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**Long-term variations in bycatch reduction device
(BRD) effectiveness and trawl catch rates in a
tropical fish assemblage**

Franz Martin Fingerlos

**For the research Degree of Master of Science in the Department of Marine and Tropical
Biology at James Cook University of North Queensland**

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Abstract

Fishing gears are often “non-selective”. That is, they frequently catch organisms that are not the target of the fishery. Such organisms are usually referred to as bycatch. Some bycatch can be sold or used, but much of it is often of no value and unwanted. Thus, bycatch is a major problem in fisheries around the world, for both the fishery and the ecosystems from which the bycatch is removed. The prawn trawling industry has in the past captured very high ratios of bycatch to target species, due to the non-selective nature of trawl gear. Fifty to ninety percent of total catch by weight in a prawn trawl net can comprise unwanted species that are mostly discarded. Reducing the amount of unwanted catch in commercial fisheries is of major importance if we are to use fish stocks wisely and conserve global biodiversity. The implementation of bycatch reduction devices (BRDs) has been a major focus of fisheries management over the last two decades.

The data for this study were obtained from field sampling in Cleveland Bay, North Queensland, Australia. The sampling was carried out by staff at James Cook University (JCU), assisted by students enrolled in a fisheries science subject at JCU. The equipment, sampling techniques and general area of sampling remained consistent over the entire fifteen year (1998-2012) study period. Prawn trawls were carried out for two or three days in the first two weeks of March for 12 of the 15 years, using the same vessel, the same net configurations, the same trawling procedures, and in the same inshore area of Cleveland Bay. Individual trawl times of 10 minutes duration could be expressed as a swept area of each trawl net of 6000m², and this also remained consistent over the entire study. For ease of comparison with the broader literature all catch rates are expressed as kg/hectare in this thesis. Thus, it was possible to obtain a long-term, scientific prawn trawling data set in which sampling procedures remained consistent.

The first data chapter of this thesis examined the effects of the Jones-Davis BRD on catch rates of prawns and fish bycatch. A total of 244 trawls (each of 10 minutes duration) were carried out in 12 separate years between 1998 and 2012 by the JCU research vessel, James Kirby, rigged as a twin otter trawler. The two prawn trawling nets had different configurations: Turtle excluder device (TED) only (control net); TED and Jones-Davis BRD. The effect of the BRD on catch rates of four major teleost families, all other fish, penaeid prawns, and total fish catch was assessed. The BRD reduced catch rates of total fish bycatch, Carangidae, Sciaenidae and “Other Fish” significantly during the study. A significant year (time) effect on catch rates was detected for all variates investigated, but no significant interaction between the BRD and year of sampling was detected for any variates. Thus, the effect of the BRD was consistent across all years. Prawn retention using this device was high, with no significant difference in prawn catch rates between the two nets. The BRD reduced catch rates of total fish (by 23.7% +/- SE 4.91), carangids (trevallies) (by 65.9% +/- SE 3.48) and sciaenids (croakers) (by 24.6% +/- SE 6.82) significantly. The BRD was an effective tool in reducing catch rates of fast, strong swimming semi-pelagic and pelagic fish species, but ineffective in reducing catch rates of slow swimming benthic and demersal fish species (e.g. Leiognathidae, *Saurida spp.*). Slow swimming benthic species comprise the majority of the tropical inshore fish assemblage.

Averaged over the 15 years of the present study, the 23.7 % reduction in total fish catch means that the Jones-Davis BRD was ineffective in eliminating 76.3% of the fish bycatch in this tropical fish assemblage in Cleveland Bay. Attempts to design BRDs specifically for tropical Indo-West Pacific conditions have reduced fish bycatch in prawn trawl nets by 20-40%, at best. Thus, this study of the Jones-Davis BRD in Cleveland Bay is consistent with these other studies, even when the BRD was not designed for local conditions. The second data chapter of this thesis investigated the effect of environmental factors on inter-annual

variations in trawl catch rates in a tropical fish assemblage in Cleveland Bay over 15 years. Environmental factors investigated included rainfall, tidal state and moon phase. Rainfall two years prior to sampling affected total fish catch rates significantly ($p < 0.05$; R^2 adj. = 0.305), again with catch rates enhanced by rainfall. The two year lag effect may be due to a two year lag in recruitment of fish to the fishery. This potential delayed recruitment into the fishery could be explained by migration of sub-adults out of the estuaries and mangrove forests into the near shore habitats where trawling occurred, combined with fish growing to a catchable size. Catch rates of prawns (mostly banana prawns, *Fenneropenaeus merguinesis*) were not affected by rainfall in this study. Catch rates of prawns and *Saurida* spp. (Lizardfish) were significantly higher on the full moon ($p < 0.05$). The effects of environmental drivers like rainfall and moon phase on inter-annual variations in catch rates of both prawns and fish in this tropical fish assemblage could only have been detected by consistent, long-term sampling.

The third data chapter of this thesis examined inter-annual variations in the species composition of the tropical fish assemblage in Cleveland Bay from 1998 to 2012. A total of 160 teleost species were recorded in the 244 trawls made over 15 years. A Multi-dimensional scaling (MDS) analysis of the species presence-absence data demonstrated that there was no long-term, systematic shift in the species composition of the tropical fish assemblage in Cleveland Bay over the 15 years of study. That is, there was no evidence that the assemblage structure changed from one state to another, despite inter-annual variations in species composition and environmental conditions. In addition, there was little evidence of differences in species composition of the trawl catch between nets fitted with and without the Jones-Davis BRD.

Long term monitoring studies are an important tool in fisheries research. Usually such studies involve collection of catch rate data from commercial fisheries. Gear efficiencies and fishing

effort often vary in long-term studies of commercial fisheries. Long-term scientific, fishery-independent, surveys using completely consistent methods are rarer, often because funds to support them are limited. The long-term consistency of sampling in this study was critical in revealing inter-annual variations in performance of a BRD, identifying environmental drivers of trawl catch rates, and demonstrating the long-term consistency of species composition of a tropical fish assemblage.

Chapter 1

General Introduction:

The role of biodiversity in maintaining ecosystem services for a growing human population is considered important to the food supply of current and future human generations. Local species richness may enhance ecosystem stability and productivity (Loreau et al., 2001; Worm et al., 2006; Worm and Branch, 2012). Such effects are more difficult to demonstrate at a landscape level (Halpern et al., 2009; Kappel et al., 2012a). Management of the world's ocean resources is a challenge, given the geographically large extent and taxonomically complex nature of these resources (Levin et al., 2009; Worm et al., 2006). Implementing local findings on large spatial scales has proven to be more difficult in marine than in terrestrial ecosystems (Hendriks et al., 2006). The marine environment provides many goods and services, one of the most important being food supply for a growing human population (Holmlund and Hammer, 1999; Peterson and Lubchenco, 1997; Smith et al., 2010). An ever-increasing proportion of the human population lives near coasts, and this human population pressure often results in loss of services due to pollution and overfishing (Crain et al., 2009; Kappel et al., 2012b). This can have very clear detrimental effects on marine ecosystems. Overexploitation of marine resources, pollution of the marine environment, and habitat destruction can be directly linked to changes in marine biodiversity (Adger et al., 2005; Dulvy et al., 2006; Worm et al., 2006). Detecting the extinction of marine species is often difficult, but a very convincing literature documenting rapid decline of populations, species diversity, or entire functional groups in the marine environment exists (Pauly and Christensen, 1995; Pauly et al., 2002; Pauly et al., 2000; Pauly et al., 2005; Worm et al., 2006; Worm et al., 2005). It is now widely accepted that since the late 19th century, commercial fisheries have substantially reduced the abundance of target populations and

affected associated fish assemblages (Pauly and Christensen, 1995; Pauly et al., 1998; Pauly et al., 2005; Worm et al., 2006). Fishing affects the life history parameters of both targeted and non-targeted marine species and in some circumstances has resulted in local extinction of certain species (Jennings et al., 2005; Pope et al., 2006; Rogers and Ellis, 2000; Tonks et al., 2008).

Approximately 148 million tonnes of fish were supplied to global markets from fisheries and aquaculture in 2010 (FAO, 2012a). Sustained growth in fish production and improved distribution channels have led to a growth in world fish supply in recent decades with an average growth rate of around 3.2% per year (FAO, 2012). Per capita human fish consumption has doubled in the last 40 years with an average of 18.6kg in 2010 (FAO, 2012). While inland fisheries have increased output over the last few years, marine fisheries production has been declining in the last two decades (Pauly et al., 2002; Pauly and Froese, 2012; Watson et al., 2012) even though fishing effort has increased and technological advances and biological knowledge of the target species have been improving (FAO, 2012). Watson et al (2012) recently showed that, on a global scale, marine fishing effort has increased 10-fold in the past 60 years, but catch per unit effort has halved over the same period. Global marine fisheries production in 2010 was estimated to be around 77.4 million tonnes. The proportion of under- exploited fish stocks has decreased gradually since the early 1970's and currently only around 13% of global fish stocks are not fully exploited, with 57% of stocks being fully exploited and around 30% being over-exploited (FAO, 2012; Froese et al., 2012). The ever increasing demand for fish has been compensated for by rapidly expanding aquaculture production, which has expanded 12-fold in the last three decades, currently producing around 60 million tonnes of fish, molluscs and crustaceans (FAO, 2012). Declining marine catches combined with increasing numbers of overexploited fish stocks clearly shows that the state of the world's marine fisheries is worsening (FAO, 2012a; Firth

and Hawkins, 2011; Pauly, 2009; Pauly and Froese, 2012; Pauly et al., 2005; Watson et al., 2012). This overexploitation of marine resources causes negative ecological impacts, reduces fisheries production, and results in negative social and economic consequences (Christensen, 2005; FAO, 2012a; Pauly and Froese, 2012; Worm et al., 2006).

Overfishing of particular fish species has been seen as the major driver for collapsing fish stocks around the world (Christensen et al., 2003). Originally fish stocks were managed to attain sustainable harvest of the target species (Brewer et al., 2006). More recently there has been a greater interest in managing the whole ecosystem (Christensen and Pauly, 2004; FAO, 2012b; Foley et al., 2010; Levin et al., 2009). This approach does not focus on sustainable harvest of the target species alone, but also on the sustainable management of the associated ecosystem, and the social and economic benefits that can be derived from such approaches (Levin et al., 2009; Worm et al., 2006). In order for such management to be implemented, a thorough understanding of ecosystem processes and dynamics is desirable. Thus, we need to better understand which parts of ecosystems are not only important in social and economic terms, but in a more holistic, ecological context. The growing commitment to an ecosystem-based approach to fisheries means that fisheries managers need to take into account the wider environmental impacts of fishing (Thrush and Dayton, 2002; Tillin et al., 2006). Fisheries management must ensure that fishing effects are sustainable not only in terms of the target species but in terms of ecosystem maintenance (Pikitch et al., 2004; Pope et al., 2006). Alterations in species composition of marine ecosystems may alter the functional diversity of communities and lead to modifications of ecosystem function (Solan et al., 2004; Tillin et al., 2006).

A major problem of many fisheries is the disturbance and alteration of habitat associated with the fishery (Dayton et al., 1995; Olsgard et al., 2008; Thrush and Dayton, 2002). This is particularly important for fisheries that target species that are closely associated with the

benthic environment (Thrush et al., 1998; Thrush and Dayton, 2002). Demersal trawling can have major habitat modification effects, and can be non-selective in terms of catch composition at times (Christensen, 2005; Dayton et al., 1995; Olsgard et al., 2008; Schratzberger et al., 2002; Schratzberger and Jennings, 2002; Tillin et al., 2006). Demersal bottom trawls can remove a large proportion of the benthic flora and fauna in trawled areas (Auster and Langton, 1999; Auster et al., 1996; Olsgard et al., 2008; Sainsbury, 1998; Wells, 2007). Long-term trawling on the North West shelf of Australia was shown to remove up to 90% of large sponges (Sainsbury, 1998; Sainsbury et al., 1993). This removal of epibenthos also affects the associated fish community and Sainsbury et al. (1997) reported that over 20 years the catch rate of large sponges in trawls on the North West Shelf of Australia fell nearly 400 fold. This drop in catch rate of sponges was associated with a 4-5 fold decrease in catch rates of reef fish such as lutjanids and lethrinids (Burrige et al., 2006). Studies have shown that a single prawn trawl in the Australian Northern Prawn Fishery can remove around 5-20% of most classes of epibenthos (Burrige et al., 2003). Cumulative trawling over many years could clearly have dramatic impacts on entire ecosystems (Auster and Langton, 1999; Pitcher et al., 2000). While marine benthic communities on soft substrata are capable of coping with intermittent disturbances such as severe storms, they are less likely to recover from chronic long-term disturbance caused by bottom trawling (Burrige et al., 2006; McConnaughey et al., 2000; Tillin et al., 2006).

Demersal trawling, particularly trawling that targets prawns in tropical regions, can be a non-selective form of fishing (FAO, 2012). The potential impacts of discarding extremely high amounts of unwanted non-target organisms has been the focus of major political, scientific and conservation debates over the last 30 years (Alverson et al., 1994; Andrew and Pepperell, 1992; Brewer et al., 1998; Brewer et al., 2008; Broadhurst et al., 2012b; Broadhurst et al.,

2008; Kelleher, 2009; Poiner et al., 1998; Robins et al., 1999; Robins and McGilvray, 1999; Rochet et al., 2011). While many trawl fisheries may capture large quantities of unwanted species, none are as non-selective as certain demersal finfish and prawn trawls, which account for up to 50% of world fisheries discards (Alverson, 1997; FAO, 2012a; Kelleher, 2009).

Prawns not only provide a valuable source of income to countries and communities but are also a valuable source of protein (FAO, 2012). Prawn production (both wild caught and farmed) was valued at \$324.1 million to the Australian economy in 2009- 2010 (Beare et al., 2010). The shrimp and prawn industries continue to be the largest single marine fisheries trade commodity in value terms, accounting for around 15% of the total value of internationally traded fishery products in 2010 (FAO, 2012).

Unfortunately, the tropical prawn trawl fisheries are also the single most non-selective fisheries worldwide in terms of catching large quantities of non-target organisms and discarding them, mostly dead, back into the oceans (FAO, 2012). Typically the ratio of non-target to target catch rates is between 5:1 and 10:1, and tropical prawn trawling accounts for 27% of all global marine fishery discards by weight. In contrast, prawn trawling accounts for around 2% of global marine fishery production in terms of weight (FAO, 2012).

Non-target organisms captured during commercial fishing are collectively termed bycatch (Andrew and Pepperell, 1992). Bycatch is defined as the indiscriminate capture of all non-target organisms and non-living materials (debris) while fishing (Earlys, 2007). Discards are the unusable or unwanted part of the bycatch that is subsequently thrown back into the sea often dead or dying (Davies et al., 2009). One of the most significant conservation issues in

fisheries is the effect of removal of bycatch on marine communities and ecosystems (Davies et al., 2009). Estimating the quantity of global fisheries bycatch is often difficult and controversial (Alverson et al., 1994; Alverson, 1997; Davies et al., 2009; Kelleher 2009). Defining exactly what is bycatch can be difficult when there are ambiguities in what is target and what is not in many trawl fisheries (Davies et al., 2009). Even though traditionally all catch other than prawns has been described as bycatch, socio-economic factors, coupled with overfished marine stocks, particularly in developing countries, have led to a very broad usage of the term bycatch in many fisheries (Davies et al., 2009; FAO, 2012). A lack of, or inconsistent, documentation of removal of fish biomass around the world's fisheries (Froese et al., 2012; Zeller et al., 2007; Zeller et al., 2006; Zeller et al., 2005) makes it difficult to fully understand the extent of the impact that bycatch removal has on the environment and associated ecosystems. Davies et al., (2009) has put forward a new definition of bycatch and defines it as the catch that is either unused or unmanaged.

Discards, especially in the tropical prawn trawling industry, often suffer very high mortality rates (Hill and Wassenberg, 2000). Some of the bycatch species encountered during prawn trawling operations are threatened or endangered species such as turtles, certain shark species, dugongs, sea snakes, sea horses, corals and some fish species (Earlys, 2007). Hill and Wassenberg (2000) showed that in the Northern Prawn Fishery only 12% of total discards might survive. Their study also estimated that only around 2% of teleost fish survive after being discarded after trawling and that their chance of being eaten by marine predators was increased. Reduction of unwanted catch and the increased utilisation of bycatch have led to a significant decrease in discards over the last decade around the world (FAO, 2012; Zeller et al., 2005). This can mainly be attributed to the use of more selective fishing gears (especially BRDs), introduction of bycatch and discard regulations, and the improved enforcement of

regulatory measures (Brewer et al., 2008; Broadhurst et al., 2008; FAO, 2012a; Isaksen et al., 1992; Ocean-Watch, 2004).

In the last 20 years much of the focus of the bycatch problem has been on trying to minimize catch rates of non-target species inside trawl nets, by modifying the fishing gear and thus using more selective fishing methods. The most common modifications are TEDs that aim to eliminate the catch of marine megafauna (e.g. turtles, sharks, rays etc.) and BRDs that aim to reduce the catch of non-target teleost fish (Earys, 2007). While many scientific studies have investigated the effectiveness of different BRDs around the world, (Andrew and Pepperell, 1992; Brewer et al., 1998; Broadhurst et al., 2012b; Broadhurst et al., 2008; Glass, 2000; Robins-Troeger, 1994; Robins-Troeger et al., 1995; Robins and McGilvray, 1999) to date there have been no fishery-independent studies of the effectiveness of BRDs over the long term.

This thesis explores a long-term (1998-2012), fishery-independent, trawl sampling data set that used consistent methods over a 15 year period in a tropical bay in northern Australia.

The aims of this thesis are:

1. To quantify inter-annual variations in the performance of a BRD in a tropical bay where prawns are the target of the fishery and all teleost fish are considered bycatch.
2. To identify what environmental factors (e.g. rainfall, moon phase) may affect the trawl catch rates of both prawns and fish over a 15 year period of sampling.
3. To examine inter-annual variations in the species composition of the tropical fish assemblage and investigate if the BRD affects the species composition of the trawl fish catch.

Chapter 2

The effectiveness of the Jones-Davis bycatch reduction device (BRD) in a tropical fish assemblage, in Cleveland Bay, Townsville, Australia.

Introduction

Until recently the primary aim of management of harvest fisheries was to maximise sustainable yield of the target species (Brewer et al., 2006). The capture of non-target organisms by fishing gears has become a global issue in the past few decades, from the points of view of both sustainability of fisheries and conservation of marine ecosystems (Macbeth et al., 2005b). Prawn trawling fisheries not only capture marine megafauna of conservation interest (e.g. turtles, sharks, rays) but also have the highest ratio of bycatch to target species of any fishery (Alverson, 1997; Alverson et al., 1994; Courtney et al., 2006; Eayrs, 2007). Trawling can account for more than one-third of total global bycatch and discards (Courtney et al., 2006; Davies et al., 2009; Pascoe, 1997). The weight of the bycatch in many tropical trawl fisheries usually exceeds the weight of the target species (penaeid prawns) and can encompass hundreds of different species, mostly teleosts fish (Courtney et al., 2006; Gray et al., 1990; Kennelly et al., 1998; Steele et al., 2002; Stobutzki et al., 2001; Van der Geest, 2000; Watson et al., 1990; Ye et al., 2000). One of the most successful techniques to minimize bycatch of harvest fisheries is to improve the selectivity of fishing gear (Brewer et al., 1997; Macbeth et al., 2005a).

BRDs and TEDs are used in modern fisheries in order to reduce the catch of non target species (Brewer et al., 2006; Courtney et al., 2006; Hall and Mainprize, 2005a; Isaksen et al.,

1992). Originally BRDs were designed to exclude the catch of marine turtles (Tillman, 1992). Later BRD technology focused on excluding larger fish and elasmobranches, and more recently technological improvements have aimed at excluding smaller bycatch taxa such as small finfish, while retaining target species catch (Broadhurst et al., 2004; Macbeth et al., 2005b; Mitchell et al., 1995; Pichot et al., 2009; Richards and Hendrickson, 2006; Rogers et al., 1997).

In recent years much of the research on BRDs has focused on their physical design and upon behavioural factors of target and non-target species that affect the performance of BRDs (Fennessy and Isaksen, 2007; Gabr et al., 2007a, b; Graham et al., 2004; Kim and Whang, 2010; Macbeth et al., 2007; Pichot et al., 2009). BRDs that separate species by their behaviour in the net operate on the principle that fish, unlike weakly swimming invertebrates, have certain characteristic responses to towed trawls (Broadhurst, 2000). Depending on their swimming capability, fish will either avoid trawling gear all together or will be herded within the net (Wardle, 1993, 1983; Wardle and Bailey, 1987; Wardle et al., 1995). Depending on their swimming abilities, fish will eventually tire and fall back towards the cod-end (Broadhurst, 2000). The development of BRDs has focused on this herding effect in terms of choosing an appropriate place for fish escapement from the net (Broadhurst, 2000; Chapman, 1964; Wardle, 1993, 1989). Prawns on the other hand have limited responses to towed trawls. They are not capable of maintaining avoidance responses for more than a short period of time and are quickly forced against the meshes and towards the cod end (Broadhurst, 2000).

Much research has focused on altering water flow in parts of the cod end to facilitate fish escapement (Broadhurst et al., 2012a; Broadhurst et al., 2002; Engaas et al., 1999; O'Neill et al., 2005). BRDs are often placed in areas of reduced water flow, where fish have the chance to escape through openings in the net. This is particularly important in prawn fisheries that occur in turbid water, as visual stimuli might be less important in inducing behavioural

escape responses (Engaas et al., 1999). Reduced water flow is not only important in inducing behavioural escape responses but also in allowing an increased number of fish to be physiologically able to escape (Broadhurst, 2000; Broadhurst et al., 2002; Wardle, 1993, 1983, 1989). Differences in body shape, swimming ability, and myotomal muscle distribution all play important roles in the ability of fish species to escape the trawl net through a BRD (Sfakiotakis et al., 1999; Videler, 1993; Wardle and Bailey, 1987; Wardle et al., 1995).

Most pelagic and semi-pelagic fish have a high percentage of red muscle fibres, a carangiform or sub-carangiform swimming mode and a very streamlined body permitting endurance swimming at high speeds (Sfakiotakis et al., 1999; Wardle et al., 1995). Red muscle fibres are aerobic and allow fish to maintain high cruising speeds for prolonged periods (Wardle et al., 1995). Benthic and benthopelagic species have myotomes dominated by white muscle fibres which use anaerobic phosphorylation and produce lactic acid as a by-product (Videler, 1993). This build up of lactic acid reduces muscle contraction capacity and is characteristic of fish with weak sustained swimming ability (Videler, 1993).

