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Ecological Impacts of Floodgates on Estuarine
Faunal Assemblages

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Abstract
Floodgates and drainage networks, in addition to natural channels and tributaries, have been developed extensively throughout agricultural lands on the estuarine floodplains of NSW for drainage of agricultural fields and pastures. A characteristic and unfortunate consequence of draining and exposing previously waterlogged floodplains to oxygen, is the oxidation of reduced pyritic soils to produce sulfuric acid. Mass fish kills and poor water quality are dramatic and well documented effects of floodgate and drain management of floodplains in northern NSW, following intensive rain and high flush events that release stored reservoirs of acidic water from upstream of floodgates.

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Ecological Impacts of Floodgates on Estuarine Faunal Assemblages

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CMA CATCHMENT MANAGEMENT AUTHORITY
Ecological Impacts of Floodgates on Estuarine Tributary Fish Assemblages

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Photographs by Tom Heath and Pia Winberg

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Contents

LIST OF FIGURES ........................................................................................................................................ IV

LIST OF TABLES ........................................................................................................................................ VII

EXECUTIVE SUMMARY ............................................................................................................................ 1

Key findings ................................................................................................................................................ 3

Recommendations......................................................................................................................................... 3

Research questions to inform further remediation management: ............................................................ 4

1 INTRODUCTION ...................................................................................................................................... 1-5

1.1 Background ........................................................................................................................................ 1-5

1.1 Development of acid sulfate soils .................................................................................................... 1-6

1.2 The ecological and economic significance of estuarine floodplains .............................................. 1-8

1.3 The development of agriculture and urban development on coastal floodplains .......................... 1-8

1.4 Function and impacts of floodgates .................................................................................................. 1-10

1.4.1 Impacts of water control structures on water quality and vegetation ........................................ 1-10

1.4.2 Impacts of water quality on freshwater and marine ecology .................................................... 1-11

1.5 Research on flow control structures and fish ecology ...................................................................... 1-12

1.6 Modified floodgate designs and smart gates (bi-directional floodgates) ........................................ 1-14

1.7 Objectives of this study .................................................................................................................... 1-15

2 METHODS .............................................................................................................................................. 2-16
2.1 Location ................................................................................................................... 2-16

2.2 Experimental Design ................................................................................................. 2-20

2.3 Sampling techniques .................................................................................................. 2-21

2.4 Water Quality .............................................................................................................. 2-22

2.5 Metal content in fish tissue ......................................................................................... 2-23

2.6 Data Analysis ................................................................................................................ 2-24

3 RESULTS ....................................................................................................................... 3-25

3.1 Water Quality ............................................................................................................... 3-25

3.1.1 Impact of floodgates ............................................................................................... 3-25

3.1.2 Impact of rehabilitated smart gates ......................................................................... 3-31

3.2 Changes in fish & invertebrate assemblages ................................................................. 3-34

3.2.1 Overall assemblage ................................................................................................. 3-34

3.2.2 Differences between assemblages at the different types of gates .............................. 3-36

3.2.3 Fish and invertebrate assemblage differences between upstream and down stream sites for floodgates and smart gates ................................................................................. 3-38

3.2.4 Metal content in fish tissue ..................................................................................... 3-44

4 DISCUSSION .................................................................................................................. 4-47

4.1 Impacts of uni-directional floodgates ......................................................................... 4-47

4.1.1 Water Quality ........................................................................................................... 4-47

4.1.2 Ecology ..................................................................................................................... 4-49
4.2 Aquatic changes associated with bi-directional smart gates ......................................................... 4-52

4.2.1 Water Quality .......................................................................................................................................... 4-52

4.2.2 Ecology .................................................................................................................................................. 4-53

4.2.3 Metal accumulation in fish tissue ............................................................................................................. 4-54

5 CONCLUSION ............................................................................................................................................... 5-55

6 REFERENCES ............................................................................................................................................... 6-56

APPENDIX I: OYSTER SHELL DISSOLUTION AS A SIMPLE MONITORING TOOL FOR ACID SULFATE SOIL IMPACTS ON WATER QUALITY .................................................................................. 5

INTRODUCTION ............................................................................................................................................... 5

METHODS ..................................................................................................................................................... 6

Lab based analysis .......................................................................................................................................... 6

Field based trials ........................................................................................................................................... 7

RESULTS ...................................................................................................................................................... 8

DISCUSSION .................................................................................................................................................. 12
List of Figures

Figure 1-1. 22 floodgate structures across the floodplain and alongside Broughton Creek (excerpt SCC 2002) ...........................................................................................................................................1-6

Figure 1-2. A typical coastal drainage system (from Naylor et al. 1995) .................................................................1-7

Figure 1-3. (a) salt burnt pastures that reduce productivity for cattle farmers, (b) agricultural drains that have been constructed to enable the cultivation of cattle pastures ...........................................................................................................1-9

Figure 1-4. Change in flood mitigation drain water quality (pH) following the installation of Smart gate (modified bi-directional floodgates) in the Shoalhaven [from Indraratna et al (2006a)] .................................1-14

Figure 2-1. Location of the Shoalhaven River in (a) southern NSW, and (b) the main tributary Broughton Creek surrounded by agricultural lands that are classified as(c) high risk for Acid Sulfate Soils. Map source: Shoalhaven City Council .................................................................................................................................2-17

Figure 2-2. tributaries and gated drains along Broughton Creek and associated floodplain, with the six sampled sites indicated. ........................................................................................................................................2-18

Figure 2-3. (a) Typical agricultural tributary with minimal riparian vegetation, (b) milky white tributary water above a floodgate associated with aluminium leaching (c) iron affected water and residue upstream of a floodgate. .........................................................................................................................2-19

Figure 2-4. (a) Emergent mangrove and seagrasses were abundant along the foreshores of tidally inundated sites below floodgates and in upstream ungated tributaries. (b) low saline tolerant grass and reeds dominated gated tributaries. ........................................................................................................................................2-19

Figure 2-5. Experimental design for faunal and water quality sampling ........................................................................2-20

Figure 2-6. Floodgate structure and sampling location both up and downstream of the gates, within 50m of each other ...........................................................................................................................................2-21

Figure 2-7. Seine net sampling technique. ..................................................................................................................2-22
Figure 3-1. Average of water quality variables (a) temperature, (b) pH, (c) dissolved oxygen and (d) salinity upstream (white bars) and down stream (grey bars) of floodgated and non-gated tributaries. Standard Error bars shown..........................................................3-26

Figure 3-2. Mean (a) aluminium, (b) iron and (c) manganese concentrations (mg/L) downstream (grey bars) and upstream (white bars) of unidirectional floodgated and non-gated sites (n=16, SE bars shown) 3-29

Figure 3-3. (a) Aluminium content within the drain solution. (b) White aluminium flocculation covering drain bottom and surrounding vegetation. Photos upstream of uni-directional floodgates..................3-30

Figure 3-4. (a) Iron flocculation disturbance during drain sediment movement. (b) Iron stains on floodgate structure and concrete pitting. Photo taken upstream of uni-directional floodgate. .........................3-30

Figure 3-5. Chlorophyll a concentration (µg/L) in drain water downstream and upstream of uni-directional floodgate (FG) and non-gated (NG) sites. .................................................................3-31

Figure 3-6. a, b, c. Mean flood mitigation drain (FMD) pH (a), dissolved oxygen (b) and salinity (c) downstream and upstream of bi-directional Smart gate, uni-directional floodgate and non-gated sites; before (B) and after (A) Smart gate mechanical rehabilitation (n= 18; bars ± 1 SE). .................................3-33

Figure 3-7. Comparison of species composition of faunal assemblages at each of the drains sampled along Broughton Creek. Data was transformed to presence absence data to compare species composition. Samples pooled across up and down stream and times, n=48. .................................................................3-34

Figure 3-8. Differences in abundance and composition between assemblages (a) downstream and (b) upstream across all gate types; smart gates (SG), floodgates (FG) and non-gated drains (N). n=48. .....3-36

Figure 3-9. MDS plot illustrating the similarity between assemblages upstream (blue triangles) and downstream (green triangles and dashed circle) of smart gates, floodgates and open tributaries. ........3-37

Figure 3-10. MDS plot of upstream faunal assemblage for each gate type before and after the rehabilitation of smart gates. Data points represent centroids for replicate seine hauls. (n =6)...............3-44
Figure 3-11. MDS representing the relative concentrations of 13 metals found in fish tissue sourced from upstream of smart gates (SU) and all other sites (upstream non-gated tributaries (NU) and downstream of both non-gated tributaries (ND) and smart gates (SD)).

Figure 3-12. Metal content in fish tissue collected upstream of smart gates (SU) compared to upstream of non-gated tributaries (NU) and at all downstream sites for smart gates (SD) and non-gated tributaries (ND). (a) nickel, (b) lead, (c) chromium, (d) iron, (e) copper, (f) manganese, (g) mercury, (h) aluminium, (i) arsenic.
List of Tables

Table 3-1. Minimum, mean and maximum water quality variables downstream (DS) and upstream (US) of uni- directionnal floodgates (FG) and no-gates (NG). .............................................................................................................3-27

Table 3-2. Minimum, mean and maximum metal concentration downstream (DS) and upstream (US) of uni-directional floodgates (FG) and no-gates (NG). .............................................................................................................3-28

Table 3-3. Minimum, mean and maximum water quality variables downstream (DS) and upstream (US) of bi-directional smart gates; before and after mechanical rehabilitation........................................................................3-32

Table 3-4. Species and the abundance sampled using seine nets in Broughton Creek tributaries. ..........3-35

Table 3-5. Permutational Analysis of Variance (PERMANOVA) table for the full model of untransformed multivariate data of fish assemblages upstream and downstream of the three gate types (Smart gates, Floodgates and No gates). As a significant interaction between the types of floodgates was found, the GaxLo pairwise comparison identifies which of the gate types differs up stream from down stream. P(MC), the MonteCarlo Probability, is used as n=3. .............................................................................................................3-38

Table 3-6. Simper Analysis identified that the following species contributed to differences upstream and downstream of all tributaries, as well as differences between upstream assemblages of floodgates, smart gates and non-gated tributaries. .............................................................................................................3-39

Table 3-7. Univariate analysis of species, species richness, abundance and diversity of assemblages upstream and down stream of all smart gates, floodgates and non-gated tributaries (n = 24). ..........3-39
Executive Summary

Floodgates and drainage networks, in addition to natural channels and tributaries, have been developed extensively throughout agricultural lands on the estuarine floodplains of NSW for drainage of agricultural fields and pastures. A characteristic and unfortunate consequence of draining and exposing previously waterlogged floodplains to oxygen, is the oxidation of reduced pyritic soils to produce sulfuric acid. Mass fish kills and poor water quality are dramatic and well documented effects of floodgate and drain management of floodplains in northern NSW, following intensive rain and high flush events that release stored reservoirs of acidic water from upstream of floodgates.

Less well understood are the consequences of chronic, long-term, ecological and sub-lethal effects of floodgate operation where dramatic flooding and massive fish kills are less common. In northern river estuaries there is evidence of increases in red spot disease in fish, loss of acid sensitive crustaceans, destruction of fish eggs, reduced spawning success, reduced feed, growth abnormalities and reduced oyster vigour and survival. In addition it is appreciated that floodgates also present barriers for fish passage and in particular, the recruitment of juvenile stocks of both commercial and recreational stocks, however few studies quantify the broader ecological impacts such as changes to whole faunal assemblages, particularly the diversity and relative abundance of fish & invertebrates.

There are 22 floodgates along Broughton Creek alone; a tributary to the Shoalhaven River which has at least 47 floodgates. Broughton Creek is recognised as a hotspot for acid sulfate soil impacts on water quality, and trials of modified floodgate structures (smart gates) were undertaken in 2000 to allow for tidal buffering of acidic floodplain runoff in the agricultural drains and tributaries. These structures fell into disrepair work was undertaken in 2009 by Shoalhaven City Council to rehabilitate functionality. This study is aligned with the rehabilitation of the smart gates to once again demonstrate improved water quality, but more importantly to determine the ecological effects of traditional floodgates and smart gates compared with open drains and tributaries.

In this study, 29 taxa were sampled in high abundance in naturally open drains and tributaries of Broughton Creek; including ecologically and commercially important species. It was demonstrated that floodgates essentially eliminate natural fish and invertebrate life from small tributary habitats and create reservoirs of highly acidic water with high metal content and metal flocculation. Only 4 species (Flathead Gudgeon, Glass Gobies, Blue Eye, crabs) in very low numbers were ever sampled at sites upstream of
floodgates. In addition, the primary production of small phytoplankton is also significantly reduced upstream of floodgates.

The consequences of this are that the 47 floodgates across the Shoalhaven River and Broughton Creek floodplains, collectively have a significant impact on at least 5 important commercial and/or recreational species, most notably prawns but also bream, mullet, tailor and luderick. It is grossly estimated that there may be a reduction of 1400 prawns for every ML of water upstream of floodgates. In addition the ecosystem and food chain that supports commercial and recreational species is severely degraded in direct proximity to floodgates, and potentially smaller but chronic water quality effects may impact the broader river ecosystem and species such as oysters.

In contrast, smart gates, that were not fully functional but partially leaky, provided for some tidal buffering and improved water quality conditions. This was reflected in increased abundance and diversity of fish and invertebrates (12 total species), that approached that of natural drains and tributaries (18 total species). However there was still significantly lower abundance and diversity in species composition compared to natural conditions. Notably the introduced species Eastern Gambusia, or Mosquito Fish, was only found at smart gate locations where low salinity and still water conditions may have provided suitable habitat for recruitment of this species.

