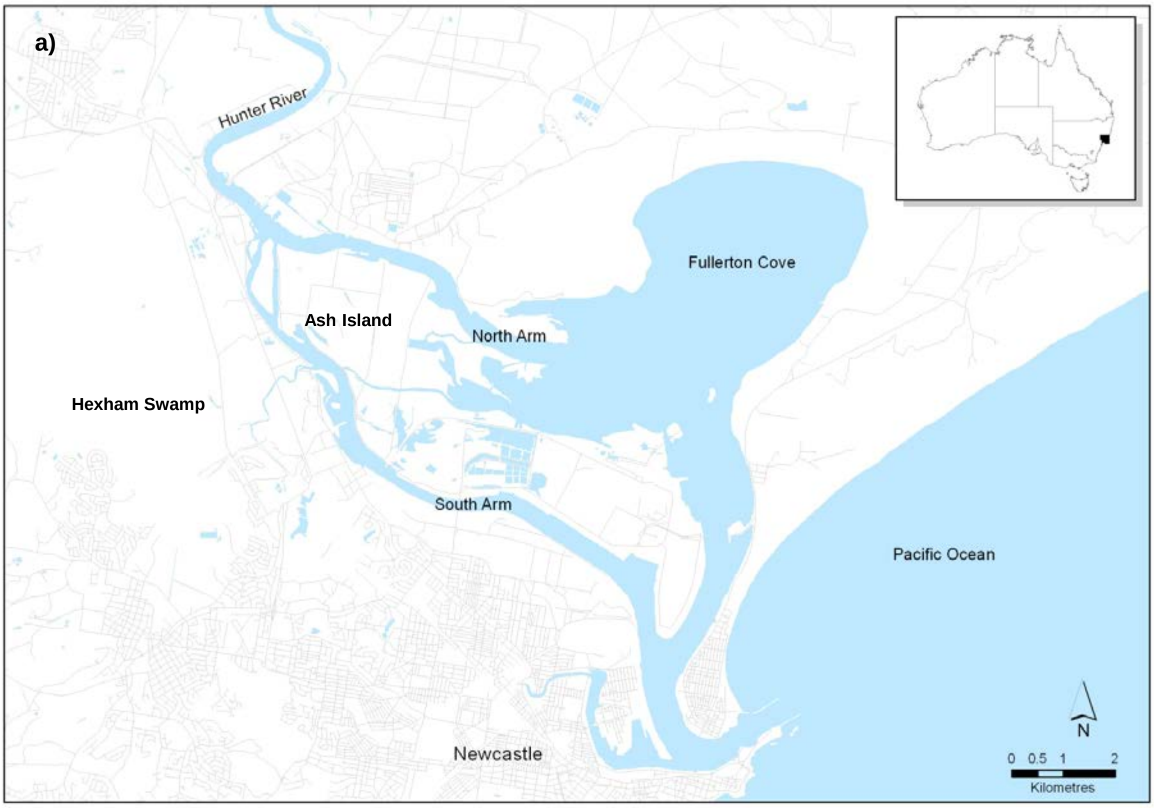


Figure 2 Map of study area showing a) the location of Hexham Swamp relative to the lower Hunter River, and b) the location of study tidal creeks



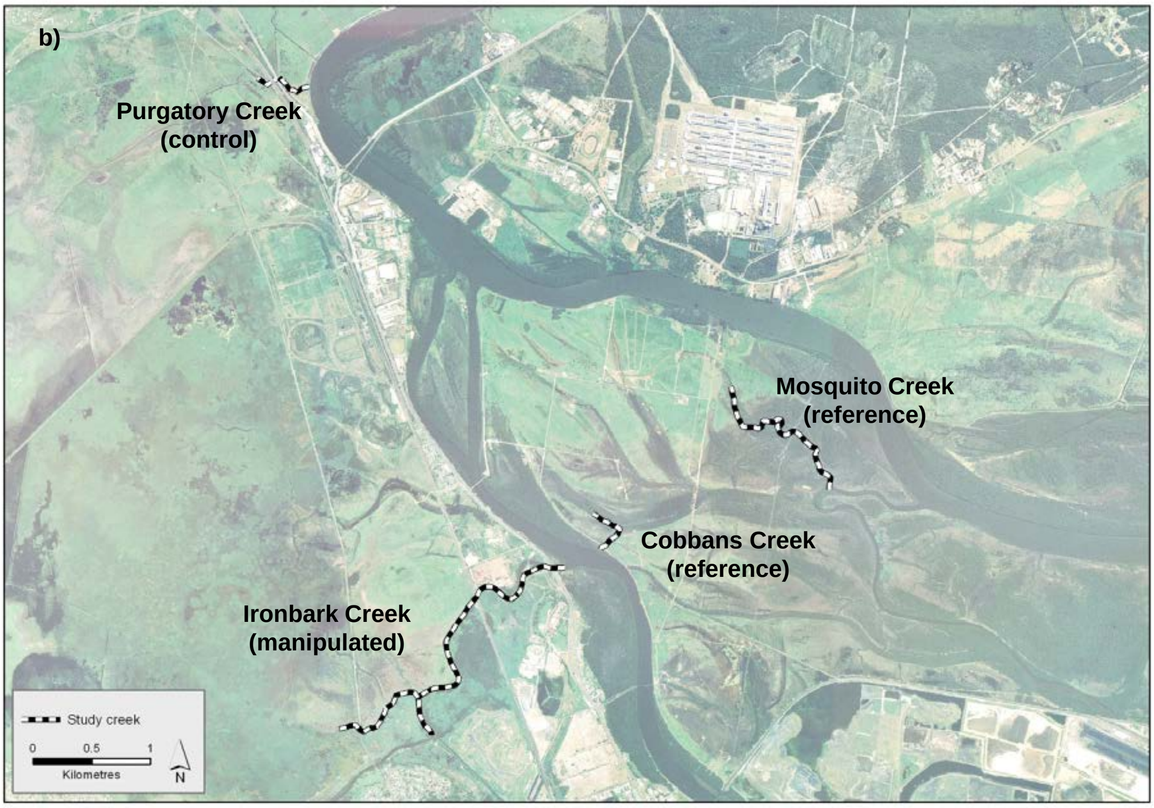
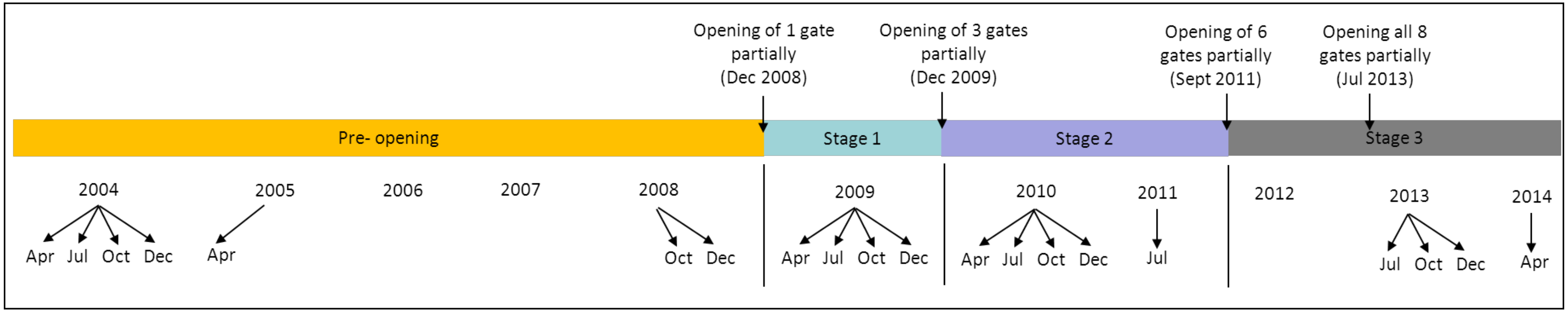


Figure 3 Sampling locations within a) Ironbark Creek, b) Purgatory Creek, c) Cobbans Creek and d) Mosquito Creek

Refer to Table 1 for explanation of location codes.

Figure 4 Sampling times throughout the 11-year study in relation to the staged opening of Ironbark Creek floodgates



Fish and crustacean sampling

Fish and decapod crustaceans were collected during daylight hours. Each of the two locations within a site was sampled using four seine net hauls (10 m headline x 1.5 m drop x 3 mm stretch mesh), performed in a 'U'-shape and pursed onto the shore (Figure 5). Hauls were spaced no closer than ~10 m apart. A separate pilot study using up to eight seine hauls was performed on various occasions. This verified that four hauls were adequate in capturing the vast majority of species present, with species seldom added with subsequent hauling (Bruce Pease, NSW DPI, unpublished data). Other studies employing this sampling method in other estuaries of NSW have also found that three seine hauls captured 86% of species present at one location (Kroon and Ansell 2006). Seine netting began shortly before high tide and was completed shortly after to coincide with maximum depth and minimum velocity. The original design included fyke netting in upper sites. However, this was subsequently removed from the design, because it was very time consuming and the data obtained did not affect the interpretation of assemblage composition (Craig Boys, NSW DPI, unpublished data).

Fish and decapods were placed in buckets of estuarine water following capture. Fork length was recorded for fish and carapace length for decapod crustaceans. Length data was not analysed in this report, but is archived in a Fisheries NSW database. Larger individuals that could easily be identified to species level were processed on site and released alive. All other individuals were euthanased by lethal dose of Ethyl-p-amino benzoate (Benzocaine) (100 mg L-1) and each seine net catch was preserved in a separate labelled bag of formalin solution for later processing in the laboratory (Figure 6).

Water quality measurements

At the time of sampling, pH, dissolved oxygen, temperature and salinity were measured at the surface and bottom of the water column at each sampling site using a handheld Horiba U10 water quality meter. Samples were averaged across depths.

Figure 5 Seine net being used to collect fish and crustacean samples

Figure 6 Preserved fish and crustacean samples after sorting in the laboratory

Hypothesis testing and statistical analyses

The catches from four seine hauls at each site were pooled to give a single sample per location, per site within a creek. In total, 18 location samples (two locations per nine sites) were analysed per sampling occasion, totalling 360 samples from the 20 sampling occasions spread across the 11-year study.

A suite of exploratory multivariate techniques (within the PRIMER v6 statistical package) were used to determine whether the fish and crustacean assemblages within Ironbark Creek were affected by the presence of floodgates, and subsequently, whether any noticeable change occurred that could be attributed to the staged opening of the gates. Prior to multivariate analysis, the data were fourth-root transformed to ensure that the rarer, less abundant species also contributed to assemblage dissimilarity, rather than differences being dominated purely by the most abundant species (e.g. ambassids, eleotrids, shrimps) (Clarke and Green 1988).