No single BRD is effective in reducing the catch of all teleost bycatch species found in all fisheries (Gorman, 1997). The design and deployment of BRDs has to be matched to the particular fish assemblage and environmental conditions encountered in the fishery (Broadhurst et al., 2012b). Hence, most BRDs were designed for a particular fishery and often designs are altered when used in another part of the world (Broadhurst, 2000). Brewer et al. (1998) tested sixteen different BRD designs in a Northern Prawn Fishery. Each design provided a degree of bycatch reduction and prawn retention, but the performance of different designs was strongly affected by weather and fishing procedures (Brewer et al., 1997; Broadhurst, 2000). However, some BRDs specifically a combination of the fish eye (Broadhurst, 2000; Rogers et al., 1997; Watson and Taylor, 1996) and Nordmore grid (Broadhurst and Kennelly, 1996; Broadhurst et al., 1996; Richards and Hendrickson, 2006)

were effective in consistently reducing fish bycatch by 25% with minimal loss of prawns (Brewer et al., 1998). The fish eye BRD was also reported to be effective in the Caribbean Shrimp Fishery with bycatch reduction rates of 30.7% (Balmori-Ramírez et al., 2003). Bycatch weight reduction of 32% was reported in the New South Wales prawn fishery using the Morrison Soft TED (Andrew et al., 1993), and Robins-Troeger (1994) found that the same device reduced bycatch by 29% in Morten Bay, Queensland.

The Jones-Davis BRD has a hard TED (aluminium grid) to exclude large bodied bycatch (turtles, sharks, rays) and then uses a radial escape section and a fish scarer to initiate escape behaviour of strong-swimming teleost bycatch (Broadhurst et al., 2012a; Rogers et al., 1997; Watson et al., 1999). The radial escape section was originally based on a BRD component called a “finfish separator” (Watson et al., 1990). The performances of various designs of the radial escape section have varied significantly, with reductions in fish bycatch of between 20-40% by weight (Broadhurst et al., 2012a). The Jones-Davis device has had different levels of success in different places. Watson and Foster (1997) reported a 58% reduction in teleost bycatch when trawl nets were equipped with the Jones-Davis BRD in the Gulf of Mexico. Van der Geest (2000) reported a 19% reduction in teleost bycatch in Cleveland Bay northern Australia. Reductions in teleost bycatch of 43.9% and 33.5% using two modifications of the Jones-Davis BRD (a Double hoop Jones-Davis; a Modified Jones-Davis) were reported in the Gulf of Mexico and in the Caribbean of Colombia (Foster and Scott-Denton, 2004; Manjarres et al., 2008).

The testing of BRDs in different environments and under different conditions is important for the future development and optimization of these devices, so that optimal designs are used in each fishery (Salini et al., 2000). Environmental and physical aspects of the fishery, and physiological and behavioural aspects of the fish are important considerations for choosing the most effective BRD for a particular fishery. Efficient functioning of BRDs will reduce

bycatch, reduce fishing costs and duration of fishing and help minimize trawling impacts on the environment (Brewer et al., 1996; Foster and Vincent, 2010). The aim of this chapter was to investigate the effectiveness of the Jones-Davis BRD in the Cleveland Bay prawn trawl fishery, northern Australia. Data from 12 separate years of trawling over a 15 year period (1998-2012) with a BRD net and a control net will be analysed to examine the inter-annual variations in efficiency of the BRD.

Materials and Methods:

Site Description:

Cleveland Bay, located at $19^{\circ} 10' S$ and $146^{\circ} 50' E$ (Figure 2.1), is approximately 17km long, 25km wide and covers an area of approximately 300km^2 (Cruz-Motta and Collins, 2004; Sinclair, 1991). The bay is shallow with an average slope of 0.7m/km dropping to only 15m depth at its outermost limit (Hardy, 1991). Cleveland Bay is enclosed by Cape Cleveland to the southeast, the mainland to the southwest and Magnetic Island to the northwest. The bay is open to the north east and is the harbour for north Queensland's largest city, Townsville (Reichelt and Jones, 1994). The city has an important port industry where large volumes of mineral goods and other resources are transported. Access to the port is via a 13km long dredged sea channel maintained to a depth of approximately 11m, known as the Platypus channel (Hardy, 1991; Sinclair, 1997).



Figure 2.1: Location of the study site in Cleveland Bay ($19^{\circ} 10' S$ and $146^{\circ} 50' E$), Townsville, Australia

Cleveland Bay sediment is affected by terrigenous material contained in runoff from two major river systems, the Burdekin River and the Ross River (Carter et al., 1993; Orpin et al., 2004; Wolanski and Jones, 1981). The mean annual rainfall in Townsville is 1146mm (Australian Bureau of Meteorology). An average 80% of the annual precipitation occurs during the months from December to March (Walker, 1981b). Rivers and creeks rise and fall rapidly in response to heavy rains in the wet season, while during the dry winter many creeks cease to flow all-together (Walker, 1981a; Wolanski and Jones, 1981). Cleveland Bay receives direct runoff from the Ross River with a catchment area of 998km² and an approximate average annual water discharge of $0.49 \times 10^9 \text{m}^3$ and from Alligator Creek which has a catchment of 265km² (Lambrechts et al., 2010; Walker, 1981b). The Burdekin River, the second largest river system in Australia, is located 80 km south east of Cleveland Bay with an approximate average annual discharge of $9.8 \times 10^9 \text{m}^3$ (Walker, 1981b). The Burdekin River has been found to be an important contributor of fine sediment to Cleveland Bay (Furnas, 2003). During the wet season, increased river discharge lowers coastal salinity levels, increases dissolved nutrient concentrations and increases suspended sediment loads, leading to an increase in turbidity levels (Carter et al., 1993; Orpin et al., 2004; Wolanski and Jones, 1981). During the dry season from around May to October rainfall and river discharge are minimal while water temperatures remain relatively high leading to an increase in evaporation and elevated salinity levels (Walker, 1981a; Walker, 1981b; Wolanski and Jones, 1981).

The substratum composition is relatively homogenous throughout Cleveland Bay, consisting of mud and sandy mud (Carter et al., 1993). Bottom mud is stable, being resuspended only under rare strong swell and storm events. However its distribution has been greatly influenced by the dumping of dredged materials from Townsville harbour since the late 70's (Jing and Ridd, 1997; Lambrechts et al., 2010; Wolanski et al., 1992). There is a net transport

of fine sediment from the dredged material dump site at the outer north-east part of the bay to the more inshore waters leading to an increase of suspended solids and turbidity (Wolanski and Gibbs, 1992; Wolanski et al., 1992).

The epibenthic community in Cleveland Bay is relatively homogeneous and has been reported to be resilient to disturbances such as dredged material disposal (Cruz-Motta and Collins, 2004; Sondita, 1997; Watson et al., 1990). The soft bottom benthic macrofaunal community has been reported to be homogenous with slight variations in fish assemblage structures possibly related to depth (Van der Geest, 2000). The demersal community is diverse with 175 fish species recorded in the bay, with the fish assemblage dominated by Leiognathidae (Cabanban, 1991; Sondita, 1997; Van der Geest, 2000). The bay is also an important shark nursery for at least eight different species from two different families (Carcharhinidae and Sphyrnidae) (Simpfendorfer and Milward, 1993).

Sampling Design:

The study had two factors: Presence/ Absence of a Jones-Davis BRD, and time (Year of sampling). The first treatment involved fitting one of two trawl nets on the pair trawler James Kirby with a Jones-Davis by-catch reduction device. The other net was a control net. Both nets were standard nets used in the commercial tropical prawn trawling fishery around northern Australia. Mesh size was 4.5cm from knot to knot. Wooden otter boards 1.5m by 1m were used to open the trawl net laterally, the top of the mouth of the net had 3 floats, and the bottom of the mouth of the net had a tickler chain. Mesh size of the codend was 4.5cm and codend length was 100 meshes. Both trawl nets (BRD+ and BRD-) were set up in twin configuration and sampled at the same time, the net fitted with the BRD was sampling on the

port side and the control net was positioned on the starboard side of the vessel. The same nets and BRD were used for the entire study but repairs were made whenever deemed necessary. During trawling, nets were sampling at a distance of 15-25m apart from each other. Port and starboard biases were assumed to be insignificant in the current study and nets were assumed to be sampling the same population of fish independently of one another as the scale of schooling demersal fish is usually less than this distance (Van der Geest, 2000). The second treatment, time, involved sampling for 2-3 days in the first week of March for 12 of 15 consecutive years, from 1998 – 2012. In three of these years (2000, 2001 and 2005) sampling was not carried out due to technical problems or unavailability of the vessel. Therefore, data of only 12 years will be analysed here. All trawling occurred during daylight between 8:00am and 4:30pm. The number of days trawled each year, the number of trawls per year and the number of replicate groups of trawls in a year are listed in Table 2.1. Trawls were conducted in front of the mouth of Ross River between 19° 13' to 19° 16'S and 146° 49' to 146° 53'E at depth of 2.5-6.5m (Figure 2.1).

Year	Days Trawled	Total Number of trawls	Number of replicate groups of trawls
1998	2	16	4
1999	2	16	8
2002	2	20	6
2003	2	20	6
2004	2	16	5
2006	3	27	9
2007	2	24	7
2008	2	21	7
2009	2	24	6
2010	2	24	6
2011	2	20	6
2012	2	24	7

Table 2.1: Number of trawling days per year, total number of trawls per year and number of replicate groups of trawls per year (number of individual student trips spread over the number of trawling days in a year) from 1998-2012.

Field work was conducted from the RV James Kirby research vessel. The James Kirby is a modified twin otter trawler, 19.5m long and 5.2m wide. Trawl nets used during sampling were standard nets used in the commercial prawn fishery and had a 4.5cm diamond mesh. Trawls were conducted at haphazardly chosen locations just outside the mouth of Ross Creek. Each trawl lasted for ten minutes at a constant speed of 4.2 km/hour, with each net sweeping an area of 6000m² during this time unit (Cabanban, 1991; Van der Geest, 2000). Catch rates will be expressed as kg/ha in the remainder of the thesis. The trawl duration of ten minutes was chosen as it allowed for a greater number of replicate trawls to be completed in one day and has been shown to be sufficient to sample a representative component of the fish assemblage of the Cleveland Bay (Cabanban, 1991; Van der Geest, 2000). Scientific short term trawls have been shown to be representative in terms of sampling size structure and species composition of commercial long term trawls (Hannah and Jones, 2007; Wassenberg et al., 1998). However, it is acknowledged that short duration trawls may under sample some species of larger, faster swimming species of teleost fish.

After each trawl, catch was sorted on deck by students and Prof. Garry Russ. Catch was sorted into six different categories: Prawns (*Fenneropenaeus merguensis*); Carangidae; Leiognathidae; *Saurida* spp.; Sciaenidae and other fish (the latter category consisted of all other species of fish not included in the four fish categories listed). On trawls which landed a very high biomass, sub-samples were taken, and the total weight of different categories in the trawl was estimated by multiplying up to account for sub-sampling. Once the catch was sorted, wet weight biomass (to +/- 0.05kg) of each category was measured using butchers hook scales. Replicate trawl samples, once sorted and weighed, were then frozen for laboratory analysis.

The Jones-Davis By-Catch Reduction Device

The Jones-Davis BRD was designed in the South-eastern United States by Leroy Jones and Harry Davis. The device has both active and passive components that aim to reduce the catch of non-target species (Figure 2.2). The passive BRD is the TED. The TED uses a physical sorting method to release larger species of bycatch such as sharks, rays and turtles from the net. It consists of an aluminium grid that is placed approximately two thirds of the way into the trawl net and sits at an angle of approximately 45°. An escape opening is placed at the base of the grid allowing the bigger organisms to escape while target species are retained in the net.

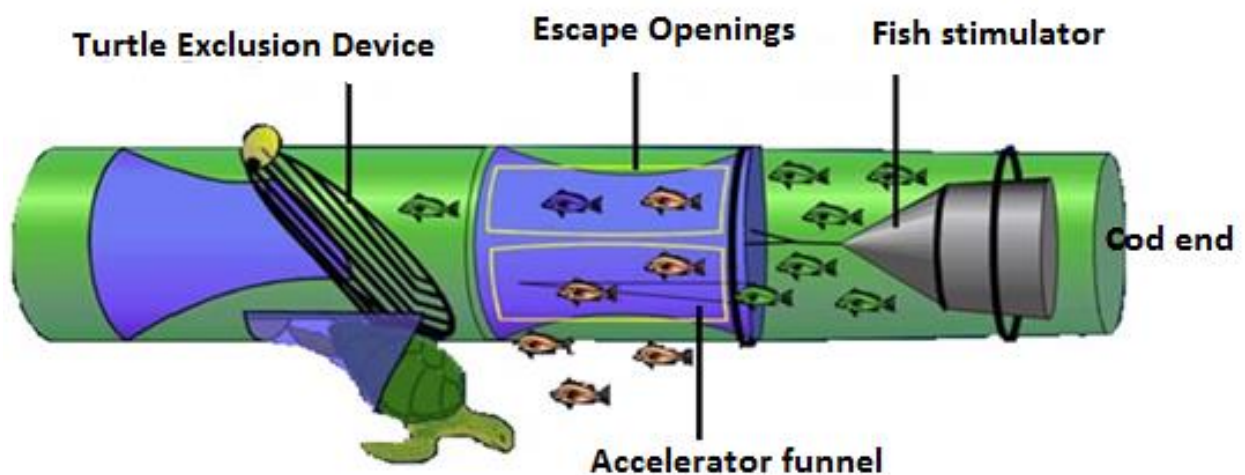


Figure 2.2: The Jones-Davis Bycatch Reduction Device.

The active component of the Jones-Davis BRD is designed to minimize the bycatch of smaller species, such as fish. It was originally designed to minimize capture of juvenile red snapper (*Lutjanus campechanus*) and mackerel in the South-eastern United States (Foster, 1999; Nance and Scott-Denton, 1997; Watson et al., 1999). The device aimed at utilizing the superior swimming capability of red snapper and other demersal fish species, compared to prawns, to enhance escapement from the trawl nets (Watson et al., 1999). It was designed to

actively scare the fish and use their natural predatory avoidance response to escape the net, and also to create pockets of reduced water flow velocity to facilitate this escapement (Foster, 1999). Three main components of the Jones-Davis BRD were used to achieve this. Firstly an accelerator funnel, which concentrates the catch into a smaller area just forward of the fish stimulator (Figure 2.2). The funnel opens onto the fish stimulator which provides a visual and physical stimulus to trigger a predatory avoidance response that overrides the optomotor response (stabilization during free locomotion through an involuntary displacement from a straight course) exhibited by some fish species (Arnold, 1974; Van der Geest, 2000; Wastson and Foster, 1997). The trawl net surrounding the funnel has escape openings placed longitudinally along the trawl net, that allow strong swimming fish to swim forward against the direction of water flow, around the accelerator funnel, and either up or down to exit the net via the escape openings (Figure 2.2). In contrast to some other BRD designs, the escape openings used in the Jones-Davis BRD have no mesh on them which reduces injury to the escaping fish (Gabr et al., 2007a; Van der Geest, 2000).

Laboratory Analysis:

All prawn and fish bycatch were identified to at least the level of genus, and usually to species, in the laboratory.

Effectiveness of the Jones-Davis BRD

The effectiveness of the Jones-Davis BRD was tested using a factorial ANOVA with year and presence/absence of the BRD as the fixed two factors and total wet weight biomass (kg/ha) the dependant variable. This enabled the testing of year and BRD effects on biomass, and interaction between these two factors. Statistical tests were conducted at 95% confidence

using Statistica 10. Biomass data from all 12 years were used to analyse trends in catch rates of Carangidae, Leiognathidae, Sciaenidae, Prawns, *Saurida* spp., “total” fish and “other” fish catch.

Data transformations

Data from three or four consecutive trawls in a day were pooled to minimize the occurrence of zero values in the data set and overcome occasional high variance from trawl to trawl for some variates, particularly *Saurida* spp., Carangidae and Leiognathidae catch rates. This number of trawls was chosen as each within day trip would usually conduct 3 to 4 trawls (Table 2.1). Thus, the same data recorder would record the data from three or four consecutive trawls. Ten minute trawls were pooled to reduce the high small-scale spatial and temporal variability in catch rate that occurred, mostly caused by the high frequency of schooling fish in the fish assemblage. The pooling of trawls, whilst not optimal, was a necessary compromise in making a sampling unit chosen for a short-term teaching exercise and using it in a long-term research project.

Bias that could have resulted from changing environmental conditions between trawls was also assumed to be negligible as three to four trawls were carried out in less than two hours. Data were then natural log transformed in order to pass various statistical assumptions of the ANOVAs: homogeneity of variances and normal distribution of residuals. Outlier data points were removed in some years for four of the dependant variables (Total Fish, Other Fish, Sciaenidae and *Saurida* spp.), in order to pass the assumptions of homogeneity of variances. These outliers were due to extremely high small-scale spatial variations in fish catch for trawls in 2011 for “total fish”, “other fish” and Sciaenidae, and the high numbers of zero values in the “*Saurida* spp.” data set, as well as one very high catch of Sciaenidae in 2012.

Results

BRD effectiveness

Catch of Prawns

Total prawn catch increased by an average of 10% (+/- SE 5.05) in the BRD net over the 12 years measured (Figure 2.5). This increase was greatest in 2010 and 1999 with 41.4% and 40.1% increase in catch in the BRD net respectively, and lowest in 2004 with a decrease in prawn catch of 10.9% (Figure 2.3). This increase in catch in the BRD net was not significant ($p>0.05$, ANOVA) over the 12 years measured (Table 2.2). Prawn catch was highly variable over time ranging from 0.04kg/ha (+/-SE 0.01) in 2010 to 3.96kg/ha (+/-SE 0.0.53) in 2008 in the BRD net, and from 0.03kg/ha (+/-SE 0.008) to 4.05kg/ha (+/-SE 0.62) in the control net in the same years respectively (Figure 2.4). Average prawn catch was 1.32kg/ha (+/-SE 0.40) in the BRD net and 1.25kg/6000m² (+/-SE 0.39) in the control. The prawn catch contributed between 0.4% and 33.7% of the total catch by weight, with an average of 10% (+/-SE 3.27) in the BRD net, and between 0.2% and 30.3% with an average of 7.9% (+/-SE 2.77) in the control net (Figures 2.18, 2.19).

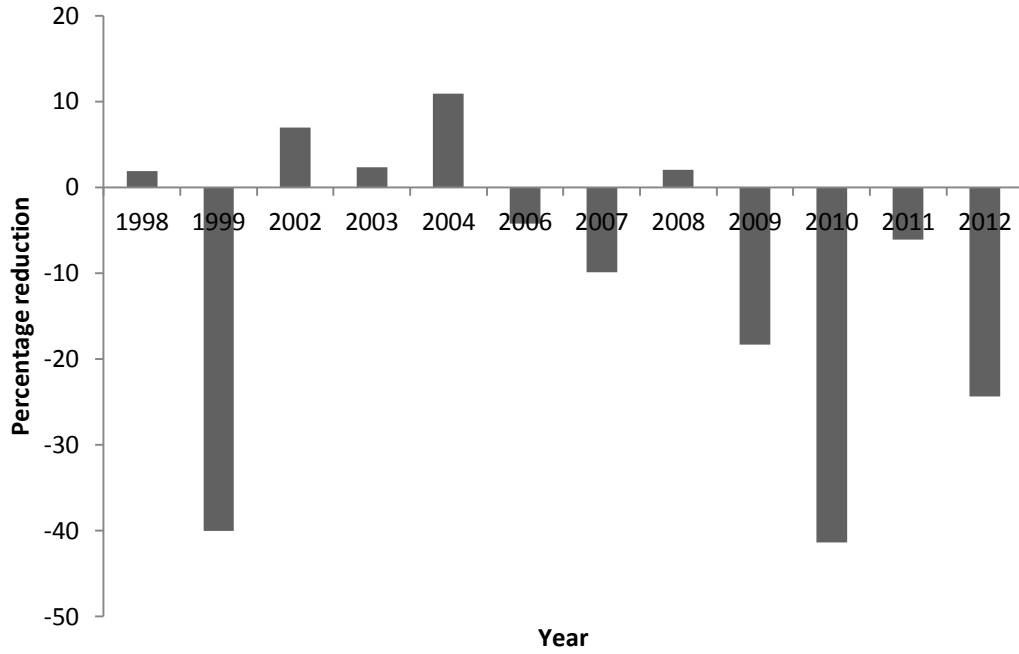


Figure 2.3: Percentage difference in catch of Prawns for the BRD and Control trawl nets for from 1998- 2012.

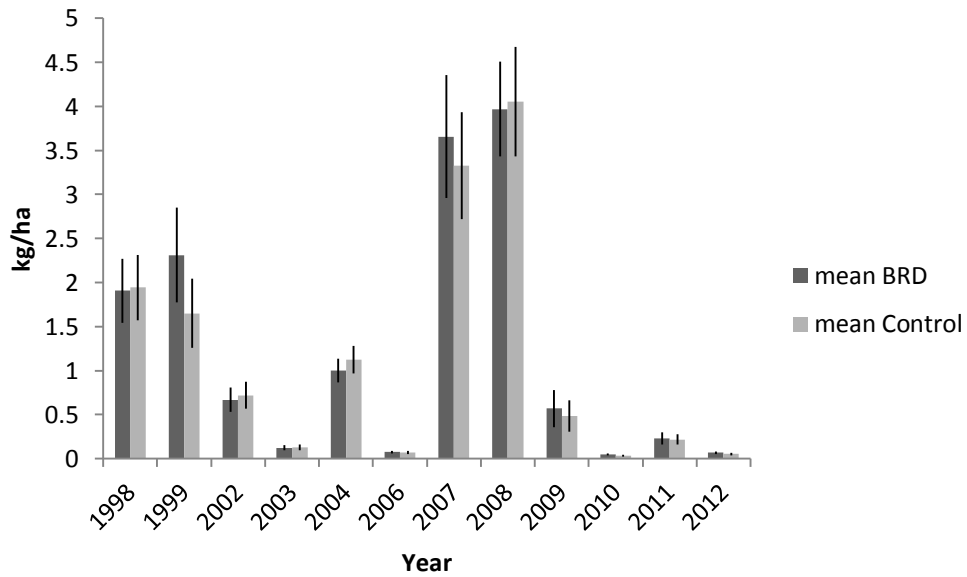


Figure 2.4: Mean (+/- SE) catch of Prawns per hectare in the BRD and control net from 1998-2012.