Despite improved conditions and ecological assemblages upstream of leaky smart gates, there were still periods of severe drops in pH and release of metals. This study demonstrated that at least 9 metals released into drain water were biologically available and taken up in the food chain. Flathead Gudgeon tissue, sampled upstream of leaky smart gates, was found to have significantly elevated concentrations of iron (>400%), mercury (>200%), lead (1000%), nickel (130%), manganese (280%), copper (280%) and chrome (700%) compared to upstream sites in non-gated tributaries, as well all downstream sites. Although this fish species is small and not a recreational or commercial catch, it represents the bottom of the food chain and had levels of lead that exceeded acceptable Australian food safety standards (FSANZ standard 1.4.1).
**Key findings**

- Floodgates have severe impacts on species diversity and abundance across the full range of macro fish and invertebrates sampled, primarily as a result of highly acidic and high metal content water quality conditions.
- Important commercial species such as prawns are strongly impacted by both flood and smart gate structures, with a potential loss of 1400 prawns for every ML of water upstream of floodgates.
- Leaky gate structures (smart gates) that provide some tidal buffering of upstream water provide significant improvement of water quality and increases in diversity and abundance of fish and invertebrates. However there are still ecological consequences if the tidal buffering and migration access is not adequate including:
  - Some reduced diversity and abundance of fish and invertebrates
  - Absence of important commercial and recreational species (prawns)
  - Provision of artificial conditions that potentially favour introduced species such as Eastern Gambusia
  - Depression of pH with elevated levels of 9 metals that are biologically available and enter the food chain, including unsafe levels of lead.

**Recommendations**

The cumulative impact of approximately 47 floodgates in and along the Shoalhaven River and tributaries remains to be quantified. The capacity of the river to dilute and flush out poor water quality and provide for recruitment of important species is unknown and such information would add further support to the management decisions that have to be made to achieve practical and effective remediation of estuarine and tributary conditions. However this study unequivocally demonstrates significant negative effects on the ecological and food production systems of Broughton Creek, and therefore provides support for the removal, modification or replacement of all floodgate structures in Broughton Creek and the Shoalhaven River to restore effective bi-directional tidal flow conditions. Consideration of experiences elsewhere in NSW with other bi-directional floodgate structures or systems should be considered and additional research questions are provided that will inform the development of remediation strategies for the Shoalhaven River.
Research questions to inform further remediation management:

1) Quantify the cumulative impact of all floodgate structures along the floodplains of the Shoalhaven River and associated tributaries.

2) Identify more appropriate modified floodgate structures and/or method of drain water level maintenance, suited to conditions in the Shoalhaven, that will provide for further improved fish and invertebrate assemblages, including prawn migration, as well as no bioaccumulation of metals in the food chain. The ecological benefits of simpler float controlled floodgates should be assessed as well as fully open drains with weirs and intermittent closure regimes should be considered (see NSW Fisheries Floodgate Design Workshop (2002) and Department of Industry and Investment (2009)).

3) Determine if modified flood mitigation structures facilitate the recruitment of pest species such as Eastern Gambusia.

4) Determine how other compatible technology, such as permeable reactive barriers, may further improve ground water quality and subsequent ecological assemblages where control of acid sulfate soils by removal of floodgates is not possible.

5) Acid sulfate soil effects through flood mitigation infrastructure is a major, but not sole cause, of deteriorating water quality in NSW estuaries. Other sources of pollution and impacts on ecological assemblages across the catchment should be identified and quantified to support strategic planning for and remediation of the important ecological and food production systems of the Shoalhaven River.
1 Introduction

1.1 Background

The development of flood mitigation drains and floodgates along acid sulfate soil influenced coastal floodplains throughout NSW, has resulted in poor water quality conditions through the discharge of sulfuric acid into the drains and natural waterways. Floodgates are usually vertical concrete structures with simple steel flaps sealing large rectangular openings in the concrete. These openings permit outflow of freshwater from the floodplain but prevent ingress of estuarine water. If floodplain soils contain pyrite, then drainage of the soil provides for oxidation of the pyrite and rainwater washes the oxidised form as sulfuric acid into the drains and tributaries (Indraratna et al., 1999). The increased acidification of aquatic estuarine environments impacts the water chemistry, including the release of metals into the water, and is understood to impact aquatic ecology including fish mortality, preventing migration and reducing recruitment of stocks (Sammut et al, 1996).

Recognising that the extent of floodgates is severely affecting specific waterways, hotspots of acid sulfate soil impacts have been identified in NSW for targeted remediation, one of which is Broughton Creek along the Shoalhaven River (SCC 2002). Across the floodplain of the Shoalhaven River, Nowra NSW, there are many floodgates structures with at least 22 in Broughton Creek alone (Figure 1-1).

One initiative for remediation is the modification of floodgate structures to allow for tidal ingress into flood mitigation drains, thus improving water quality upstream by allowing tidal seawater buffering of these waterways. One such floodgate technology was designed and trialled in Broughton Creek by engineers at the University of Wollongong and is referred to as a “smart gate” (Glamore, 2003). Smart gates are essentially a re-engineering of traditional floodgates through computer controlled opening and closing of the steel flaps on floodgates in response to water physical and chemical variables. Water height and chemical probes send signals to a computer that controls a motor driven steel flap.

Although the smart gates were shown to provide for improved water quality through an increase in average pH (Glamore & Indraratna, 2004), the smart gates were not maintained adequately and have since fallen into disrepair. Despite this, the gates remained slightly leaky with some inflow of tidal water. Shoalhaven City Council committed to rehabilitating the smart gate functionality for two gates in 2009, to once again determine that substantial water quality improvements were made and, in addition, to determine whether there were ecological benefits as a consequence. This study was designed to coincide with the smart gates rehabilitation by Shoalhaven City Council and to determine what ecological impacts flood and smart gates have on a whole fish and invertebrate assemblages of Broughton Creek and
tributaries, including determining whether metals released into the drains are actually in biologically available forms and taken up in the food chain. The ecological benefits of flood gate modifications are not often considered, and assemblage and metal accumulation effects are a novel outcome approach used in this study that extend knowledge beyond that of changes to water chemistry.

Figure 1-1. 22 floodgate structures across the floodplain and alongside Broughton Creek (excerpt SCC 2002).

1.1 Development of acid sulfate soils

Acid sulfate soils (ASS) are commonly defined as soils containing iron sulfide (sulfidic) minerals, mainly in the form of pyrite (FeS₂), in either a reduced or oxidised state. In Australia, the acid sulfate soils of most concern are those present within floodplain lowlands, swamps, estuarine embayments and wetlands, developed during the Holocene-age (<10000 years BP), following the last major sea level rise (Dent 1986; van Breeman 1980; Dent, 1986; Naylor et al. 1995; Sammut et al. 1996; White et al. 1997; Cook et al. 2000; Sammut 2000). During this major sea level rise, new coastal landscapes were formed through rapid sedimentation. Bacteria in this organically rich, waterlogged sediment reacted under reduced conditions with sulfate from tidal waters and iron from the sediment, to form iron disulphide (iron pyrite)
(Dent, 1986; Cook et al. 2000; Sammut 2000). Within the low-lying coastal floodplain environment, iron sulfides are present in a layer of waterlogged soil, generally below the natural water table.

Coastal floodplains disturbed through flood mitigation drain development, as described above, result in pyrite oxidation when the water level drops below the pyritic soil layer (Figure 1-2) (Amongst others, Dent, 1986; Naylor et al. 1995; Indraratna et al. 1999, 2002, 2006). Pyrite oxidation occurs when the pyritic soil layer and the associated sulphide minerals are exposed to atmospheric oxygen. The reaction involves the conversion of pyrite (FeS₂) to iron II (FeII) and sulfate (SO₄) in the presence of oxygen and moisture. The by-product of this reaction, hydrogen (H⁺) ions, subsequently produces acidic pore water (Stumm & Morgan 1996; Indraratna et al. 2001; Cook et al. 2000; Glamore 2003). This oxidation generates the production of acid sulfate soils, consequently resulting in the build up of sulfuric acid, iron and aluminum in the floodplain sediment (Nordstrom, 1982; Cook et al. 2000; Glamore 2003; Kroon & Ansell 2006). Discharges of these chemicals into the flood mitigation drains and waterways of the surrounding coastal floodplains, subsequently occurs following rain events.

![Figure 1-2. A typical coastal drainage system (from Naylor et al. 1995)](image-url)
1.2 The ecological and economic significance of estuarine floodplains

The aquatic habitats associated with estuarine floodplains are significant regions for fish migration and recruitment of both commercial and non-commercial species (Pollard 1976; Blaber 2000; Kroon & Ansell 2006). This is because many fish species of ecological and economic importance utilise coastal floodplain habitats during their lifecycle (Gilanders et al. 2003; Kroon & Ansell 2006). Numerous studies have indicated that subtropical mangroves, seagrass and salt-marsh areas within estuaries provide valuable nursery habitats for juvenile fish (Stephenson and Dredge 1976; Blaber & Blaber 1980; Morton et al. 1987; Morton 1990, 1992; Pollard & Hannan 1994), in addition to being important feeding and spawning grounds for adult fish (Blaber 1980; Bell et al. 1984; Morton et al. 1987; Morton 1990, 1992). Less well researched habitats of importance for fish development in the estuarine environment are tributaries (Kroon & Ansell 2006). Favourable light and nutrient regimes with resultant high primary productivity, and physical refuge from predatory species are provided within the estuarine tributaries (e.g. shallow water, vegetation presence, rocks, roots) and offer nursery and migratory habitats (Gray et al. 1996; Newton 1996; Esteves et al. 2000; Vasconcelos et al. 2007; Martinho et al. 2008). The protection of fish nursery refuges and feeding habitats has both ecological and economic importance, as their degradation could have significant deleterious impacts on fish stock recruitment, maintenance and species conservation.

1.3 The development of agriculture and urban development on coastal floodplains

Increased anthropogenic demands on land and resources have far reaching impacts on coastal floodplains worldwide. Systematic alterations in hydrological regimes, agricultural and urban development, the presence of introduced species and declines in water quality reduce the natural ecological integrity of flood plains (Bildstein et al. 1991; Corfield 2000; Kroon & Ansell 2006). The development of drainage systems associated with land reclamation for agricultural purposes, on low-lying coastal land, is a common practice world wide, further reducing natural ecological integrity and exacerbating the loss of habitat (Pons 1973; Maguire 1982; Tuong 1993; Dick 2000; Johnston et al. 2005a,c; Johnston et al. 2009). In Australia, coastal floodplains have an extensive record of agricultural use following European settlement (Willet et al. 1993; Sammut et al. 1995, 1996; White et al. 1997, 2007). This is due to a favourable climate, plentiful soil water and young, fertile soils present within the Australian coastal floodplain environments, increasing the productivity of the system (King 1948; Lin 2004).
Physical re-engineering of coastal floodplains, for agriculture, has resulted in alterations to the hydrological dynamics of coastal floodplains. Under natural conditions, many coastal floodplains remain inundated for much of the year or are subject to regular tidal flooding (Glamore 2003). This prolonged inundation results in an elevated natural ground water level, producing saturated soils with low oxygen content, within the large expanses of wetlands and low-lying sulfidic sediment environment (Lin et al. 2004; White et al. 2007). An increased investment in floodplain agriculture, however, required the development of flood mitigation processes, to reduce frequent inundation events as submersion and salt burns affect the productivity of pastures (Figure 1-3a). This flood mitigation generally involved re-engineering the floodplain and the natural hydrology of the region (Naylor et al. 1995; Thom 2002). An extensive network of constructed drains and modified water courses were consequently developed on the coastal floodplains of eastern Australia (Pollard & Hannan 1994; Johnston et al. 2005). Flood mitigation drains were designed to rapidly remove surface water after flooding events and also significantly lower the natural water table in order to increase agricultural productivity. Natural meandering channels were straightened, cleared and deepened, and adjoining secondary drains were constructed (Naylor et al. 1995; Glamore 2003) (Figure 1-3b).

![Figure 1-3](image)

**Figure 1-3.** (a) salt burnt pastures that reduce productivity for cattle farmers, (b) agricultural drains that have been constructed to enable the cultivation of cattle pastures.

As a result of the agricultural exploitation of productive land, coastal floodplains of Australia have endured the systematic deterioration of water quality, consequently reducing the fitness of native flora and fauna inhabiting the local environment (Corfield 2000). White et al. (1997) estimated that drainage...
modifications could increase discharge by at least two orders of magnitude. Prior to flood mitigation drain development, flood waters following inundation events may have resided within the floodplain system for up to 100 days. Modelling of flood flow drainage patterns, following drain modifications, indicated that all flood waters drained from floodplain environments within 5 days (White et al. 1997). This effectively alters a previously slow release of high nutrient, sulfur rich and oxygen poor water into a pulse event. An increase in drainage discharge and changes to the land use of sulfidic coastal floodplains, has a significant impact on water table dynamics, reducing soil saturation and raising oxygen levels, consequently leading to the development of active acid sulfate soils on the coastal floodplain (Willett et al., 1993; Naylor et al. 1995; Sammut et al., 1995, 1996; Cook et al. 2000; Kroon & Ansell 2006).

1.4 Function and impacts of floodgates

Along the coastal fringe of Australia, the majority of water control structures are one-way, tidal restricting floodgates (Glamore 2003). Floodgates are utilised in flood mitigation channels, near the discharge point to allow drainage outflow and restricting extensive upstream tidal ingress inundating low-lying coastal floodplains during high tide or flood events. These structures however, often operate as complete barriers to restrict saltwater inflow and tidal exchange (Cappo et al. 1998; Kroon and Ansell 2006). An unintended consequence of floodgates, in acid sulfate soil environments, involves flood mitigation drains behaving as reservoirs retaining highly acidic waters which consequently discharge into the waterways either as a pulse during flood events or as a consistent input during ebb tide cycles (Johnston, 2005c; Sammut et al., 1996; Whiter et al. 1997; Glamore 2003; Johnston et al. 2005a).