Patterns in the fish and crustacean assemblage data were interpreted visually using non-metric multidimensional scaling ordination (nMDS; Kruskal and Wish 1978), based on Bray-Curtis dissimilarities between samples (Bray and Curtis 1957). Before dissimilarities were calculated, the average abundance of each species across the two locations at each site for each sampling occasion was calculated, resulting in one data point per site, per time of sampling. From the ordination, it was possible to track the change in assemblage composition at each site within Ironbark Creek over time, relative to the assemblages in the control and reference sites.

Nonparametric permutational analyses of variance (PERMANOVA; Type III sum-of squares) (Anderson 2001, McArdle and Anderson 2001) was used to determine if assemblage composition within Ironbark Creek sites changed significantly among the different stages of floodgate opening. If the test of main effects showed that a change had occurred, pairwise comparisons among each stage were subsequently performed to determine when these changes happened within each site. For upper Ironbark Creek sites (which showed the biggest assemblage response), the Similarity Percentages (SIMPER) procedure (Clarke 1993) was used to determine the relative contribution of the suite of species to the significant assemblage change identified by the PERMANOVA.

To examine changes in guilds of species based on their requirement to move between different parts of the estuary, the following functional groups were used (after Boys et al. 2012a).

- Estuarine–marine (E–M): saltwater species that require access to either estuarine or oceanic waters.
- Freshwater–estuarine (F–E): euryhaline species that can occupy both freshwater and saltwater.
- Freshwater (F) species: those typically confined to freshwater tributaries of estuaries.

Mean species richness (number) was calculated for these functional groups, as well as for total species and commercially important species. The results were presented as interaction plots to allow differences among sites and stages of floodgate opening to be compared. Mean interaction plots were presented in a similar way for the water quality parameters, to determine whether any changes may have been associated with floodgate opening.

Under a scenario where the Ironbark Creek floodgates had degraded the aquatic assemblage and water quality within Hexham Swamp, and gate opening rehabilitated the swamp, the following response was hypothesised. Before gate opening, the Ironbark Creek sites above the floodgate would differ from the sites below the floodgate and the two un-gated reference creeks. Once the floodgates were opened, the assemblage and/or water quality would be expected to become more similar to the sites below floodgates and in the un-gated reference creeks. A positive response to floodgate opening would be further supported if no such change was observed above the Purgatory Creek floodgate, which remained closed as a control.

Reporting against OEMP detrimental impact

As part of the OEMP, it was necessary to determine whether a detrimental threshold of a 75% reduction in species diversity or abundance was exceeded at any time following the opening of the floodgates. In this report, species richness (i.e., the total number of species) was reported rather than diversity. This was because preliminary investigation revealed that both richness and diversity (Shannon H') showed the same pattern of response, and richness is an easier concept to interpret than a relativised diversity measure.

Using a statistical control chart approach (Anderson and Thompson 2004), the mean change in species richness and abundance (Ln(N+1) transformed) was plotted for the upper and lower sites of Ironbark Creek. A significant change was concluded to have occurred if at any time the mean values fell outside of the 95% confidence interval for the pre-opening mean value. Changes were then judged against a 75% value of this mean pre-opening condition. Since the focus was on those species expected to benefit from floodgate opening, only estuarine–marine species were included in the analysis.

Results

Catch summary

In total, 54 fish species (136,433 individuals) and 14 decapod species (235,081 individuals) were sampled from tidal creeks in the lower Hunter River (Appendix 1). Ambassid, goby, shrimp and prawn were the most abundant types of taxa caught, with goby being the most diverse family (18 different species). Two-thirds (66%) of the species caught were primarily estuarine–marine dwelling, with the remaining third (34%) being species capable of inhabiting both freshwater and estuarine habitats. Close to a third (31%) of the species caught were juveniles of commercially important species, which accounted for 13% of the total abundance. The most abundant commercial species sampled were Flat-tail and Sea mullet (*Liza argentea* and *Mugil cephalus*); School, Greasyback and Eastern king prawn (*Metapenaeus macleayi, M. bennettae* and *Melicertus plebejus*) and Yellowfin bream (*Acanthopagrus australis*).

Assemblage changes associated with floodgate opening

In Ironbark Creek, the composition of the assemblage differed significantly among sites (upper, lower, below) and among the different stages of floodgate opening (PERMANOVA, Table 2). The assemblage at sites below and just above (lower sites) the floodgate varied through time, although by Stage 3 (with eight gates open) the composition had returned to the pre-opening stage (Table 2). Throughout the entire study, these two Ironbark Creek sites remained similar to control sites below Purgatory Creek floodgate and the un-gated reference sites (Figure 7b & c).

The major change observed in assemblage composition throughout the study was at the upper Ironbark Creek site. At this site the change did not occur until Stage 3, but it persisted, and by the end of the study the assemblage was significantly different from before floodgate opening (Table 2). The shift in assemblage composition was from one similar to that above the control floodgate at the start of the study, to one that had become similar to the un-gated reference creeks by the end (Figure 7d). Notably, this change in assemblage composition was not seen at the lower control site above Purgatory Creek floodgate, where the gates remained closed. At this site, the assemblage remained distinctly dissimilar in composition to that below the floodgate and that of the un-gated reference creeks throughout the entire study (Figure 7a).

The permanent shift in assemblage composition that occurred in the upper Ironbark Creek site once all eight floodgates were opened was partly driven by a significant increase in the number of species present (a doubling in richness, or eight new species in Stage 3 alone) (Figure 8). Most of the increase was in estuarine–marine dwelling species, but freshwater–estuarine species also increased. More commercially important species were also found. The additional species that were not present prior to Stage 3 are shown in Table 3, and notably included the commercially important Eastern king prawn. As a result of species addition, by the end of the study upper Ironbark sites had similar species richness to below-floodgate and reference sites. The change did not occur in these upper sites until all eight gates were opened, and a similar response was not observed in the control sites above the floodgate that remained closed (Purgatory Creek).

As well as more species being present in the upper Ironbark Creek site following the Stage 3 gate opening, many species changed significantly in abundance. SIMPER revealed that 25 of the 41 species found at this site explained 92.13% of the dissimilarity in assemblage composition between Stage 3 and the previous stages (Table 3). The catch per unit effort (CPUE) of each of these 25 species at each location and time before and after Stage 3 opening was consistent enough to suggest that they explained the assemblage change. Fifteen of these 25 species were estuarine–marine dwelling, and all of these increased in abundance. Ten species were freshwater–estuarine dwelling, and 50% of these declined in abundance. Included in the species that increased in abundance in the upper site were commercially important species, such as School prawn (15 times more), Eastern king prawn, Yellowfin bream (62 times more), Flat-tail mullet (10 times more) and Silver biddy (*Gerres subfasciatus*) (19 times more). The five freshwater–estuarine species that decreased in abundance were Empire gudgeon (*Hypseleotris compressa*), Flathead gudgeon (*Philypnodon grandiceps*), Sea mullet, Pink shrimp (*Acetes sibogae australis*) and the invasive Mosquitofish (*Gambusia holbrooki*).

Table 2 Results of PERMANOVA analysis showing the effect of site and floodgate opening stage on assemblage composition

Source	df	SS	MS	Pseudo-F	p
Site	2	24,859	12,429	14.08	0.001
Stage	3	13,602	4,534.2	5.1364	0.001
Site x stage	6	7,649.6	1,274.9	1.4443	0.015
Residual	47	41,489	882.8		
Summary of pairwise comparisons for factor Stage within site (p<0.05)					
Ironbark Below	Before = Stage 3 ≠ Stage 1 = Stage 2				
Ironbark Lower	Before = Stage 3 ≠ Stage 1 ≠ Stage 2				
Ironbark Upper	Before = Stage 1 = Stage 2 ≠ Stage 3				

df = degrees of freedom; SS = sum of square; MS = mean squares; p = probability

Table 3 Results of SIMPER analysis (Bray-Curtis dissimilarity, 4th-root transformed) showing the contribution that each of the 41 species made to the change in assemblage composition found at the upper Ironbark Creek site after Stage 3 opening of the floodgates. Species are ranked from biggest to smallest % contribution to dissimilarity. The change in mean catch per unit effort (CPUE: per four seines at a location) (untransformed) is shown to give an indication of the 'real' % change in abundance. Those species where the % change could not be defined (nd) were not present prior to Stage 3