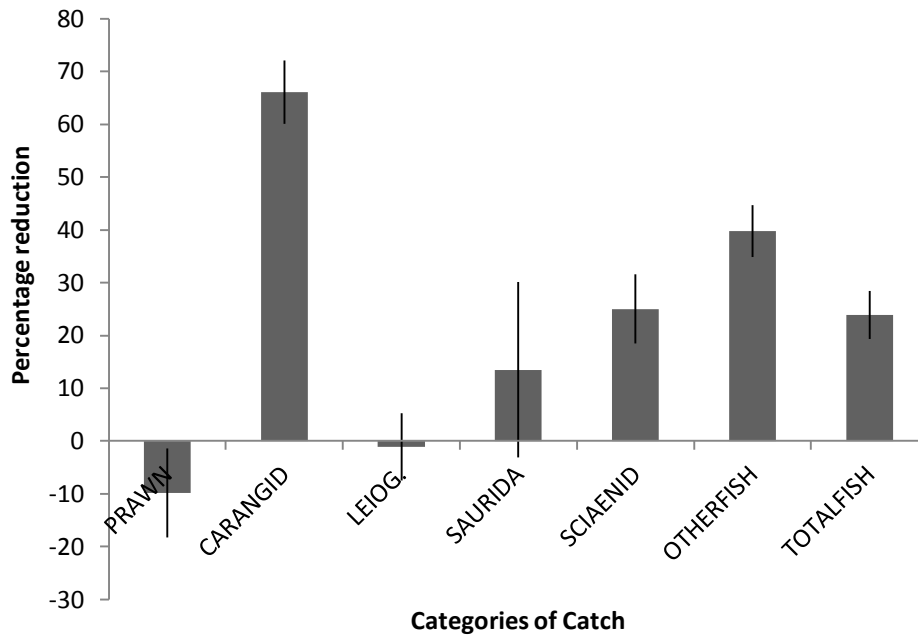


Figure 2.5: Mean (+/- SE) percentage reduction of major categories of catch in the BRD net (relative to the control net) from 1998-2012.

	BRD	Year	BRD * Year	Effect
Prawn	0.653	<0.001	0.999	+ 10%
Total Fish	0.000	<0.001	0.563	- 23.7%
Carangidae	<0.001	0.001	0.728	- 65.9%
Sciaenidae	0.005	<0.001	0.381	- 24.7%
<i>Sauridaspp.</i>	0.447	<0.001	0.980	- 12.6%
Leiognathidae	0.774	<0.001	0.981	+ 1.5%
Other Fish	<0.001	<0.001	0.873	- 39.5%

Table 2.2: Univariate ANOVAs testing effects of BRD and Year of Sampling for all variates and direction of change of variates due to the BRD (as %). The denominator of all F values is 129. For more detailed ANOVA results see Appendix 1, Tables 1.1-1.8.

Total fish catch

The Jones-Davis BRD reduced total fish bycatch by an average of 23.73% (SE+/- 4.9) measured for 12 separate years in the 15-year (1998-2012) study period (Figure 2.5). This reduction was largest in 2012 (52.8%), and smallest (0.3%) in 2006 (Figure 2.6). This reduction over the 12 years of measurements was significant ($p < 0.001$, ANOVA, Table 2.2). Total fish catch was highly variable ranging from a mean of 7.71kg/ha (SE+/- 0.7) in 2003 to

30.82kg/ha (SE+/- 7.55) in 2011 in the control net, and 7.43kg/ha (SE+/- 7.03) in 2012 and 17.17kg/ha (SE+/- 3.12) in 2011 in the BRD net (Figure 2.7).

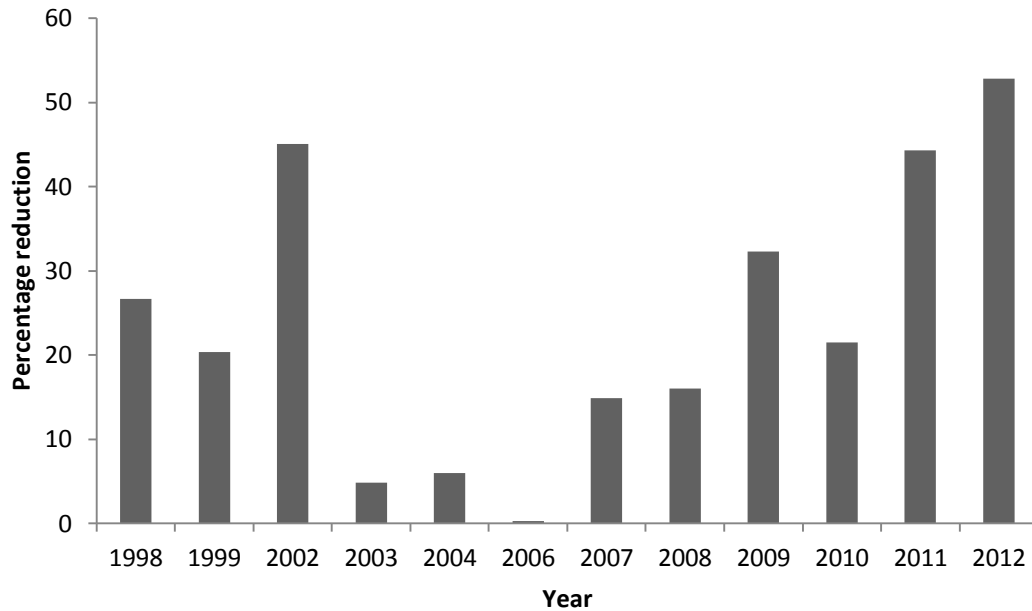


Figure 2.6: Percentage difference of BRD and Control trawl nets for total fish catch from 1998- 2012.

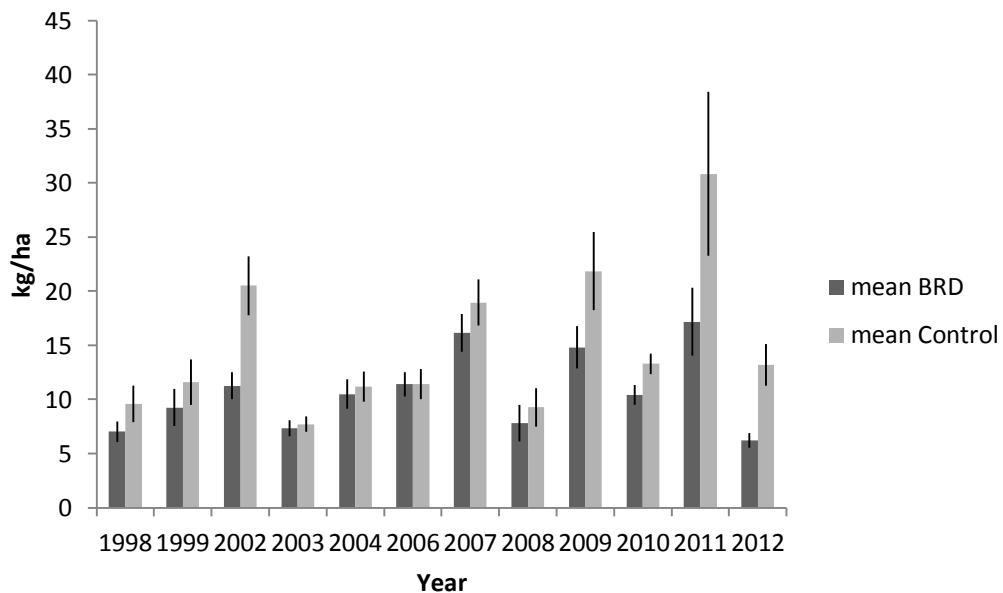


Figure 2.7: Mean (+/- SE) Total fish catch per hectare in the BRD and control net from 1998- 2012.

Catch of Carangidae

Total catch of Carangidae (Trevallies or Jacks) was reduced by an average of 65.9% (SE \pm 3.48) over the 12 years measured (Figure 2.5). This reduction was largest in 1998 and 2002 with 78.8% and 78.9% respectively, and smallest in 2009 with 45% less fish caught in the BRD net (Figure 2.8). This reduction was highly significant ($p < 0.001$, ANOVA, Table 2.2) over the 12 years of measurements. Catch of Carangidae ranged from a mean of 0.17kg/ha (SE \pm 0.043) in 2010 to a mean of 0.07 kg/ha (SE \pm 0.05) in 1998 in the control net, and a mean of 0.07 kg/ha (SE \pm 0.06) in 2008 to a mean of 0.29kg/ha (SE \pm 0.05) in 2009 in the BRD net (Figure 2.9). Catch of Carangidae contributed between 0.5% and 2% to the total fish biomass caught in the BRD net, and between 0.9% and 6.1% in the control net, with an average of 1.25% in the BRD net and 3.2% in the control net (Figures 2.18, 2.19).

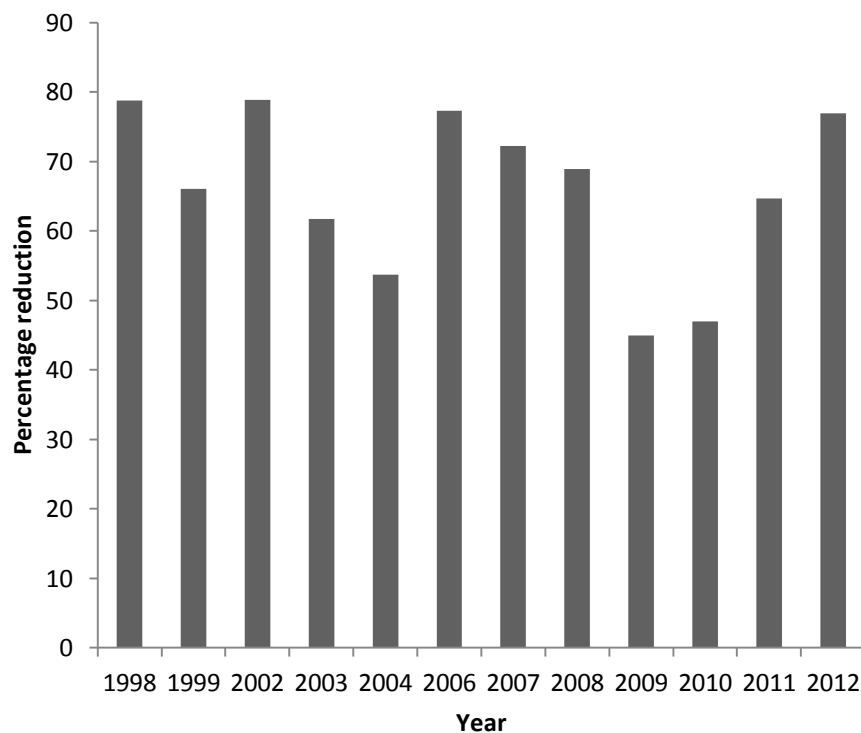


Figure 2.8: Percentage difference in catch of Carangidae between the BRD and Control trawl nets from 1998- 2012.

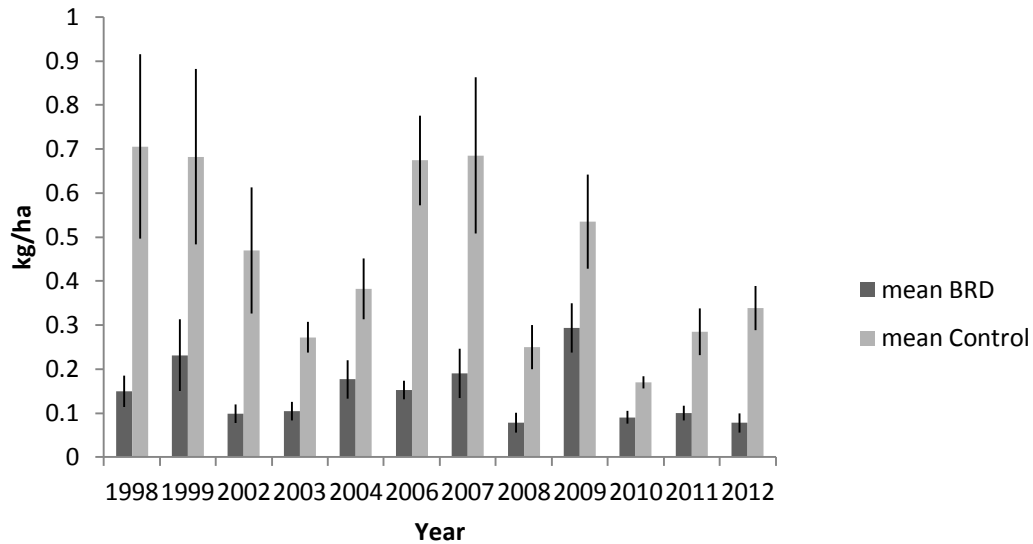


Figure 2.9: Mean (\pm SE) catch of Carangidae per hectare in the BRD net and the control net from 1998- 2012.

Catch of Sciaenidae

Total catch of Sciaenidae (croakers or jewfish) was reduced by an average of 24.6% (\pm SE 6.82) over the 12 years measured (Figure 2.5). This reduction was largest in 2010 with 56.5% and smallest in 1998 with 14.1% less Sciaenidae in the BRD net (Figure 2.10), although the catch of this family was very small in 2010. The reduction of 24.6% over the study period was significant ($p < 0.05$, ANOVA, Table 2.2). The catch of Sciaenidae ranged from 0.013kg/ha (\pm SE 0.01) in 2010 to 1.69kg/ha (\pm SE 0.89) in 2012 in the BRD fitted net, and from 0.03kg/ha (SE \pm 0.01) to 3.31kg/ha (SE \pm 1.47) in the same years, respectively, in the control net (Figure 2.11). Catch of Sciaenidae contributed between 0.13% and 23.74% to the total fish biomass with a mean of 5.8% (\pm SE 1.86) in the BRD fitted net, and between 0.23% and 22.8% in the control net, with a mean of 5.61% (\pm SE 1.75) (Figure 2.18, 2.19).

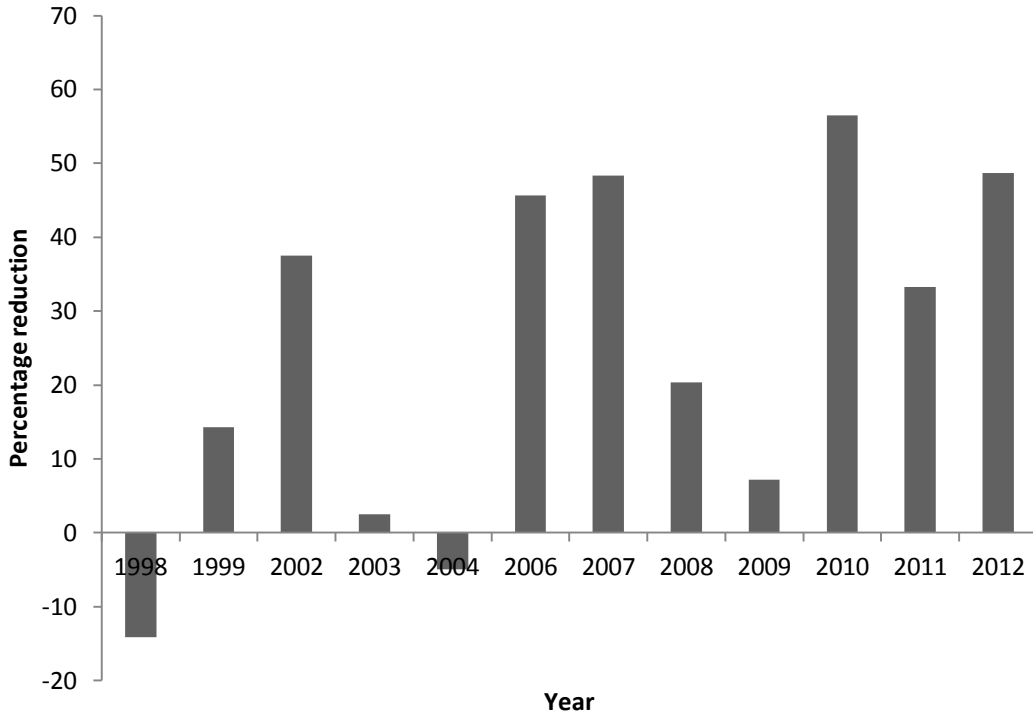


Figure 2.10: Percentage difference in catch of Sciaenidae for the BRD and Control trawl nets from 1998- 2012.

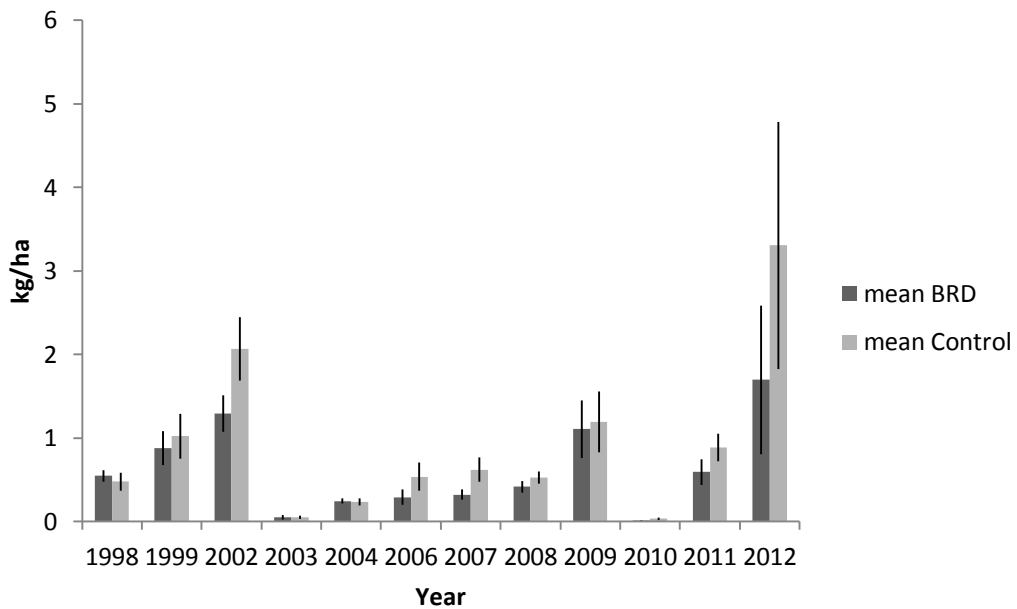


Figure 2.11: Mean (+/- SE) catch of Sciaenidae per hectare in the BRD and control net from 1998- 2012.

Catch of *Saurida* spp.

Total catch of *Saurida* spp. decreased by an average of 12.6% (\pm SE 5.96) in the BRD net over the 12 years measured (Figure 2.5). This reduction was largest in 2003 with 38.2% less fish caught, and smallest in 2006 with an increase in catch of 32% in the BRD net (Figure 2.12). This overall mean reduction of 12.6% was not significant ($p > 0.05$, ANOVA, Table 2.2) over the 15 year study period. *Saurida* spp. catch ranged from 0.01kg/ha (\pm SE 0.007) in 2012 to 0.65kg/ha (\pm SE 0.09) in 2007 in the BRD net, and from 0.01kg/ha (\pm SE 0.01) to 0.88kg/ha (\pm SE 0.11) in control net, in the same years respectively (Figure 2.13). Mean *Saurida* spp. catch was 0.28kg/ha (\pm SE 0.05) in the BRD net and 0.33kg/ha (\pm SE 0.07) in the control. The catch contributed between 0.14% and 4.1% of the total catch with an average of 2.27% (\pm SE 0.36) in the BRD equipped net, and between 0.09% and 4.4% with an average of 2.12% (\pm SE 0.39) in the control net (Figures 2.18, 2.19).

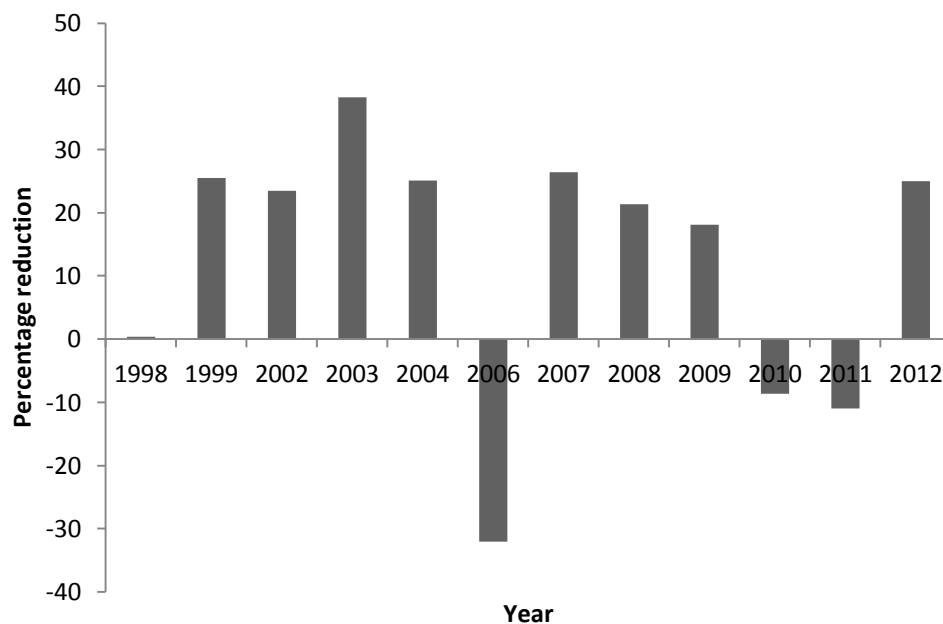


Figure 2.12: Percentage difference in catch of *Saurida* spp. for the BRD and Control trawl nets from 1998- 2012.

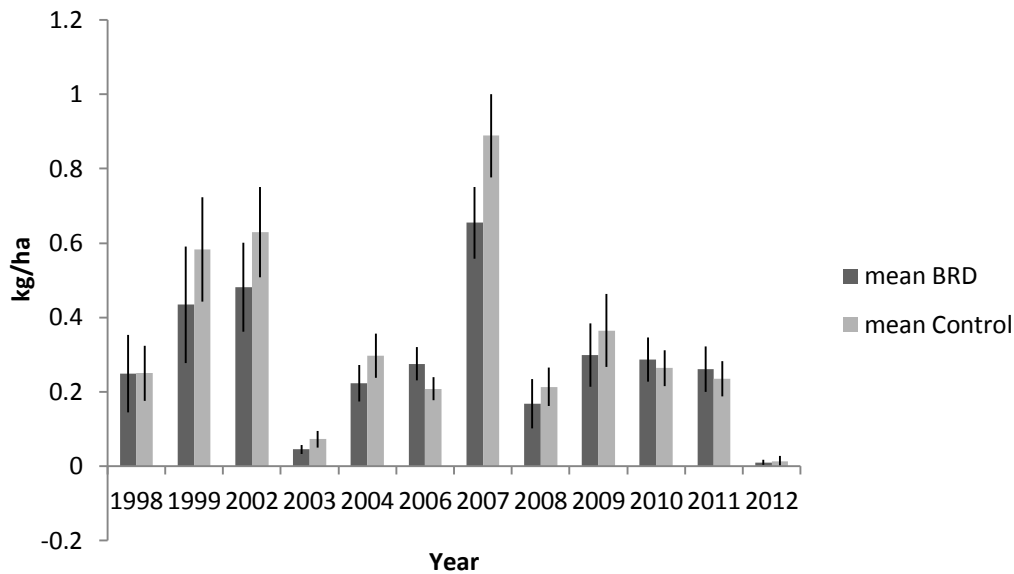


Figure 2.13: Mean (\pm SE) catch of *Saurida* spp. per hectare in the BRD and control nets from 1998-2012.