1.4.1 Impacts of water control structures on water quality and vegetation

The discharge of acid sulfate soil constituents, following rain events, has an immediate and degrading impact on water quality, aquatic ecology, and terrestrial ecology of the surrounding floodplain and waterways (Sammut et al. 1996; White et al. 1997; Wilson et al. 1999; Cook et al. 2000; Johnston et al. 2004). Drainage outflow events with high levels of acidity (Sammut et al. 1996; Wilson et al. 1999; Blunden 2000; Cook et al. 2000) and low dissolved oxygen (Pressey & Middleton 1982; NSW Fisheries 2001; Johnston et al. 2005c) are well documented. These negative impacts are exacerbated through the construction of water control structures, such as dams and floodgates (Sammut et al. 1996; Glamore 2003; Glamore & Indraratna 2004; Indraratna et al. 2006; Kroon & Ansell 2006; Johnston et al. 2009).
The relationship between floodgate structures and water pollutants, such as sulfuric acid, iron and aluminium, within flood mitigation drains and waterways results in the degradation of aquatic habitats within coastal floodplains (Beck et al. 2001; Kroon & Ansell 2006). These impacts include decreased native plant growth (Dent 1986) and the promotion of aquatic weed growth (Cappo et al. 1998; White et al. 1997). The acidity of water within flood mitigation drains also has a direct economic impact on engineering and agricultural infrastructure through corrosion.

The lack of flushing and poor water quality conditions associated with floodgate structures are also thought to contribute to the accumulation of monosulfidic black oozes in drain basal sediments in areas with acid sulfate soils (Sullivan et al. 2002; Johnston et al. 2005c). Monosulfidic black oozes are organic material enriched in iron monosulfides, which when brought into suspension react rapidly to consume dissolved oxygen, resulting in the acute deoxygenation of flood mitigation drains and waterways (Bush et al. 2004; Johnston et al. 2005c).

Decreases in natural inundation along coastal floodplains, have favoured floodplain vegetation species adapted to extended dry periods (Pressey & Middleton 1982; White et al. 1997). Water intolerant vegetation species have replaced the original water tolerant species in the emergent zone (Lin et al. 2004). Large areas once dominated by reeds and rushes have now been replaced by grass species. Some of these species have limited tolerance to inundation and salinity. The resultant high vegetation mortality increases sediment migration into flood mitigation drains during flood events (Johnston et al. 2005). Floodgates are also effective in preventing the upstream establishment of fringing mangrove vegetation in the drains above them. The main ecological effects of these flood mitigation works have been to generally degrade the overall quality of available fish habitat consequently contributing to increases in generalized aquatic habitat disturbance (Pollard & Hannan 1994).

1.4.2 Impacts of water quality on freshwater and marine ecology

Export of sulfuric acid from acid sulfate soil is likely to have a major effect on inshore fisheries and breeding grounds especially in periods of inundation following drought, where large volumes of low pH water can be flushed and or leached into sensitive aquatic habitats (Brown et al. 1983; Easton 1989; Sammut et al. 1996; Corfield 2000). The acidification of waterways will have both lethal and sublethal effects on fish and invertebrate effects on populations. Such reduction in water quality can result in high fish and invertebrate mortality due to asphyxiation resulting from the extremely low dissolved oxygen levels, while surviving fish are more susceptible to epizootic ulceration syndrome (commonly known as red spot disease) which has a direct economic impact on fisheries and population fitness (Callinan et al.
The majority of reports examining floodgate impacts on faunal assemblages within tributaries, focus on significant fish kill events (Pollard & Hannan 1994; Sammut et al. 1996; NSW Fisheries 2001; Bush et al. 2004; Lin et al. 2004; Kroon & Ansell 2006). However, it needs to be understood that all impacts to fish and aquatic assemblages may not be lethal at the scale of major fish kills. Sub-lethal and significant ecological effects are anticipated through physiological stress, toxic effects, high oxygen demand, susceptibility to diseases, and may have cryptic impacts such as slow declines in productivity of recreational and commercial fisheries and aquaculture.

While chronic acid sulfate discharge may create chemical barriers to migration and ecosystem maintenance (Kroon 2005; Kroon & Ansell 2006), a few studies have demonstrated that floodgate structures are also a major barrier to physical migration and consequent recruitment of fish and invertebrate species within the drainage systems (Pollard & Hannan 1994; Halls et al. 1998, 1999; Kroon & Ansell 2006). Australian native fish species require unimpeded access along waterways in order to survive and reproduce. Both fresh and saltwater fish migrate upstream and downstream at different times in their lifecycles to access food and shelter, avoid predators, and to aggregate during spawning events (NSW DPI 2005). The construction of flood mitigation drains and floodgate structures blocks fish passage, reducing migration and dispersal, consequently denying juvenile fish and invertebrates access to feeding and spawning grounds upstream of these structures. Species of fish that can be affected include Australian Bass, Sea Mullet, Yellow Fin Bream and School Prawns (Kroon & Ansell 2006). The anthropogenic alteration of coastal floodplain habitats is likely to affect population genetics and dynamics of fisheries species, leading to local extinctions of some species (Sammut et al. 1995; NSW Fisheries 2001; Bush et al. 2004; Kroon & Ansell 2006). Some barriers can also create excellent habitat for pest species to proliferate, such as Common Carp and Eastern Gambusia (Mosquito Fish). Barriers create still-water pools that are favoured by these species, allowing them to out-compete native fish for food and shelter (NSW DPI 2005).

1.5 Research on flow control structures and fish ecology

Despite the dominance of water chemistry research in and around floodgates and acid sulfate soils, some research has been undertaken in and around flood mitigation drains of coastal floodplains, as well as in lab based experiments, to assess the effects of acid sulfate soil affected water on species (eg. Dent 1986; Sammut et al. 1995, 1996; White et al. 1996, 1997; Glamore 2003; Indraratna et al. 1999, 2002, 2006, 2008)}.
Dove & Sammut, 2007a, 2007b). In particular, the northern rivers region of NSW (Richmond and Clarence Rivers) experiences pulse flooding rains more regularly than in the southern rivers of NSW and there are numerous reports documenting fish kills and consequent fisheries closures associated with acid sulfate soil inflow (Pollard & Hannan 1994; Sammut et al. 1996; NSW Fisheries 2001; Bush et al. 2004; Lin et al. 2004; Kroon & Ansell 2006). Mass mortality of fish can happen rapidly in large flooding events in these areas due to the mobilisation of large quantities of monosulfidic black ooze into the waterways which consumes large quantities of dissolved oxygen, asphyxiating whole fish populations (NSW Fisheries 2001; Bush et al. 2004).

However different climatic factors, different drainage regimes, agricultural uses and soil properties, all influence the scale and types of effects of acid sulfate soils often rendering research findings as site specific, and also effectively invisible such as sub-lethal changes to the ecological assemblage structure. Sub-lethal effects on eco-systems and slow chronic effects are less well understood and researched, though some research has addressed the occurrence of redspot disease (Sammut, 1995) and the effect on oyster health (Dove & Sammut, 2007a, 2007b). Pollard & Hannan (1994) and Kroon & Ansell (2006) appear to be the only research to focus on whole fish ecology, at scales below mass mortality events, within acid sulfate soil affected environments. Both studies examined the potential role of tidal floodgates in depleting estuarine and coastal fisheries stocks along coastal floodplains of Northern NSW. Through the utilisation of quantitative sampling, Kroon & Ansell (2006) identified a significant difference in juvenile fish abundance, biomass and assemblage between drainage systems with and without floodgate structures. Pollard & Hannan (1994) had previously established similar floodgate structure impacts. Flood mitigation drains downstream of floodgates and drains where gates were not present were found to contain larger proportions of both commercial fish species, as well as the majority of species with marine-estuarine affinities. The findings of Pollard & Hannan (1994) and Kroon & Ansell (2006) emphasize the importance of natural water flows within coastal floodplain environments for good management of estuarine ecosystems.

The southern rivers regions of NSW do not experience as large fish mortality events as the northern rivers, in part due to less frequent flood events, and therefore not much is known about the sub-lethal effects of acid sulfate soils on the ecology of southern rivers assemblages. In the southern rivers, most research on acid sulfate soils has focused on water chemistry.
1.6 Modified floodgate designs and smart gates (bi-directional floodgates)

Marine water has the capacity to buffer or neutralise acidic water though bicarbonate (HCO$_3^-$) and carbonate (CO$_3^{2-}$) buffering. The buffering reaction of bicarbonate with a strong acid, such as sulfuric acid from acid sulfate soils, forms weak carbonic acid, reducing water acidity and therefore increasing pH. Because floodgates block tidal marine water intrusion, the carbonate/bicarbonate buffering effect of marine water cannot provide for buffering or acid neutralisation (Indraratna et al. 1999, 2002, 2006a; Glamore 2003; Johnston et al. 2009), and instead reservoirs of acid are created upstream of floodgates. Substantial improvements in water quality have been reported when leakage of one-way floodgates allows tidal incursion within acid-affected flood mitigation drains (e.g. Blunden 2000). Consequently, the restoration of tidal influx via modified bi-directional floodgates is identified as an important step in the remediation of acid sulfate soil effects through improving geochemical, hydrodynamic and acid transport conditions within acid sulfate soil affected drains (Indraratna et al. 2002, 2006a; Glamore 2003; Glamore & Indraratna 2004). Tidal exchange has been identified to decrease the ‘acid reservoir effect’, resulting in increased drain water pH (Figure 1-4), increased dissolved oxygen levels, a decrease in the hydraulic gradient between the drain and groundwater and diminished aluminium flocculation (Pollard & Hannan 1994; Portnoy & Giblin 1997; Dick & Osunkoya 2000; Indraratna et al. 2002, 2006a; Glamore 2003; Glamore & Indraratna 2004).

![Figure 1-4. Change in flood mitigation drain water quality (pH) following the installation of Smart gate (modified bi-directional floodgates) in the Shoalhaven [from Indraratna et al (2006a)].](image)

1-14 | P a g e
Through the development of engineered floodgate modifications, water quality can be restored and, with improved fish passage, breeding and feeding habitats for fish populations of commercial and ecological importance could be improved (Pollard & Hannan 1994; Kroon & Ansell 2006). Research into the effects of modified floodgate structures however, has focused on water and soil quality improvements (e.g. Glamore 2003; Glamore & Indraratna 2004; Indraratna et al. 1999, 2002, 2006; Johnston et al. 2004, 2005a,b,c, 2009; Sammut et al. 1996; Sammut 2000). Such studies have demonstrated, for example, the mechanisms and benefits of carbonate/bicarbonate buffering (Stumm & Morgan 1996; White et al. 1996, 1997; Glamore 2003; Indraratna et al. 1999, 2002, 2006) through carbonate loaded permeable reactive barriers and partially or intermittently open floodgates. As a hotspot for acid sulfate soil impacts, Broughton Creek along the Shoalhaven River has been a central focus for acid sulfate soil research and tidal remediation utilising bi-directional “Smart gate” structures (Pease 1995; Sbeghen 2995; Glamore 2003; Glamore & Indraratna 2004; Indraratna et al. 1999, 2002, 2006a, 2006b). The installation of bi-directional “Smart gate” structures within flood mitigation drains associated with Broughton Creek (tributary of Shoalhaven River) has resulted in the establishment of full tidal flushing within the primary drain and consequently significant water and soil quality improvement (Glamore 2003; Glamore & Indraratna 2004; Indraratna et al. 1999, 2002, 2006a, 2006b).

Despite the demonstrated water quality improvements following trials of smart gates, the extent of benefits to the ecology of the waterways from such modifications is unknown, and little research has been done elsewhere on the sub-lethal effects of drains from acid sulfate soil areas. This is despite the extensive engineering of the floodplain for agriculture, including the development of drainage networks and floodgates. There is consequently a need to undertake further research to assess the relationship between acid sulfate soil discharge, floodgate structure operation, and their impact on fish diversity and migration within the aquatic habitats of coastal floodplains and their tributaries. Unlike the rivers of northern NSW, there is little documented information relating to large fish kills or the sub-lethal effects within the Shoalhaven River relating to acid sulfate soil constituent and floodgate structure impacts.

### 1.7 Objectives of this study

Despite evidence of improved water quality following installation of 5 smart gates in Broughton Creek, the gates were quite technically complicated, not well maintained and fell into disrepair. In addition, it is not well understood how the water quality improvements translate into improved aquatic ecosystems and faunal assemblages. Therefore, rehabilitation of the smart gates and establishment of the ecological benefits was required prior to investment of ongoing maintenance and building of further modified floodgate structures across the floodplains.
Improvement of catchment management for fish and invertebrate communities in the Shoalhaven River is recognised as important in the Shoalhaven Estuarine Management Plan and by NSW Fisheries (Gehrke et al. 2001). In particular, threatened fish species such as Australian Grayling (Protroctes maraena) and Macquarie Perch (Macquaria australasica) are native to the Shoalhaven River, and drainage channels and associated tributaries are important for the recruitment of recreational and commercial species (eg. prawns). In addition, the oyster industry downstream in the Shoalhaven estuary has indicated that improvement of river water quality and ecology will be of benefit to the industry, and supports the investigation of smart gates as part of the solution.

Shoalhaven City Council undertook rehabilitation of the smart gates in 2009 and this study was designed to coincide with that. It provided an opportunity to measure and compare the conditions of faunal assemblages affected by floodgates, improved and modified floodgates (smart gates) and fully open drains. The specific objectives of the study in Broughton Creek were to:

1) Determine the effect of traditional floodgates structures on estuarine faunal assemblages (fish and invertebrates) in comparison to fully open or unblocked drains/tributaries.
2) Determine the potential ecological benefits as a result of smart gates operation with some tidal influx.
3) Determining whether metals released into the drains are actually in biologically available forms and taken up in the food chain.