		SIMPER[b]			Mean CPUE[c]		
Common name	**Scientific name[a]**	**Av. diss**	**Diss/SD**	**% cont.**	**B, S1 & S2**	**S3**	**% change**
Grass shrimp	*Palaemon debilis*[E–M]	6.22	5.28	10	0.7	289.3	41,229
School prawn	*Metapenaeus macleayi*[E–M d]	5.19	2.23	8.36	18.2	292.6	1,508
Swan River goby	*Pseudogobius olorum*[F–E]	4.44	3.12	7.14	1.9	122.9	6,368
Largemouth goby	*Redigobius macrostoma*[E–M]	4.14	3.2	6.66	1.3	94.6	7,177
Glass goby	*Gobiopterus semivestitus*[E–M]	3.55	1.71	5.71	46.5	185.6	299
Pink shrimp	*Acetes sibogae australis*[F–E]	3.47	1.5	5.58	237.2	147.8	–38
Port Jackson glassfish	*Ambassis jacksoniensis*[E–M]	2.82	1.4	4.54	29.4	48.9	66
Yellowfin bream	*Acanthopagrus australis*[E–M d]	2.58	2.58	4.15	0.2	12.6	6,200
Glassfish	*Ambassis spp*[E–M]	2.58	1.44	4.15	1.5	32.8	2,087
Striped gudgeon	*Gobiomorphus australis*[F–E]	2.35	1.53	3.77	12.0	98.4	720
Mangrove goby	*Mugilogobius paludis*[E–M]	2.03	2.04	3.26	0.3	7.3	2,333
Sea mullet	*Mugil cephalus*[F–E d]	1.66	1.22	2.66	33.8	1.8	–95
Southern blue-eye	*Pseudomugil signifer*[F–E]	1.64	1.29	2.64	4.1	12.8	212
Mosquitofish	*Gambusia holbrooki*[F–E]	1.62	1.3	2.61	39.9	14.9	–63
Flat-tail mullet	*Liza argentea*[E–M d]	1.59	1.08	2.56	0.8	8.8	1,000
Empire gudgeon	*Hypseleotris compressa*[F–E]	1.54	1.36	2.47	30.7	12.4	–60
Silver biddy	*Gerres subfasciatus*[E–M d]	1.51	1.43	2.43	0.1	2.0	1,900
Eastern king prawn	*Melicertus plebejus*[E–M d]	1.38	0.98	2.22	0.0	4.6	nd
Tamar River goby	*Afurcagobius tamarensis*[F–E]	1.31	1.11	2.11	0.9	3.4	278
Flathead gudgeon	*Philypnodon grandiceps*[F–E]	1.24	1.56	1.99	17.9	2.3	–87
Pistol shrimp	*Alpheus spp*[E–M]	1.04	0.98	1.67	0.0	1.0	nd
Grapsid crab	*Parasesarma erythrodactyia*[E–M]	0.88	0.97	1.42	0.0	0.5	nd
Half-bridled goby	*Arenigobius frenatus*[E–M]	0.87	0.94	1.39	0.0	0.5	nd
Striped shrimp	*Macrobrachium intermedium*[F–E]	0.83	0.98	1.34	0.0	0.8	nd
Eastern fortescue	*Centropogon australis*[E–M]	0.81	0.96	1.31	0.0	0.9	nd
Estuary perchlet	*Ambassis marianus*[E–M]	0.64	0.62	1.03	0.3	1.0	233
Exquisite sand goby	*Favonigobius exquisitus*[E–M]	0.48	0.63	0.78	0.2	0.1	–50
Greasyback prawn	*Metapenaeus bennettae*[E–M d]	0.47	0.68	0.76	0.3	0.1	–67
Sand whiting	*Sillago ciliata*[E–M d]	0.45	0.63	0.73	0.1	0.1	0
Australian bass	*Percalates novemaculeata*[F–E d]	0.45	0.57	0.72	0.0	0.6	nd
Tarwhine	*Rhabdosargus sarba*[E–M d]	0.44	0.63	0.71	0.03	0.4	1,233
Dwarf flathead gudgeon	*Philypnodon macrostomus*[F–E]	0.44	0.65	0.71	0.3	0.0	–100
Shrimp	*Palaemonidae spp*[E–M]	0.38	0.56	0.62	0.3	0.0	–100
Bridled goby	*Arenigobius bifrenatus*[E–M]	0.34	0.57	0.55	0.0	0.13	nd
Goldfish	*Carassius auratus*[F–E]	0.2	0.25	0.32	0.7	0.0	–100
Short-finned eel	*Anguilla australis*[F–E d]	0.16	0.37	0.26	0.1	0.0	–100
Fire Tailed gudgeon	*Hypseleotris galii*[F–E]	0.1	0.26	0.15	0.1	0.0	–100
Long-finned eel	*Anguilla reinhardtii*[F–E d]	0.09	0.25	0.14	0.03	0.0	–100
Golden goby	*Glossogobius biocellatus*[E–M]	0.08	0.25	0.13	0.03	0.0	–100
Tailor	*Pomatomus saltatrix*[E–M d]	0.08	0.25	0.13	0.03	0.0	–100
False spider crab	*Amarinus lacustris*[F–E]	0.08	0.26	0.12	0.03	0.0	–100

a *E-M* = estuarine–marine, *F-E* = freshwater–estuarine; **b** 4th-root transformed; **c** untransformed; **d** = commercially important. N.B. Only those species appearing above the dashed line have sufficiently high consistency ratios (Diss/SD) to suggest that they consistently contribute to dissimilarity, i.e. those species below the dashed line are not good discriminators.

Figure 7 nMDS ordination showing a) assemblage differences between all sites at each time of sampling: plots b–d are the same ordination, but only show the trajectory of assemblage change for each site in Ironbark Creek; b) below the floodgate; c) at the lower site above the floodgate; and d) at the upper site above the floodgate. For ease of interpretation, ordination points have been removed for control and reference creeks in plots b-d and replaced with shading to represent the spread of points. The ordination was performed using Bray-Curtis similarity on 4th-root transformed data

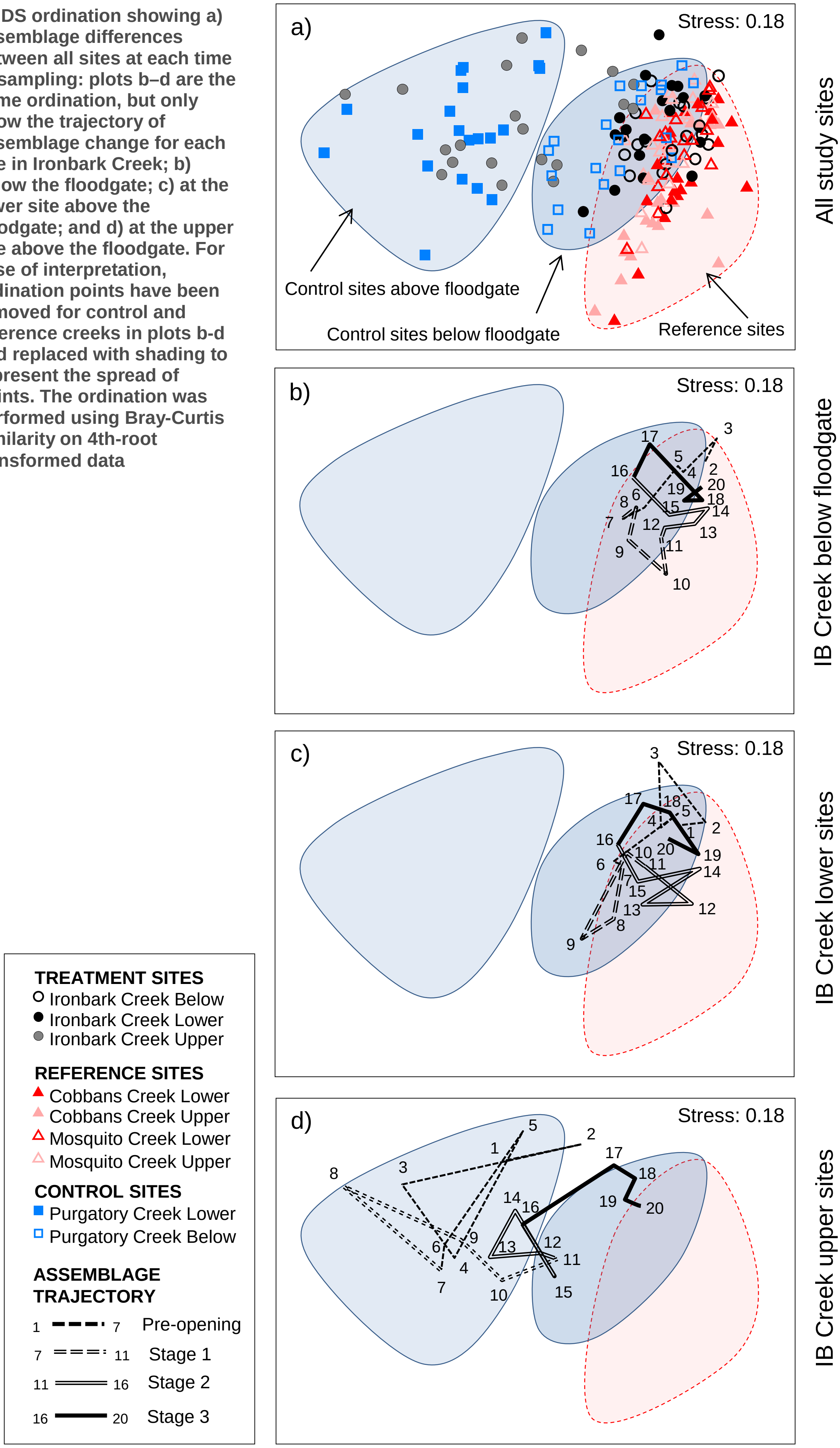
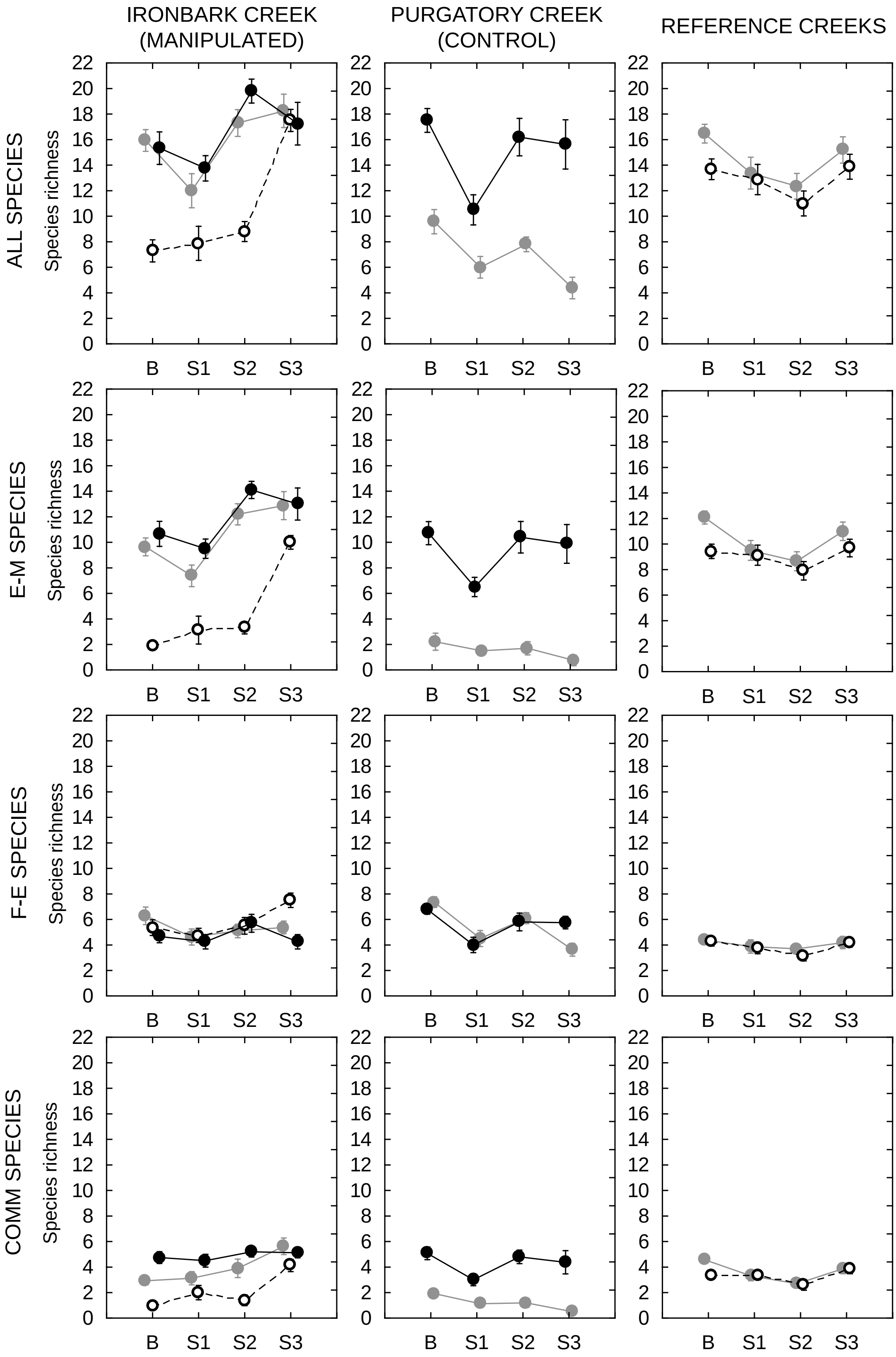


Figure 8 **Mean (±SE) species richness plots for each site at each phase of floodgate opening. From left to right are Ironbark, Purgatory and the combined reference creeks. From top to bottom are all, estuarine–marine (E–M), freshwater–estuarine (F–E) and commercially important (comm) species. Lines represent upper (dashed), lower (grey) and below-floodgate (black) sites. B = before; S1–S3 = Stage 1–3**



Water quality changes associated with floodgate opening

Water quality improved in Ironbark Creek as a result of floodgate opening. The biggest response was seen at the upper Ironbark Creek site, where pH, salinity and dissolved oxygen all increased (Figure 9). While pH and dissolved oxygen improved with every subsequent stage of floodgate opening, salinity did not improve in upper sites until Stage 3. The increase in salinity at this site during Stage 3 was large, changing from between 10 and 15 ppt to ~25 ppt following opening: comparable to un-gated reference creeks. The upper Ironbark Creek site was also the only site found to be acidic, although this was only before floodgate opening.