Catch of Leiognathidae

The catch of Leiognathidae (Ponyfish) increased by an average of 1.5% (\pm SE 7.02) in the BRD net over the 12 years measured (Figure 2.5). The effect of the BRD on the catch of Leiognathidae ranged from a reduction of 32.8% in 1998 to 58.8% increase in 2006 (Figure 2.14). There was no statistically significant difference in leiognathid catch between the two nets ($p > 0.05$, ANOVA, Table 2.2) over the 12 years studied. Mean catch of Leiognathidae ranged from 1.01kg/ha (\pm SE 0.16) in 2012 to 7.35kg/ha (\pm SE 1.3) in 2004 in the BRD net, and from 1.12kg/ha (\pm SE 0.2) in 2012 to 7.45kg/ha (\pm SE 1.1) in 2007 in the control net. Mean catch of Leiognathidae was 5.18kg/ha (\pm SE 0.61) in the BRD net and 5kg/ha (\pm SE 0.5) in the control (Figure 2.15). The catch contributed between 14.3% and 73.8% to the total catch with an average of 42.3% (\pm SE 5.9) in the BRD net and, and between 7.8% and 55.1% with an average of 33.4% (\pm SE 4.61) in the control net (Figure 2.18, 2.19).

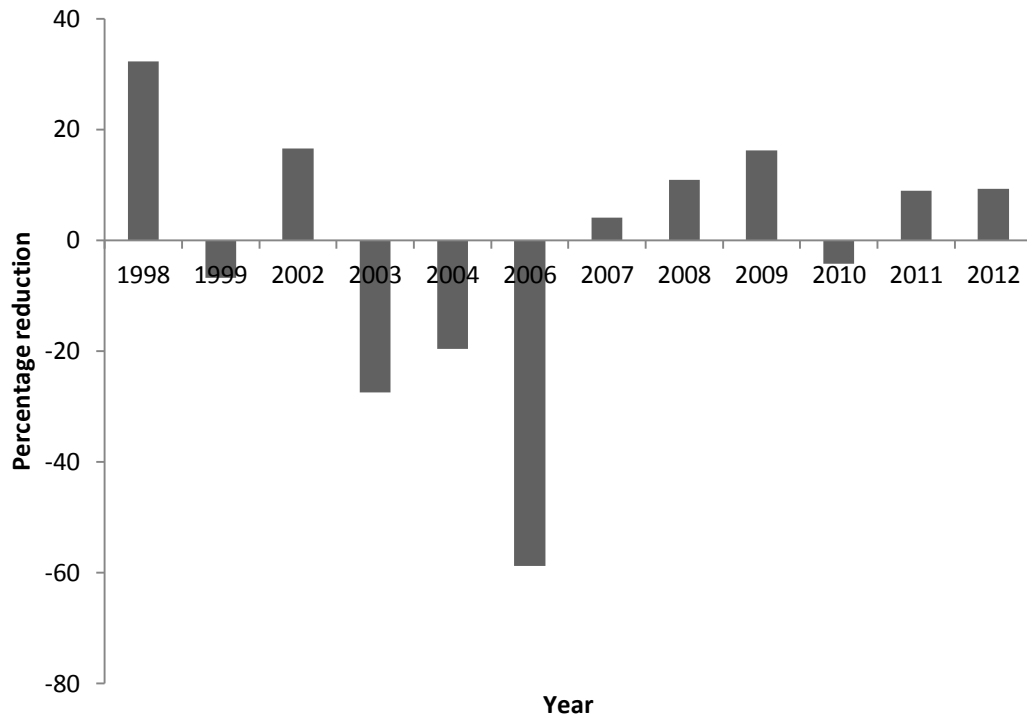


Figure 2.14: Percentage difference in catch of Leiognathidae for the BRD and Control trawl nets from 1998- 2012.

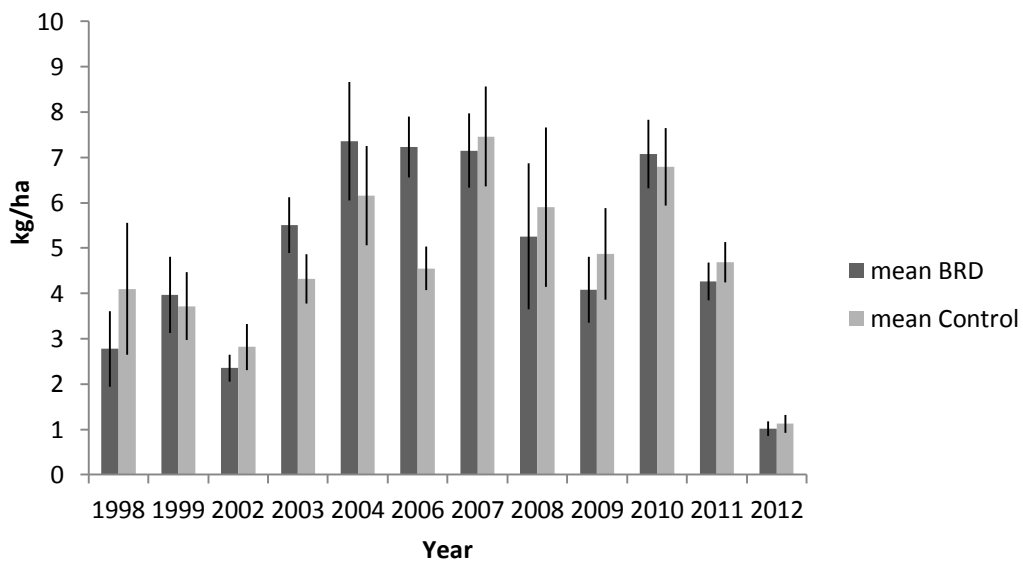


Figure 2.15: Mean (+/- SE) catch of Leiognathidae per hectare in the BRD and the control net from 1998- 2012

Catch of Other fish

Catch of the “Other fish” component was decreased by an average of 39.5% (\pm SE 3.51) in the BRD net over the 12 years measured (Figure 2.5). This decrease was largest in 2012, with a decrease of 55.5%, and smallest in 1998 with 18.2% less fish caught in the BRD net (Figure 2.16). This decrease in catch was highly significant ($p < 0.0001$, ANOVA, Table 2.2) over the 12 years measured. Catch of “Other fish” ranged from 1.88kg/ha (\pm SE 0.12) in 2008 to 11.95kg/ha (\pm SE 3.07) in 2011 in the BRD net, and 2.39kg/ha (\pm SE 0.18) to 24.74kg/ha (\pm SE 7.33) in the control net, in the same years respectively (Figure 2.17). Average catch of “other fish” was 4.68kg/ha (\pm SE 0.9) in the BRD net and 8.24kg/ha (\pm SE 1.9) in the control net. The catch of “other fish” contributed between 15.9% and 68.7% of the total fish catch with an average of 38.2% (\pm SE 5.25) in the BRD net, and between 17.9% and 79.7% with an average of 47.6% (\pm SE 5.36) in the control net (Figure 2.18, 2.19).

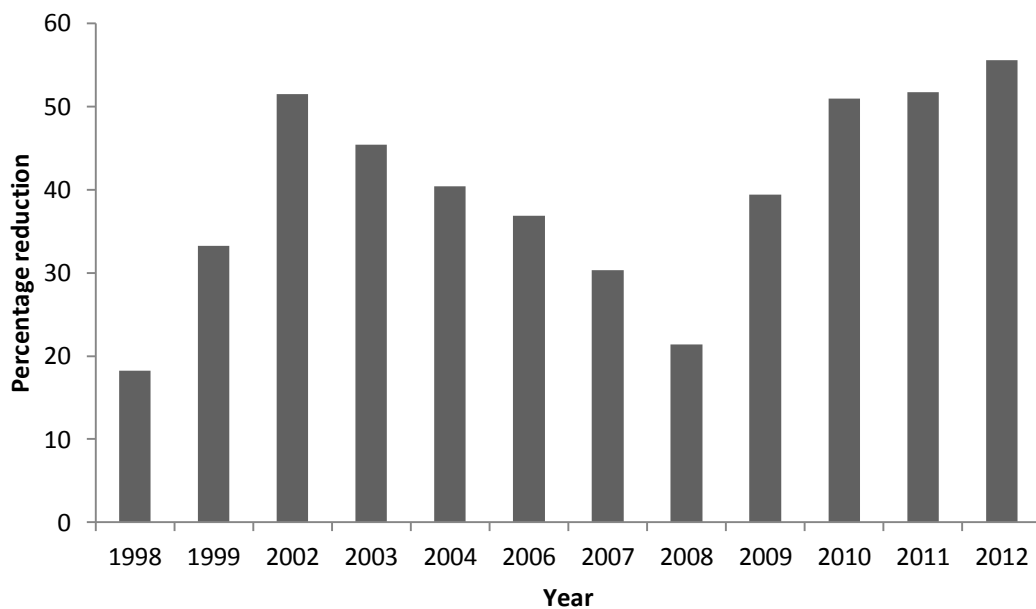


Figure 2.16: Percentage difference in catch of “Other fish” for the BRD and Control trawl nets from 1998- 2012.

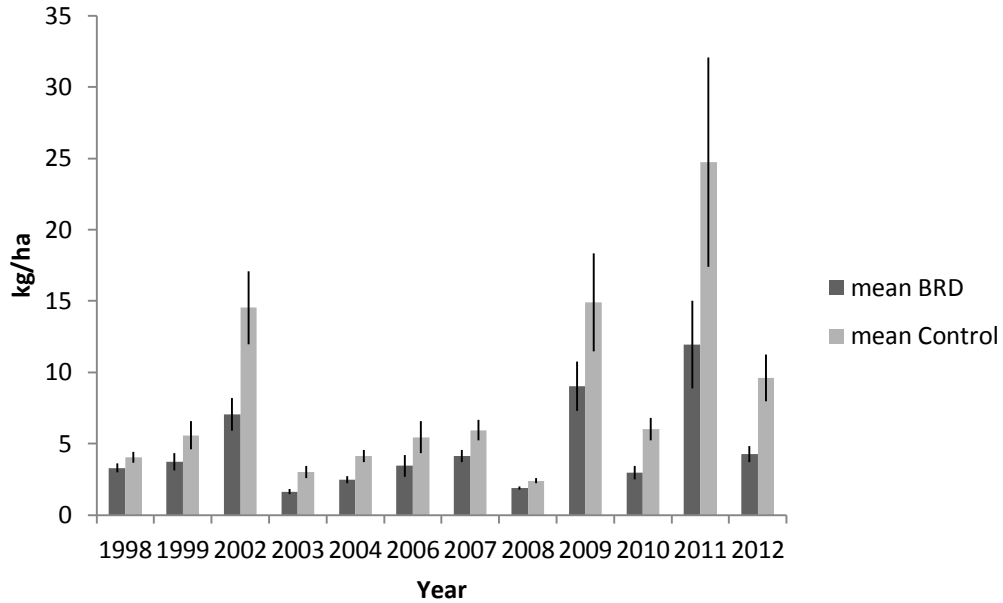


Figure 2.17: Mean (+/- SE) catch of “Other fish” per hectare in the BRD and control net from 1998-2012.

The Interaction between the BRD and Year Factors

There were no significant interactions between the effect of the BRD and year of sampling for any variates ($p > 0.05$, Table 2.2). This indicates that the BRD effect did not vary significantly with year of sampling, despite considerable variation in the performance of the BRD among years (e.g. Figures 2.6, 2.7).



Figure 2.18: Catch composition (as % by weight) of the six major groups in the BRD equipped trawl net.

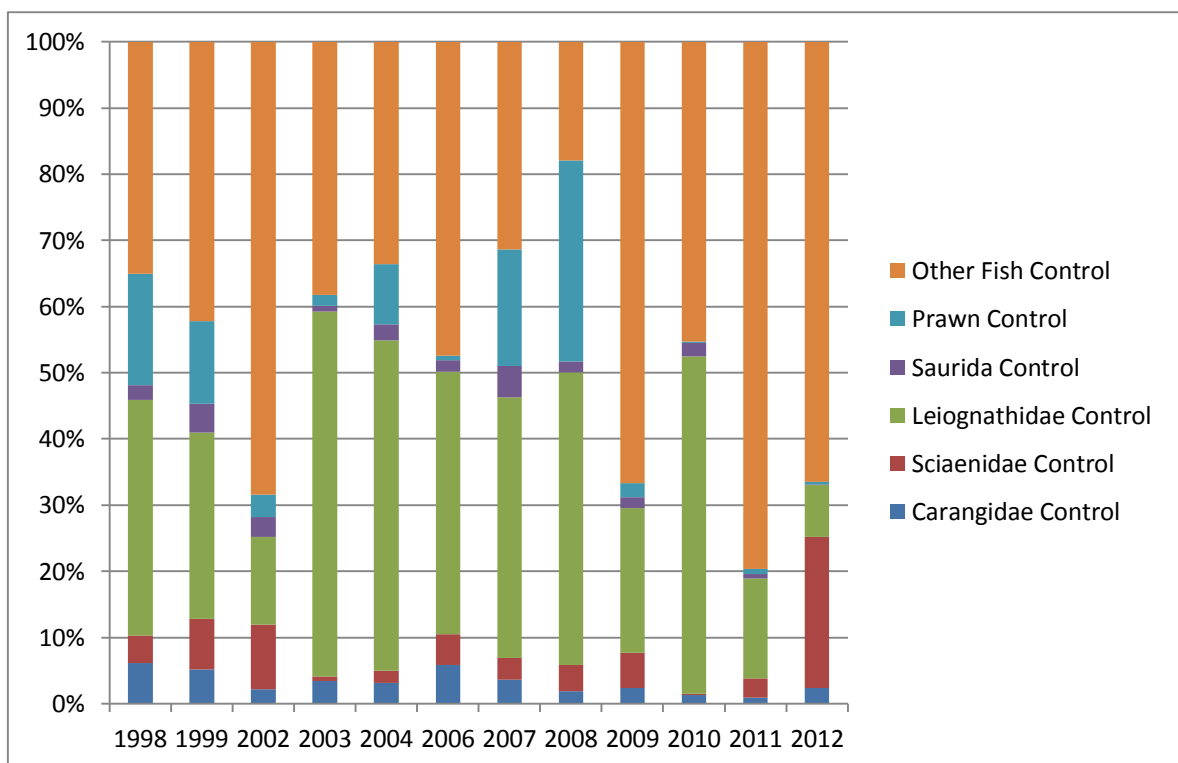


Figure 2.19: Catch composition (as % by weight) of the six major groups in the control trawl net.

Discussion:

There was no significant difference in prawn catch rate between the nets with and without the Jones-Davis BRD, averaged across 12 years of measurement in the 15 year study. This is an important result, since it demonstrates that the BRD has no significant effect on catch of the target of the fishery. Thus, stakeholders in the industry are far more likely to accept and use BRDs. Some studies have actually recorded increased prawn catches in BRD-fitted nets (Broadhurst and Kennelly, 1997). Van der Geest (2000) found a statistically significant increase in target species catch (banana prawns) of 18% using the Jones-Davis BRD in Cleveland Bay over three years. Brewer et al. (1998) suggested that weather conditions (such as rough seas) have a significant effect on prawn retention rate during a study assessing several different BRDs, including the radial escape section which is part of the Jones-Davis BRD. Weather conditions were not recorded during our study but it is likely that changes in oceanic conditions, mainly wave action, played a role in BRD performance. Wave records over time could not be obtained for Cleveland Bay as monitoring systems were affected by cyclone activity in the region. Trawling speed was kept constant at 4.2 km/hr in the present study.

Broadhurst and Kennelly (1997) found a significant increase in catches of the prawn *Penaeus plebejus* (5.5% to 14%) when using a composite-panel netting BRD in the New South Wales prawn fishery. Brewer et al. (1998) found an increase in target species catch rate using several different BRD designs (Super Shooter, Radial Escape Section, Fisheye) in the Gulf of Carpentaria. Fennessy and Isaksen, (2010) reported a 3% increase in prawn catch using square mesh panels in the Indian Ocean off Mozambique. One hypothesis to explain increases in target species catch in BRD equipped nets are that the reduction of total fish bycatch biomass maintains a wider wing spread of the net, effectively increasing the swept

area of the net (Brewer et al., 1997). Interannual variations in prawn catch were considerable in this study (Figure 2.3). This temporal variation will be addressed in the next chapter.

The Jones-Davis BRD significantly reduced total fish catch rate by 23.7% (+/- SE 4.9) in the trawl samples made in Cleveland Bay, Australia. There was clear evidence that the Jones-Davis BRD selectively removed bigger, stronger swimming fish from the trawl net (e.g. carangids, "Other Fish") but was far less successful in removing smaller, weak swimming fish (e.g. leiognathids, *Saurida* spp.). Pelagic and semi pelagic species such as carangids gain the most benefit from the BRD, in fact the majority of carangids were able to escape the trawl nets, with mean catch rate reductions in the BRD equipped net of 65.9% (SE +/- 3.48). Benthic, weaker-swimming species such as the leiognathids and *Saurida* spp. did not benefit from the presence of the BRD in the trawl net. Results suggest that strong swimming fish taxa are the ones that benefited most by the inclusion of the Jones-Davis BRD.

Differences in weather and environmental conditions especially wave height and frequencies have been reported to alter BRD performance (Broadhurst, 2000; Crawford et al., 2011). Broadhurst et al. (2012) found that otter trawls can have large variations in wing spread during deployment. Differences in deployment of the trawls between years could explain some of the variations in BRD performance. Other potential influences on interannual variations in catch rates will be examined in chapter three.

BRD performance was analysed only at the family level. Species level catch rate data of bycatch were considered beyond the scope of this project. Catch rates were expressed in kg/ha, with all weights determined on board, immediately following each trawl. There was no time, neither on board, nor in the laboratory, to sort the bycatch to species level and then weigh each species.

Catch rate of carangids was reduced most by the Jones-Davis BRD. Carangids are a fast swimming fish, having a carangiform swimming mode and a streamlined, laterally compressed body shape (Gemballa and Treiber, 2003; Sfakiotakis et al., 1999). They have a high percentage of red, aerobic, slow twitch muscle enabling them to maintain high cruising speeds for a prolonged period of time, but can also use their anaerobic white muscles for fast burst swimming (Gemballa and Treiber, 2003; Wardle et al., 1995). While lactic acid build-up in white muscle hinders continuous swimming (Videler, 1993), the physiology of carangids enables them to maintain their cruising speed and accelerate to high burst speeds, increasing their chances of reacting to the fish-stimulator, swimming forward in the net into the current, and finding their way out of the net via the escape openings (Figure 2.2).

The catch rate of sciaenids was reduced by nearly 25% in this study, with reductions of 33% or more in 6 of the 12 years studied (Figure 2.10). This reduction was statistically significant over the 12 years measured. Results suggest that the Jones-Davis BRD could be an effective tool in reducing sciaenid bycatch. Sciaenids are strong swimming, demersal predators that are capable of high burst speeds in order to catch their prey (Wardle et al., 1995). This would have allowed a significant percentage to react to the fish stimulator and escape via the escape openings in the BRD.

Being small benthic ambush predators, *Saurida* spp. are adapted for short-burst swimming and are physiologically incapable of sustaining high cruising speeds over prolonged periods of time (Sfakiotakis et al., 1999; Wardle et al., 1995). This makes them extremely vulnerable to bottom trawling and hence they are a common bycatch in tropical prawn trawl fisheries (Blaber et al., 1990; Kennelly, 1995; Van der Geest, 2000). Even though the efficiency of the

Jones-Davis BRD in removing this family was highly variable, there was no significant overall reduction in catch rate of *Saurida* spp. in this study.

The Family Leiognathidae was a dominant component of trawl catch in most years in Cleveland Bay (Figures 2.18, 2.19). Leiognathids have physiological adaptations specific to their demersal life style, making them extremely vulnerable to capture by prawn trawling (Staunton-Smith et al., 1999; Wootton, 1990). They are weak swimmers and do not have the musculature required for continuous, fast cruising or burst swimming like carangids or scombrids do (Sepulveda and Dickson, 2000; Sfakiotakis et al., 1999; Wardle et al., 1995). Thus, leiognathids are unable to swim against strong currents and thus are unlikely to be able to react to the fish stimulator and swim forward in the Jones-Davis BRD to escape via the escape openings (Figure 2.2). Van der Geest (2000) reported no significant reduction in leiognathid catch in the Jones-Davis BRD in a three-year study in Cleveland Bay

The remainder of the fish assemblage (termed “other fish”) was reduced significantly in all years and over the 12 years of measurement. This very heterogeneous group, consisting of over 100 species of teleost fish (Van der Geest, 2000) contains some species with large body size and good swimming ability (e.g Pomadasyidae, Teraponidae). Some of these species with large bodied individuals would have had a substantial influence on trends (BRD +/-, annual) in this group. Such groups are capable of sustained swimming and are likely to react to the fish stimulator and be able to escape via the escape openings in the Jones-Davis BRD. This “other fish” result shows that the BRD is effective at reducing bycatch of certain fish species encountered in the tropical fish assemblage in Cleveland Bay.

This study indicates that the Jones-Davis BRD preferentially excludes strong-swimming fish species and operates relatively consistently over time. The Jones-Davis BRD was originally designed to exclude the bycatch of red snapper in the Gulf of Mexico, where this BRD reduced total fish biomass by 58% in the shrimp fishery there (Wastson and Foster, 1997). A three year study by Van der Geest (2000) found a 19% reduction of fish bycatch over a three year period in Cleveland Bay, a result similar to the present study which reduced bycatch by nearly 24% for 12 separate years over a 15 year period. The data from two of the three years of Van der Geest (2000) study are included in the current study. These results suggest that the Jones-Davis BRD, while effective in a sub-tropical fish assemblage only has a very limited effectiveness on a tropical fish assemblage. Van der Geest (2000) came to a similar conclusion. These differences are most likely due to the differences in fish community composition between the sub-tropical Gulf of Mexico and the tropical waters of Cleveland Bay. While most of the assemblage in Cleveland Bay is dominated by small, slow-swimming demersal fish species, the Gulf of Mexico is dominated by large, strong-swimming, semi-pelagic and pelagic species such as snapper, mackerel and croaker (Nance and Scott-Denton, 1997; Van der Geest, 2000; Wastson and Foster, 1997). Foster (1999) reported decreases in fish catch of 58% using the Jones-Davis BRD in the Gulf of Mexico shrimp trawl fishery for which the BRD was originally designed. Foster and Scott-Denton (2004) also reported reduction in bycatch of 43.9% and 33.5% using two modifications of the Jones-Davis BRD (a Double hoop Jones-Davis; a Modified Jones Davis). Rogers et al. (1997) reported reductions between 55% and 22% testing eight different BRD designs in the inshore waters of Louisiana, however all designs also lead to a significant reduction in target species catch. Brewer et al. (1998) reported reductions of small fish bycatch between 15% and 40% in the Gulf of Carpentaria using several different BRD designs (Super Shooter, Nordmore Grid; Radial escape section; Fisheye; Square- mesh window). A modification of the Nordmore

Grid involving a rope grid led to reductions of major fish bycatch species of 36% - 50% in the temperate Gulf of Maine pink shrimp fishery (He and Balzano, 2007, 2011). The radial escape section BRD reduced between 20% and 40% of the fish bycatch (Brewer et al., 1998). Courtney et al. (2006) found a 24% decrease in bycatch using a radial escape section BRD in the southern Queensland King prawn (*Penaeus plebejus*) fishery during a nine day research trial. Garcia-Caudillo et al. (2000) compared the radial escape section BRD with TED against a control only using a TED in the Gulf of California shrimp fishery and found reductions in bycatch of up to 43%. Steele et al. (2002) evaluated the effects of an extended mesh funnel (similar to a radial escape section) with TED in the Florida shrimp fishery and found that for a range of different net sizes, bycatch was reduced by between 18% and 60%. While these studies suggest that the radial escape section is quite efficient at reducing bycatch, they also reported a significant loss of target species catch (García-Caudillo et al., 2000; Rogers et al., 1997; Steele et al., 2002). It is acknowledged however that many of the studies listed above did not use a TED in the Control net. The TED used in the control net in the current study would have removed some very large and strong swimming fish in both nets, hence results can only draw inferences about bycatch reduction of teleost fish that were not affected by the TED. The results of these various studies compared to the present study suggest that the Jones-Davis device is partially effective in reducing fish bycatch while not reducing prawn catch.