2 Methods

2.1 Location

Broughton Creek, near the town of Berry (34° 48'S, 150° 41'E), on the south coast of NSW, Australia (Figure 2-1a,b), is a large, tidal tributary of the lower Shoalhaven River. The agricultural region has undergone significant hydrological alterations due to the construction of flood mitigation drainage networks. An active acid sulfate soil layer is situated approximately one metre below the soil surface throughout the study region (Figure 2-1c). The flood mitigation drains (approximately 3.5m deep x 8m wide) discharge these acid sulfate soil constituents into Broughton Creek.
Bi-directional “smart gates” were installed within five of the Broughton Creek flood mitigation drains in 2000 to replace uni-directional floodgates, with the intent of providing for controlled water exchange within the floodplain tributaries or drains. The smart gates however, were not maintained and consequently reverted to uni-directional floodgate function. Considerable technological maintenance of the electronic, computer-controlled floodgates operations is required.

In early June 2009, two of the smart gates were repaired, providing an opportunity to examine the ecological benefits associated with tidal ingress into flood mitigation drains. Smart gate rehabilitation was undertaken by Shoalhaven City Council with the original engineer designers of the smart gates. The field based water probes and communication with the computer controlled floodgate flaps were repaired.
Flood mitigation drains were selected for this study on the basis of drain location, presence of floodgate structures and operational status. Along a 3 km section of the Broughton Creek system, six flood mitigation drains were selected for examination: two drains without floodgates (reference drains), two drains with bi-directional smart gates targeted for mechanical rehabilitation, and two drains with non-operational smart gates (uni-directional/tidal restricting floodgates) (Figure 2-2). Typical acid sulfate soil affected waterways have been documented through pH measurements and visual indicators such as water colour from high aluminium or iron content (Figure 2-3a-c). Vegetation upstream of uni-directional floodgates reflected the water quality and tidal exchange differences. Macrophytic grass and reed species with low saline tolerance dominated drains upstream of floodgates (Figure 2-4) in comparison to mangroves and seagrass beds at all other sites.

Figure 2-2. tributaries and gated drains along Broughton Creek and associated floodplain, with the six sampled sites indicated.
Figure 2-3. (a) Typical agricultural tributary with minimal riparian vegetation, (b) milky white tributary water above a floodgate associated with aluminium leaching (c) iron affected water and residue upstream of a floodgate.

Figure 2-4. (a) Emergent mangrove and seagrasses were abundant along the foreshores of tidally inundated sites below floodgates and in upstream ungated tributaries. (b) Low saline tolerant grass and reeds dominated gated tributaries.
2.2 Experimental Design

The experimental model used for sampling fauna was designed to determine if there was an effect on fish and invertebrate assemblages in tributaries upstream of floodgates and smart gates compared to non-gated tributaries, and also whether there are any effects on assemblages directly downstream of floodgates. Sampling was undertaken four times during the months of March and May (total eight times) in 2009, and again in July and September 2009, after the Smart gates were rehabilitated (Figure 2-5).

![Figure 2-5. Experimental design for faunal and water quality sampling.](image)

Sampling events in each month were grouped into four periods of four, performed within three weeks of each other. Time intervals between periods were >21, days. These intervals were included in order to reduce the influence of temporal variations, including rainfall events and tidal cycles, and to also effects of disturbance from sampling.

Sampling from March and May was used to determine the assemblage similarities between the all locations, gate types and upstream and downstream sites. All data was used in a before/after, control/impact (BACI) experimental design to determine if there were ecological changes within the smart gate drain following rehabilitation in June.
2.3 Sampling techniques

At each of six flood mitigation drains, two locations were selected as sampling sites. One site was located upstream and the other downstream in the four drains with barriers (Figure 2-6). The two flood mitigation drains without floodgates (reference drains), similarly had “upstream” and “downstream” sites, spatially separated by >50m, to mirror those sites in which floodgates were present.

Figure 2-6. Floodgate structure and sampling location both up and downstream of the gates, within 50m of each other.

Flood mitigation drain water depth/height varied with the tide. In addition water discharge from sites upstream of uni-directional floodgates only occurred during low tides. Consequently sampling was undertaken an equal number of occasions under each tidal condition, i.e. eight high tide events (1.32 m average height) and eight low tide events (0.44 m average height).

Fish and invertebrate collection was trialled using fyke and seine nets and bait traps. Seine netting proved the most effective (Figure 2-7) while the other technique results are not reported here. Flood mitigation drain fish assemblages were quantitatively sampled using a fine mesh seine net and dip net at all sites. All data collection was performed in situ, facilitating immediate return of fish into the aquatic habitat and reducing fish sampling mortality. Fish were identified to species level where possible (Briggs, McDowall et al. 1996; Yearsly, Last et al. 1999), and the total number of individuals per species was recorded. In instances where an individual species’ abundance exceeded 200, the total species count was rounded to the nearest 10 individuals, to minimise time out of water. The nets were well rinsed/flicked and inspected between and prior to each sample in order to ensure no animals collected from another seine or site remain stuck to the net.

Fine mesh seine net (6 m headline x 2m drop x 5 mm stretch mesh) sampling was performed using a similar methodology to that utilised in previous studies (West and King 1996; Rotherham and West 2002; Kroon and Ansell 2006; Miles 2007; Shoalhaven City Council 2009). The net was set from the flood mitigation drain bank in a U shape, before being pursed onto the bank. During each seine sample, the
headline was maintained on the surface, and the lead-line on the drain bottom. This retained the structure of the net, while minimising the potential for fish to escape. A total seine volume of 180m3 was sampled per drain site per sampling event. Due to the morphological variation between individual flood mitigation drains and sites, the majority of sampling involved 3 seine nets per site, however due to the configuration of some sites, only one large seine was performed. From each site 3 seine replicates were developed with 60 m3 sample volume per seine. When multiple seine samples were required at a site during a sampling event, it was ensured that seine replicates did not overlap. This reduced the environmental degradation associated with multiple samples, while increasing diversity of aquatic habitats sampled.

Figure 2-7. Seine net sampling technique.

2.4 Water Quality

In order to minimise disturbance of the water column and drain sediment, water quality measurements and samples were collected from drain mid depth prior to biota sampling. Water quality parameters included pH, dissolved oxygen (DO, mg.L⁻¹), salinity (ppt) and temperature using a Yeo-Kal Intelligent Water Quality Analyser (Model 611). Calibration of all sensors, utilising standard certified solutions, took place between each sampling period (4 x sampling event).

Water samples were also collected for chlorophyll a in pre-washed one litre containers. All water samples were immediately stored on ice in a dark container and processed on return to the laboratory. A 500mL aliquot of each water sample was filtered through a glass fibre filter paper (Metrigard, 1.2μm) in a filtration tower. Following filtration, the filter papers were folded and individually placed in aluminium foil to exclude light. The foil wrapped filter papers were then stored at -80°C, until analysis was performed. Individual filter papers were placed in 50mL sterile screw top centrifuge tubes. Pigments were
extracted in 30mL of 90% acetone for 12-24 hrs in 4°C refrigeration. Following extraction, the samples were centrifuged for 15 minutes at 3000 rpm, before decanting a sample of the supernatant into a 5cm spectrophotometric cell. The absorption value of each solution was analysed using a spectrophotometer (Shimadzu UV1700) following the manufacturer’s methodology. The wavelength absorption at wavelengths 630, 647, 664 and 750 nm were subsequently used to calculate the chlorophyll a concentration in each sample according to Parsons et al. (1984):

\[
\text{Chl a} = \frac{11.85 \times (E_{664} - E_{750}) - 1.54 \times (E_{647} - E_{750}) - 0.08 \times (E_{630} - E_{750})}{V_1 \times V_2 \times L} \times 1000
\]

Where: Chl a = Chlorophyll a (μg/L)

\[V_1 = \text{Volume of acetone extract (30mL)}\]

\[V_2 = \text{Volume of seawater filtered (500mL)}\]

\[L = \text{Path length of spectrophotometer cell (5cm)}\]

Additional water samples were collected in sterile 30mL jars for metal analysis. The samples were frozen and transported within polystyrene eskies to Environmental Analysis Laboratory (EAL), a commercial, NATA accredited (ISO/IEC 17025 chemical testing) laboratory affiliated with Southern Cross University. At this external facility, a metal analysis suite was performed with emphasis placed on aluminium (Al), iron (Fe), and manganese (Mn) concentrations. The methodology for analysis was performed according to APHA (2005), in which samples were acidified with nitric acid and filtered through 0.45μm cellulose acetate filter. Metals/salts were subsequently analysed (mg/L) by ICP-MS (Inductively Coupled Plasma - Mass Spectrometry) or ICP-OES (Inductively Coupled Plasma - Optical Emission Spectrometry).

2.5 Metal content in fish tissue

The fish species *Philypnodon grandiceps* was the species most consistently sampled upstream at both smart gate and open tributary locations and was therefore selected as the species of choice for analysis of metal accumulation in fish tissue. There were no fish that could be reliably sampled upstream of floodgates, and these sites were therefore excluded from analysis.

Three fish samples were taken both upstream and downstream of all smart gate and open tributary sites, placed on ice immediately and frozen at the laboratory before wet weight analysis of whole homogenised fish. Metal content was determined at the NATA accredited Environmental Analysis Labs using 1:3 nitric
acid digestion (APHA 3120 ICPMS and ICPOES). A suite of 13 metals (Ag, Ar, Pb, Cd, Cu, Mn, Ni, Se, Zn, Hg, Fe, Al) were tested for.

2.6 Data Analysis

Multivariate fish assemblage data was analysed in Primer 6+ with the PERMANOVA module for nested hierarchical statistical tests (Plymouth Marine Labs, 2008). Data was visualized in non-metric Multidimensional Scaling (MDS) plots with the use of a dummy variable (1) to replace the high number of zeros in the data set. The 3-way and nested multivariate data set was analysed using both untransformed and transformed data ($4^{th}$ root and presence-absence) to determine whether compositional or abundance effects were more apparent. Where there was no difference between the two transformations then only untransformed data is presented here. As sampling replicates were $n = 3$, Monte Carlo Probability values were used in the analysis. Pair-wise a-posteriori comparison was done with the Permanova module using a multivariate analogue of the t-test. To determine which fish species contributed to any assemblage differences between sites, a SIMPER, Similarity Percentages, analysis was done in Primer.

Univariate analysis was done for water quality variables, faunal abundance and species diversity indices (Species Richness ($s$) and Shannon-Wiener ($H'$)) and selected species from SIMPER analysis. Hierarchical and 3-way Analyses of Variance (ANOVAs) were done using the GMAV 5 software (Underwood, University of Sydney, Australia), and assumptions of ANOVA were examined prior to analysis. Normality was tested visually and homogeneity of variance tested using Cochran’s (C), with transformations to achieve homogeneity of variance where required. Where groups differed significantly ($p < 0.05$), a-posterior comparisons among means were done with Student-Newman-Keuls (SNK) tests.
3 Results

The results of this study considered the local rainfall pattern between March and August 2009, in which rainfall was recorded on 69 days with a total of 361.7 mm. The rainfall observed during this study was half the long term mean for all years between these months (736.8 mm) (Commonwealth of Australia. Bureau of Meteorology 2009). Whilst lower than the average, the rain pattern provided constant, minor discharge into flood mitigation drains with no distinct flood events associated with extreme, pulse discharges.

3.1 Water Quality

3.1.1 Impact of floodgates

Temperature

Water temperature varied across sites during the study by 12.6 to 14.5°C (Table 3-1). The lowest mean temperature was upstream of uni-directional floodgates, however all upstream sites were significantly cooler than their respective downstream sites (ArcSin(%) transformed C = 0.1467. F = 45.72; P = 0.0002) (Figure 3-1).

pH

The drain water upstream of uni-directional floodgate structures was acidic with a mean pH of 5.5 and a minimum value of 3.91 (Table 3-1). This pH range was significantly lower (Figure 3-1b) than the downstream floodgate site and non-gated sites ($\times^{10}$Constant transformed C = 0.1709. F =4.03; P = 0.0002). Drain sites downstream of uni-directional floodgates and along non-gated systems had relatively constant pH values with a mean around 7.17 pH.

Dissolved Oxygen (DO)

Mean dissolved oxygen levels were not significantly different (C = 0.1440. F = 2.56; P = 0.0596) between drain sites (Figure 3-1c). All mean DO levels were relatively similar with a value greater than 6.69 mg/L (Table 3-1).
Mean salinity (ppt) was significantly lower (7.2 ppt) ($C = 0.0894 \ F = 4.77; \ P = 0.0038$) upstream of uni-directional floodgates in comparison to all other sites (Figure 3-1d). Mean salinities at downstream and upstream non-gated drains had similar values which fluctuated with the tide (Table 3-1).

Figure 3-1. Average of water quality variables (a) temperature, (b) pH, (c) dissolved oxygen and (d) salinity upstream (white bars) and downstream (grey bars) of floodgated and non-gated tributaries. Standard Error bars shown.
Table 3-1. Minimum, mean and maximum water quality variables downstream (DS) and upstream (US) of uni-directional floodgates (FG) and no-gates (NG).

<table>
<thead>
<tr>
<th>Water Variable</th>
<th>Floodgate Drains</th>
<th>Non-gated Drains</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Downstream</td>
<td>Upstream</td>
</tr>
<tr>
<td></td>
<td>Min</td>
<td>Mean</td>
</tr>
<tr>
<td>pH</td>
<td>6.40</td>
<td>7.10</td>
</tr>
<tr>
<td>DO (mg/L)</td>
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<td>7.46</td>
</tr>
<tr>
<td>Salinity (ppt)</td>
<td>2.12</td>
<td>14.40</td>
</tr>
</tbody>
</table>

Metals

Aluminium (Al)

Mean concentrations of aluminium (Figure 3-2a) were significantly higher (Ln(X) transformed C = 0.2805, F = 5.03; P = 0.0041) upstream of uni-directional floodgates compared to all other sites. Sites downstream of drains with floodgates also appeared to be impacted by the elevated upstream Al concentrations with significantly higher concentration than at non-gated sites (SNK P < 0.01). The mean aluminium concentration upstream of uni-directional floodgates was eleven times greater than any other sample site, with a mean value of 5.54 mg/L and a maximum concentration of 17.17mg/L (Table 3-2). The mean aluminium concentration downstream of the floodgate structure was 0.53 mg/L, while the non-gated drain environment presented lower concentrations of 0.15 mg/L at downstream, and 0.21 mg/L at upstream sites.