In contrast to the upper Ironbark Creek site, the water quality at the lower Ironbark Creek site appeared less affected by the floodgate, and therefore remained similar to comparably located sites in the reference creeks throughout all stages. An upstream pH and salinity gradient before floodgate opening was below > lower > upper, but this gradient was significantly reduced by Stage 3. The main impact observed at Purgatory Creek was a slight lowering of salinity upstream of the floodgate during most stages of the study.

In comparison with the other water quality parameters, water temperature appeared minimally affected by floodgate presence or opening. Changes in water temperature over time reflected estuary-scale changes that were consistent at all sites throughout the study.

Assessment of OEMP detrimental thresholds

Species richness fell below the 75% detrimental reduction level on only one occasion (Figure 10). This occurred at the upper Ironbark Creek site on the first sampling occasion after the initial Stage 1 opening. With the exception of one occasion in Stage 1, it was not until Stage 3 that species richness in the upper site was significantly greater than before floodgate opening. Consistent with previous reports, species richness did not significantly increase at the lower site.

The abundance of estuarine–marine species also fell below the 75% detrimental reduction trigger value on one occasion (Figure 11 and Figure 10). As with species richness, this occurred at the upper Ironbark Creek site on the first sampling occasion after the initial Stage 1 opening. Although abundance varied substantially over time, a significant increase in abundance was only sustained for consecutive sampling times at the upper site during Stage 3.

Figure 9 **Mean (±SE) water quality plots at each site at each phase of floodgate opening. From left to right are Ironbark, Purgatory and the combined reference creeks. From top to bottom is pH, salinity, dissolved oxygen and temperature. Lines represent upper (dashed), lower (grey) and below-floodgate (black) sites. B = before; S1–S3 = Stage 1–3**

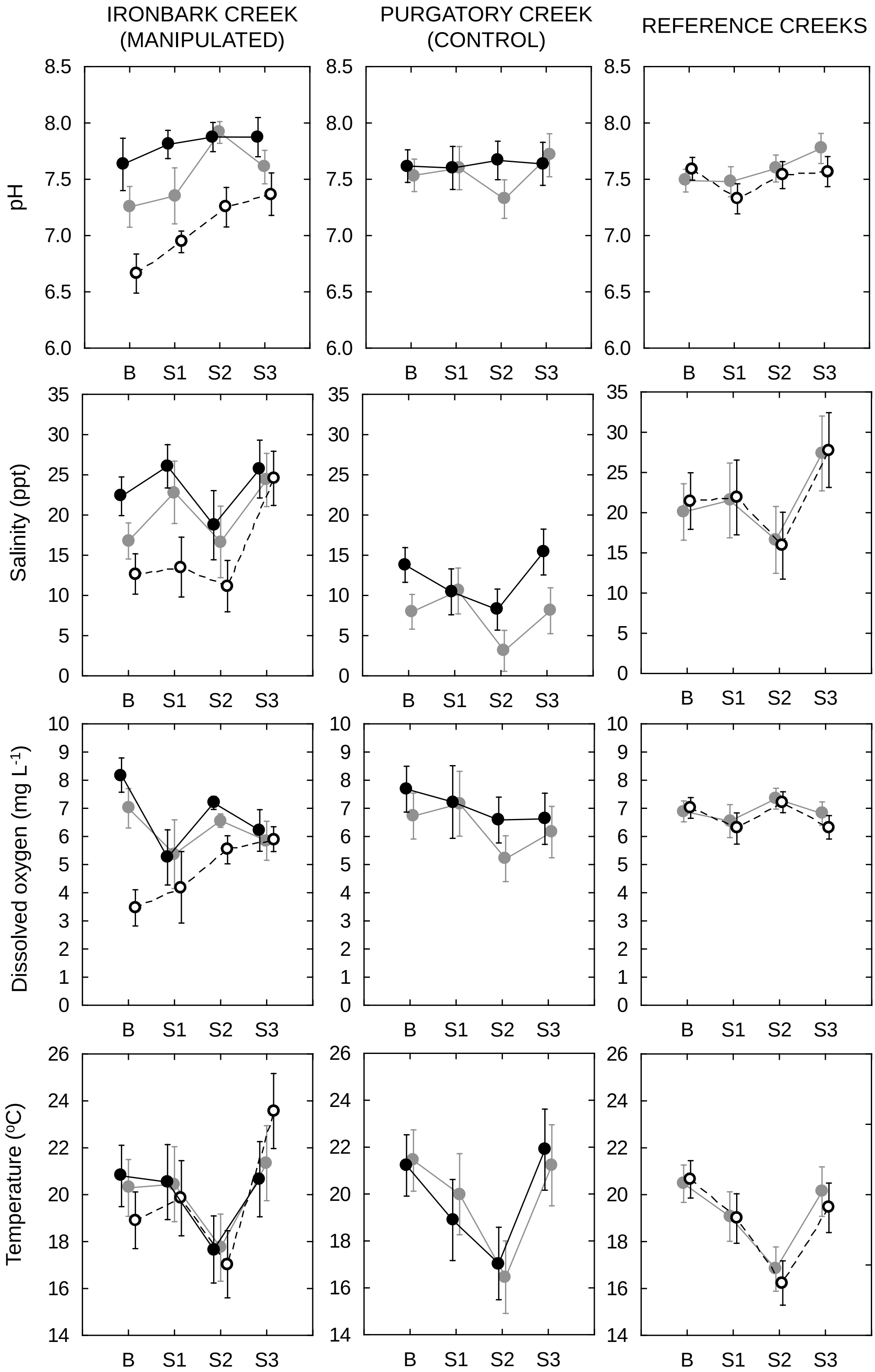


Figure 10 Change in mean species richness (freshwater–estuarine species excluded) in a) upper Ironbark Creek and b) lower Ironbark Creek sites at each sampling date throughout the study. The solid black lines shows the mean of the pre-opening values and the dashed lines the upper and lower 95% confidence intervals of that mean. The 75% reduction threshold as identified in the OEMP is shown