Even though bycatch of some families of teleost fish were reduced significantly, the Jones-Davis BRD did not eliminate the majority of fish bycatch from trawl nets. More than 75% of fish were still retained in the trawl nets, of which the majority would be discarded dead over the side of the vessel after a lengthy sorting process to remove the target group, the penaeid prawns. Even though the Jones-Davis BRD might be a valuable tool in temperate trawl

fisheries that have a similar fish assemblage to the Gulf of Mexico, the device was not very effective in tropical waters where the fish assemblage is dominated by small benthic and demersal species. The Jones-Davis BRD would need to be modified in order to exclude more of the slow swimmers in tropical fish assemblages, characteristic of northern Australia and Indo-West Pacific region. This represents a considerable challenge. Some improvements have been achieved in BRD technology, such as research on optical, acoustic and data processing improvements in order to pin point problems associated with benthic trawls (Graham et al., 2004). Bar spacing for example in the Nordmore grid BRD has been investigated to improve bycatch reduction potential while retaining a significant catch of target species (Silva et al., 2012). In order to improve BRD performance, research has focused on investigating BRD technologies in flume tanks prior to testing in the fishery to better understand the relationship between codend shape, grid design and water velocity as well as saving time in field studies if problems related to water flow parameters should occur (He and Balzano, 2011; Heales et al., 2008; Kvalsvik et al., 2006; O'Neill et al., 2005; O'Neill et al., 2003). Quantifying basic gear-related characteristics, aimed at reducing catch of non targeted species while maintaining catch of target species can save considerable time and effort in the long term development of more selective gears.

In addition to modifying the BRD design to remove fish from the net while it is fishing, development has focused on improving the separation of target species and bycatch after the catch is landed on board the trawler (eg. Broadhurst et al., 2004; Kim and Wang, 2010; Macbeth et al., 2005a; Pichot et al., 2009). The invention of post trawl sorting mechanisms such as flotation and hopper tanks is a valuable development tool, potentially increasing survival rates of bycatch species (Ocean-Watch, 2004). It has been suggested that the use of hopper tanks significantly increases post trawl bycatch survival rates (Ocean-Watch, 2004).

Deployment of post trawling sorting tanks should only be applied in combination with efficient BRD technology (Broadhurst et al., 2008). Further improvements in bycatch reduction are needed, especially in the tropical prawn trawling fishery, in order to promote long term sustainable fisheries and help to sustain global biodiversity.

Chapter 3

Interannual Variations in Trawl Catches in Cleveland Bay and Environmental Correlates of the Variations.

Introduction

The primary purpose of monitoring fish assemblages is to describe and possibly help understand why they change or remain the same. In fisheries such monitoring assists in making estimates of sustainable catches (Friedlander and Parrish, 1998; Gaertner et al., 1998). Monitoring fish stocks over long time periods also allows us to make inferences about long term fishing effects as well as climatic, and environmental effects on the fish assemblage (Condie et al., 2012). A key principle of ecosystem-based fisheries management is to view the fish assemblage holistically rather than focusing on single species abundance trends (Toole et al., 2011). Data from long term studies are important in assessing and predicting the effect of environmental change on the community and such studies are important in whole ecosystem based management of fisheries (Dichmont et al., 2006; Harris and Poiner, 1991; Trenkel et al., 2007).

Marine ecosystems and communities are often described as open and variable systems that are strongly influenced by external (e.g. climatic, oceanographic) factors, often operating on the process of recruitment (Caley, 1995; Caley et al., 1996; Pittman et al., 2004). Species diversity and abundance estimates can change temporally due to natural processes. (Condie et al., 2012). Changes in climate and exploitation of resources both put stress on marine systems, creating greater temporal variability in abundance of components of the fish assemblage (Meynecke and Lee, 2011; Rotherham et al., 2011). Accurate information is

required to draw inferences on the temporal dynamics of abundance of both commercially exploited species and the fish community as a whole (Trenkel et al., 2004). Anthropogenic degradation of aquatic environments throughout the world has led to international accords and legislation which have focused on increased environmental reporting and accountability in the management of these ecosystems (Machias et al., 2004; Miranda et al., 2005; Sheaves et al., 2007). Monitoring and management requirements have become a foundation of environmental policy in many countries such as Australia, the USA, South Africa and Europe (Hallett et al., 2012).

Many studies of aquatic ecosystems and communities are conducted to determine if changes result from natural environmental changes or human impacts (Albouy et al., 2012). Estuaries and adjacent coastal areas are highly dynamic environments in which physical and chemical characteristics can change in a matter of hours, daily, seasonally and annually. Consequently estuarine associated fish assemblages often exhibit large year to year variations in abundance, species composition and size structure (Hagan and Able, 2003). Most studies investigating changes in estuarine associated fish assemblages are on scales of months to years since funding that supports sampling is often granted on such time scales (Trenkel et al., 2007). Long term studies (> 3 years) have shown that most estuarine associated fish assemblages can undergo significant changes in community structure and that such change can sometimes be explained by events such as major rainfall, droughts, severe storms and temperature fluctuations, and that long term climatic changes can alter fish assemblage structure significantly (Adjeroud et al., 2002; Warwick et al., 2002). Fish assemblage structure and biodiversity are often affected by inter-annual variations in breeding success and larval survival and recruitment, these processes themselves often being associated with biological and physical processes (Moyano and Hernandez-Leon, 2011).

Many environmental factors have been shown to affect abundances of fish populations and thus structure of fish assemblages all over the world. Extreme weather events such as cyclones can have dramatic effects on assemblage structure of fishes over a very short period of time. Cyclones affect fish assemblages directly by increasing turbidity and water movement, decreasing atmospheric pressure leading to an increased tidal height and an increased freshwater input particularly in coastal systems, and indirectly by re-suspension of organic matter, nutrients and suspended solids which in turn leads to an increase in primary production (Adjeroud et al., 2002; Alvarez et al., 2012; Condie et al., 2012; Costa et al., 2007).

Many studies have focused on temperature as a factor affecting fish assemblages (Meynecke and Lee, 2011; Zeeberg et al., 2008). Temperature has been shown to play an important role in the life cycle of many coastal marine species (Gillson et al., 2009; Meynecke and Lee, 2011). It is generally agreed that warmer waters can increase productivity as a result of increased primary production and an increase in metabolic activity in fish (Meynecke and Lee, 2011). This is however species-specific as some species might not be as tolerant to temperature fluctuations as others (Meynecke and Lee, 2011). Temperature effects on fish assemblages in temperate estuaries have been shown in South Africa but are less distinct in tropical climates (Meynecke and Lee, 2011; Whitfield, 2005). Other studies have also found significant inter-annual variations in fish assemblages due to changing temperatures (Condie et al., 2012; Garcia et al., 2003; Lin and Shao, 1999; Meynecke and Lee, 2011; Travers et al., 2012). Changes in primary and secondary production, changes in spatial distribution due to habitat shifts as a result of altered temperature and changes in recruitment have all been proposed as mechanisms relating estuary fish catches with temperature (Meynecke and Lee, 2011).

Changes in freshwater input into estuaries and other coastal marine environments can affect mortality, growth, development and movement of certain organisms (Gillanders and Kingsford, 2002; Loneragan and Bunn, 1999; Robins et al., 2005; Vinagre et al., 2007). The effects of freshwater flows on marine fish and fisheries can be direct or indirect through importing nutrients, sediment and detritus, changing salinity levels in estuarine and coastal areas, altering turbidity, dissolved oxygen concentrations and affecting olfactory cues of fish (Gillson et al., 2009; Gillson and Suthers, 2012; Gowns and James, 2005; Loneragan, 2009). Production of phytoplankton and zooplankton are stimulated by nutrient supplies which in turn influence larval, juvenile and adult fish populations (James et al., 2012; James et al., 2008; Russ and McCook, 1999). Similarly, detritus can act as a substrate for microorganisms after it is broken down or can directly benefit detritivorous fish (Russ and McCook, 1999). Sediment input into near shore environments replenishes habitats as they are continuously eroded away by ocean currents and tidal changes, and provides shelter and refuge for many fish species as a result of increased turbidity, but also increases feeding success in suspension-rich water (Lamberth et al., 2009). Freshwater flow into the marine environment is also an important cue for migrations of many fish species in and out of the estuarine environment, either for feeding, shelter or spawning (Loneragan, 2009; Miller et al., 2009; Robins et al., 2005; Whitfield, 2005; Yako et al., 2002). Water flow levels can change due to environmental factors such as precipitation and rainfall but also due to anthropogenic factors like construction of dams that will strongly influence the amount of freshwater flow into estuarine and coastal systems (Gillanders and Kingsford, 2002; Sheaves et al., 2007). Dams not only alter the amount of freshwater input but also the timing of flow of water to coastal systems (Gillanders and Kingsford, 2002).

Lunar phase has also been shown to affect catch rates of many commercially important fish and invertebrate species (Chatterji et al., 1994; Hernández-León, 2008; Libini and Khan, 2012; Poisson et al., 2010). However these relationships are often species-specific and hence it is difficult to draw general inferences. Few studies are designed to measure the catch rates of any trawl species over a lunar cycle. However, catch rates and moon phase are related for many species that are not trawled. For example catch rates of Atlantic herring in the Gulf of Mexico are highest during the dark phase of the moon, (Anthony and Fogarty, 1985); butterfish catch rates in the Gulf of Mexico peak during the first quarter moon (Render and Allen, 1987); several crab species in India have highest catch rates during the new moon and the new and full moons (Chatterji et al., 1994). Catch rate of endeavour prawns (*Metapenaeus endeavouri*) were found to be higher during the full moon and Tiger prawns (*P. esculentus* and *P. semisculatus*) were found to peak at last quarter moon, in the Gulf of Carpentaria prawn fishery (Salini et al., 2001). Griffiths (1999) reported significantly higher catches of *Penaeus plebejus* during periods of the full moon in Shellharbour Lagoon, New South Wales. Significant differences in the composition of a mangrove fish assemblage have been reported during different moon phases in a tropical estuary in Brazil (Ramos et al., 2011).

The fish fauna of Cleveland Bay, Townsville Australia has been well studied (Cabanban, 1991; Sondita, 1997; Van der Geest, 2000). However studies of long term variations in the trawl bycatch fish assemblage are not available. Most previous studies of this tropical estuary have concentrated on the biology and seasonal movements of a single species or variations of fish assemblage structures over a short period of time (Cabanban, 1991; Sondita, 1997; Van der Geest, 2000). Long term variations in the bycatch fish assemblage are an important aspect for fisheries management.

The aims of this chapter were:

1. To quantify inter-annual variations in abundance of the most important target and bycatch taxa in a trawl fishery in Cleveland Bay over a 15 year period (1998-2012).
2. To investigate any possible relationships between environmental factors and inter-annual variations in trawl catches.

Materials and Methods:

Statistical Procedures

The same data set used in Chapter 2 was used to investigate inter-annual variations in catch rates for the most important target and bycatch groups in the trawl fishery in Cleveland Bay (Prawns, Carangidae, Sciaenidae, *Saurida* spp, Leiognathidae, “Other Fish” and total fish). However, this chapter uses data only from the control (non-BRD) net sampled in Cleveland Bay over the study period.

Interannual Variations in catch rates of major target and bycatch groups in the trawl fishery.

A total of 244 ten minute trawls were carried out in inshore Cleveland Bay (see Figure 2.1) between 1998 and 2012. Inter-annual variations in catch rates of major groups were analysed using a one-Way Analysis of Variance to test for differences among years. Wet weight biomass catch rate (expressed as kg/ha) was the dependant variable and year (1998-2012, n = 12 sampling years over a 15 year period) the categorical predictor (main factor) variable. Statistical tests were conducted at a significance level of 0.05 in Statistica 10. Data from all 12 years and from each individual trawl were used to analyse inter-annual variations in catch rates of Prawns, Carangidae, Leiognathidae, *Saurida* spp., Sciaenidae, total and “other” fish.

Three outliers were removed in the Total fish and Sciaenidae data set to improve assumptions of the ANOVA. These outliers were due to three particularly high catch rates in individual trawls for these two groups. These outliers were due to extreme small scale variability in catch rates of schooling fish.

Data were natural log transformed after adding $0.1\text{kg}/6000\text{m}^2$ to all catch data in order to eliminate the occurrence of zero values. This ensured more robust analyses, increased homogeneity of variances, normality of residuals and produced lower variances around the means. Note that data were analysed using catch rate per 6000m^2 but will be expressed in the thesis as kg/ha.

Environmental effects on catch rates of target and bycatch groups in the trawl fishery

Environmental variables investigated during this study included temperature, tide (Tidal state and Tidal height), Southern Oscillation Index (SOI), rainfall and moon phase. Data on Tidal state and Tidal height were obtained from the Australian Institute of Marine Science (AIMS) monitoring stations. Rainfall and SOI data were obtained from the Bureau of Meteorology (BOM) website. Most of the variables tested had no significant effect on catch rates. Only variables that had significant effects on catch rates will be reported in this chapter. Only rainfall and moon phase had significant effects on catch rates and thus will be reported in detail below. Ten minute trawls were pooled to reduce the high small-scale spatial and temporal variability in catch rate that occurred, mostly caused by the high frequency of schooling fish in the fish assemblage. The pooling of trawls, whilst not optimal, was a necessary compromise in making a sampling unit chosen for a short-term teaching exercise and using it in a long-term research project.

Rainfall effects on catch rates of target and bycatch groups in the trawl fishery.

Mean overall catch rate per year (i.e. all samples collected in March of any year pooled) for all major groups was used to determine the effect of rainfall (1,2,3 and 4weeks prior to trawling date) as well rainfall of prior wet seasons on the fish assemblage. The mean was used rather than the raw data in order to minimize effects of small scale spatial variability in catch rates. Total wet season rainfall one or two years prior to the sampling date was used to investigate the lag effect of rainfall on catch rates of major groups. Data were analysed by a multiple regression and a multivariate test of significance, to determine how much of the variation in catch rates was explained by rainfall 1-4 weeks prior to sampling and either 1 or 2 years prior to sampling. No transformation was used to investigate the effect of rainfall on catch rates of major groups.

Lunar effects on catch rates of major groups caught in trawls:

The effect of lunar phase on catch rates of major groups was investigated using a one-Way ANOVA, lunar phase being the main factor and catch rate (kg/6000m²) the variate of interest. Lunar phase (Full Moon, New Moon, Last Quarter and First Quarter) was defined as any day on or 3 days either side of the particular moon phases. Catch rates were again grouped by observer (i.e. individual trips on the trawler sampling 3-4 trawls). This helped to reduce zero catch rate values. Data were natural log transformed in order to pass the assumptions of the ANOVA. Because all assumptions were not met for the prawn data set, a non-parametric test (Kruskal-Wallis) was performed on the untransformed data. In inferring lunar effects in this Chapter, the explicit assumption is made that lunar phase had a stronger effect on than any year effect.

Results:

Prawns

Prawns are the major target group of the trawl fishery in Cleveland Bay. Prawn catch rates were highly variable among years (ANOVA, $p < 0.01$, Table 3.1, Figure 3.1). Mean catch rates varied from 0.03kg/ha (+/-SE 0.01) in 2010 to 4.05kg/ha (+/- SE 0.62) in 2008 (Figure 3.1). Prawn catch contributed between 0.2% and 30.3% of the total trawl catch by weight, with an average of 7.9% (+/-SE 3.27) (Figure 3.2). Differences in catch rate varied by 117-fold among years (Figure 3.1). There were no significant differences in catch rates among 1998, 1999 and 2004 (Tukeys Tests, $p > 0.05$) but catch rates in these years differed significantly from those in all other years (Tukeys tests, $p < 0.01$). No significant difference in catch rate was observed between 2007 and 2008 ($p > 0.05$) but catch rates in these two years were significantly different from those in all other years ($p < 0.01$). The five years in which catch rates were particularly low (2003, 2006, 2010, 2011 and 2012) did not differ significantly from each other (Tukeys tests, $p > 0.05$) but were significantly different from all other years (Tukeys tests, $p < 0.01$) (for all between year effects see Tukeys HSD, Appendix 2, Table2.8).

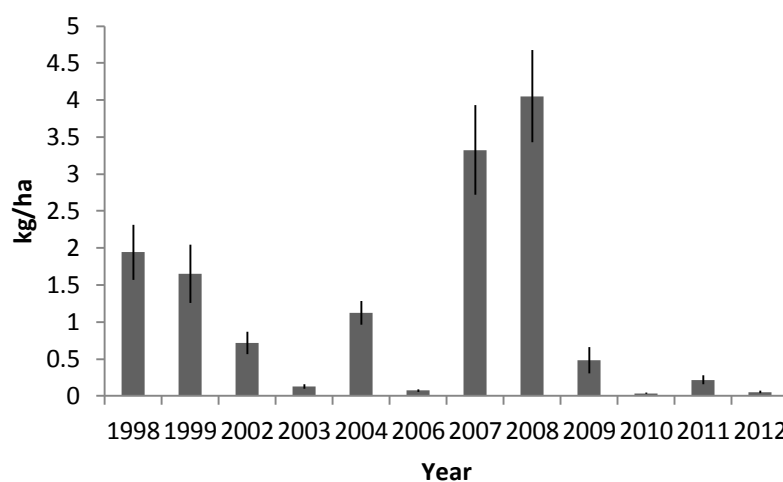


Figure 3.1: Mean (+/-SE) catch rate (kg/ ha) of prawns in the control net from 1998-2012. Years not sampled were 2000, 2001, 2005.

	SS	Degrees of Freedom	MS	F	p
Prawn	254.826	11	23.166	56.094	<0.001
Total Fish	27.644	11	2.513	7.505	<0.001
Carangidae	13.825	11	1.256	3.492	<0.001
Sciaenidae	91.472	11	8.315	10.467	<0.001
<i>Saurida</i> spp.	46.102	11	4.191	11.409	<0.001
Leiognathidae	65.165	11	5.924	10.135	<0.001
Other Fish	77.778	11	7.071	14.020	<0.001

Table 3.1: Univariate ANOVAs testing the effect of year of sampling on catch rate (kg/ha) for all variates. The denominator of all F values is 244. (For more detailed ANOVA results see Appendix 2, Table 2.7).

	SS	Degrees of Freedom	MS	F	p
Total.Fish	112.742	1	9.395	6.094	0.033
Carangidae	0.131	1	0.011	1.772	0.212
Sciaenidae	2.277	1	0.189	5.328	0.043
Prawns	5.928	1	0.494	2.258	0.163
Leiognathidae	8.977	1	0.748	3.397	0.095
<i>Saurida</i> spp.	0.243	1	0.021	0.079	0.783
Other.Fish	70.394	1	5.866	14.499	0.003

Table 3.2: Multiple regression results for the effect of 2 year lagged rainfall on catch rate (kg/ha) of all variates. The denominator for all F values is 10.

Catch Rate of Total Fish

Catch rate of Total fish was highly variable among years ($p < 0.01$, Table 3.1, Figure 3.3). Mean catch rates varied from 7.71kg/ha (SE +/-0.7) in 2003 to 30.82kg/ha (SE +/-7.55) in 2011 (Figure 3.3). Catch rates of Total fish varied 3.9-fold among years (Figure 3.3). There was no statistically significant difference between the four highest catch rate years (2002, 2007, 2009 and 2011) (Tukeys HSD – Appendix 2, Table 2.9; tests at 0.05, and Figure 3.3).

The years 1998, 2004, 2010 and 2012 did not differ significantly from any other years with the exception of 2011 (Tukeys HSD tests at 0.05, Figure 3.3).

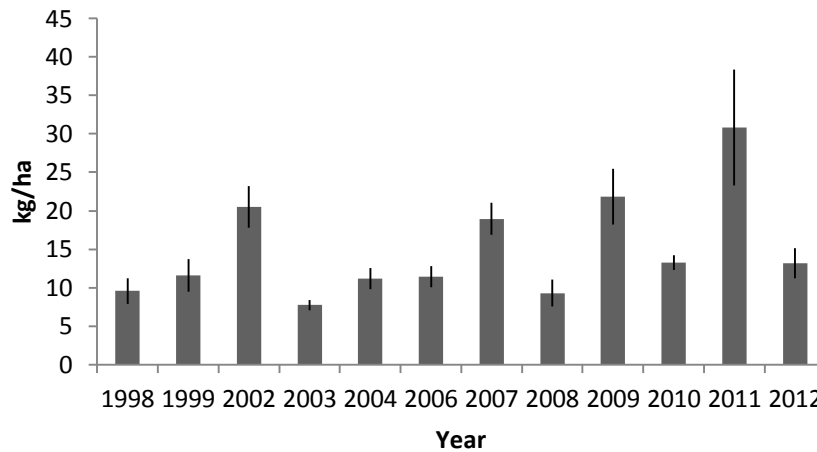


Figure 3.2: Mean (+/- SE) catch rate (kg/ha) of Total Fish in the control net between 1998 and 2012. Years not sampled were 2000, 2001, 2005.

Catch Rate of Carangidae

The catch rate of Carangidae (kg/ha) was relatively consistent from 1998 to 2012. The catch rate of Carangidae was relatively low and ranged from a mean of 0.17kg/ha (SE +/- 0.01) in 2010 to a mean of 0.7kg/ha (SE +/- 0.21) in 1998 (Figure 3.4). Carangidae catch made up between 0.9% and 6.1% of the total fish catch with an average of 3.2% (Figure 2.19). Differences in catch rates varied up to 4.1-fold. Three years were significantly different from the others (Table 3.1, Figure 3.4). The catch rate in 2006 was significantly higher than in 2008, 2010 and 2011 (Tukeys HSD tests: $p < 0.05$, Appendix 2, Table 2.10), and the catch rate in 2010 was significantly lower than in 1998 (Tukeys HSD test, $p < 0.05$).