Other visual indicators consistent with high aluminium concentrations included cloudy, white drain water both upstream and to some extent downstream of uni-directional floodgates (Figure 3-3) and precipitated aluminium flocs upstream, covering areas of the drain bottom and surrounding vegetation.
Iron (Fe)

Mean iron concentration (Figure 3-2b) upstream of the uni-directional floodgates was three times greater than all other sites with a mean concentration of 2.30 mg/L and a maximum concentration of 9.18 mg/L (Table 3-2). Water downstream of all drains had similar mean iron concentrations of 0.37 mg/L (floodgate) and 0.32 mg/L (no-gate), while the upstream non-gated site had a mean concentration of 0.61 mg/L.

Elevated levels of iron were visually apparent upstream of uni-directional floodgates with iron flocs easily disturbed from bottom sediment (Figure 3-4). Such iron precipitates were observed at all upstream sampled sites, however a greater concentration was observed upstream of the uni-directional floodgates during sampling. Floodgate structures were also iron stained and cement pitted, both upstream and downstream.

Manganese (Mn)

The mean concentration of manganese (Figure 3-2c) was significantly higher upstream and downstream of uni-directional floodgates in comparison to open drains (Ln(X) transformed C = 0.1505. F = 4.29; P = 0.0093). Manganese concentrations were greatest upstream of uni-directional floodgates with a mean concentration of 0.48 mg/L while downstream of floodgates had a mean concentration of 0.24 mg/L (Table 3-2). Sites upstream and downstream within non-gated drains both had low concentrations of 0.05 mg/L (downstream) and 0.07 mg/L (upstream).

Table 3-2. Minimum, mean and maximum metal concentration downstream (DS) and upstream (US) of uni-directional floodgates (FG) and no-gates (NG).

<table>
<thead>
<tr>
<th>Metals in tissue</th>
<th>Floodgate Drains</th>
<th>Non-gated Drains</th>
</tr>
</thead>
<tbody>
<tr>
<td>(mg/L)</td>
<td>Downstream</td>
<td>Upstream</td>
</tr>
<tr>
<td></td>
<td>Min</td>
<td>Mean</td>
</tr>
<tr>
<td>Aluminium</td>
<td>0.09</td>
<td>0.53</td>
</tr>
<tr>
<td>Iron</td>
<td>0.04</td>
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</tr>
<tr>
<td>Manganese</td>
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<td>0.24</td>
</tr>
</tbody>
</table>
Figure 3-2. Mean (a) aluminium, (b) iron and (c) manganese concentrations (mg/L) downstream (grey bars) and upstream (white bars) of unidirectional floodgated and non-gated sites (n=16, SE bars shown)
Figure 3-3. (a) Aluminium content within the drain solution. (b) White aluminium flocculation covering drain bottom and surrounding vegetation. Photos upstream of uni-directional floodgates.

Figure 3-4. (a) Iron flocculation disturbance during drain sediment movement. (b) Iron stains on floodgate structure and concrete pitting. Photo taken upstream of uni-directional floodgate.
**Chlorophyll a**

The mean concentration of chlorophyll a (Figure 3-5) was not significantly different between drain sites (ln(X) transformed $C = 0.1828$, $F = 1.56; P = 0.2047$). Sites downstream had similar mean chlorophyll a concentrations of 4.88 µg/L (floodgate) and 4.60 µg/L (no-gate). The lowest mean concentration of chlorophyll a, 1.18 µg/L, was observed upstream of uni-directional floodgates. Although possessing the greatest mean concentration of 9.08 µg/L, chlorophyll a upstream non-gated drain sites was highly variable between sampling events.

![Figure 3-5. Chlorophyll a concentration (µg/L) in drain water downstream and upstream of uni-directional floodgate (FG) and non-gated (NG) sites.](image)

**3.1.2 Impact of rehabilitated smart gates**

**pH**

Mean pH before and after the rehabilitation of Smart gate operations remained relatively constant upstream and downstream of flood mitigation drains with uni-directional floodgates and no-gates (Figure 3-6a). A significant decline in water quality was however present upstream of bi-directional smart gates following rehabilitation ($x^10$ transformed $C = 0.2445$, $F = 10.40; P = 0.0001$). Prior to renewed tidal exchange, mean pH remained relatively constant downstream (pH7.11) and upstream (pH6.85) of the Smart gate structure. A similar downstream mean pH was observed following Smart gate rehabilitation.
with a pH of 6.92. A decline in pH of 1.47 upstream of the bi-directional Smart gate however, increased water acidity to pH 5.45 following mechanical rehabilitation.

\textit{Dissolved Oxygen (DO)}

Mean dissolved oxygen concentrations (Figure 3-6b) illustrated no significant difference between sites before or after Smart gate rehabilitation ($C = 0.2946, F = 10.08; P = 0.1942$). Although mean DO increased both upstream and downstream of the bi-directional barrier following gate rehabilitation, mean DO remained lower upstream of smart gates in comparison to downstream sites (Table 3-3).

\textit{Salinity}

Mean salinity (ppt) was not significantly different ($10\times$Constant transformed $C = 0.3490, F = 0.43; P = 0.6308$) between sites before or after Smart gate rehabilitation (Figure 3-6c), despite the downstream mean salinity increase to 18.50 ppt after rehabilitation. Mean salinity remained constant upstream of smart gates around 14.60 ppt (Table 3-3).

Table 3-3. Minimum, mean and maximum water quality variables downstream (DS) and upstream (US) of bi-directional smart gates; before and after mechanical rehabilitation.

<table>
<thead>
<tr>
<th>Water Variable</th>
<th>Downstream</th>
<th>Upstream</th>
</tr>
</thead>
<tbody>
<tr>
<td>Before pH</td>
<td>Min: 6.30</td>
<td>Mean: 7.11</td>
</tr>
<tr>
<td></td>
<td>Min: 3.33</td>
<td>Mean: 6.85</td>
</tr>
<tr>
<td>After pH</td>
<td>Min: 6.10</td>
<td>Mean: 6.92</td>
</tr>
<tr>
<td></td>
<td>Min: 3.21</td>
<td>Mean: 5.45</td>
</tr>
<tr>
<td>Before DO (mg/L)</td>
<td>Min: 3.30</td>
<td>Mean: 6.77</td>
</tr>
<tr>
<td></td>
<td>Min: 2.70</td>
<td>Mean: 5.24</td>
</tr>
<tr>
<td>After DO (mg/L)</td>
<td>Min: 7.00</td>
<td>Mean: 7.82</td>
</tr>
<tr>
<td></td>
<td>Min: 5.00</td>
<td>Mean: 7.26</td>
</tr>
<tr>
<td>Before Salinity (ppt)</td>
<td>Min: 2.39</td>
<td>Mean: 14.94</td>
</tr>
<tr>
<td></td>
<td>Min: 4.29</td>
<td>Mean: 14.68</td>
</tr>
<tr>
<td>After Salinity</td>
<td>Min: 14.42</td>
<td>Mean: 18.50</td>
</tr>
<tr>
<td></td>
<td>Min: 10.39</td>
<td>Mean: 14.65</td>
</tr>
</tbody>
</table>
Figure 3.6. a, b, c. Mean flood mitigation drain (FMD) pH (a), dissolved oxygen (b) and salinity (c) downstream and upstream of bi-directional Smart gate, uni-directional floodgate and non-gated sites; before (B) and after (A) Smart gate mechanical rehabilitation (n= 18; bars ± 1 SE).
3.2 Changes in fish & invertebrate assemblages

3.2.1 Overall assemblage

Over 42,000 fish and invertebrates, including 29 taxa were sampled using the techniques of seine, bait and dip nets both upstream and downstream of the three types of gates. The number of specimens retrieved using bait and dip nets was too small to allow for a robust statistical analysis, and therefore only results from seine net sampling is presented here. Fish represented 25 of the species, prawns were the most abundant and, together with crabs, crustaceans represented over 80% of the overall abundance. Other invertebrates, polychaetes, molluscs and jellyfish (cnidarians), were a minor component of samples (Table 3-4).

There was minimal difference between the species composition of assemblages at any of the tributaries of Broughton Creek, indicating that the selected tributaries were representative of a collective assemblage of species common to Broughton Creek (Figure 3-7).

![Graph showing species composition](image)

Figure 3-7. Comparison of species composition of faunal assemblages at each of the drains sampled along Broughton Creek. Data was transformed to presence absence data to compare species composition. Samples pooled across up and down stream and times, n=48.
Table 3-4. Species and the abundance sampled using seine nets in Broughton Creek tributaries.

<table>
<thead>
<tr>
<th>Species</th>
<th>number</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dendrobranchiata spp. (Prawn)</td>
<td>35215</td>
</tr>
<tr>
<td>Gobiopterus semivestitus (Glass Goby)</td>
<td>3838</td>
</tr>
<tr>
<td>Pseudomugil signifer (Southern Blue-Eye)</td>
<td>1000</td>
</tr>
<tr>
<td>Ambasss spp. (Glassfish)</td>
<td>883</td>
</tr>
<tr>
<td>Philypnodon grandiceps (Flathead Gudgeon)</td>
<td>605</td>
</tr>
<tr>
<td>Brachyura spp (Crab)</td>
<td>392</td>
</tr>
<tr>
<td>Philypnodon spp. (Dwarf Flathead Gudgeon)</td>
<td>243</td>
</tr>
<tr>
<td>Afurcagobius tamarensis (Tamar River Goby)</td>
<td>112</td>
</tr>
<tr>
<td>Mugil cephalus (Sea Mullet)</td>
<td>94</td>
</tr>
<tr>
<td>Redigobius macrostoma (Largemouth Goby)</td>
<td>47</td>
</tr>
<tr>
<td>Monodactylus argenteus (Batfish)</td>
<td>45</td>
</tr>
<tr>
<td>Gambusia holbrooki (Eastern Gambusia)</td>
<td>40</td>
</tr>
<tr>
<td>Girella tricuspidata (Luderick)</td>
<td>34</td>
</tr>
<tr>
<td>Anguilla reinhardtii (Long Finned Eel)</td>
<td>31</td>
</tr>
<tr>
<td>Acanthopagrus australis (Yellowfin Bream)</td>
<td>19</td>
</tr>
<tr>
<td>Sprattus novaehollandiae (Silver Sprat)</td>
<td>19</td>
</tr>
<tr>
<td>Potamalosa richmondi (Freshwater Herring)</td>
<td>16</td>
</tr>
<tr>
<td>Bivalvia (Freshwater Mussels)</td>
<td>16</td>
</tr>
<tr>
<td>Gerres ovatus (Silver-Biddy)</td>
<td>8</td>
</tr>
<tr>
<td>Pomatomus saltatrix (Tailor)</td>
<td>6</td>
</tr>
<tr>
<td>Syngnathinae spp. (Pipefish)</td>
<td>4</td>
</tr>
<tr>
<td>Scyphozoa spp. (jellyfish)</td>
<td>4</td>
</tr>
<tr>
<td>Gobiomorphus australis (Striped Gudgeon)</td>
<td>3</td>
</tr>
<tr>
<td>Polychaeta</td>
<td>3</td>
</tr>
<tr>
<td>Hypseleotris compressa (Empire Gudgeon)</td>
<td>2</td>
</tr>
<tr>
<td>Achlyopa nigra (Black Sole)</td>
<td>2</td>
</tr>
<tr>
<td>Platyccephalus fuscus (Dusky Flathead)</td>
<td>2</td>
</tr>
<tr>
<td>Centropogon australis (Fortescue)</td>
<td>1</td>
</tr>
<tr>
<td>Tetractenos hamiltoni (Common Toadfish)</td>
<td>1</td>
</tr>
<tr>
<td>Galaxias maculatus (Common Jollytail)</td>
<td>1</td>
</tr>
</tbody>
</table>
3.2.2 Differences between assemblages at the different types of gates

Assemblages downstream of all gate types were not significantly different to each other with similar and diverse species composition and abundance. In contrast there was a significant difference in species abundance and composition upstream, between the different gates types; leaking smart gates and uni-directional floodgates (t = 3.644; P(MC) = 0.02), uni-directional floodgates and non-gated drains (t = 3.7154; P(MC) = 0.01) and smart gates and non-gated tributaries (t = 2.0857; P(MC) = 0.05). Upstream of non-gated drains and all downstream sites showed similar diverse and abundant assemblage composition (Figure 3-8).

Consequently, there were significant differences in assemblages on either side of the gates but not upstream in non-gated drains. Smart gate assemblages differed upstream and downstream in composition, and composition and abundance (Figure 3-9, Table 3-5), as did floodgate assemblages. However floodgates hardly had any fauna upstream at all with only 4 species (Flathead Gudgeon, Glass Gobies, Blue Eye, crabs) ever sampled at sites upstream of floodgates. Smart gates had a total number of 12 species upstream which approached the species richness of open tributaries of 18 species, however there was low abundance and differences in species composition, notably the introduced species Eastern Gambusia only at Smart gate locations. In contrast, there was no significant difference in assemblages up and downstream of non-gated tributaries.

Figure 3-8. Differences in abundance and composition between assemblages (a) downstream and (b) upstream across all gate types; smart gates (SG), floodgates (FG) and non-gated drains (N). n=48.
Figure 3-9. MDS plot illustrating the similarity between assemblages upstream (blue triangles) and downstream (green triangles and dashed circle) of smart gates, floodgates and open tributaries.
Table 3-5. Permutational Analysis of Variance (PERMANOVA) table for the full model of untransformed multivariate data of fish assemblages upstream and downstream of the three gate types (Smart gates, Floodgates and No gates). As a significant interaction between the types of floodgates was found, the Ga x Lo pairwise comparison identifies which of the gate types differs upstream from downstream. P(MC), the MonteCarlo Probability, is used as \( n = 3 \). 