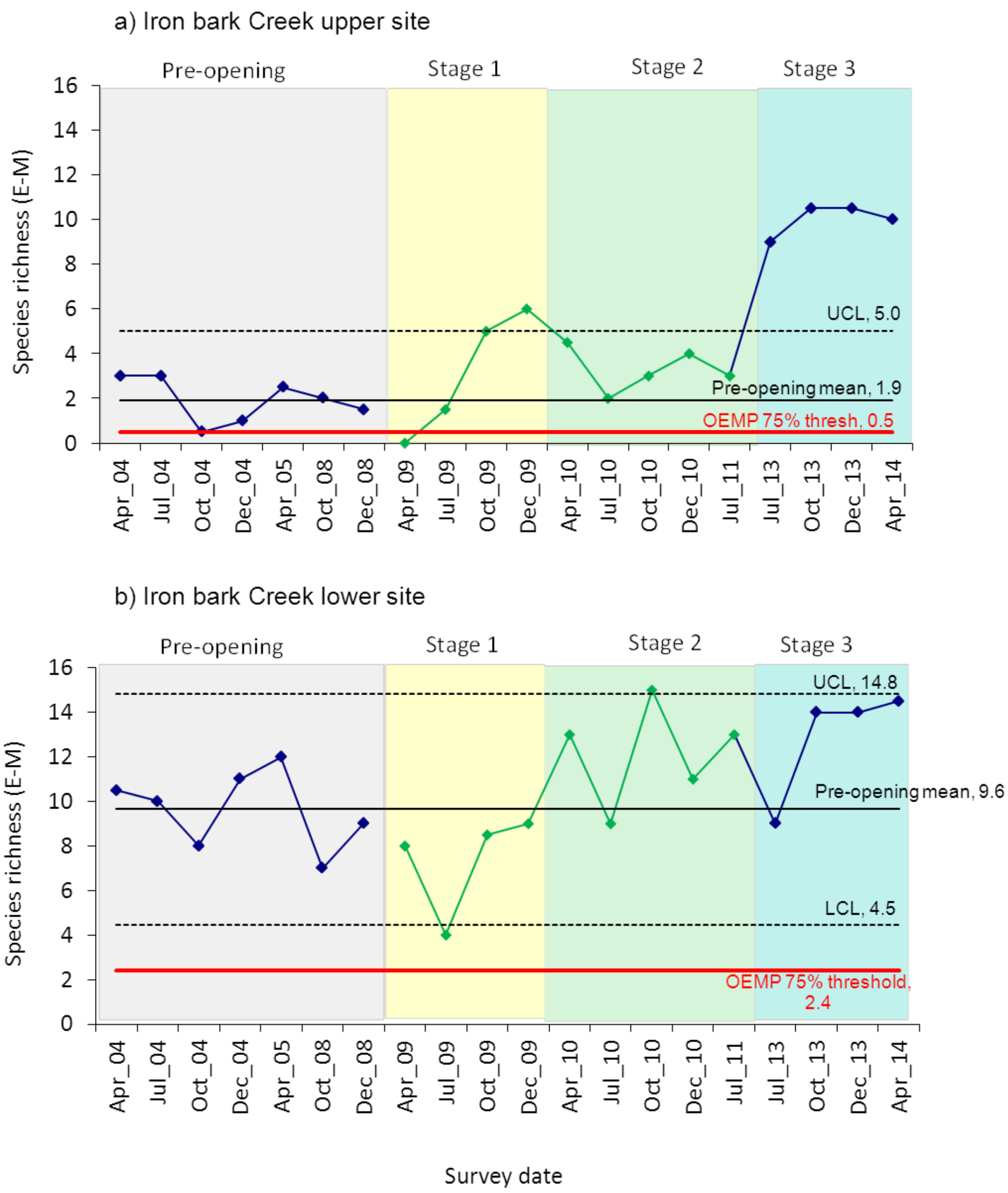
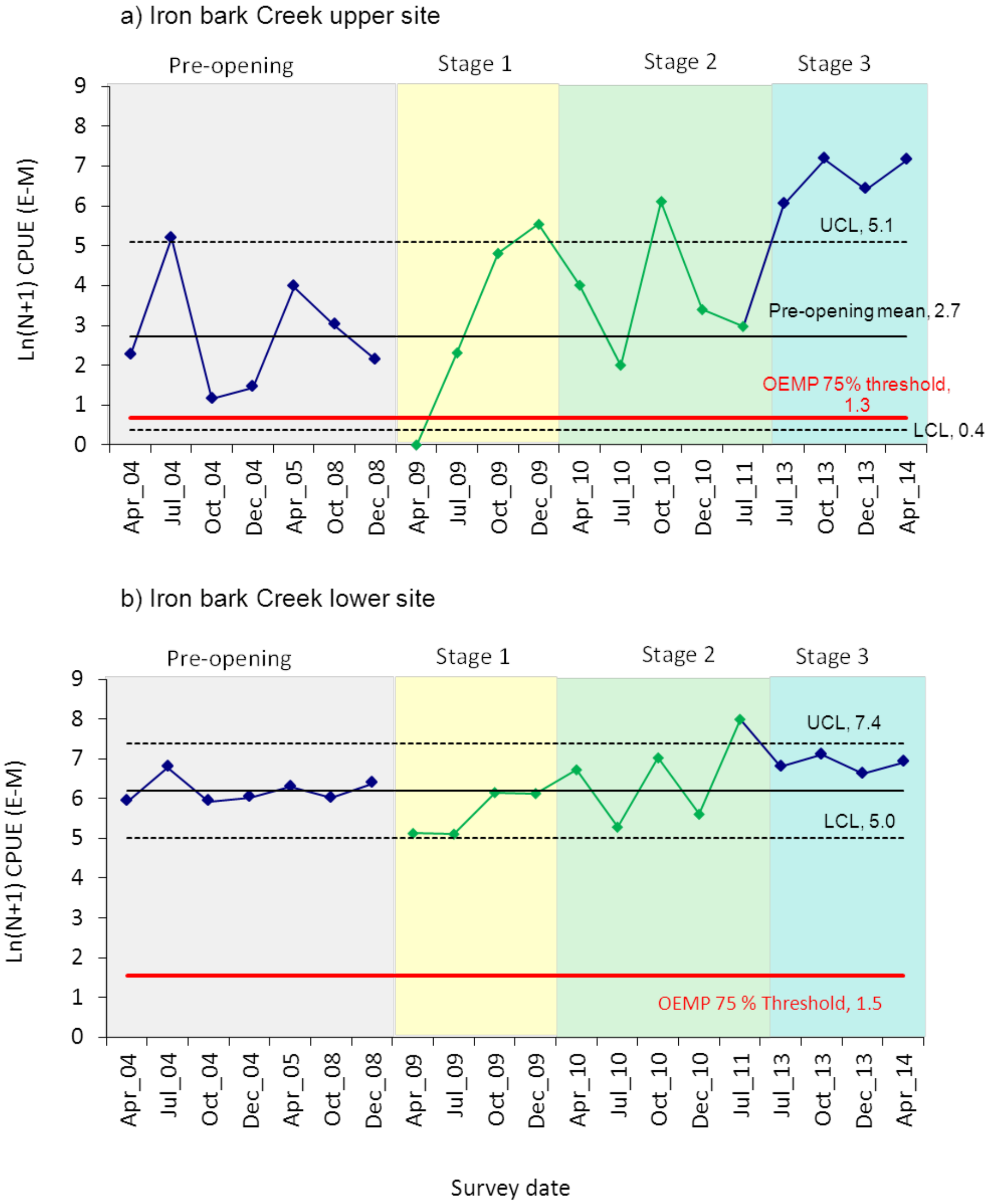


Figure 11 Change in mean catch per unit effort (CPUE) [Ln(N+1) transformed] (freshwater–estuarine species excluded) in a) upper Ironbark Creek and b) lower Ironbark Creek sites at each sampling date throughout the study. The solid black lines shows the mean of the pre-opening values and the dashed lines the upper and lower 95% confidence intervals of that mean. The 75% reduction threshold as identified in the OEMP is shown



Discussion

Impact of Ironbark Creek floodgate on Hexham Swamp

Prior to the Stage 1 opening of the Ironbark Creek floodgates, the assemblage at the lower site immediately above the floodgate was not significantly different from the assemblage below the floodgate or in lower sites of reference creeks. This suggests that the gates had either been opened at some stage to allow fish passage, or display a certain degree of ‘leakiness’, as observed at floodgates in other estuaries (Kroon and Ansell 2006).

Despite the evidence of passage in the immediate vicinity of the Ironbark Creek gates prior to the Stage 1 opening, the gates had clearly been affecting the assemblage over larger spatial scales in Hexham Swamp. The upper Ironbark Creek site was quite distinct in assemblage composition from the lower and below gate sites, primarily driven by a lower abundance of estuarine–marine dwelling species at the upper site. This study adds to the growing body of evidence showing that reduced tidal flushing of coastal wetlands can significantly reduce species diversity (e.g. Raposa and Roman 2003, Eberhardt et al. 2011, Boys et al. 2012a, Boys and Williams 2012a).

A reduction in estuarine–marine species was not associated with greater numbers of freshwater species. Entirely freshwater species were absent in tidal creeks within Hexham Swamp, as has been reported in floodgated creeks in the Macleay and Clarence Rivers (Boys et al. 2012a). Instead, floodgated creeks seem to be favoured by species that can tolerate the large variations in salinity levels that can occur in tidally restricted wetlands.

A reduction in species in aquatic assemblages can affect food webs and ecosystem functioning (Petchey et al. 2004). For example, species such as the Pistol shrimp (*Alpheus* spp.) and Striped shrimp (*Macrobrachium intermedium*) form important food sources for fish and wading birds (Miranda and Collazo 1997), but their distribution was restricted to the lower site prior to floodgate opening. Similar effects on shrimp species have been reported in floodgated creeks elsewhere (e.g. *Acetes sibogae australis*: Kroon and Ansell 2006, Boys et al. 2012a). Many other estuarine–marine species were absent or in much reduced abundance within Hexham Swamp before the floodgates were opened. In a general sense, restricting the movement of fish into wetlands can have ramifications for the larger coastal marine ecosystem, because fish migration can play an important role in exporting the productivity of wetlands (e.g. carbon and nitrogen) to estuaries and the marine environment (Deegan 1993).

Although floodgates pose a physical barrier to fish passage for at least a large proportion of time, many of the species impacts observed in Hexham Swamp may have been caused by reduced water quality and estuarine habitat values upstream of the floodgates. Numerous examples in the literature describe how floodgates can degrade estuarine habitats, particularly the physicochemical environment in which these species live (Roman et al. 1984, Pollard and Hannan 1994, Kroon and Ansell 2006, Boys et al. 2011). By restricting the tidal flushing of Ironbark Creek, the floodgates significantly decreased the area of occupiable habitat within Hexham Swamp, making the upper site less suitable and restricting species distribution to the lower site just above the floodgate.

On average, the pH was acidic in the upper site prior to floodgate opening. Acidification can result when coastal wetlands with acid sulphate soils are floodgated and artificially drained (Sammut et al. 1996): a scenario common throughout the lower Hunter River floodplain. Chronic acidification make habitats less suitable for species to occupy and affect fish growth and reproductive behaviour (Beamish et al. 1975, Sammut et al. 1995). Episodic acidification events with a rapid onset can also result in fish kills and predispose fish to fatal bacterial and fungal diseases, such as red spot disease (Sammut et al. 1995, Sammut et al. 1996, Sanaullah et al. 2001). These effects are amplified when lower pH is combined with other sub-optimal water quality parameters, such as lower dissolved oxygen and salinity (Lilley et al. 1998), which also compromise fish immunity – as was observed in Ironbark Creek.

Assemblage responses to the opening of Ironbark Creek floodgates

The opening of Ironbark Creek floodgates coincided with a significant improvement in both species diversity and abundance. Positive responses mainly involved estuarine–marine dwelling species, which indicates that connectivity with the estuary is an important factor mediating the presence of this guild in tidal wetlands. Benefits were observed across multiple trophic levels, from detritivores such as shrimps and prawn (e.g. Grass shrimp, *Palaemon debilis*; School prawn and Eastern king prawn) to opportunistic generalists that feed on small invertebrates and fish (e.g. Yellowfin bream and a variety of gobiid species).

Increasing the diversity and abundance of species across multiple trophic levels will improve the complexity of food webs. In turn, this can increase ecosystem function and resilience to buffer ecosystems against disturbance (Hooper et al. 2005, Fischer et al. 2006). By maintaining connectivity between tidally active wetlands and the rest of the estuary, biota can move freely throughout the landscape and ensure that resource, genetic and process-based linkages are maintained (Lundberg and Moberg 2003).

Ironbark Creek improved as potential nursery habitat for commercially important species once the floodgates were opened, with more juvenile Yellowfin bream, Flat-tail mullet, Silver biddy, School prawn and Eastern king prawn being sampled. Connectivity between juvenile and adult habitats is critical to the value of estuarine and marine nurseries (Beck et al. 2001). Benefits gained from additional nursery habitat in tidal wetlands are expected to translate into greater productivity of commercial and recreational estuarine and offshore fisheries. Studies investigating long-term fisheries data from the subtropics demonstrate that catches tend to be highest in estuaries of greatest tidal wetland habitat availability and connectivity (Turner 1992, de Graaf and Xuan 1999, Manson et al. 2005a, Manson et al. 2005b, Meynecke et al. 2008, Meynecke 2009).