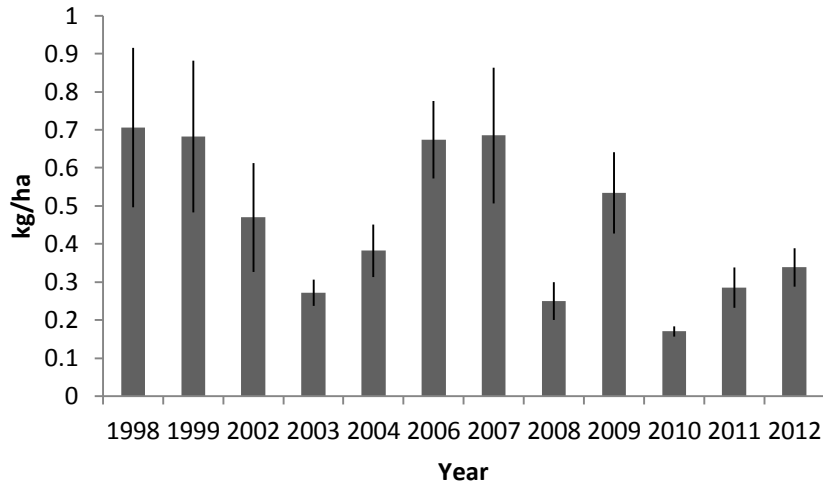


Figure 3.3: Mean (+/- SE) catch rate (kg/ha) of Carangidae in the control net from 1998-2012. Years not sampled were 2000, 2001, 2005.

Catch Rate of Sciaenidae:

Catch rates of Sciaenidae were highly variable among years (Table 3.1, $p < 0.01$, Figure 3.5). The catch rates of Sciaenidae ranged from 0.03kg/ha (SE +/- 0.01) in 2010, to 3.31kg/ha (SE +/- 1.47) in 2012 (Figure 3.5). Catch rates of Sciaenidae contributed between 0.23% and 22.8% of the total fish catch with a mean of 5.61% (+/- SE 1.75) (Figure 2.19). Catch rates of Sciaenidae varied up to 103-fold among years. The relatively high mean catch rate in 2012 was due to one particular trawl that captured 31.6kg/ha. This trawl also accounted for the high variance in catch rate for this particular year. Catch rate in 2012 was significantly higher than in 2003-2006 and 2010. Catch rate of Sciaenidae in 2002 was significantly higher than all years except for 1999, 2009, 2011 and 2012 (Figure 3.5). No significant difference in catch rate (kg/ha) was found between the years 2004 and 2009. The very low catch rate in 2010 was significantly less than for all years except 2003-2006 (Tukeys HSD Tests at 0.05, Appendix 2, Table 2.11).

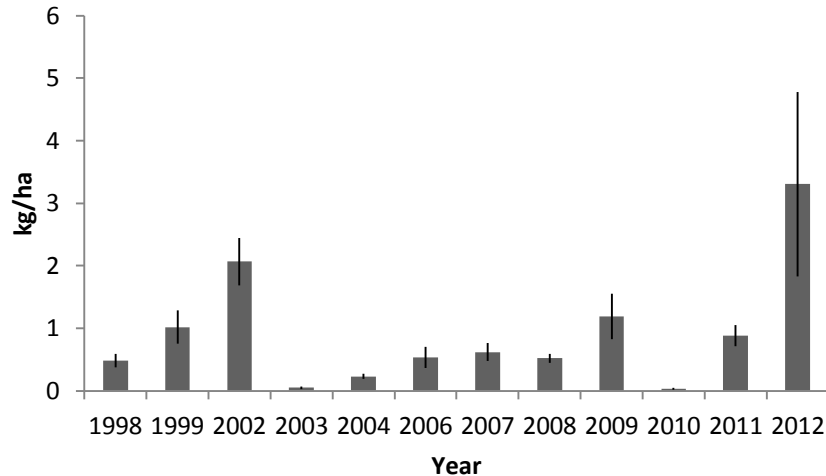


Figure 3.4: Mean (+/- SE) catch rate (kg/ha) of Sciaenidae in the control net from 1998-2012. Years not sampled were 2000, 2001, 2005.

Catch Rates of *Saurida* spp.:

The catch rates of *Saurida* spp. varied significantly among years (Table 3.1, $p < 0.01$; Figure 3.6). Catch rates varied from 0.01 kg/ha (+/-SE 0.01) in 2012 to 0.88 kg/ha (+/-SE 0.11) in 2007 (Figure 3.6). Catch of *Saurida* spp. contributed between 0.09% and 4.4% with an average of 2.12% (+/-SE 0.39) to the total fish catch (Figure 2.19). Catch rates of *Saurida* spp. varied up to 64-fold among years. Significantly higher catch rates of *Saurida* spp. were recorded in 2007 than in all years except 1999 and 2002 (Figure 3.6). Catch rates of *Saurida* spp. were significantly lower in 2012 than in all years except for 2003 (Figure 3.6). There was no significant difference in catch rates of *Saurida* spp. in the years 1998, 2003-2006 and 2008-2011 (Tukeys HSD: $p > 0.05$, Appendix 2, Table 2.12).

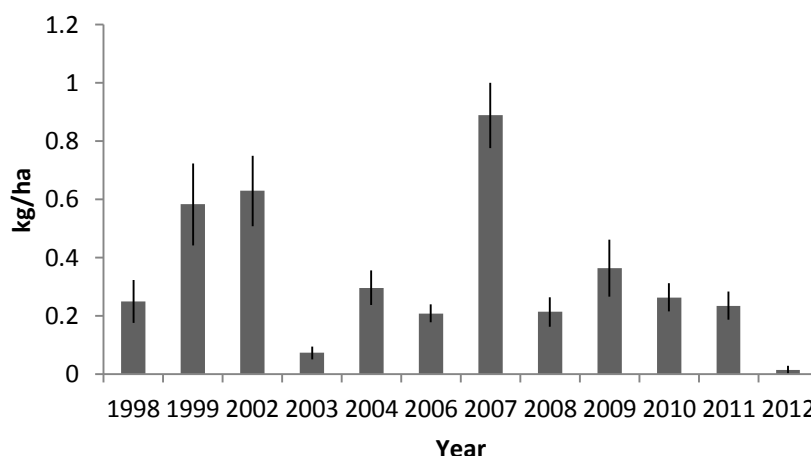


Figure 3.5: Mean (\pm SE) catch rate (kg/ha) of *Saurida* spp. in the control net from 1998-2012. Years not sampled were 2000, 2001, 2005.

Catch Rates of Leiognathidae:

The catch rates of Leiognathidae (kg/ha) were relatively consistent over the 15 year sampling period, however some significant inter-annual variations were observed (Table 3.1, $p < 0.01$; Figure 3.7). Catch rates of Leiognathidae ranged from 1.12kg/ha (\pm SE 0.2) in 2012 to 7.45kg/ha (\pm SE 1.1) in 2007 with a mean catch of 5.03/ha (\pm SE 0.5) (Figure 3.7). Catch of Leiognathidae contributed between 7.8% and 55.1% with average of 33.4% of the total fish catch (Figure 2.19). Catch rates of Leiognathidae (kg/ha) varied up to 6.6-fold among years. Significantly lower catch rates of Leiognathidae were recorded in 2012 than in other years. Even though mean catch rate (kg/ha) in 2002 was more than twice that in 2012, the analysis could not detect any significant difference in catch rate between the two years (Tukeys HSD $p > 0.05$, Appendix 2, Table 2.12). Mean catch rates (kg/ha) did not differ significantly among the years 1999, 2003-2006, 2008 and 2011.

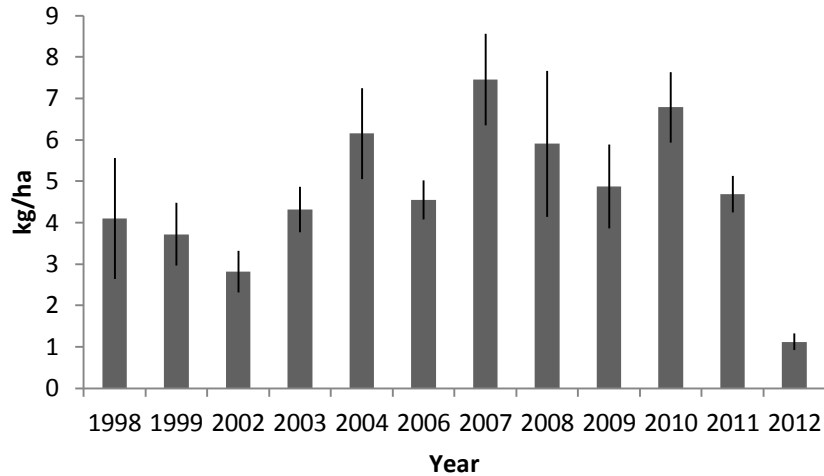


Figure 3.6: Mean (\pm SE) catch rate (kg/ha) of Leiognathidae in the control net from 1998-2012. Years not sampled were 2000, 2001, 2005.

Catch Rates of Other Fish:

The mean catch rate (kg/ha) of the “Other Fish” group varied significantly over the 15 year sampling period ($p < 0.05$, Table 3.1, Figure 3.8). Catch rates of “Other Fish” ranged from 2.39kg/ha (SE \pm 0.18) in 2008 to 24.74kg/ha (SE \pm 7.33) in 2011, with an average of 8.24kg/ha (SE \pm 1.9) (Figure 3.8). The catch of “Other Fish” contributed between 17.9% and 79.7% to the total fish catch, with an average of 47.6% (\pm SE 5.36) (Figure 2.19). Catch rates of “Other Fish” varied up to 7.3-fold among years. Catch rates (kg/ha) of this group were significantly lower in 1998 and 1999 than in 2002. The following years (2003 – 2008) were also significantly lower than 2002 and 2011 and with the exception of 2007 these years were also lower than 2009 and 2012 ($p < 0.05$; Tukeys test, Appendix 2, Table 2.13). A significantly higher catch rate was observed in 2009 than in many other years (1998, 2003, 2004, 2006 and 2008), followed by lower catch rates in 2010 with the highest catch rate observed in 2011 (Figure 3.8). No significant differences in catch rate were observed among the years 1999, 2009, 2010, 2011 and 2012 even though mean catch rates varied up to 4.04 fold among these years.

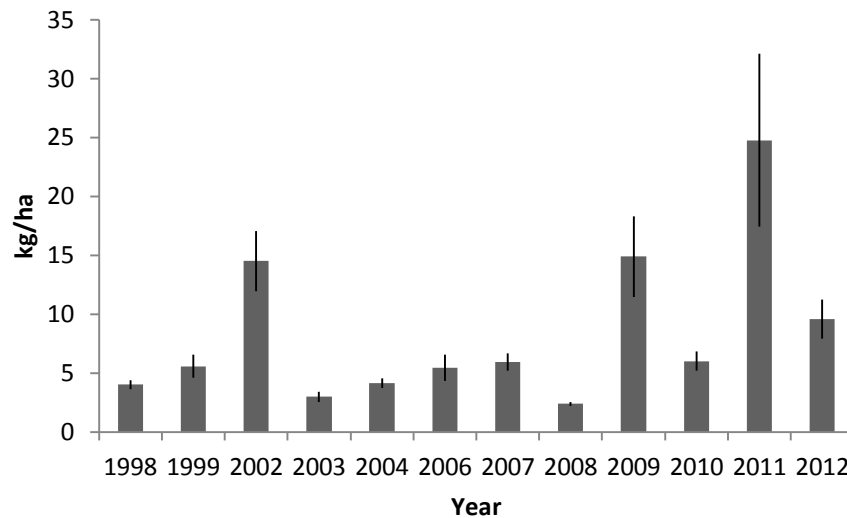


Figure 3.7: Mean (+/- SE) catch rate (kg/ha) of “Other Fish” in the control net from 1998-2012. Years not sampled were 2000, 2001, 2005.

Environmental factors affecting catch rates:

Rainfall:

Few significant relationships were found between trawl catch rates and rainfall (Appendix 2, table 2.1 -2.4).

No significant relationships were found between cumulative rainfall 1 – 4 weeks prior to the sampling dates and catch rate of any group (Appendix 2, Tables 2.1-2.7). However, some significant relationships emerged when the effect of rainfall 2 years prior to sampling was plotted against catch rate. A multiple regression explained 32% of the variance in the relationship between total fish catch rate and rainfall two years prior to sampling (Figure 3.8). The slope of the regression was significant ($p < 0.05$; Table 3.2, Figure 3.8). Catch rate of Total Fish doubled from the lowest to highest lagged rainfall recorded in the study (Figure

3.8). A similar result was obtained for Sciaenidae, with the relationship between catch rate and wet season rainfall two years prior to sampling accounting for 28.2% of the variation in the data set, and catch rate almost 10-fold higher from lowest to highest 2-year lagged rainfall ($p < 0.05$; Table 3.2, Figure 3.8). Similarly, the relationship between catch rate of “Other Fish” and wet season rainfall two years prior to sampling accounted for 55% of the variation in the data set ($p < 0.05$; Table 3.2, Figure 3.8). Catch rate of “Other Fish” increased by almost 9-fold between the lowest and highest 2-year lagged rainfall (Figure 3.8). There were no significant relationships between catch rate and lagged wet season rainfall for any of the other major groups in trawl catches in Cleveland Bay for 1998-2012 (see Appendix 2, Table 2.1 to 2.7).

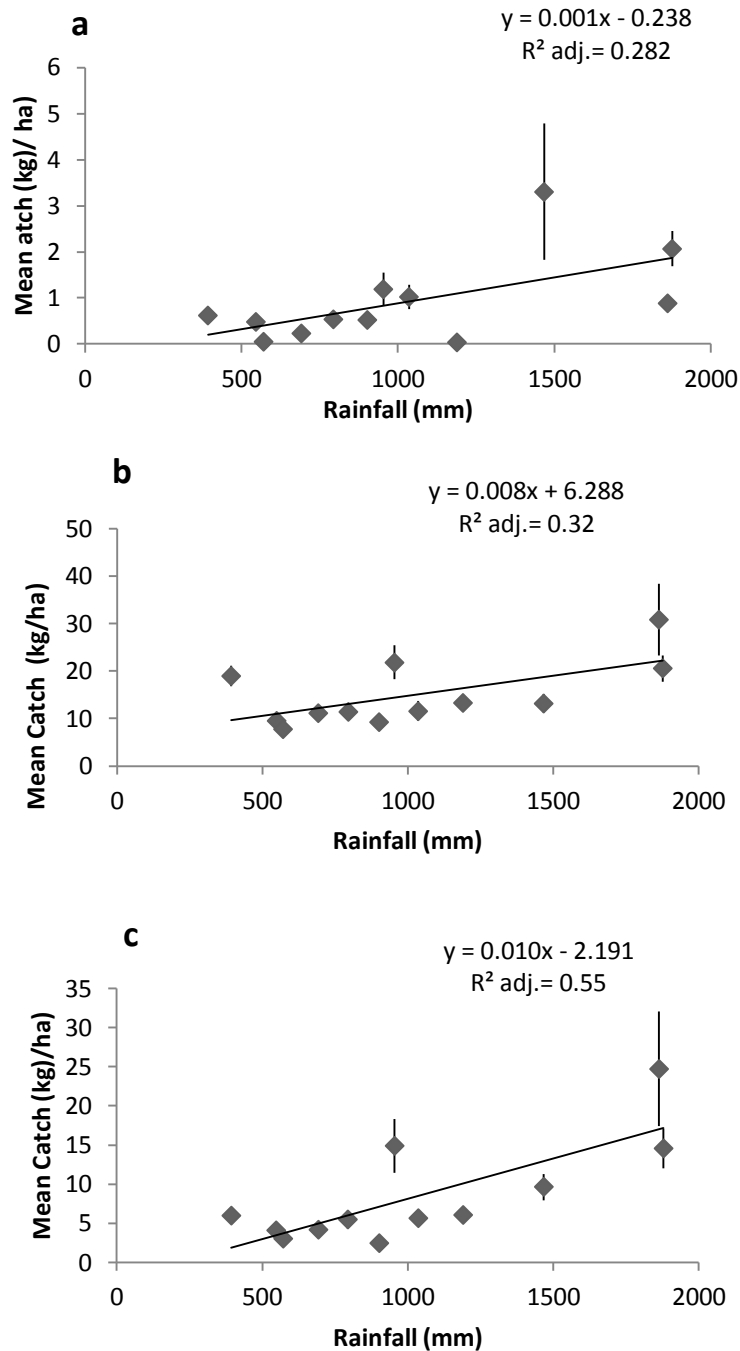


Figure 3.8: Linear Regression of mean catch rate (kg/ha) of Sciaenidae (a), ‘Other Fish’ (b) and ‘Total Fish’(c) in a sampling year against average wet season rainfall two years prior to sampling. The line of best fit explained 28.2% (Sciaenidae); 32% (‘Other Fish’) and 55% (‘Total Fish’) of the variance and was significant ($p < 0.05$) for all groups.

Moonphase:

Mean catch rate (kg/ha) of the main target of the fishery, prawns, was significantly higher during periods on or around the full moon compared to all other moon phases ($p < 0.01$) (Table 3.3, Figure 3.11). Mean catch rate for prawns was 2.96kg/ha (\pm SE 0.62) during full moon periods, 0.76kg/ha (\pm SE 0.15) during last quarter, 0.51kg/ha (\pm SE 0.17) during new moon and 1.27kg/ha (\pm SE 0.45) during the first quarter moon phases.

	Full Moon	Last Quarter	New Moon	First Quarter
Full Moon		0.011	0.006	>0.001
Last Quarter	0.011		1	1
New Moon	0.006	1		1
First Quarter	>0.001	1	1	

Table 3.3: Multiple comparison p values (2 tailed) for effects of moon phase on catch rate of prawns, Kruskal-Wallis test: $H(3, N=77) p = 0.0015$.

There were no significant differences in catch rates of prawns among the other moon phases (Figure 3.09). Mean catch rate (kg/ha) of *Saurida* spp. was significantly higher on the full moon than during periods of the new moon or the first quarter moon ($p < 0.05$) (Table 3.4, Figure 3.10).

	SS	Degrees of Freedom	MS	F	p
Total Fish	1.547	3	0.515	1.628	0.191
Carangidae	1.398	3	0.466	0.758	0.521
Sciaenidae	0.181	3	0.061	0.248	0.862
<i>Saurida</i> spp.	1.247	3	0.415	7.612	>0.001
Leiognathidae	1.828	3	0.609	1.721	0.171
Other Fish	3.139	3	1.046	1.771	0.161

Table 3.4: Univariate ANOVAs testing for the effect of moon phase on catch rate (kg/ha) for all fish variates. The denominator of all F values is 73.

Mean catch rates of *Saurida* spp. were 0.7kg/ha (\pm SE 0.16) during periods of the full moon, 0.37kg/ha (\pm SE 0.05) during the last quarter, 0.32kg/ha (\pm SE 0.09) during the new moon

and 0.15kg/ha (+/-SE 0.03) during the first quarter moon phase. There was a significant increase in catch rate of *Saurida* spp. during periods of the full moon (Figure 3.09). There was however no significant difference in catch rate of *Saurida* spp. between the last quarter and full moon (Figure 3.10). There was no significant difference in catch rates among moon phases for “total fish”, Carangidae, *Saurida* spp., Sciaenidae, Leiognathidae and “other fish” (Appendix 2: Figures 2.1-2.5, Table 2.8).

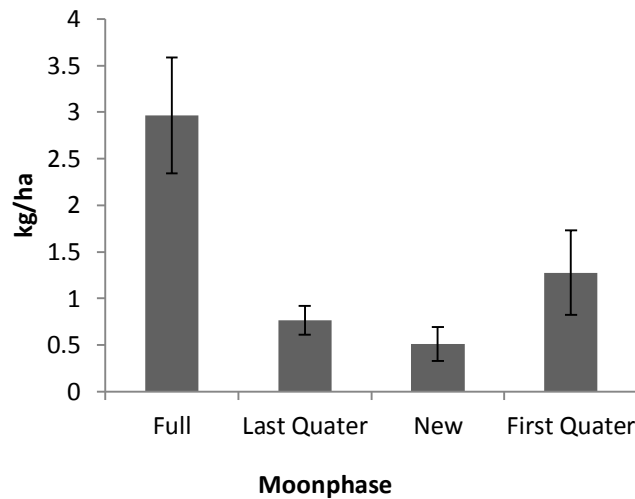


Figure 3.9: Mean (+/- SE) catch rate (kg/ha) of prawns for all 12 sampling years from 1998-2102 at different phases of the moon.

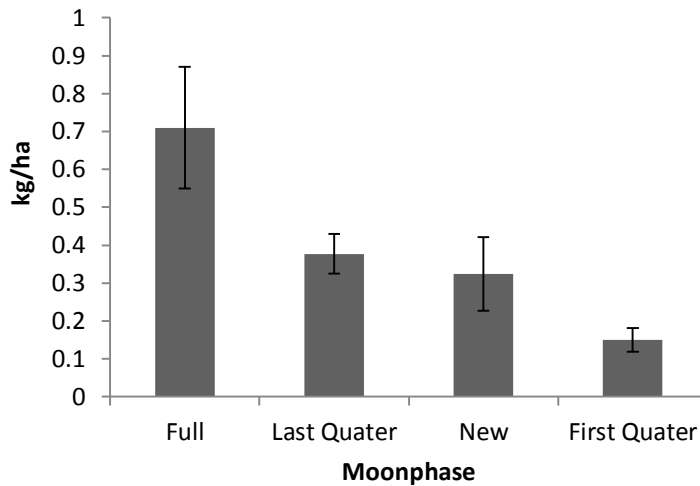


Figure 3.10: Mean (+/- SE) catch rate (kg/ha) of *Saurida* spp. for all 12 sampling years from 1998-2012 at different phases of the moon.