<table>
<thead>
<tr>
<th>Source</th>
<th>df</th>
<th>SS</th>
<th>MS</th>
<th>Pseudo-F</th>
<th>P(MC)</th>
<th>permutations</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gate (SG, FG, NG)</td>
<td>2</td>
<td>51541</td>
<td>25771</td>
<td>4.2518</td>
<td>0.002</td>
<td>15</td>
</tr>
<tr>
<td>Location (US vs. DS)</td>
<td>1</td>
<td>1.57E+05</td>
<td>1.57E+05</td>
<td>30.131</td>
<td>0.001</td>
<td>958</td>
</tr>
<tr>
<td>Day</td>
<td>7</td>
<td>46825</td>
<td>6689.3</td>
<td>2.1822</td>
<td>0.001</td>
<td>996</td>
</tr>
<tr>
<td>Sites (Ga)</td>
<td>3</td>
<td>18183</td>
<td>6061.1</td>
<td>3.5796</td>
<td>0.001</td>
<td>999</td>
</tr>
<tr>
<td>Ga x Lo</td>
<td>2</td>
<td>49561</td>
<td>24780</td>
<td>4.7438</td>
<td>0.002</td>
<td>998 *sig</td>
</tr>
</tbody>
</table>

Ga x Lo Pairwise t

<p>| | | | | | | |</p>
<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Smart Gates</td>
<td>3.7543</td>
<td>0.007</td>
<td>3 *sig</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Floodgates</td>
<td>9.1224</td>
<td>0.001</td>
<td>3 *sig</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>No gates</td>
<td>1.3672</td>
<td>0.222</td>
<td>3 ns</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

| Gate x Day           | 14   | 50559    | 3611.4   | 1.1781   | 0.172 | 997         |
| Loc x Day            | 7    | 41476    | 5925.1   | 2.0087   | 0.003 | 998         |
| Sites(Ga) x Lo       | 3    | 15671    | 5223.8   | 3.0851   | 0.001 | 997         |
| Sites(Ga) x Day      | 21   | 64372    | 3065.3   | 1.8103   | 0.001 | 996         |
| Gate x Loc x Day     | 14   | 50347    | 3596.2   | 1.2192   | 0.139 | 997         |
| Si(Ga) x Loc x Day   | 21   | 61944    | 2949.7   | 1.7421   | 0.001 | 995         |
| Residual             | 192  | 3.25E+05 | 1693.2   |          |       |             |
| Total                | 287  | 9.33E+05 |          |          |       |             |

3.2.3 Fish and invertebrate assemblage differences between upstream and downstream sites for floodgates and smart gates

3.2.3.1 Effect of Upstream vs. Downstream for smart and floodgates

The same group of taxa contributed to differences in assemblages between upstream and downstream sites of smart gates, floodgates and no gates (Table 3-6), as well as upstream and downstream of floodgates. The differences between the abundance and composition of the sites was influenced heavily by fluctuating numbers of prawns upstream of non-gated tributaries, but other taxa were obviously impacted by floodgates and smart gate structures (Table 3-7). Species richness was improved upstream of smart gates and approached that of open tributaries, however this was mostly attributed to a few species, Flathead Gudgeon, crabs and dwarf Flathead Gudgeons, as well as the introduced Mosquito Fish. The
abundance of assemblages upstream of smart gates was still significantly depressed compared to open tributaries, and prawns were notably absent but occurred in large fluctuating numbers in open tributaries.

Table 3-6. Simper Analysis identified that the following species contributed to differences upstream and downstream of all tributaries, as well as differences between upstream assemblages of floodgates, smart gates and non-gated tributaries.

<table>
<thead>
<tr>
<th>Species that contribute to assemblage differences</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dendrobranchiata spp. (Prawn)</td>
</tr>
<tr>
<td>Gobiopterus semivestitus (Glass Goby)</td>
</tr>
<tr>
<td>Pseudomugil signifier (Blue Eye)</td>
</tr>
<tr>
<td>Philypnodon grandiceps (Flathead Gudgeon)</td>
</tr>
<tr>
<td>Ambassis spp. (Glass Fish)</td>
</tr>
<tr>
<td>Brachyura spp. (Crab)</td>
</tr>
<tr>
<td>Philypnodon spp. (Dwarf Flathead Gudgeon)</td>
</tr>
<tr>
<td>Alurcagobius tamarensis (Tamar River Goby)</td>
</tr>
<tr>
<td>Monodactylus argenteus (Batfish)</td>
</tr>
<tr>
<td>Mugil cephalus (Mullet)</td>
</tr>
<tr>
<td>Redigobius macrostoma (Large Mouth Goby)</td>
</tr>
<tr>
<td>Gambusia holbrooki (Eastern Gambusia or Mosquito Fish)</td>
</tr>
</tbody>
</table>

Table 3-7. Univariate analysis of species, species richness, abundance and diversity of assemblages upstream and downstream of all smart gates, floodgates and non-gated tributaries (n = 24).

<table>
<thead>
<tr>
<th>Taxa</th>
<th>Upstream / Downstream and Gate effects</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Prawns</td>
</tr>
<tr>
<td></td>
<td>- highly variable across sampling times</td>
</tr>
<tr>
<td></td>
<td>- a trend towards high abundance and in non-gated tributaries</td>
</tr>
<tr>
<td></td>
<td>- Highly motile taxa that is effectively blocked by smart and floodgates</td>
</tr>
</tbody>
</table>
Glass Gobies
- Significantly higher abundance upstream in non-gated tributaries
- higher populations around whole tributaries with no gates, though downstream effect not significant
- one of few species found up and downstream at all sites

Southern Blue Eye
- effectively blocked from upstream migration by smart and floodgates
- variable downstream populations might be affected by upstream floodgates.

Flathead Gudgeon
- effectively blocked or low survival upstream of floodgates.
- smart gates provide adequate seepage for FHG populations to establish upstream.

Glass Fish
- temporally variable at all downstream sites
- maybe blocked from upstream migration
Crab
- Effectively better habitat provided upstream of smart gates
- blocked or unsuitable conditions upstream of floodgates although one of few taxa found at all sites

Dwarf Flathead Gudgeon
- Effectively better habitat provided upstream of smart gates
- blocked or unsuitable conditions upstream of floodgates

Tamar River Goby
- unexplained high populations downstream of smart gates
- blocked or unsuitable conditions upstream of floodgates

Batfish
- potential minor effect from blocking or low survival upstream of flood and smart gates.
**Mullet**
- potential minor effect of blocking or low survival upstream of floodgates.

**Large Mouth Goby**
- blocked or unsuitable conditions upstream of floodgates

**Glass Fish**
- highly variable with potential minor blockage upstream of floodgates and smart gates.

**Eastern Gambusia (Mosquito Fish)**
- established population of introduced pest species upstream and downstream of one smart gate site.
species richness
- reduced species richness upstream of floodgates

abundance
- reduced abundance of taxa upstream of flood and smart gates

Shannon diversity
- reduced diversity upstream of floodgates
3.2.3.2 Effect of smart gate rehabilitation on faunal assemblage

There was no significant difference in species composition upstream of bi-directional smart gates following mechanical rehabilitation (Figure 3-10) \( (t = 1.1772; P(MC) = 0.34) \).

![MDS plot of upstream faunal assemblage for each gate type before and after the rehabilitation of smart gates. Data points represent centroids for replicate seine hauls. (n =6).](image)

3.2.4 Metal content in fish tissue

Metal content in fish tissue upstream of the smart gates was significantly higher \( (p_{(perm)} = 0.002, \text{pseudo } F = 7.7) \) to that of upstream of open tributaries and downstream at all sites (Figure 3-11). Further analysis of individual metal types revealed that of the 13 metals that were screened, there were significantly elevated levels of iron (>400%), mercury (>200%), lead (1000%), nickel (130%), manganese (280%), copper (280%) and chrome (700%) compared to upstream sites in non-gated tributaries, as well as being significantly higher than in all downstream sites (Figure 3-12). There were also elevated levels of arsenic (160%) and aluminium (>200%) compared to upstream sites in non-gated tributaries, but not compared to downstream sites.
Figure 3-11. MDS representing the relative concentrations of 13 metals found in fish tissue sourced from upstream of smart gates (SU) and all other sites (upstream non-gated tributaries (NU) and downstream of both non-gated tributaries (ND) and smart gates (SD)).
Figure 3-12. Metal content in fish tissue collected upstream of smart gates (SU) compared to upstream of non-gated tributaries (NU) and at all downstream sites for smart gates (SD) and non-gated tributaries (ND). (a) nickel, (b) lead, (c) chromium, (d) iron, (e) copper, (f) manganese, (g) mercury, (h) aluminium, (i) arsenic.
4 Discussion

Consistent with other studies in Broughton Creek (Glamore & Indraratna, 2004), this study confirmed that uni-directional floodgates and drains in Broughton Creek create water quality profiles typical of acid sulfate soil affected areas. Traditional floodgates reduced the diversity and abundance of species dramatically compared to the assemblages of up to 32 species found in open tributaries. Leaking smartgates however, and despite sub-optimal conditions compared to open drains and tributaries nearby, provided for some improved water quality parameters and ecological assemblages with increases in fish abundance and diversity compared to the traditional floodgates. Unfortunately, as the repair and maintenance of smart gates is highly technical, the reparation of the smart gates did not achieve adequate flushing for a full return to open drain ecological conditions. In addition, a small drop in pH at smart gates provided for biologically available metals that were taken up in the food chain.

Some further time may be required to adequately test for recovery, however it is recommended that mechanically simpler bi-directional floodgates are tested in a similar manner to this study which demonstrates the ecological benefits of some bi-directional flow through floodgates. Such alternative bi-directional gate structures need to address the water quality and passage that is required for the return of full tributary assemblages, high abundance of prawns, and pH levels that do not create concentrations of biologically available metals in the water. In addition, simple, practical and minimal effort is a requirement for management of drain management structures.

4.1 Impacts of uni-directional floodgates

4.1.1 Water Quality

By preventing tidal exchange, uni-directional floodgates provided a stagnant, poorly flushed aquatic environment, resulting in the accumulation of acid sulfate soil discharge with low pH, temperature and salinity drain water. In previous studies in Broughton creek (Pease 1994; Sbeghen 1995; Indraratna, Blunden et al. 1999; Glamore 2003; Indraratna, Golab et al. 2006a) and elsewhere (Brown, Morley et al. 1983; Callinan, Fraser et al. 1993; Sammut, Melville et al. 1995; Sammut, White et al. 1996), increased acidification of flood mitigation drains promoted the increased concentration of dissolved aluminium and iron. This is consistent with the findings of this study where aluminium was eleven times greater and iron three times greater at sites with low pH upstream of uni-directional floodgates in comparison to sites downstream of gates or in non-gated drains.
The increased exposure of drain environments to metal species such as aluminium, iron and manganese are associated with pH-dependent solubility (van Breeman 1992; Willett, Melville et al. 1993; Sammut, White et al. 1996). Exposure to increased metal concentrations, particularly aluminium and iron, has deleterious effects on the aquatic biota and habitat of flood mitigation drains (Driscoll, Baker et al. 1980; Sammut, Melville et al. 1995). Similar to Lin and Melville (1992) and Sammut et al. (1995), the acidic waters upstream of uni-directional floodgates had the greatest concentration of soluble aluminium. These waters appeared cloudy white with flocs of aluminium hydroxide covering aquatic vegetation upstream of gated drains. Such visible indicators are typical of acid sulfate soil impacted waters (Sammut, Melville et al. 1995; Sammut, White et al. 1996; Sammut 2000).

Concentrations of soluble ferrous iron fluctuated between sites and between sampling events. The greatest mean iron concentration was observed upstream of uni-directional floodgates associated with the lowest pH. Soluble ferrous iron is present in acidified water with a pH less than four, however, when pH increases above four in the presence of oxygen, iron oxyhydroxides may be formed (Simpson and Pedini 1985; Sammut, Melville et al. 1995). The oxidation of ferrous iron to iron hydroxide consumes oxygen and releases hydronium ions consequently decreasing dissolved oxygen concentrations and pH (Sammut, Melville et al. 1995). When acidic waters mix with less acidic water the soluble iron within the acidic drains precipitates forming reddy-brown iron hydroxide, or oxyhydroxide flocs (Sammut, White et al. 1996; Sammut 2000). These flocs smothered the emergent vegetation and other benthos within all study sites influencing the environmental and ecological fitness of the drains.

During the current study, mean dissolved oxygen concentrations were similar at all sites. In contrast, other studies associated acid sulfate soil impacts with with corresponding declines in dissolved oxygen levels and consequent anoxic conditions (Pressey and Middleton 1982; NSW Fisheries 2001; Johnston 2005c). Deoxygenation events however did occur in Broughton Creek upstream of floodgate drains with minimum dissolved oxygen concentrations as low as 3.20 mg/L. The variability however resulted in averages that were not significantly lower than non-gated sites.

Uni-directional floodgates acted as barriers to upstream tidal exchange, separating drains into two distinct salinity regimes; one predominantly freshwater and one saline. The elimination of saline intrusion through tidal exchange prevents carbonate/bicarbonate buffering, thereby limiting acid neutralisation, while also transforming a previously saline aquatic environment into a relatively freshwater habitat (Indraratna, Blunden et al. 1999; Indraratna, Glamore et al. 2002; Glamore 2003; Indraratna, Golab et al. 2006a; Johnston 2009). Kroon and Ansell (2006) identified the dependence of salinity on the pattern of seasonal rainfall and possible floodgate leaks, allowing small intrusions of tidal water into freshwater reservoir environments. In Broughton Creek, minor leaks were noted during some high tide sampling events, in
which small amounts of saline water entered the upstream drainage systems. Nevertheless, the mean salinities upstream of uni-directional floodgates were considerably lower than sites with tidal influence, indicating the effectiveness of uni-directional floodgates in preventing extensive saline penetration into upstream sites.