Floodgate opening was also associated with reduced abundance of the invasive Mosquitofish in Hexham swamp. This supports the growing body of knowledge suggesting that reduced hydrological connection between wetlands and marine waters favours populations of invasive species, particularly *Gambusia* (Kroon and Ansell 2006, Boys and Williams 2012b, MacKenzie and Bruland 2012). Importantly, this study demonstrates that reinstatement of tidal flushing can measurably reduce the number of Mosquitofish. Such improvements are beneficial for wetland ecosystems, because in large numbers, Mosquitofish can alter food webs and affect nutrient cycling through heavy feeding on rotifer, crustacean, and insect populations, which subsequently increases phytoplankton populations (Hurlbert et al. 1972, Hurlbert and Mulla 1981). Mosquitofish also 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). This nationally protected frog species co-inhabits wetland habitats in the Hunter River estuary (Hamer et al. 2002).

The staged manner in which the eight floodgates were opened gave us a rare opportunity to investigate the extent of water quality and assemblage change in Ironbark Creek under differing degrees of tidal flushing. Even with the gates closed, water quality appeared to be minimally affected in the lower site compared with the upper site. Consequently, assemblage and water quality changes were not pronounced in the lower site following floodgate opening. In comparison, the upper site was substantially affected by sub-optimal water quality and reduced species diversity and abundance prior to opening. These metrics significantly improved once the floodgates were opened.

The improvement in dissolved oxygen and pH was incremental, coinciding with each stage of opening, whereas salinity did not improve until the final stage. It was not until Stage 3 that water quality was similar to un-gated reference creeks. Similarly, it was not until Stage 3 that assemblage composition was similar to un-gated reference creeks. These findings support those of others, who have shown that fish and crustacean assemblage responses to wetland restoration will be greatest in wetlands that are much more tidally restricted than other wetlands (Raposa and Roman 2003). It also illustrates that floodgate opening and improved passage on its own may not be sufficient to rehabilitate wetland assemblages, unless a threshold of tidal flushing is exceeded to allow sufficient improvement in water quality and habitat condition. In the case of Ironbark Creek, this meant opening all gates, rather than one or three. In a nearby creek on Ash Island in the lower Hunter (Fish Fry Creek), eight years passed after culvert removal before the creek's hydrogeology and geomorphology had changed sufficiently to support a fish and crustacean assemblage similar in composition to nearby, natural, unrestricted creeks (Boys and Williams 2012a).

School and Eastern king prawn abundance

This monitoring program never intended to target or describe the responses of specific species to floodgate opening. However, there has been particular community interest in the responses of two commercially important penaeid prawn species: Eastern king prawn and School prawn.

Hexham Swamp was historically believed to be a nursery for juvenile prawns. Oral accounts from local commercial prawn fishers indicate that the swamp used to supply the lower Hunter with significant numbers of both Eastern king and School prawn recruits (Simon Walsh 2014, NSW DPI, unpublished information). If this was the case, then Hexham Swamp may have been contributing significantly to the productivity of both the lower Hunter prawn fishery and the broader east coast of Australia. Tagging studies have already confirmed that Eastern king prawns from NSW estuaries (including the lower Hunter River) can migrate large distances north to spawning grounds off the Queensland coast, thus contributing to a stock exploited by estuarine and offshore fisheries (Ruello 1975, Montgomery 1990). It may therefore not be surprising that commercial fishers believe the productivity of the Hunter River prawn fishery reduced dramatically after the Ironbark Creek floodgates were installed.

Although the oral accounts of commercial fishers are compelling, limited data is available on the potential use of tidal creeks by penaeids in NSW estuaries. Eastern king prawn are known to associate with seagrass (Ochwada-Doyle et al. 2009), but this habitat is lacking in the Hunter River estuary. Large numbers of Eastern king prawn have been found at channel edges near mangrove and saltmarsh in several rivers, including the Hunter (Gibbs et al. 1999). This supports studies from subtropical Queensland, which highlight the potential importance of mangrove-lined wetlands as nurseries for this species (Young and Carpenter 1977, Young 1978 Halliday 1995, Skilleter et al. 2005). Unlike the Eastern king prawn, the use of tidal creeks by juvenile School prawn has been clearly demonstrated in recent studies (e.g. Boys et al. 2012a, Boys and Williams 2012a). School prawn is a euryhaline species that is distributed well into the upper Hunter River. They have a greater tolerance for lower salinities than that of the Eastern king prawn, which may make tidal creeks more suitable for the School prawn (Dall 1981).

The sampling techniques used in this study may not have been ideal for targeting penaeid prawns, since night sampling was not performed (Guest et al. 2003). Despite this, we obtained large numbers of prawns in the seine nets at times. Applying standardised sampling methods throughout the 11-year study has allowed us to compare the relative changes in prawn distribution and abundance following floodgate opening in the lower Hunter and Hexham Swamp.

Juvenile Eastern king prawn and School prawn occupy tidal creeks in the lower Hunter estuary in a seasonal manner (Figures A2.1 and A2.2 in Appendix 2). Juvenile Eastern king prawn abundances in tidal creeks peaked in spring (October), and tended to fall off rapidly, so that by summer (December) they were rarely sampled in most years. This supports the limited amount of research from temperate waters of east coast Australia, which suggest the main recruitment of Eastern king prawn post larvae to estuaries occurs between September and October (Racek 1959). After about three months in nursery habitats, they run out to sea to mature and mate (Racek 1959, Young 1975). A ‘short’ nursery estuarine phase is also supported by oral accounts of commercial prawn fishers, some of whom target schools on their out-migration (Simon Walsh 2014, NSW DPI, unpublished information). School prawn appeared to have slightly longer residency in tidal creeks of the Hunter: peaking in abundance in December, but being sampled in smaller numbers throughout the year.

As well as seasonal variation in prawn abundance, there were also clear differences between years (Figures A2.1 and A2.2 in Appendix 2). Of particular note was the strong recruitment of Eastern king prawns to the Hunter estuary in the spring of 2013. The supply of post-larvae to estuaries on the NSW coast is likely to be driven by extraneous factors, such as relative spawning success and the characteristics of the Eastern Australian Current (Montgomery 1990). Whether this or other oceanographic factors drove the 2013 abundance peak is outside the scope of this study. It does, however, highlight the potential benefit arising from the reinstatement of estuarine connectivity with Hexham Swamp.

It is clear that with the Ironbark Creek floodgates closed, less nursery habitat would have been available for both Eastern king prawn and School prawn. Before Ironbark Creek was opened, both species were rarely caught in Hexham Swamp, and their distribution was limited to a few individuals in lower reaches near the floodgate. Following Stage 1 floodgate opening, significantly more School prawns occupied the lower reaches of Ironbark Creek; after Stage 2 opening, School prawns were being sampled from upper Ironbark Creek for the first time since the study began. By Stage 3, 15 times more School prawns were being sampled from the upper site, on average, compared with previous stages (Table 3). A similar response was seen for Eastern king prawn, which were detected in significantly higher numbers in lower sites following Stage 2 opening, and penetrated into the upper site for the first time following opening of all floodgates in Stage 3. This response was undoubtedly mediated by improvements in both passage and water quality. Salinity in the upper site only increased to around 25 ppt after Stage 3, bringing it closer to the optimal range for juvenile Eastern king prawn (28–30 ppt; Dall 1981).

When peak abundances of School and Eastern king prawn were sampled in the Hunter estuary, prawns were prevented from accessing tidal creek habitats upstream of the Purgatory Creek and Ironbark Creek floodgates before Stage 1 opening. As more floodgates were opened at Ironbark Creek and tidal flushing extended further into Hexham Swamp, the extent of suitable prawn habitat also increased. Clearly, floodgates at Hexham Swamp degraded prawn habitat in the Hunter River; but, most importantly, proper floodgate management has significantly increased the extent of available habitat. This is a significant finding, because it is the first time an increase in prawn occupancy in tidal creeks has been detected following floodgate opening or culvert removal in NSW estuaries. The removal of tidal restrictions in several NSW estuaries (e.g. Hunter, Clarence, Macleay) has often led to a rapid and sustained improvements in fish and decapod diversity and abundance. However, an increase in School or Eastern king prawn abundance had not been noted until now (Boys et al. 2012a, Boys and Williams 2012a).

While the opening of Ironbark Creek floodgate has increased the availability of prawn habitat in the lower Hunter River, the degree to which such changes will contribute to improved commercial productivity is still uncertain. A study is currently underway by NSW Department of Primary Industries involving stable isotope analysis and quantitative sampling of tidal creeks for prawns. For the first time, this study should provide an indication of the nursery contribution made by un gated tidal creeks and recently rehabilitated creeks such as Ironbark Creek.

Monitoring against OEMP trigger values

On only one occasion did fish and decapod species richness and abundance decline beyond the 75% trigger value, which would denote a detrimental impact to floodgate opening under the OEMP. This occurred in the upper Ironbark Creek site in July 2009, immediately following Stage 1 floodgate opening. It was a one-off observation, from which species richness and abundance recovered quickly.

The episodic nature of the decline, along with its occurrence close to the time of floodgate opening, suggests that it was unlikely to be caused by tidally induced effects on swamp habitat (e.g. freshwater vegetation die-off). Instead, it was likely to be caused by significant amounts of rain and higher-than-usual freshwater inputs to the swamp, which resulted in fewer estuarine–marine species being collected.

Conclusions and recommendations

The construction and closure of the Ironbark Creek floodgates significantly decreased the amount of suitable habitat for aquatic biota within Hexham Swamp, contributing to a decline in the number and abundance of estuarine–marine fish and crustacean species. The opening of the Ironbark Creek floodgates has altered the species assemblage within Hexham Swamp, although significant changes only occurred after all eight gates were opened. Such a threshold response is not surprising, given that having all gates opened most closely mimics the natural, un-gated condition. This suggests that management of floodgates or other tidal restrictions for wetland rehabilitation needs to recognise that ecological responses may not follow until a certain degree of wetland flushing and water quality or habitat availability is achieved.

The current study corroborates the views of many commercial fishers that the installation of the Ironbark Creek floodgates degraded important nursery habitat for species such as the Eastern king prawn. The positive response of this and several other commercially important species to floodgate opening suggests that improved recruitment of these species is a likely outcome of wetland rehabilitation. The degree to which these positive changes translate into improved productivity of commercial and recreational fisheries will depend on the contribution of those juveniles occupying rehabilitated nursery habitats to the adult exploited population. This is the subject of ongoing research by NSW Department of Primary Industries.