Discussion

The Value of Long-term Scientific Monitoring

Most long-term studies of seasonal and temporal variation in catch rates of trawl fisheries use data collected from commercial fishing operations. This study used data collected by a standardized scientific survey. Thus, over the 15 year study, the sampling method was consistent, including using the same vessel, fishing nets, crew (for the most part), deployment of gear, the same time of the year (March), a consistent trawl duration and speed that could be converted to a standard swept area (density), a consistent general location (inshore Cleveland Bay), and collection of catch rate data for both target (prawn) and non-target (fish) organisms. This consistency of sampling of both target and non-target organisms over such a long period (1.5 decades) is rare, particularly in tropical trawl fisheries. Catch rates collected from commercial fisheries can be biased since the fishery tries to, logically, maximize catch per unit effort by fishing at the best times and places on the basis of the fishing experience of the fishers (Bailey et al., 2006; Meynecke and Lee, 2011). Another reason why such long

term data from commercial trawls are inconsistent is that the development of new fishing gear technologies over time can result in “effort creep” (sampling becoming more efficient over time due to improved technologies) unless these improvements are accounted for properly. Gear modifications improving catch rates are common in most fisheries so that long term catch rate data from commercial fisheries may be inconsistent. The use of different gears and deployment techniques between fishing vessels and/or between fishing trips will influence catch rates of target and bycatch species (O'Neill and Leigh, 2007; Pascoe et al., 2010). The use of commercial catch rate data has been criticised as an abundance measure, as it can be influenced by fishing practices of commercial fishers, the economics of the fishery, the dynamics of other species in other fisheries and changes in management practices (Dulvy et al., 2003; Maunder et al., 2006; Meynecke and Lee, 2011). This study has the distinct advantage of sampling consistency over a long time period. Below it will be argued that without this consistency over such a large time-scale, some of the key correlations of environmental factors like rainfall and moon phase with the catch rates of target and non-target organisms in this tropical trawl fishery could not have been detected.

Interannual Variations in Catch Rates in the Trawl Fishery of Cleveland Bay

General

Significant inter-annual variations in mean catch (kg/ha) were found for all groups in this study. These variations affected the overall composition of the catch each year. In most years the trawl fish assemblage (in terms of wet weight caught: kg/ha) was dominated by ponyfish (Leiognathidae) but this was not always the case, with the ponyfish catch varying from 8-55% of the total catch (Figure 2.19). The “Other Fish” component, which consisted of all species except those of Leiognathidae, Carangidae, Sciaenidae and *Saurida* spp., that is, an

assemblage of over 140 species (see Appendix 4) accounted for 17-80% of the catch from year to year (Figure 2.19). Catch rates of croakers (Sciaenidae) accounted for <1-22% of the catch (Figure 2.19). Catch rates of this group varied by a factor of over 100 among years. In contrast, catch rates of Carangidae were relatively stable over the 15 year study period.

Catch Rates of the Target of the Fishery, Banana Prawns

Catch rates of prawns varied significantly among years. While in some years almost no prawns were caught, in other years prawn catch contributed about a third of the total wet weight (kg/ha) of catch in the trawl nets (Figure 2.19). While all trawls were carried out during the month of March, earlier and later trawl dates (ie: first week of March vs. Last week of March) were not taken into consideration. Banana prawns (*Fenneropenaeus merguensis*) move from inshore nursery grounds like mangroves to offshore waters to spawn (Staples and Vance, 1986; Vance et al., 1998; Vance et al., 1990). Larvae and postlarvae then move to inshore mangrove systems where they stay for several months and move offshore again as juveniles and adults (Staples and Vance, 1986; Vance et al., 1998). Commercial fisheries often target this offshore movement that occurs between March and April in the SE Gulf of Carpentaria and in some places along the east coast of Queensland (Vance et al., 1998). Even though it is well known that this movement occurs late in the tropical wet season, different environmental factors have been suggested to trigger this migration (Rothlisberg et al., 1986; Staples and Vance, 1987, 1986; Vance et al., 1985). Commercial catch data, obtained from the Department of Agriculture, Fisheries and Forestry for Grid J21 (which includes Cleveland Bay and surrounding areas), shows that banana prawn catch rates were highly variable among years and highest catch rates (in the month of March) in the

commercial fishery did not coincide with highest catch rates in this study (Appendix 5). Catch rates of endeavour and tiger prawns in northern Queensland have been shown to correlate positively with rainfall and the Southern Oscillation Index (SOI) (Meynecke and Lee, 2011). Other studies have shown a positive correlation of prawn catches with freshwater flow or rainfall in the same or previous year (Galindo-Bect et al., 2000; Gammelsrød, 1992; Vance et al., 1985). Some studies have shown significant relationships between annual prawn catches and wet season temperature (Galindo-Bect et al., 2000; Vance et al., 1998; Vance et al., 1985). Meager (2003) also found that warmer temperatures were associated with higher prawn catches in sub-tropical Queensland. It has been suggested that increased catch rates during periods of elevated temperatures might be due to increased catch ability of prawns due to an increase in prawn activity (Meynecke and Lee, 2011).

The commercial trawl fishery for prawns in Cleveland Bay is small, usually limited to a few weeks at the beginning of the prawn season (opening March 1 each year on the East Queensland coast) and only during rare years when environmental conditions produce commercially viable catch rates. Catch rates of banana prawns during this fifteen year study in Cleveland Bay were relatively high (2-4 kg/ha) in only four of the 12 years sampled (Figure 2.4). However, peaks in catch rate were high in consecutive years (1998 and 1999; 2007 and 2008). In contrast to the very clear effects of rainfall on catch rates of banana prawns in the SE Gulf of Carpentaria (Rothlisberg et al., 1986; Staples and Vance, 1985; Vance et al., 1998; Vance et al., 1985) and on catch rates of tiger and endeavour prawns on the east coast of Queensland (Meynecke and Lee, 2011) no relationship between rainfall in a specific year and prawn catch rates in that year was found in the present study (Appendix 2, Figure 2.1-2.5). Don Battersby (personal communication from the skipper of the James Kirby) suggests that catch rates of banana prawns in trawls are highest during particular moon phases and tidal heights, and is not always related to rainfall in a particular year.

Moon phase did have a significant effect on banana prawn catch rates, with significantly higher catches during periods on and around the full moon (Figure 3.13). In inferring lunar effects in this Chapter, the explicit assumption is made that lunar phase had a stronger effect on than any year effect. Catch rates were almost 8-10-fold higher on full moons than at other moon phases (Figure 3.13). Previous studies have reported that lunar phase had significant effects on catch ability of fish and crustaceans (Chatterji et al., 1994; Griffiths, 1999; Hernandez-Leon, 2008; Lowry et al., 2007; Salini et al., 2001). Catch rates of endeavour prawns (*Metapenaeus endeavouri*) were significantly higher during phases of the full moon and tiger prawns (*Penaeus esculentus*) during periods of the last quarter moon when trawling occurred at night (Salini et al., 2001). Prawns are thought to increase their activity on dark nights around the new moon as a response to reduced risk of predation (Griffiths, 1999). It has been suggested that during phases of the full moon, eastern king prawns (*Penaeus plebejus*) bury into the sediment to avoid predators (Griffiths, 1999). Results of the present study suggest that prawn activity may be increased on days of the full moon. This could be a result of increased water movement due to high tidal changes leading to increased food availability and more active prawns, although this would apply equally to new moons. Alternatively it could be that prawns are less active and perhaps buried in the sediments during the night around the full moon, attempting to avoid nocturnal predators, and hence most of their feeding activity may occur during the day, making them more vulnerable to bottom trawling. Further research is required to understand why banana prawn catch rates are so clearly enhanced on the full moon.

Catch Rates of Total Fish

Few studies until the mid-1990s in Queensland recorded the amount of fish catch in prawn trawls. There was little interest in catch rates of fish bycatch until the bycatch issue arose in Queensland in the mid-1990s (see Chapter 2). Thus, the present study is somewhat unique in that it has recorded fish catch rates in trawls almost annually over 1.5 decades. Total fish catch was highly variable among years but there was no long term trend up or down in fish catch rates over the 15 years (Figure 3.3). Catch rates of fish were mainly driven by the abundance of ponyfish (Leiognathidae), which contributed between 8% and 55% of the fish biomass (Figure 3.7). The percentage contribution of ponyfish to total fish catch was under 20% in only 3 of 12 years (Figure 3.2). The other major contributor to mean total fish catch was the “Other Fish” group, which contributed between 17% and 80% of the fish biomass (Figure 3.2). “Other fish” catch comprised over 140 different species (Appendix 4).

Rainfall in a sampling year had no significant effect on catch rate of fish in trawls in the present study. However, rainfall in the wet season two years prior to the sampling year explained 32% of the variation in catch rate of total fish (Figure 3.10). This is first study to show a significant two-year delay effect of rainfall on catch rates of fish in prawn trawls in Queensland. Further research is required to establish if this applies to other areas of Queensland. If it does, it suggests that 2 years after heavy rainfall, Bycatch Reduction Devices (BRDs) will have a lot more fish bycatch to remove from trawl nets than in other years. This may affect inter-annual performance of BRDs (see Chapter 2). Lagged effects of rainfall on catch rates in this study are somewhat consistent with the findings of Meynecke and Lee (2011) that found a positive correlation with 2 year lagged rainfall and catch rates of Barramundi, Mud Crabs and Endeavour Prawns in Queensland. However, Meynecke and Lee (2011) did not examine fish catch rates in trawls, and rainfall in the year of sampling

accounted for more of the variation in catch rates of Barramundi, Mud Crab and Endeavour Prawns than did the 2-year lagged rainfall (see Figure 4 of Meynecke and Lee 2011). The latter result suggests a far quicker effect of rainfall on catch rates than suggested in the present study. This is the first time such a trend has been described for a whole bycatch fish assemblage in a tropical trawl fishery. Because most scientific studies of trawl bycatch only ran for several months, and in rare cases up to around three years, this lag effect could not have been detected by short term data sets. The unique length of the data set in this study allowed these unique trends to be detected.

Many fish species rely on coastal and estuarine habitats as nursery grounds and often only recruit into the coastal trawl fishery after moving away from inshore nursery grounds (Allain et al., 2007; Alvarez et al., 2012; Gantias et al., 2007; Laegdsgaard and Johnson, 1995; Robins et al., 2005; Staunton-Smith et al., 2004; Travers et al., 2012; Whitfield, 2005). Freshwater flow or rainfall are thought to increase primary productivity and hence increase survivorship of post-larvae and juvenile stages of fish as a result of increased food supply and decreased predation due to higher turbidity levels (Lamberth et al., 2009). This is hypothesized to lead to strong recruitment into the sub-adult and adult populations of coastal fish (Sondita, 1997). Rainfall as well as temperature shifts also trigger migratory patterns, inducing sub-adults and adults to move to spawning grounds offshore or inducing juveniles to move offshore into new feeding grounds (Sondita, 1997). It cannot be deduced from our results that recruitment driven by rainfall is the main driver of variations in catch rates of the fish assemblage in Cleveland Bay, however it would be beneficial to investigate this idea in the future.

It is difficult to explain patterns and interannual variations in catch rates of total fish in a species-rich tropical trawl fish assemblage. Factors affecting abundance patterns are most often species-specific. Catch rates of total fish might stay relatively constant over a prolonged period of time even though the catch rates of individual species or even groups of species

changed dramatically. Changes in community structure of the fish assemblage over time will be the focus of Chapter 4 of this thesis. Attempts have been made to link abundance of fish in coastal fish assemblages with physical and environmental variability (Zeeberg et al., 2008; Meynecke and Lee, 2011). Freshwater flow and temperature have been shown to play an important role in the life cycle of many coastal fish species on the east coast of Queensland (Blaber, 1990; Gillson et al., 2009, Meynecke and Lee, 2011). Meynecke and Lee (2011) concluded that temperature most likely affected juvenile fish when they resided in estuarine systems. Structure of fish assemblages might shift as a result of temperature changes. Temperature tolerance and ideal temperature ranges are species-specific. As metabolic demand increases with higher temperatures, certain fish species will benefit from increased temperature whereas species with narrower temperature tolerance may be negatively affected (Meynecke and Lee, 2011; Pittman et al., 2004). Effects of temperature on catch rates in commercial fisheries in Queensland usually relate to seasonal effects (Meynecke and Lee 2011). Temperature effects could not be investigated in this study, since accurate sea temperature data were not available and since sampling was always carried out in March. Thus seasonal effects of temperature on fish catch rates could not be detected. Freshwater inflow into the coastal systems is also a well-known factor affecting inshore fish assemblages (Robins et al., 2005; Staunton-Smith et al., 2004; Travers et al., 2012; Whitfield, 2005).

It has been shown that short term disturbances can lead to an increase in herbivorous reef fish abundance due to increased food availability (Russ and McCook, 1999). Coastal development and runoff from agriculture, as well as alterations to freshwater flow from the construction of dams and overfishing, have also been shown to affect abundance of fish in coastal fish assemblages (Rogers and Millner, 1996; Sheaves et al., 2007)

Catch Rates of Carangidae:

Catch rates of Carangidae were the least variable of the groups among the years investigated. Even though significant inter-annual changes in catch were recorded, catch rates were generally very low, never exceeding 0.7kg/ha. Carangidae accounted for just 1-6% of the total fish catch by weight during the study. Carangiadae are semi-pelagic predators that can be found in a range of habitats from estuaries to coral reefs (Sfakiotakis et al., 1999; Gembella and Treiber, 2003). Many carangids occupy a range of inshore habitats when juvenile and often migrate offshore to reefs as adults. Carangids can tolerate a wide range of environmental conditions, ranging from estuarine environments to open ocean conditions (Wardle et al., 1995; Gemballa and Treiber, 2003). This means that they will be less affected by changes in environmental parameters such as salinity and temperature fluctuations. This coupled with relatively low overall catches of this group may explain the small variability in catch rates over a prolonged period of time. The fact that the study found no correlation between environmental parameters and carangid catch rates further supports this suggestion.

Catch Rates of Sciaenidae

Mean catch rates (kg/ha) of Sciaenidae varied significantly and substantially among years, with catch rates of this family varying 122-fold during the study. Sciaenids are a benthopelagic predator that are found in coastal waters where they prey on small fish and invertebrates such as prawns (Lazo et al., 2010; Sasaki et al., 2001). As one of the main foods of sciaenids are small fish and prawns, it is possible that sciaenids abundance would be affected by food availability and hence prawn abundance. The present study could not find

such a correlation. However the year with the highest catch rates of sciaenids (2012) was also the year with the lowest pony fish catches, which are a potential prey item for sciaenids (Sasaki, 2001). Sondita (1997) has suggested that high numbers of predators can have enhanced effects on abundance of prey species.

Catch rates of Sciaenidae were affected by total wet season rainfall two years prior to trawling (Figure 3.11). Two year lagged rainfall explained 28.2% of the total variation in Sciaenidae catch rates. This is consistent with the findings of Meynecke and Lee (2011) who found the same trend for Barramundi on the east coast of Queensland. Examination of otoliths of sciaenids in these trawl catches in Cleveland Bay, show that many fish caught by trawls are often 1-2 years of age (Garry Russ personal communication). This supports the hypothesis that heavy rainfall may enhance nutrient input and thus productivity in the bay, resulting in enhanced larval and juvenile survivorship, and thus enhanced recruitment of these fish to the trawl sampling around 2 years later.

Catch Rates of *Saurida* spp.

Inter-annual variations in catch rates (kg/ha) of *Saurida* spp. were highly significant even though catch rates were generally low (~ 0.3 kg/ha). *Saurida* spp. contributed only a small amount (0.1 to 4% of catch by weight) of the total fish catch. Being a benthic ambush predator that often buries in the sand or mud waiting for prey, these fish are very vulnerable to bottom trawling (Blaber et al., 1990; Kennelly, 1995; Van der Geest 2000). They are known to feed on small fish and bottom dwelling crustaceans (Sondita, 1997). A possible explanation for the very low catches in 2012 was that prey availability, particularly prey like ponyfish and prawns, was also very low in this year and abundance of other predators such as Sciaenidae was high. Intraspecific competition between predators could explain this very low

catch rate of *Saurida* spp. (Sondita, 1997). Moon phase was found to have a significant effect on catch rates (kg/ha) of *Saurida* spp. Catch rates at periods on or around the full moon were significantly higher than around the new and first quarter moon phases. This could be due to the higher abundance of prawns during periods of the full moon (Figure 3.12). Increased water movement as a result of increased tidal changes during periods of the full moon might enhance food supply out of the coastal mangrove systems and estuaries, and hence give *Saurida* spp. a good opportunity to increase consumption of prey. However this would also hold true for periods around the new moon and is hence unlikely to be the major driver of this trend.

Catch rates of Leiognathidae

Leiognathidae were a major contributor (8-50% by weight of total catch) to total mean fish biomass (kg/ha) of the fish assemblage in Cleveland Bay. Inter-annual variations in catch rates were highly significant, showing that the leiognathid assemblage is strongly influenced by year to year variations in survival and recruitment. Similar results were found by Cabanban (1991) in a 3 –year study of trawl fish in Cleveland Bay. Leiognathidae are small demersal zoobenthic feeders and planktivores. They are schooling fish that occupy coastal areas and are often found over sandy and muddy bottoms (Cabanban, 1991; Sondita, 1997). They are known to often enter estuarine environments (Jones, 1985). This makes them relatively well-adapted to environmental changes such as fluctuations in temperature and salinity and no environmental variable was found to significantly explain variations in catch rates of Leiognathidae in this study. Being a small schooling fish they are common prey of a range of predatory fish, and variations in catch rates may be due to changes in predation rates and changes in recruitment success driven by changes in productivity among years. Because

they are schooling fish it is also possible that inter-annual variations in catch rates are due to haphazard capturing of large schools in some years compared to others (Cabanban, 1991).

Catch rates of “Other Fish”

The “Other Fish” group was the dominant component of fish catch in Cleveland Bay, accounting for between 18% and 80% of the total fish catch by weight during the study. Catch rates (kg/ha) of this group varied significantly among years. However there were no long term trends in catch rates of this group. Similar findings were made by Sondita (1997) in a study of trawl fish in Cleveland Bay over 3 years. The fish assemblage in Cleveland Bay is very diverse, with 175 fish species recorded in the bay (Sondita, 1997; Van der Geest, 2000; Appendix 4). Even though some of these species are extremely rare, inter-annual variations in the fish assemblage are difficult to understand when species level changes are not examined carefully. Different species will respond to different environmental cues over time and in space. The fish community in the bay is dominated by juvenile fish (Sondita, 1997), and hence changes in abundance of components of the fish assemblage are most likely related to changes in recruitment success. It has been suggested that due to the wide range of habitats that are associated with Cleveland Bay and surrounding areas (estuaries, extensive mangrove systems, seagrasses, shallow reefs, soft substrata etc.) fish populations are relatively resilient to physical and environmental disturbances due to this high habitat variability that benefits juvenile and larval fish (Sondita, 1997).

Two year lagged rainfall explained a significant (55 %) proportion of the variation in mean catch rate (kg/ha) of “Other Fish” among years. As the fish assemblage is dominated by juvenile fish, high recruitment success in certain years could explain recruitment into the fishery as juveniles two years after such events (Friedlander and Parrish 1998). Similar lagged effects have been shown on Barramundi (in the gill net fishery) by Meynecke and Lee

(2011). To better understand these inter-annual changes, research has to be focused on single species trends as each species is affected in different ways by short and long term environmental changes.

The multispecies fish assemblage in Cleveland Bay is a very dynamic one, and changes in abundances of different fish groups vary inter-annually. Long term trends in catch rates were not detected, suggesting that the fish community has remained relatively stable over time. Changes in abundance of species groups are driven by many different factors. While a lot of variations in catch rates might be related to annual fluctuations in rainfall and hence productivity, the trawl fish assemblage seems to be resilient to these changes in the long-term. Studies such as this could prove useful for fisheries management providing a good understanding of the fish community and assemblage.

Monitoring changes in the trawl bycatch fish assemblage is most informative at the species level. The present study however could only investigate broader trends for some families. Future studies should take into consideration species level data in order to detect clearer and more informative trends about changes in trawl fish bycatch catch rates. It is acknowledged that sampling for two to three days per year provides only very limited information on interannual variations in the bycatch fish assemblage. However the sampling regime of this study made it difficult to draw clearer inferences on an annual or seasonal basis. The results are consistent with local fisher knowledge of environmental factors affecting local fish and prawn populations (Battersby personal comments, skipper of the James Kirby). In the next chapter, I will examine the consistency of the trawl fish assemblage composition over time and between samples with and without a Bycatch Reduction Device.

Chapter 4

Long-term Variations in Species Composition of a Tropical Fish Assemblage

Introduction:

A wide range of interacting environmental processes, operating at multiple scales, can affect the species composition of fish assemblages (Jennings et al., 2005). These include both physical (e.g. habitat type, depth, temperature, salinity) and biological (e.g. predation, competition, food availability, recruitment variation and migration) factors. Long-term studies of variations in the species composition of temperate trawl fish assemblages exist in some northern hemisphere commercial fisheries (e.g. Daan and Gislason 2005, Jennings et al., 2002, Jennings et al., 2005) but are less common in the tropics (e.g. Travers et al, 2012). Studies have shown that the species compositions of benthic fish assemblages on soft substrata were directly linked to habitat type and depth (Fitzpatrick et al., 2012; McManus, 1986; Moranta et al., 2008). Habitat heterogeneity over a scale of hundreds of kilometres has been shown to be a surrogate of biodiversity, showing that each predefined habitat type supports a distinct fish assemblage (Harborne et al., 2008). Fitzpatrick et al. (2012) found that relative species richness and diversity of fish declined with depth while average fish length and trophic level of fish increased with depth. Toole et al. (2011) observed that seasonal shifts in fish distributions were most likely a function of juvenile settlement in nursery habitat and of ontogenetic seasonal inshore-offshore movement of individuals. Similar observations were made by Travers et al. (2012) showing that water depth plays an important role in structuring fish communities, reflecting the tendency of some species to use near shore habitats as nursery areas and often moving offshore over time.

Seagrass beds are often associated with tropical soft substrata (Poiner et al., 1989; Poiner et al., 1987). Seagrasses are amongst the most productive habitats in the world supporting diverse communities linking coastal areas to offshore areas and reefs via nutrient fluxes (Adams et al., 2006; Yeager and Arias-Gonzalez, 2008). Due to physical disturbances, shifts in benthic cover have often occurred from seagrass beds to sand and mud dominated areas (Orth et al., 2006; Waycott et al., 2005; Yeager and Arias-Gonzalez, 2008). This leads to a decrease in benthic meiofauna and shelter which in turn decreases food availability for benthic fish (Schratzberger et al., 2002; Schratzberger and Jennings, 2002; Travers and Potter, 2002).

Freshwater flow from rivers and creeks has been shown to affect fish communities in coastal waters for a variety of reasons (Costa et al., 2007; Gillanders and Kingsford, 2002; Gillson et al., 2009; Gillson and Suthers, 2012). Freshwater flow regulates physical, chemical and environmental properties of coastal waters, affecting fish during all life stages by altering habitat availability and trophic structure in estuarine and coastal systems (Darnaude et al., 2004; Gillson and Suthers, 2012; Kimmerer, 2002; Lamberth et al., 2009; Rueda and Defeo, 2003). Freshwater flow affects coastal salinity, turbidity and nutrient levels, all major potential drivers of the species composition of inshore fish assemblages (Drinkwater and Frank, 1994; Gillson and Suthers, 2012; Meynecke et al., 2008; Travers et al., 2012). Fish assemblages have also been known to vary in biomass and composition significantly with changing lunar phases in tropical estuaries and adjacent waters (Ramos et al., 2011; Salini et al., 2001). Travers and Potter (2002) reported fish species composition on seagrass beds and adjacent sand patches varied between night and day, reflecting in part a tendency for certain species to move into unvegetated areas to feed at night when the risk of predation by visual predators would be reduced.