The tidal reaches of Broughton creek, including non-gated flood mitigation drains and downstream sites at gated drains, showed reasonable water quality conditions despite the periodic inflows of acid water from floodgate drains. The dilution and/or buffering capacity of Broughton Creek tidal water were adequate to neutralize the volumes of acid soil discharge into the creek during this study, and is consistent with other studies (Hem 1985; Sammut, White et al. 1996; Indraratna, Glamore et al. 2002; Johnston 2005b; Johnston 2005c; Indraratna, Golab et al. 2006a). However as this study period did not include any flood events and only intermittent and below average rainfall, greater impacts on downstream water quality is likely during the pulse discharge of large volumes of acid sulfate soil drainage through floodgates. Green et al (2006) identified episodic precipitation as complex events, in which the evolution of acid sulfate soil water composition is unique to each event with in variations in acidity. The intensity of acid sulfate soil discharge during regular precipitation is reduced in comparison to pulse acid sulfate soil discharge events following extensive drought periods (Sammut 2000; Green, Macdonald et al. 2006).

Chlorophyll a concentrations are generally enhanced within estuaries by nutrient loading in freshwater discharges (Kalke and Montagna 1991; McKee, Eyre et al. 2000), as was the case in this study in the non-gated drain environment with full tidal exchange. However, mean chlorophyll a concentrations were depressed upstream of uni-directional floodgates in comparison to all other sites despite higher light and nutrient levels associated with agriculture (King 1948; Lin, Wood et al. 2004). This demonstrates the impact of floodgates on primary production and the ecology of drains at the base of the food chain, linked directly to water quality.

4.1.2 Ecology

Although mass fish kills are not regularly observed in the Shoalhaven River, including Broughton Creek, this study demonstrated that acid sulfate soil discharges impact whole faunal assemblages and significantly reduced the ecological diversity upstream of uni-directional floodgates. These results are the first to document whole assemblage impacts of flood mitigation drains and uni-directional floodgates on aquatic species composition within Broughton Creek, but are consistent with Pollard & Hannan (1994) who established that flood mitigation drains without floodgates contained a larger proportion of both commercial and non-commercial fish species in comparison to floodgate influenced drains. Similarly,
Kroon & Ansell (2006) identified a higher abundance and biomass of juvenile fish in non-gated drains in comparison to drains with uni-directional floodgates.

Fish and invertebrate assemblages that were present at sites with tidal exchange but absent upstream of floodgates included species of commercial and ecological significance such as *Dendrobranchiata spp.* (Prawn) and *Gobiopterus semivestitus* (Glass goby) respectively. The direct cause for these effects of floodgates on faunal assemblages may be three fold. Kroon and Ansell (2006) showed that uni-directional floodgates act as physical barriers to juvenile fish and invertebrate migration. In addition, upstream floodgate drain water quality is often not conducive to the maintenance of local aquatic assemblages, with low pH levels, high metal content and reduced salinity. Furthermore, floodgates effects were associated with a lack of fringing mangrove vegetation in upstream drains and an increase in less saline tolerant vegetation such as grasses. The reductions in natural saline tolerant fringing vegetation, including mangroves, and replacement with salt intolerant grass and tree species as observed here, has also been identified in other estuarine drain environments, creating very different available habitat for faunal assemblages and increasing drain erosion (NSW Agriculture and Fisheries 1989; Ashraf and Yasmin 1991; Pollard and Hannan 1994; Johnston, Slavich *et al.* 2005a; Kroon and Ansell 2006).

**Unsuitable Water Quality**

Fish species begin to show stress from elevated hydrogen ion concentrations when pH levels decline below 6.5. However, for most species the critical pH level is less than 5.0 (Brocksen, Marcus *et al.* 1992; Pease 1994; Russell and Helmke 2002). This relates to gill and skin damage, associated with acid water interaction, resulting in disturbances to oxygen transport and ion regulation (Brocksen, Marcus *et al.* 1992; Pease 1994; Sammut, White *et al.* 1996; Sammut 2000; Russell and Helmke 2002). Gill and skin damage reduces the capability of fish to obtain oxygen, or regulate their intake of salts and water, consequently increasing fish mortality (Pease 1994; Sammut, White *et al.* 1996; Sammut 2000). Skin damage also increases the susceptibility of fish to fungal infections and diseases, such as epizootic ulcerative syndrome, or red spot disease (Callinan, Fraser *et al.* 1993; Pease 1994; Sammut, White *et al.* 1996; Sammut 2000). A disease such as epizootic ulcerative syndrome is not the direct cause of fish mortality, but may accelerate death due to feeding disorders, decreased immunity and increased susceptibility to secondary infection (Fraser, Callinan *et al.* 1992; Callinan, Fraser *et al.* 1993; Pease 1994).

Elevated mean aluminium and iron concentrations were observed during the current study, upstream of floodgates. These concentrations are considered directly damaging to fish biology, being the primary cause of lethal and sub-lethal impacts (Driscoll, Baker *et al.* 1980; McDonald 1983; Sammut, White *et al.*
1996; Hyne and Wilson 1997; Sammut 2000; Canli and Guluzar 2003; Green, Macdonald et al. 2006). The sulfuric acid present within the Broughton Creek drainage network, particularly upstream of unidirectional floodgates, can react with a number of sedimentary minerals to increase soluble concentrations of pH-dependent elements, including iron and aluminium, to levels that are toxic to biota, particularly fish under direct acid stress (Driscoll, Baker et al. 1980; Pease 1994; Sammut, Melville et al. 1995; Lin, Bush et al. 2001; Russell and Helmke 2002). Driscoll, Baker et al. (1980), Sammut, Melville et al. (1996) and Green, Macdonald et al. (2006) demonstrated that inorganic aluminium, even at very low concentrations (<3 μm), is responsible for lethal and sub lethal fish impacts in acidic water. In conditions similar to those observed upstream of floodgates, inorganic monomeric aluminium accumulates on the negatively charged gill surfaces of fish and invertebrates (Norrgren, Wicklund Glynn et al. 1991; Sammut, Melville et al. 1995; Sammut 2000; Dove and Ogburn 2007). Through the displacement of calcium, this increases gill permeability causing an ionic imbalance and physiological stress, resulting in increased fish mortality (Sammut, Melville et al. 1995).

High concentrations of soluble ferrous iron and red-brown flocs of iron hydroxide were observed upstream of unidirectional floodgates within the Broughton Creek system. Such deposits can coat the gills of fish and invertebrates, impairing gas exchange (Simpson and Pedini 1985; Sammut, Melville et al. 1995; Sammut 2000; Russell and Helmke 2002), while also increasing egg and larval mortality by smothering benthic habitats (Simpson and Pedini 1985; Sammut, Melville et al. 1995; Sammut, White et al. 1996). Simpson and Pedini (1985) also noted that iron hydroxide encrusts benthic and emergent vegetation reducing photosynthetic processes and rendering them inedible to meiofauna. This was supported in this study with reduced chlorophyll a concentrations apparent in the floodgated drains.

Reduced tidal exchange upstream of unidirectional floodgates impeded saline water intrusion resulting in relatively freshwater environments. Although aquatic organisms inhabiting estuarine environments are tolerant to fluctuations in salinity (Remmert 1980), the consistently low salinity observed upstream of the Broughton Creek floodgates may also contribute to lower fish biomass, abundance, and species richness in comparison to the natural tidally influenced drains (Swales 1982). Low salinities have a negative effect on larval survival and growth in several crab species, while also impacting juvenile fish growth rates (Anger 1985; Anger 1996; Spivak 1999; Labonne, Morize et al. 2009). Estuarine fish populations, in particular, can show high recruitment variability as a result of the complex physical environment they inhabit (North and Houde 2003; Nicholson, Jenkins et al. 2008). Nicholson, Jenkins et al. (2008) observed high levels of physiological deformities in Black bream (Acanthopagrus butcheri) larvae, influenced by low aquatic salinity levels. The effects of salinity on embryo survival can vary depending on temperature, suggesting an interaction between the two environmental factors (Hassel, Coutin et al. 2008; Nicholson, Jenkins et al. 2008).
Changes to Habitat

Reduced tidal exchange increases adult mangrove mortality and reduces the establishment of mangrove propagules (State Pollution Control Commission 1978). Pollard and Hannan (1994) observed similar losses of mangroves above floodgates, due to greatly reduced tidal exchange. The reduction in mangroves has been associated with the reduction of highly productive fish nursery and refuge habitats (Stephenson and Dredge 1976; Blaber and Blaber 1980; Morton, Pollock et al. 1987; Bell and Pollard 1989; Morton 1990; Pollard and Hannan 1994), the degradation of drain bank structure and increased sedimentation through increased erosion (Dent 1986; White, Melville et al. 1997; Cappo, Alongi et al. 1998; Johnston, Slavich et al. 2005a).

4.2 Aquatic changes associated with bi-directional smart gates

Prior to smart gate rehabilitation, the smart gate structures leaked, providing some buffering capacity, and showed significantly higher species abundance and diversity in comparison to uni-directional floodgates. Species abundance and richness was 10 fold higher upstream of smart gates in comparison to floodgates. The assemblage composition and abundance approached but remained significantly lower than that of non-gated drains. Similarly, other studies have reported water quality improvements when uni-directional floodgates leak upstream during high tide events (Pollard, Middleton et al. 1991; Pollard and Hannan 1994; Blunden and Indraratna 2000; Kroon, Bruce et al. 2004; Kroon 2005; Johnston 2005b; Indraratna, Golab et al. 2006a).

It was hypothesised that following smart gate rehabilitation, upstream smart gate and non-gated drain assemblages would become even more similar with renewed tidal exchange. However, the tidal exchange and mechanic of the gates did not appear to be immediately successful with no observations of the gates opening, observation of increasing water sealing of the gates (i.e. reduced leakage) and some indications of reduced water quality and diversity. This however is simply an artefact of the practical difficulties associated with the maintenance and control of the current mechanical model of the smart gates, and again justification for a simpler smart gate mechanism. A further consideration is that it may take a year or more for some of the fish assemblages to recover through natural recruitment.

4.2.1 Water Quality

Changes in water quality at sites upstream of bi-directional smart gates were not consistent with the expected restoration of tidal exchange, but rather a reversion to a closed floodgate condition. The
significant decrease in mean pH for example, is not consistent with previous smart gate studies in Broughton Creek, in which pH increased with improved tidal exchange (Pease 1994; Sbeghen 1995; Indraratna, Blunden et al. 1999; Indraratna, Glamore et al. 2002; Glamore 2003; Glamore and Indraratna 2004; Indraratna, Golab et al. 2006a).

The extent and stability of water quality improvement is dependent upon the frequency, magnitude and duration of upstream tidal flows. Frequent and prolonged tidal exchange, associated with diurnal tide cycles promotes stable improvements in drain water quality (Johnston 2005b). Both bi-directional smart gates leaked before mechanical modification due to infrastructure corrosion, but appeared to have an improved seal with no leakage following repair. This apparent reduction in tidal exchange is supported by the measured decrease in mean pH and relatively low salinity levels indicative of no tidal exchange.

The current mechanical model of smart gates is complex and therefore, with a lack of maintenance and operation, they quickly fall into disrepair. Simpler floodgate models exist that reliably open and close under tidal influence with float systems rather than computer controlled and motorised systems. Such alternative models of bi-directional floodgates should be further explored in this way to demonstrate effective improvement of ecological systems and low maintenance requirements. Comparisons of different flood gate modifications and designs have been compared and an adaptation of a suitable one for Broughton Creek conditions should be considered (NSW Fisheries Floodgate Design Workshop (2002), NSW Industry & Investment (2009)).

4.2.2 Ecology

The diverse faunal assemblage upstream of bi-directional smart gates, in comparison to uni-directional floodgates, was evident prior to remediation of smart gates and most probably due to minor but constant leakage through the gates. This observed leakage and consequent improvement of water quality provided for passage and suitable conditions for some species. Following the rehabilitation of bi-directional smart gates, the leakage was effectively sealed and malfunctioning of the gates appeared to persist. Thus the mechanics of the smart gates proved too complex for effective, practical operation. A further consideration is that environmental and ecological drain quality may require prolonged tidal exchange to restore more normal conditions within upstream drains.

This current study illustrated similar low faunal assemblage and species diversity at sites upstream of uni-directional floodgates. Although species assemblages at sites upstream of bi-directional smart gates were not as great as that observed at open, non-gated drains, they were greater than sites upstream of uni-directional floodgates. As an ecological management tool, this emphasizes that although the renewed tidal
exchange is an improvement in comparison to uni-directional floodgates, full tidal exchange associated with open flood mitigation drains is required in order to retain faunal assemblages similar to that exhibited in Broughton Creek.

Ecological evidence suggests that biological factors such as environmental preferences and diadromous life cycle contribute to riverine fish assemblage variations (McDowall 1993; Townsend and Hildrew 1994; McDowall 1998; Gehrke, Gilligan et al. 2001; Gehrke and Harris 2001; Joy and Death 2001; Talmage, Perry et al. 2002; Miles 2007), and therefore changes to drain and tributary habitat conditions can have consequences on the productivity of estuaries. Miles (2007) investigated the migration activity of *Potamalosa richmondia* (Freshwater herring), observing the downstream migration of adults from freshwater to marine areas during winter, to spawn. During the current research, this species was sampled downstream of smart gates before the rehabilitation of smart gates in autumn. *P. richmondia*, however not collected during winter ‘after’ sampling. The differences in species composition between before and after sampling may consequently be attributable to seasonal migratory behaviour.