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Appendices

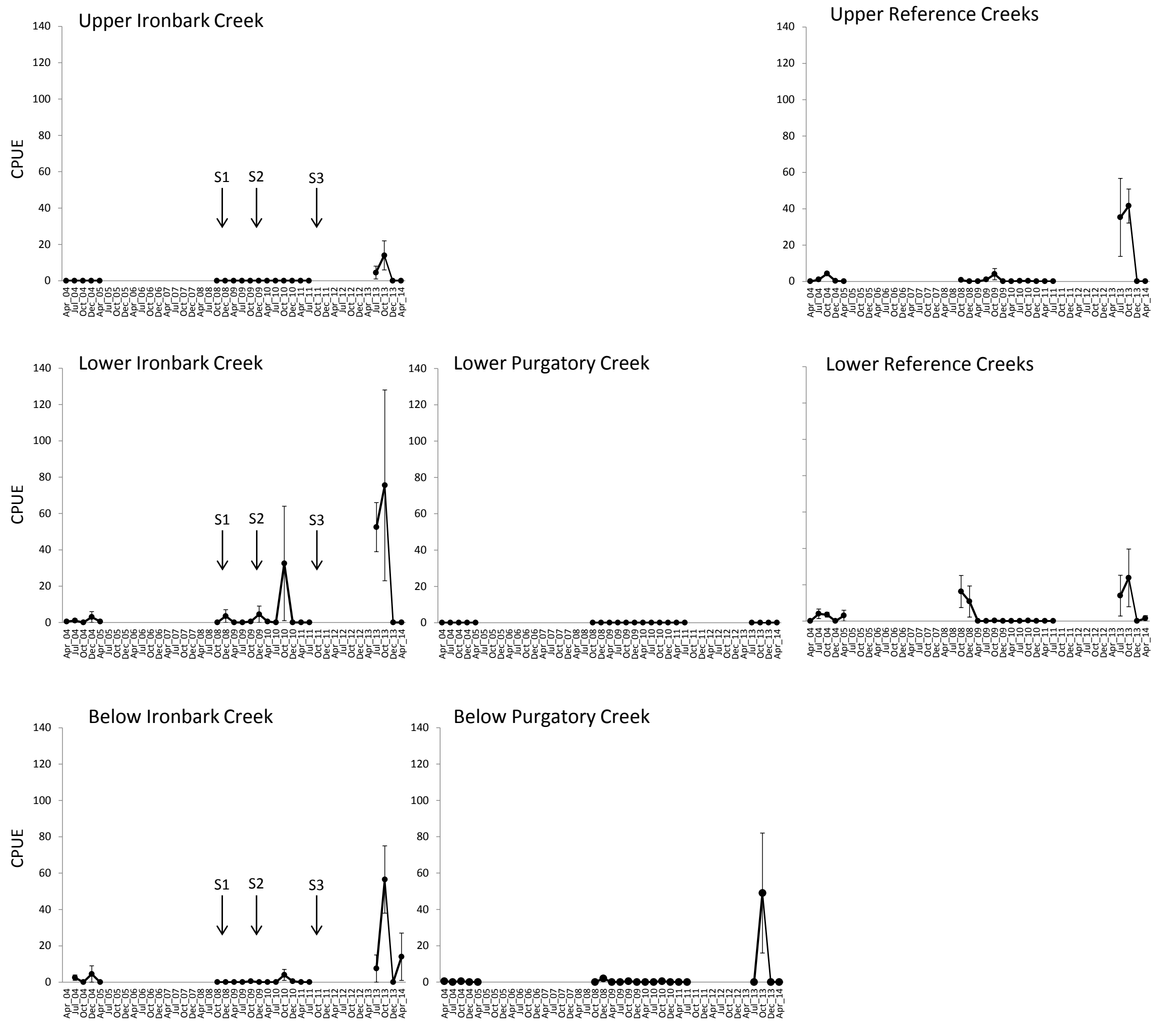
Appendix 1 – Catch summary

Family	Scientific Name †	Common Name	Grand Total	Ironbark Creek Total	Ironbark Creek Below	Ironbark Creek Lower	Ironbark Creek Upper	Cobbans Creek Total	Cobbans Creek Lower	Cobbans Creek Upper	Mosquito Creek Total	Mosquito Creek Lower	Mosquito Creek Upper	Purgatory Creek Total	Purgatory Creek Below	Purgatory Creek Lower
FISH																
Ambassidae	*Ambassis jacksoniensis* [E-M]	Port Jackson Glassfish	**11,243**	**6,499**	2,784	2,384	1,331	**1,896**	1,651	245	**680**	358	322	**2,168**	2,139	29
	Ambassis marianus [E-M]	Estuary perchlet	**219**	**52**	11	24	17	**7**	6	1	**12**	10	2	**148**	147	1
	Ambassis spp [E-M]	Glassfish	**7,793**	**6,230**	3,522	2,399	309	**1,076**	992	84	**366**	272	94	**121**	121	0
Anguillidae	*Anguilla australis* [F-E *]	Short-finned eel	**15**	**3**	0	1	2	**0**	0	0	**0**	0	0	**12**	0	12
	Anguilla reinhardtii [F-E *]	Long-finned eel	**7**	**1**	0	0	1	**0**	0	0	**1**	0	1	**5**	1	4
Clupeidae	*Herklotsichthys castelnaui* [E-M]	Southern Herring	**19**	**0**	0	0	0	**1**	1	0	**9**	9	0	**9**	9	0
	Hyperlophus vittatus [E-M *]	Sandy Sprat	**841**	**56**	15	41	0	**507**	121	386	**1**	1	0	**277**	277	0
	Potamalosa richmondia [F-E]	Freshwater herring	**3**	**0**	0	0	0	**0**	0	0	**0**	0	0	**3**	3	0
Cyprinidae	*Carassius auratus* [F-E]	Goldfish	**92**	**22**	0	0	22	**0**	0	0	**0**	0	0	**70**	9	61
	Cyprinus carpio [F-E]	Common carp	**28**	**0**	0	0	0	**0**	0	0	**0**	0	0	**28**	1	27
Dasyatidae	*Dasyatis spp* [E-M]	Stingray	**2**	**0**	0	0	0	**0**	0	0	**0**	0	0	**2**	2	0
Engraulidae	*Engraulis australis* [E-M *]	Australian anchovy	**3**	**1**	1	0	0	**1**	0	1	**0**	0	0	**1**	1	0
Galaxiidae	*Galaxias maculatus* [F-E]	Common jollytail	**7**	**1**	1	0	0	**0**	0	0	**0**	0	0	**6**	0	6
Gerreidae	*Gerres subfasciatus* [E-M *]	Silver biddy	**447**	**285**	188	78	19	**61**	25	36	**14**	12	2	**87**	52	35
Girellidae	*Girella tricuspidata* [E-M *]	Luderick	**23**	**0**	0	0	0	**1**	1	0	**12**	6	6	**10**	10	0
Gobiidae	*Acanthogobius flavimanus* [F-E]	Yellowfin goby	**128**	**38**	26	12	0	**5**	4	1	**56**	44	12	**29**	29	0
	Afurcagobius tamarensis [F-E]	Tamar River Goby	**6,496**	**3,073**	1,074	1,943	56	**765**	595	170	**990**	691	299	**1,668**	1,667	1
	Arenigobius bifrenatus [E-M]	Bridled Goby	**1,853**	**621**	315	305	1	**100**	92	8	**902**	499	403	**230**	230	0
	Arenigobius frenatus [E-M]	Half-bridled goby	**288**	**102**	62	36	4	**81**	34	47	**100**	51	49	**5**	5	0
	Arenigobius spp [E-M]	*Arenigobius* undefined	**2**	**2**	0	2	0	**0**	0	0	**0**	0	0	**0**	0	0
	Cryptocentroides gobiodes [E-M]	Oyster goby	**86**	**29**	14	15	0	**31**	27	4	**13**	7	6	**13**	13	0
	Favonigobius exquisitus [E-M]	Exquisite sand goby	**3,726**	**2,755**	1,947	800	8	**371**	263	108	**289**	257	32	**311**	311	0
	Glossogobius biocellatus [E-M]	Golden Goby	**4**	**4**	1	2	1	**0**	0	0	**0**	0	0	**0**	0	0
	Gobiomorphus australis [F-E]	Striped gudgeon	**2,274**	**1,469**	47	251	1,171	**2**	2	0	**4**	3	1	**799**	111	688
	Gobiopterus semivestitus [E-M]	Glass Goby	**34,327**	**9,640**	4,860	1,806	2,974	**10,173**	7,283	2,890	**9,484**	5,611	3,873	**5,030**	5,018	12
	Hypseleotris compressa [F-E]	Empire gudgeon	**7,215**	**1,339**	15	243	1,081	**6**	5	1	**2**	1	1	**5,868**	54	5,814
	Hypseleotris galii [F-E]	Firetailed Gudgeon	**4**	**2**	0	0	2	**0**	0	0	**0**	0	0	**2**	1	1
	Mugilogobius paludis [E-M]	Mangrove Goby	**756**	**201**	41	92	68	**69**	46	23	**455**	179	276	**31**	10	21
	Philypnodon grandiceps [F-E]	Flathead gudgeon	**6,378**	**960**	55	315	590	**43**	5	38	**55**	26	29	**5,320**	3,747	1,573
	Philypnodon macrostomus [F-E]	Dwarf flathead gudgeon	**195**	**8**	0	0	8	**0**	0	0	**0**	0	0	**187**	6	181
	Pseudogobius olorum [F-E]	Swan River goby	**17,620**	**6,704**	2,333	3,327	1,044	**2,296**	1,576	720	**5,327**	2,031	3,296	**3,293**	1,271	2,022
	Redigobius macrostoma [E-M]	Largemouth goby	**13,249**	**4,053**	824	2,429	800	**1,515**	555	960	**4,506**	2,250	2,256	**3,175**	2,832	343
	Taenioides purpurascens [E-M]	Eel Goby	**13**	**10**	9	1	0	**0**	0	0	**3**	1	2	**0**	0	0
Monodactylidae	*Monodactylus argenteus* [E-M]	Silver batfish	**9**	**1**	1	0	0	**0**	0	0	**1**	0	1	**7**	4	3
Mugilidae	*Liza argentea* [E-M *]	Flat-tail mullet	**4,408**	**1,770**	1,168	508	94	**1,079**	822	257	**1,060**	569	491	**499**	494	5
	Mugil cephalus [F-E *]	Sea Mullet	**3,832**	**2,575**	1,289	190	1,096	**13**	10	3	**323**	205	118	**921**	826	95
Paralichthyidae	*Pseudorhombus jenysii* [E-M *]	Small-toothed flounder	**21**	**8**	2	6	0	**8**	7	1	**0**	0	0	**5**	5	0
Percichthyidae	*Percalates colonorum* [E-M *]	Estuary perch	**1**	**0**	0	0	0	**0**	0	0	**0**	0	0	**1**	0	1
	Percalates novemaculeata [F-E *]	Australian bass	**21**	**10**	0	5	5	**0**	0	0	**7**	6	1	**4**	4	0
Platycephalidae	*Platycephalus fuscus* [E-M *]	Dusky flathead	**22**	**13**	7	6	0	**3**	2	1	**3**	3	0	**3**	3	0
Poeciliidae	*Gambusia holbrooki* [F-E]	Mosquitofish	**5,034**	**1,741**	31	314	1,396	**8**	0	8	**66**	4	62	**3,219**	189	3,030
Pomatomidae	*Pomatomus saltatrix* [E-M *]	Tailor	**26**	**3**	2	0	1	**6**	2	4	**1**	0	1	**16**	16	0
Pseudomugilidae	*Pseudomugil signifer* [F-E]	Southern blue-eye	**4,575**	**613**	103	277	233	**1,066**	669	397	**2,846**	1,135	1,711	**50**	50	0
Scatophagidae	*Scatophagus argus* [E-M]	Spotted scat	**2**	**0**	0	0	0	**0**	0	0	**1**	0	1	**1**	1	0
Scorpaenidae	*Centropogon australis* [E-M]	Eastern fortescue	**119**	**27**	8	12	7	**15**	12	3	**60**	15	45	**17**	17	0
	Notesthes robusta [F-E]	Bullrout	**1**	**0**	0	0	0	**1**	0	1	**0**	0	0	**0**	0	0
Sillaginidae	*Sillago ciliata* [E-M *]	Sand Whiting	**66**	**17**	4	9	4	**11**	8	3	**22**	17	5	**16**	16	0
Sparidae	*Acanthopagrus australis* [E-M *]	Yellowfin bream	**2,748**	**1,081**	516	459	106	**1,211**	1,123	88	**319**	171	148	**137**	136	1
	Rhabdosargus sarba [E-M *]	Tarwhine	**172**	**80**	9	67	4	**14**	14	0	**53**	17	36	**25**	25	0
Terapontidae	*Pelates sexlineatus* [E-M *]	Eastern striped grunter	**1**	**0**	0	0	0	**0**	0	0	**0**	0	0	**1**	0	1
	Terapon jarbua [F-E]	Grunter	**5**	**3**	2	1	0	**2**	1	1	**0**	0	0	**0**	0	0
Tetraodontidae	*Tetractenos glaber* [E-M]	Smooth toadfish	**11**	**7**	7	0	0	**4**	4	0	**0**	0	0	**0**	0	0
	Tetractenos hamiltoni [E-M]	Common toadfish	**2**	**0**	0	0	0	**2**	2	0	**0**	0	0	**0**	0	0
		TOTAL ABUNDANCE OF FISH	**136,432**	**52,109**	**21,294**	**18,360**	**12,455**	**22,450**	**15,960**	**6,490**	**28,043**	**14,461**	**13,582**	**33,830**	**19,863**	**13,967**
		TOTAL NUMBER OF FISH SPECIES	**53**	**43**	**36**	**34**	**31**	**36**	**33**	**30**	**35**	**31**	**32**	**46**	**42**	**25**
DECAPOD CRUSTACEA																
Alpheidae	*Alpheus spp* [E-M]	Pistol Shrimp	**1,586**	**672**	283	381	8	**264**	189	75	**502**	335	167	**148**	148	0
Grapsidae	*Grapsidae Spp* [E-M]	Marsh crab	**102**	**40**	22	18	0	**8**	5	3	**26**	11	15	**28**	20	8
	Parasesarma erythrodactyia [E-M]	Grapsid crab	**46**	**22**	13	5	4	**6**	5	1	**11**	7	4	**7**	7	0
Hymenossomatidae	*Amarinus lacustris* [F-E]	False Spider Crab	**317**	**30**	15	14	1	**21**	20	1	**31**	21	10	**235**	16	219
Ocypodidae	*Ocypode Spp* [E-M]	Ghost Crab	**14**	**4**	4	0	0	**6**	5	1	**1**	0	1	**3**	3	0
	Ocypode Spp [E-M]	Ghost Crab	**1**	**0**	0	0	0	**0**	0	0	**0**	0	0	**1**	1	0
Palaemonidae	*Macrobrachium intermedium* [F-E]	Striped shrimp	**47**	**9**	1	2	6	**14**	0	14	**12**	9	3	**12**	12	0
	Macrobrachium novaehollandiae [E-M]	Long-armed prawn	**1**	**0**	0	0	0	**0**	0	0	**1**	1	0	**0**	0	0
	Palaemon debilis [E-M]	Grass shrimp	**39,796**	**14,100**	6,184	5,579	2,337	**14,783**	7,817	6,966	**7,685**	4,171	3,514	**3,228**	3,154	74
	Palaemonidae spp [E-M]	Shrimp	**15,103**	**7,018**	3,756	3,253	9	**1,430**	531	899	**6,137**	2,088	4,049	**518**	439	79
Penaeidae	*Melicertus plebejus* [E-M *]	Eastern king prawn	**1,335**	**566**	180	349	37	**382**	191	191	**281**	118	163	**106**	106	0
	Metapenaeus bennettae [E-M *]	Greasyback prawn	**2,038**	**610**	146	453	11	**132**	93	39	**939**	458	481	**357**	317	40
	Metapenaeus macleayi [E-M *]	School prawn	**32,082**	**14,709**	2,650	9,136	2,923	**3,130**	1,459	1,671	**8,250**	3,862	4,388	**5,993**	5,654	339
Sergestidae	*Acetes sibogae australis* [F-E]	Pink Shrimp	**142,613**	**56,165**	39,048	8,346	8,771	**19,660**	17,254	2,406	**25,607**	18,561	7,046	**41,181**	41,047	134
		TOTAL ABUNDANCE OF DECAPODS	**235,081**	**93,945**	**52,302**	**27,536**	**14,107**	**39,836**	**27,569**	**12,267**	**49,483**	**29,642**	**19,841**	**51,817**	**50,924**	**893**
		TOTAL NUMBER OF DECAPOD SPECIES	**14**	**12**	**12**	**11**	**10**	**12**	**11**	**12**	**13**	**12**	**12**	**13**	**13**	**7**