### 4.2.3 Metal accumulation in fish tissue

Despite improvements to the fish assemblage diversity and abundance, the smart gate construction and degree of water exchange did not allow for full buffering of water upstream of the gates. Therefore, the pH was still depressed upstream of smart gates although it fluctuated widely. The lower pH facilitated the release of metals in a biologically available form, upstream of smart gates, and resulted in elevated levels of 9 metal types. In the case of lead, the content as mg/kg wet weight exceeded that of acceptable Australian & New Zealand Food Standards code for contaminants and natural toxicants (standard 1.4.1, 2010).

Although the fish species used in the study (*Phylipnodon grandiceps* or Flathead Gudgeon) is small and not consumed by humans, the fact that poor buffering of drain waters provided a point of entry for bioaccumulation of metals into the ecological food chain has consequences for ecological health and potentially human health. This species is likely to be preyed upon by a wide range of species including bass, flathead, flounder, tailor and other species (Allan Lugg, NSW Fisheries, personal comment) that are consumed by humans.
5 Conclusion

Floodgates created highly acidic and metal laden water reservoirs upstream of gate structures. This and the physical barrier of the gates resulted in significant and severe reductions (to zero for most species) in abundance and diversity across a suite of at least 30 fish and invertebrate species, compared to open drains and tributaries. Of note is that 5 of these species are of recreational and/or commercial fishing importance, and that although the cumulative scale of impacts across all 47 floodgates in the Shoalhaven River and Broughton Creek are unknown, it has the potential to be huge with socio-economic consequences. Such estimates could be calculated from GIS data and ground truthing samples of drain and tributary dimensions as well as further assessment of the extent of species migration in the tributaries. A very simple estimate based on the impact on prawn populations is that there is a loss of 1400 prawns for every ML of water upstream of all 47 floodgates.

Despite improved conditions and ecological assemblages upstream of leaky smart gates, there were still periods of severe drops in pH and release of metals. This study demonstrated that at least 9 metals released into drain water were biologically available and taken up in the food chain. Flathead Gudgeon tissue, sampled upstream of leaky smart gates, was found to have significantly elevated concentrations of iron (>400%), mercury (>200%), lead (1000%), nickel (130%), manganese (280%), copper (280%) and chrome (700%) compared to upstream sites in non-gated tributaries, as well all downstream sites. Although this fish species is small and not a recreational or commercial catch, it represents the bottom of the food chain and had levels of lead that exceeded acceptable Australian food safety standards (FSANZ standard 1.4.1). This is a cause for concern given that many predatory fish species are actively targeted by recreational and commercial fishers in Broughton Creek and the Shoalhaven estuary.

The consequences of these findings support the need to remove, modify or replace floodgates structures in Broughton Creek and the Shoalhaven River estuary floodplains with adequate bi-directional flow systems in agricultural drains. This effort must be supported with further research to:

- determine the extent of ecological assemblage improvement (preferable not significantly different to that of open drains)
- provide for the return of commercially important species (e.g. prawns)
- confirm that metals are no longer biologically available in the aquatic food chain
- demonstrate that inundation of pastures does not unreasonably affect farmers
6 References


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Appendix I: Oyster Shell Dissolution as a simple monitoring tool for acid sulfate soil impacts on water quality

Introduction

Monitoring for acid sulfate impacts can be as simple as observing metal flocculation and water colour in severe cases, however where there are intermittent or more cryptic low pH values, it is of interest to determine where in a catchment the most affected acid sulfate soil run-off occurs. This information has application for land managers to target priority areas for remedial measures, and also to monitor that on-ground works achieve better outcomes. However pH monitoring requires long temporal measurements of daily fluctuations, an effort which can become very costly with installations of expensive analytical instruments. This project undertook a pilot trial to test if an alternative and less costly monitoring tool could be applied, on the understanding that oyster shell dissolves measurably at pH<7.

Variations in estuarine and marine water pH, have been shown to have deleterious effects on a number of bi-valve species including dissolution of shell (Loosanoff and Tommers 1947; Babmer 1990; Dove and Ogburn 2007; Dove and Sammut 2007b; Marshall, Santos et al. 2008). Bamber (1987; 1990), and a number of bivalve species were unable to tolerate water acidity levels below pH 7. Shell calcium carbonate dissolution increases the rate of mortality, while reducing growth rates and increasing disease (Kuwatani and Nishii 1969; Bamber 1987; Bamber 1990; Dove 1997; Dove and O’Connor 2007b). Direct impacts to Saccostrea glomerata, associated with exposure to acid sulfate soil discharge, include increased mortality, reduced growth, shell dissolution and soft tissue damage (Dove and Ogburn 2007; Dove and Sammut 2007a; Dove and Sammut 2007b). As dissolution of shell can be measured simply as a loss of shell weight, it may be possible to use oyster shell as a simple monitoring tool for relative comparisons of sites in acid sulfate soil affected waterways.

The aim of this project was to determine if drain acidity associated with acid sulfate soil discharge can be monitored using a simple method of natural calcium carbonate dissolution. The use of Saccostrea glomerata (Sydney rock oyster) shell dissolution rates as a monitoring tool for acidity within flood mitigation drain systems was trialed in the lab under controlled conditions and in the field. This project hypotheses that S. glomerata shell dissolution will be measurably greater in lower pH aquatic environments. The investigation and development of a bio-monitoring system will provide an inexpensive environmental based monitoring approach to acid sulfate soil impacts.
Methods

The rate of calcium carbonate dissolution, associated with the shells of *Saccostrea glomerata* (Sydney rock oyster), was examined in both the laboratory and field. *S. glomerata* shell lids were supplied by a number of commercial oyster growers located along the south coast of NSW, in areas not immediately impacted by ASS-affected waters (Dove & Sammut 2007). Following collection, the oyster lids were dried and cleaned of shell inhabiting organisms, including barnacles and bryozoans, as these organisms have differing dissolution rates to the oyster shell substrate (Smith 2009). The weight of each shell lid was individually obtained (d = 0.001 g) utilising a Kern 572 precision balance. The lids were subsequently positioned within a pocket of a mesh sample bag, oriented in the same direction. The sample bags (910mm x 200mm) were constructed from fibre glass fly-screen and sewn, using polyester and fish line threads, with 9 (laboratory bags) or 10 (field bags) pockets in each bag. Shell lids were placed in individual pockets to ensure there was no contact and equal distance between shells and to allow individual shell weight loss to be determined (Fig. 1).

![Cleaned and pre-weighed Sydney Rock Oyster shell to be suspended in mesh.](image)

**Figure 1.** Cleaned and pre-weighed Sydney Rock Oyster shell to be suspended in mesh.

**Lab based analysis**

Five adjusted pH seawater treatments (pH: 4, 5, 6, 7, and seawater control of 8) were repeated in 80L glass tanks. These tanks were replicated three times (n=15). The seawater for this experiment was collected at Jervis Bay (Vincentia) with a salinity of 33 ppt and sterilized using chlorine to prevent epibiota from settling. A 5% solution of sulfuric acid (type) was added to each tank until the required pH was attained. The control treatment was established as the natural pH of the saline water over the period of the study (7.99pH). A Yeo-Kal Intelligent Water Quality Analyser (Model 611) measured pH during the experiment. Further addition of sulfuric acid solution to tanks occurred where required. All other environmental variables, including temperature, light and salinity remained constant.

When the water pH was stable, a nine pocket oyster bag (as described above) was installed in each tank (Figure 2). The bags were attached, by fishing line, to support rods above the tank so that all bags were
positioned mid-depth in each tank. The oyster shell lids remained in the tanks for four weeks, with individual oyster weights measured every two weeks utilising the precision balance (Kern 572). Before measuring weight, each shell was lightly wiped with paper towel to dry shells and remove dendrite material which may have accumulated during settlement.

Figure 2: Oyster shells suspended in pH controlled environments in aquaria.

Field based trials

Two oyster bag replicates (each of 10 pockets) were installed at each upstream flood mitigation drain field site and two Broughton Creek reference sites. Bags were attached to 1.80m hardwood stakes using zip ties and u-shaped nails, 20cm from the top and 50cm from the bottom (Fig. 3). The bags and stakes were installed at each site during a low tide event, in order to ensure that a minimum of seven S. glomerata shell lids were submerged at all times as water depth, and the upper shells allowed for the influence of tides. The water pH for each site was also measured using a standard pH probe. The oyster shell lids remained at each site for six weeks, during which time the individual weight of each oyster shell was examined every two weeks. Before measuring weight, each bag was sluiced in the drain water to remove large particulate matter. Each shell was further wiped lightly with paper towel to dry shells and remove fine dendrite material, metal flocculation and algal material which may have accumulated during settlement.
Figure 3: Oyster mesh bags in field. Field bags were suspended vertically to account for the potential influence of tidal variation.

Results

Laboratory based

*Saccostrea glomerata* shell weight loss (Fig. 4) was significantly greater at lower pH treatments in comparison to higher pH treatments (ArcSin(%) transformed $C = 0.1097$. $F = 14.34$; $P = 0.0004$). Oyster shells exposed to water with a pH of 4 lost on average 0.49 g of shell, while water with a pH 5 lost on average 0.36 g of shell over the four week treatment period which was significantly greater than the higher pH treatments (pH 6 – 8). *S. glomerata* shells exposed to the alkaline pH 8 water (seawater control) retained the majority of shell weight losing 0.046 g of mean shell weight.

Similarly, mean percent shell weight loss (Fig. 4) was significantly greater at lower pH water treatment (%Sqrt(x +1) transformed $C = 0.1231$. $F = 12.02$ $P = 0.0008$). *S. glomerata* shells exposed to pH 4 water lost mean 7.85% of mean weight, in comparison to the next highest, pH 5 treatment, which lost mean 5.77% of shell weight. The lowest mean percentage weight loss was in pH 8 water where 0.74% of mean shell weight was lost.
Figure 4. Total mean *Saccostrea glomerata* shell weight loss (bar), and mean percentage weight loss (line) in laboratory pH treatments (n = 27; bars = ± 1 SE).

**Field based**

The mean pH of sites upstream bi-directional Smart gate and uni-directional floodgate drains (Fig 5) were relatively similar with a mean pH of 5.45 and 5.74 respectively. The non-gated upstream drain sites and the Broughton Creek sites also had consistent pH with a mean pH of 7.23 at non-gated sites and 7.14 at sites in Broughton Creek. These differences between gate influenced and non-gated sites were considered sufficient to warrant a field based experiment assessing oyster shell weight loss and acidity.
The exposure of *Saccostrea glomerata* shells to varying water acidities within the Broughton Creek system (Fig. 6) showed no significant differences between drain water acidity and mean shell weight loss ($\ln(X)$ transformed $C = 0.1557, F = 5.99, P = 0.0878$). The mean shell weight loss for Smart gate, creek and to some extent non-gated treatments were consistent with the laboratory results. A significant reduction in shell weight of 0.32 g was observed upstream of bi-directional smart gates with a low pH. In contrast, however, upstream of uni-directional floodgates with similar acidic conditions, did not conform to the predicted dissolution rate based on laboratory results. The mean shell weight loss upstream of uni-directional floodgates, 0.18 g, had an almost identical mean weight loss to that at non-gated drains where 0.17 g was lost. The Broughton Creek oyster shells had the lowest mean weight loss of 0.12 g over the four week treatment period.

Low shell dissolution rates upstream of uni-directional floodgates can be attributed to large quantities of iron and aluminium flocculation, sediment material and biofilms were observed on the shell surface following immersion (Fig. 7). Attempts to remove the majority of settlement material was performed prior to weighing the shells, however, not all material could be removed.

The differences in percentage weight loss between sites (Fig. 6) was similarly not significant ($\sqrt{X+1}$ transformed $C = 0.1764, F = 6.14, P = 0.0852$). The greatest mean percentage weight loss was upstream of
the bi-directional smart gates, where 4.6% of an average 6.29 g shell was lost. The lowest mean percent weight loss was in Broughton Creek, where 1.7% of initial *S. glomerata* shell weight was lost.

Figure 6. Total mean *Saccostrea glomerata* shell weight loss (bar), and mean percentage weight lost (line) at field treatment sites (Smart gates (SG), Floodgates (FG), Non-gated (NG), and Broughton Creek (CK)) (n = 18; bars = ± 1 SE).
Discussion

The laboratory shell dissolution experiment showed a significant and linear reduction of *Saccostrea glomerata* shell weight with decreasing pH. Aggressive calcium carbonate dissolution at low pH values of 3 and 4, illustrate the dramatic impact of acidified sea water on *S. glomerata* shells. The quantity of calcium carbonate dissolving from *S. glomerata* shells, may have a substantial impact on the overall structure and integrity of *S. glomerata* shells. With an average initial weight of 6.29 g, shells exposed to low pH water (pH 3-4) lost 5-7% total weight over the four week treatment period, which may be a considerable impact on shell structure.

The variation in *S. glomerata* shell dissolution in the field, at sites with similar pH to those in the laboratory, illustrates the influence of not only acidic water, but other environmental variables on shell dissolution within the Broughton Creek system. Although shells upstream of smart gates and floodgates were exposed to similar acidic conditions, the mean weight loss for shells at Smart gate sites was greater than that observed at upstream floodgate sites. This variation may relate to environmental differences between sites reducing the dissolution rate and consequently total weight loss.
Schroeder (1969) and Reaves (1986) indicated variations in calcium carbonate dissolution, in association with variations in ion and organic matter concentrations. Large quantities of iron and aluminium flocculation, dendrite material and biofilms were observed on the surface of shells at sites within Broughton Creek. Attempts to remove the majority of this material were performed prior to weighing shells, however, not all material could be removed. It is therefore considered that, total mean weight loss was potentially reduced due to the presence and concentration of organic material. These flocs also indirectly influence weight loss, through a reduction in shell surface area exposed to acid water.

The use of oyster shell dissolution and weight loss may provide for a novel and cost effective method for community groups to measure the pH fluctuation in estuaries. Further analysis is consequently required to establish the impact of organic matter, particularly metal flocs, on S. glomerata shell dissolution between sites. An examination of the effects of tidal exchange and the possible effects of environmental weathering is also required, in order to understand if such processes may also influence shell weight loss.