E–M = estuarine–marine dwelling; F–E = occupies fresh and saltwater; F = purely freshwater dwelling. * = commercially important species.

Appendix 2 – Prawn CPUE plots

Figure A2.1 Change in mean (±S.E.) catch per unit effort (CPUE) of juvenile Eastern king prawn (*Melicertus plebejus*) in lower Hunter River study sites



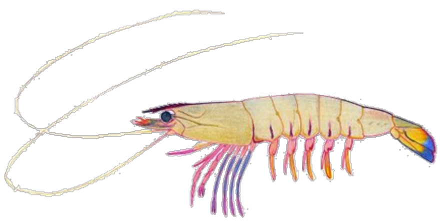
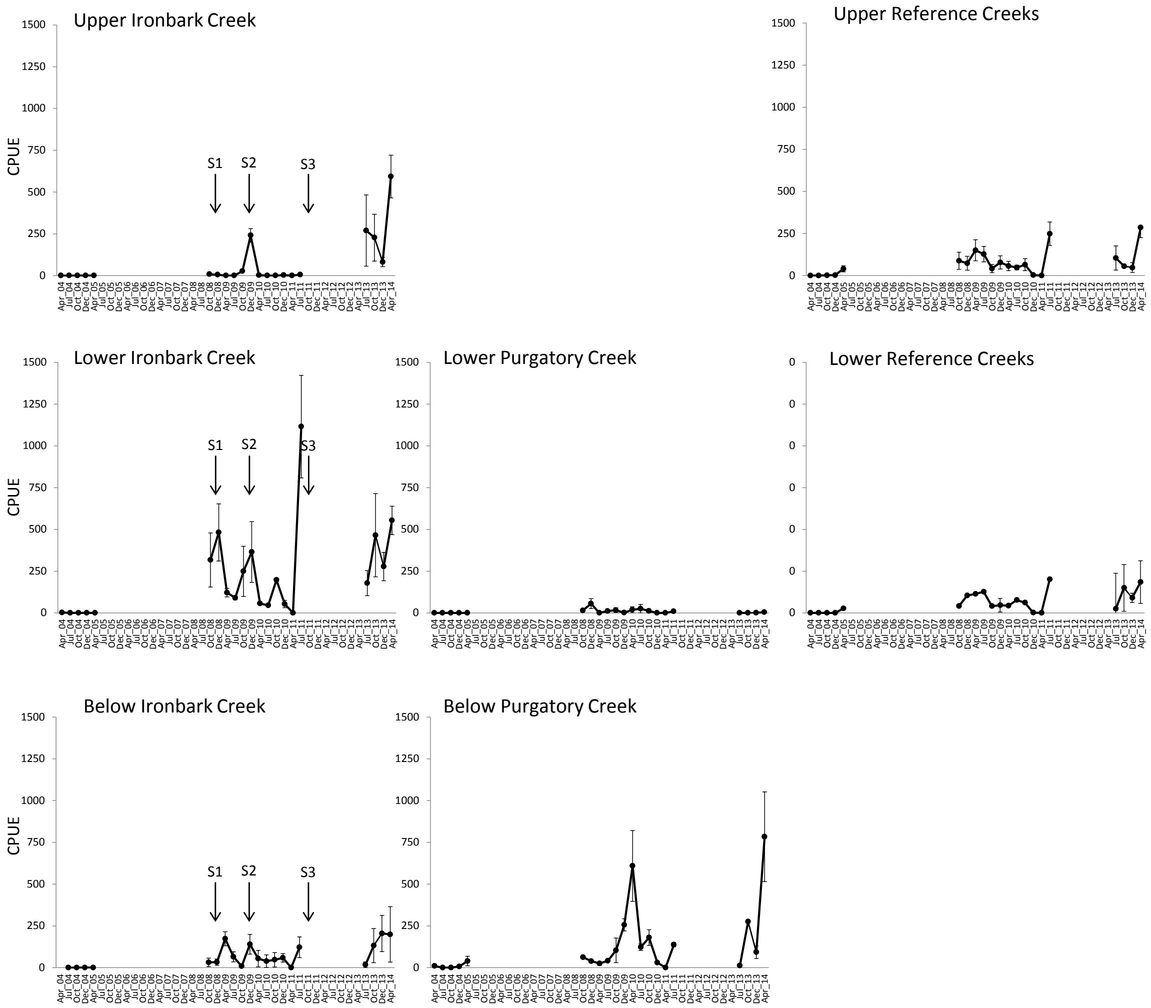
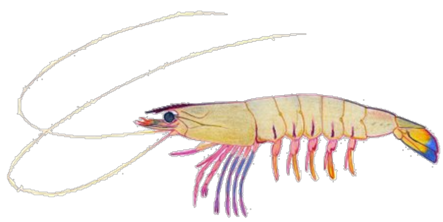


Figure A2.2 Change in mean (±S.E.) catch per unit effort (CPUE) of juvenile School prawn (*Metapenaeus macleayi*) in lower Hunter River study sites





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