Chronic and widespread disturbances to benthic habitat are caused by bottom trawling (Auster and Langton, 1999; Auster et al., 1996; Christensen, 2005; Dayton et al., 2003; Dayton et al., 1995; Freese et al., 1999; Thrush et al., 1998; Thrush and Dayton, 2002). Trawling can decrease abundance and cover of benthic macrofauna, with larger and habitat forming macrofaunal species like sponges and gorgonians showing greater sensitivity to such disturbance than smaller organisms (Schratzberger et al., 2002; Schratzberger and Jennings, 2002). This can result in smaller, short-lived benthic organisms with a high turnover to occupy areas that are trawled frequently (Jennings et al., 2005; Schratzberger et al., 2002; Schratzberger and Jennings, 2002). Changes in benthic habitat structure and complexity can lead to changes in recruitment of particular fish species, as many prefer to recruit into a particular habitat type, thus affecting the species composition of the fish assemblages in the long run (Feary et al., 2007; Toole et al., 2011).

Fish communities are generally composed of both resident and transient species (Belmaker, 2009; Boisnier et al., 2010). Resident species occupy the same habitat during their juvenile and adult life stages, while transient or temporary species occupy different habitats, or are not confined to the same habitat during different ontogenetic stages (Boisnier et al., 2010). Large proportions of fish species found in shallow water soft sediment habitats can be transient or temporary. Fluctuations in the abundance of these species can affect the species composition of tropical inshore fish assemblages on soft substrata substantially (Boisnier et al., 2010; Sondita, 1997). Short-term, temporal changes in species composition of shallow water fish assemblages have been reported to also be affected by tides and diel factors (Blaber et al., 1990; Primo et al., 2012; Reis et al., 2011).

Seasonal changes in species composition of fish assemblages have been reported by many studies (Alvarez et al., 2012; James et al., 2012; Jung et al., 2010; Moranta et al., 2008; Rotherham et al., 2011; Toole et al., 2011; Wantiez et al., 1996; Watson et al., 1990; Wilson

and Sheaves, 2001). Seasonal variations in species composition of fish assemblages are often correlated with temperature and salinity levels, as well as changes in nutrient inputs and hence productivity, leading to changes in recruitment success, predatory/prey interactions, fish growth and larval and spawning migrations (Darnaude et al., 2004; Gillson and Suthers, 2012; Kennen et al., 2005; Kimmerer, 2002; Lamberth et al., 2009; Sheaves et al., 2007; Travers et al., 2012; Vinagre et al., 2007; Vonesh and Kraus, 2009).

Inter-annual variations in the species composition of tropical inshore fish assemblages have often been attributed to changes in environmental factors such as temperature, salinity and nutrients, but also to changes in recruitment success. (Caley, 1995; Caley et al., 1996; Kennen et al., 2005; Lamberth et al., 2009; Machias et al., 2004; Sheaves et al., 2007; Vonesh and Kraus, 2009).

Sondita (1997) reported that the shallow water fish community in Cleveland Bay remained relatively stable over a period of three years but had distinct changes in relative abundance and species composition among seasons. The present study examines the question of inter-annual variations in species composition of the tropical inshore benthic fish assemblage in Cleveland Bay over a much longer time period. Trawling can at times be a quite non-selective fishing gear and thus may be useful to sample the species-rich tropical bycatch fish assemblage here. Bycatch reduction devices (see Chapter 2), by definition, aim to make trawl nets more selective in the way they fish. Thus, one might predict that BRDs may affect the species composition of trawl catches.

The specific aims of this chapter are:

1. To measure variations in the species composition of the tropical demersal fish assemblage in Cleveland Bay over a 15 year period (1998-2012).
2. To examine if the Jones-Davis bycatch reduction device (see Chapter 2) affects the species composition of the fish catch in prawn trawls in Cleveland Bay.

Materials and Methods:

Field Sampling:

The same field sampling methods described in Chapter 2 were used in this chapter. All fish species captured by trawling in March from 1998-2012 were frozen and identified later in the laboratory. Fish were identified to species level where possible using Sainsbury et al. (1985) and Gloerfelt-Tarp and Kailola (1984). A very reliable list of species trawled in Cleveland Bay, made into a visual guide to Families for students, was also available, compiled by Michael Fogg based on very detailed studies by Cabanban (1992), Sondita (1997) and Van der Geest (2000). In addition, keys to the Family Leiognathidae (Cabanban, 1992) and Carangidae were used to help identify fish to species level. Since identification of individual fish was carried out by students, under close supervision by Professor Garry Russ, only identifications thought to be reliable were used for analysis in this chapter. A full list of all species caught and identified with confidence is provided in Appendix 4. As for Chapter 2, fish captured in both trawl nets (Jones-Davis BRD/ Control) were used in the analysis. This allowed comparison of the species composition of the fish assemblage between a BRD net and a non-BRD net. No weight or abundance data for individual species were collected either on the vessel or in the laboratory. Thus, the analysis was based on presence/ absence of species among years, and between BRD and control (no BRD) nets. It is acknowledged that

abundance data for each species would be a much more effective way of determining relative changes in fish assemblage structure but the sampling regime only recorded presence/absence data. Thus, interpretations on changes in the fish assemblage structure in Cleveland Bay are very conservative. Data from all trawls carried out on a particular day were pooled. Thus, species composition of trawl catches was investigated at the level of day, not individual trip within a day or trawl. As a result all interpretations referred to as ‘interannual variations’ in the fish assemblage were based on observations made at only one particular weekend in March each year. Thus, interpretations about changes among years have to be interpreted with caution. Table 2.1 in Chapter 2 provides a table of replicate trawls per year and day.

Data Analysis:

Data were analysed using the statistical package PRIMER 6. To examine differences in species composition of the fish assemblage among years, an ordination technique was used. As multivariate data were in the form of presence/absence, no transformations were needed for a Multi-dimensional Scaling analysis using the Bray-Curtis similarity index. SIMPER analyses were used to investigate which species accounted for the dissimilarities among years (1998-2012) or between BRD+ and BRD- nets, in terms of presence absence of species. The species accounting for the top 25% of cumulative dissimilarities among years or between BRD+/BRD- nets was chosen arbitrarily to discern which species were most important in determining multivariate differences. The top 25% of species was thought to be sufficient as a guide to which species explained species compositional differences among years over the 15 year study period (1998-2012) and between BRD+ and BRD – nets.

Results:

Species Recorded in the Study

A total of 160 species were recorded in this study. Twenty-three were observed on only one occasion in 15 years (Appendix 3, Table3.13). Only 5 species were caught on every day over the 15 year study period (*Caranx bucculentus*; *Leiognathus bindus*; *Secutor ruconius*; *Thryssa hamiltonii*; *Upeneus sulphureus*).

Inter-Annual Variations in Species Composition of the Fish Assemblage

The species composition of the fish assemblage in Cleveland Bay remained relatively stable in multivariate space (Figure 4.1). The Multi-dimensional scaling (MDS) plot showed that the multispecies assemblage did not have any systematic shift in species composition over the 15 years of sampling (Figure 4.1). The MDS showed that species composition changed among years but had no long term trend or specific direction over time in multivariate space (Figure 4.1). Certain years such as 1998 showed large variations in community composition between trawling days. This year had the least replicate trawls per day. Species composition in years such as 2008, 2010 and 2012 formed tight clusters, showing that the species composition in these years was more distinct compared to other years.

To emphasize the lack of trend or direction of change of species composition of the fish assemblage in multivariate space, it is informative to examine subsets of the data. When just 2010-2012 data were examined the MDS suggested a systematic temporal trend (2D- Stress: 0.13) in species composition in multivariate space over the three years (Figure 4.2). The same applied to the data from just the period 2006-2008 (Figure 4.3; 2D-Stress: 0.11). Adding the 2009 data reversed what appeared to be a systematic directional change in species

composition in multivariate space (Figure 4.4; 2-D Stress: 0.15). Adding the 2010 data showed that the species composition of the fish assemblage had returned to the general area of multivariate space from which it began in 2006 (Figure 4.5; 2-D Stress: 0.17). Thus, the species composition of the fish assemblage showed no evidence of any systematic, directional shift of species composition in multivariate space over the longer-term. The BRD had no clear, consistent effects on the species composition of the fish assemblage in Cleveland Bay during the study (Figure 4.6). In 8 of 12 years there appeared to be evidence of some differentiation of species composition between BRD and control nets (1999, 2003, 2004, 2007-2010 and 2012), in 4 of 12 years this was not clear (1998, 2002, 2006, 2011) (Figure 4.6).

Species Accounting for Variations in Species Composition of the Fish Assemblage among years:

Year	Dissimilarity to all other years	# of species accounting for top 25% dissimilarity	Comments
1998	36.58%	17	Only year in which <i>Torquigener pallimaculatus</i> was present.
1999	33.13%	16	-
2002	33.33%	17	Catches of <i>Sillago analis</i> , <i>Sillago maculata</i> particularly common
2003	33.82%	17	-
2004	34.11%	18	-
2006	35.49%	17	-
2007	34.53%	16	-
2008	37.24%	16	-
2009	33.85%	15	<i>Lagocephalus spadiceus</i> contributed the most dissimilarity (2.52%) of any species in any given year
2010	36.61%	18	-
2011	34.40%	18	-
2012	35.66%	15	<i>Saurida undosquamis</i> absent from all trawls

Table 4.1: Average dissimilarity (expressed as a percentage) between a specific year and all other years and the number of species accounting for the top 25% of the dissimilarity identified by SIMPER. (For detailed SIMPER tables see Appendix 3, Tables 3.1-3.12)

Variation in Species Composition between BRD and CONTROL nets

The results of the MDS (Figure 4.6) and SIMPER analysis (Table 4.2) showed that 19 species account for 25.84% of the dissimilarity between the BRD net and the control net over the 15 year study period. Four of the top six species that accounted for differences in species composition of the fish assemblage in the BRD and control nets were fast, strong swimming carangids and scombrids (*Scombermorus commerson*, *Rastrelliger kanagartha*, *Alectis indicus* and *Scomberoides tol*) (Table 4.2). In 3 of these 4 cases, the species was more commonly recorded in the control net (Table 4.2).

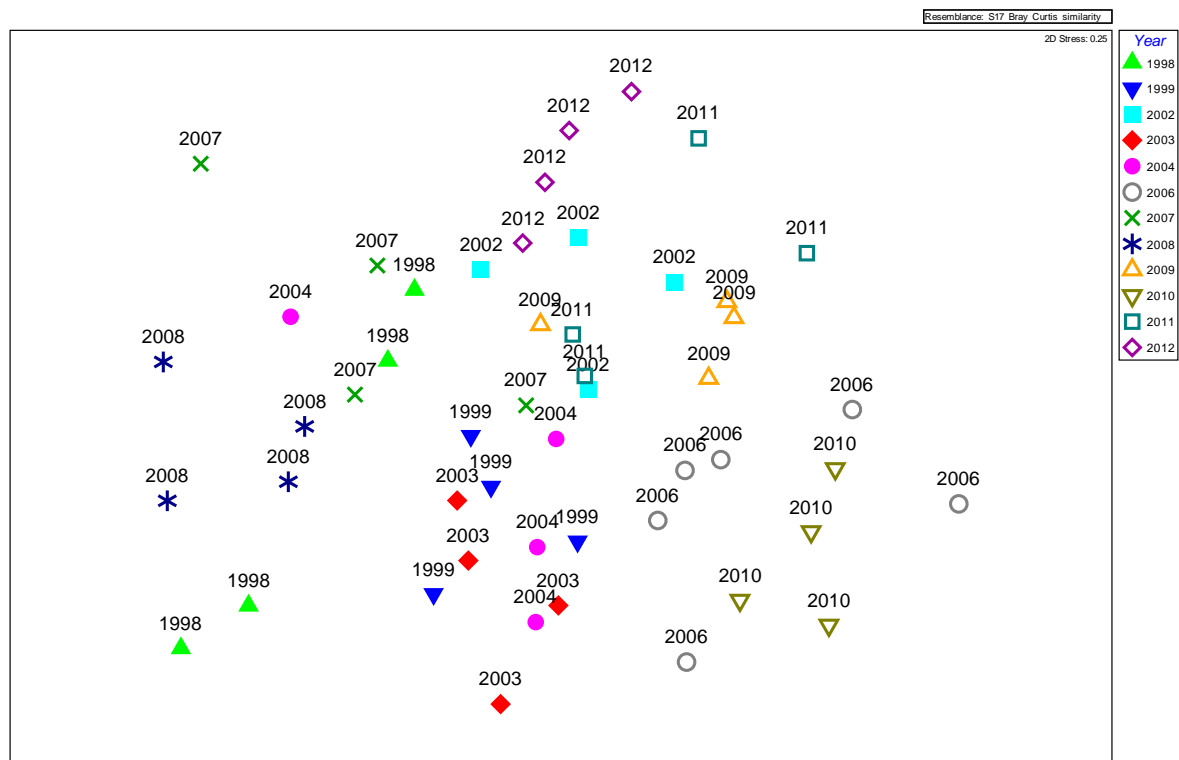


Figure 4.1: Ordination of the multispecies fish assemblage in Cleveland Bay from 1998 – 2012. Multi-dimensional Scaling was undertaken using the Bray Curtis similarity index. Different years are colour coded and represented by different symbol types. A symbol type within a year (usually 4 replicates within a year) shows the trawl samples on separate days of sampling. To see which symbols relate to BRD +/- see Figure 4.6.

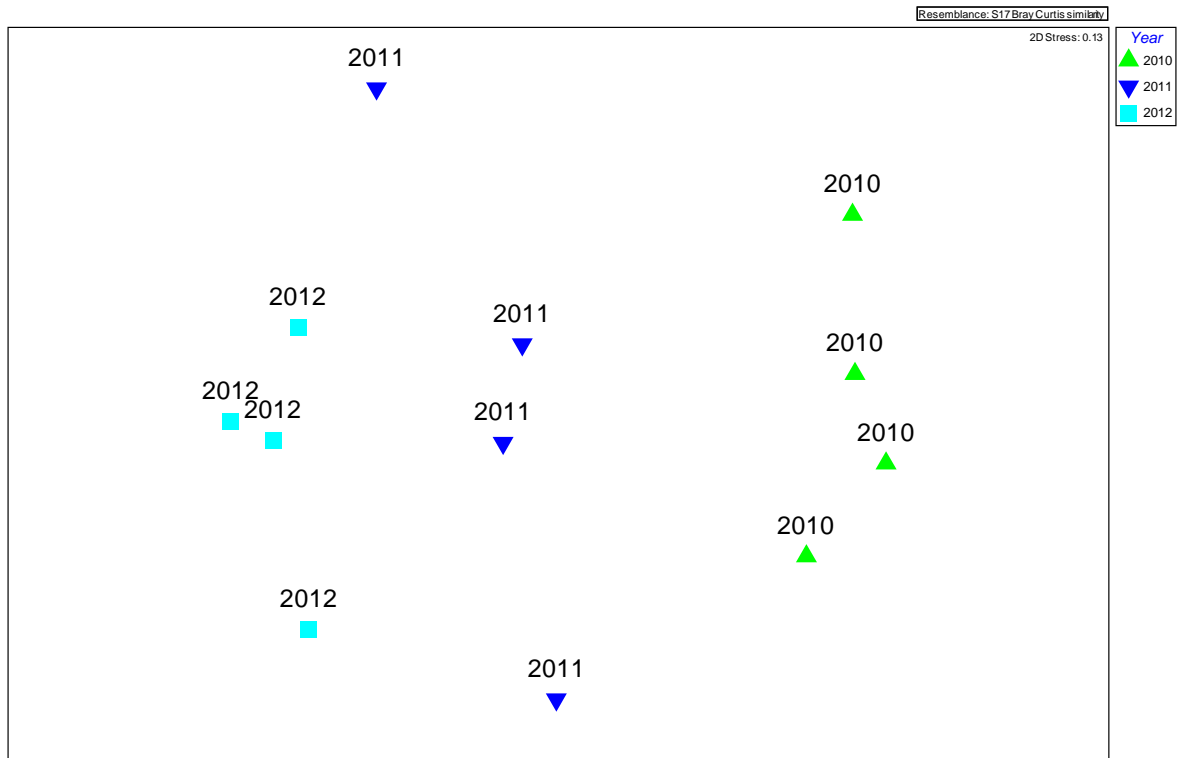


Figure 4.2: Ordination of the multispecies fish assemblage in Cleveland Bay from 2010 – 2012. Multi-dimensional Scaling was undertaken using the Bray Curtis similarity index. Different years are colour coded and represented by different symbol types. A symbol type within a year shows the trawl samples on separate days of sampling. To see which symbols relate to BRD +/- see Figure 4.6.

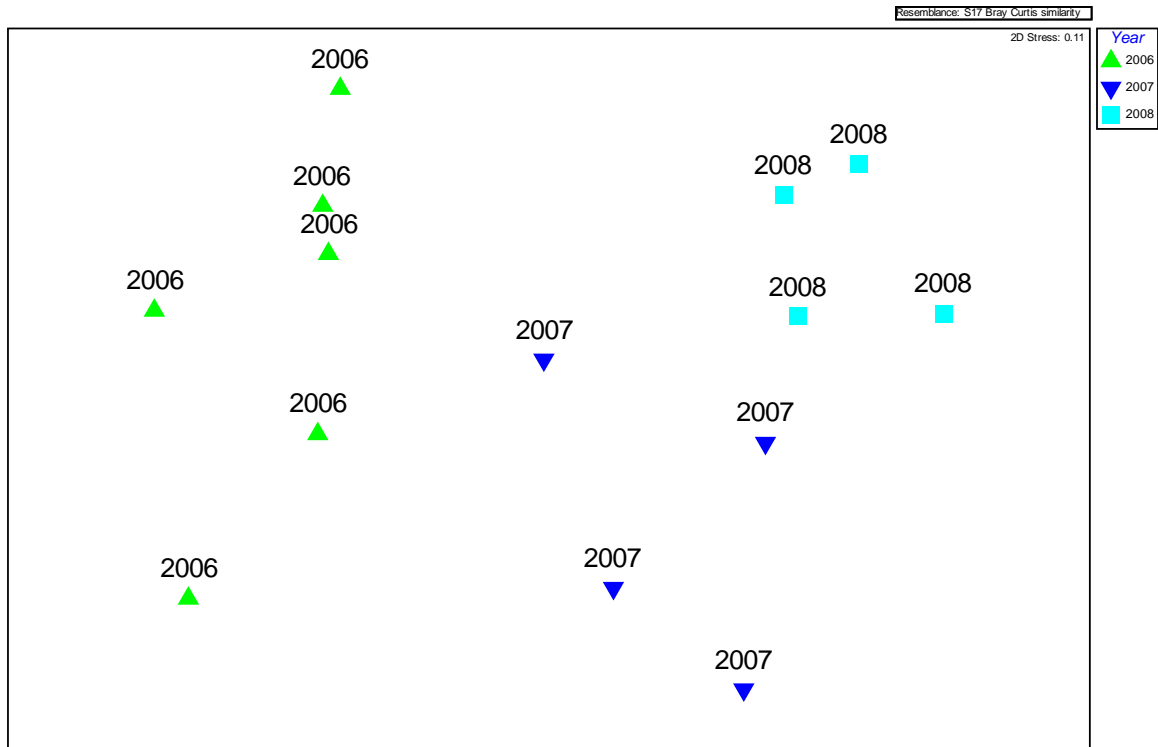


Figure 4.3: Ordination of the multispecies fish assemblage in Cleveland Bay from 2006–2008. Multi-dimensional Scaling was undertaken using the Bray Curtis similarity index. Different years are colour coded and represented by different symbol types. A symbol type within a year shows the trawl samples on separate days of sampling. To see which symbols relate to BRD +/- see Figure 4.6.

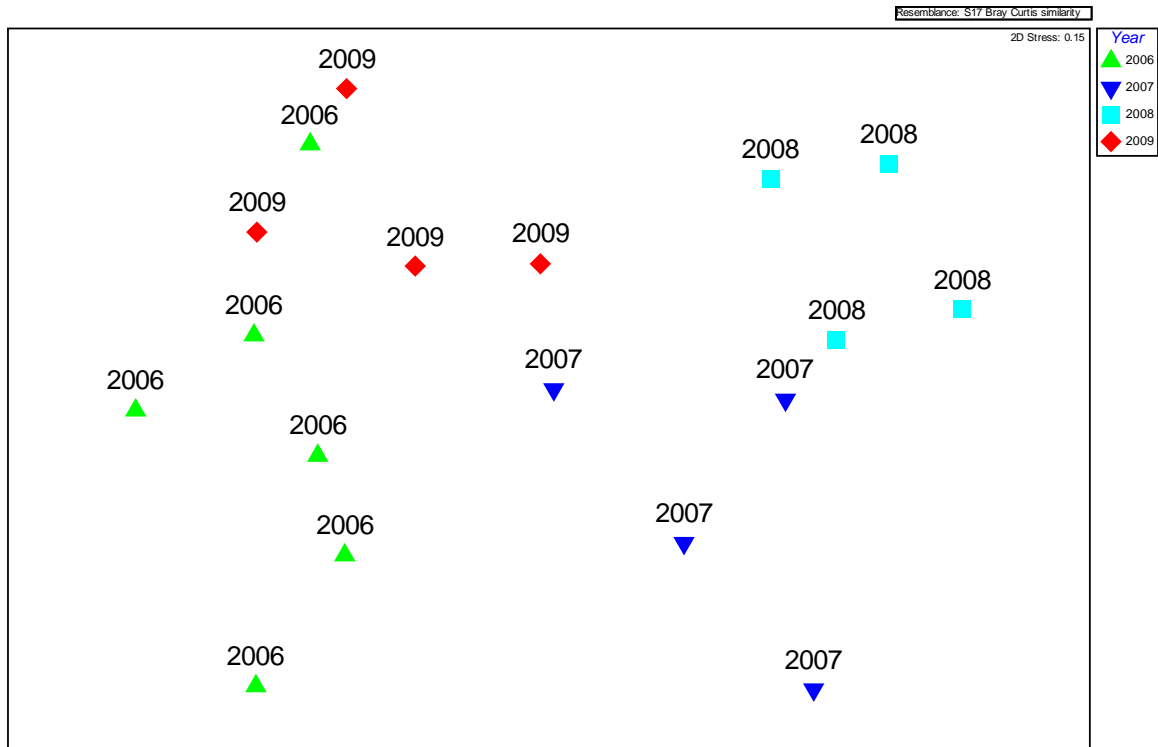


Figure 4.4: Ordination of the multispecies fish assemblage in Cleveland Bay from 2006 – 2009. Multi-dimensional Scaling was undertaken using the Bray Curtis similarity index. Different years are colour coded and represented by different symbol types. A symbol type within a year shows the trawl samples on separate days of sampling. To see which symbols relate to BRD +/- see Figure 4.6.

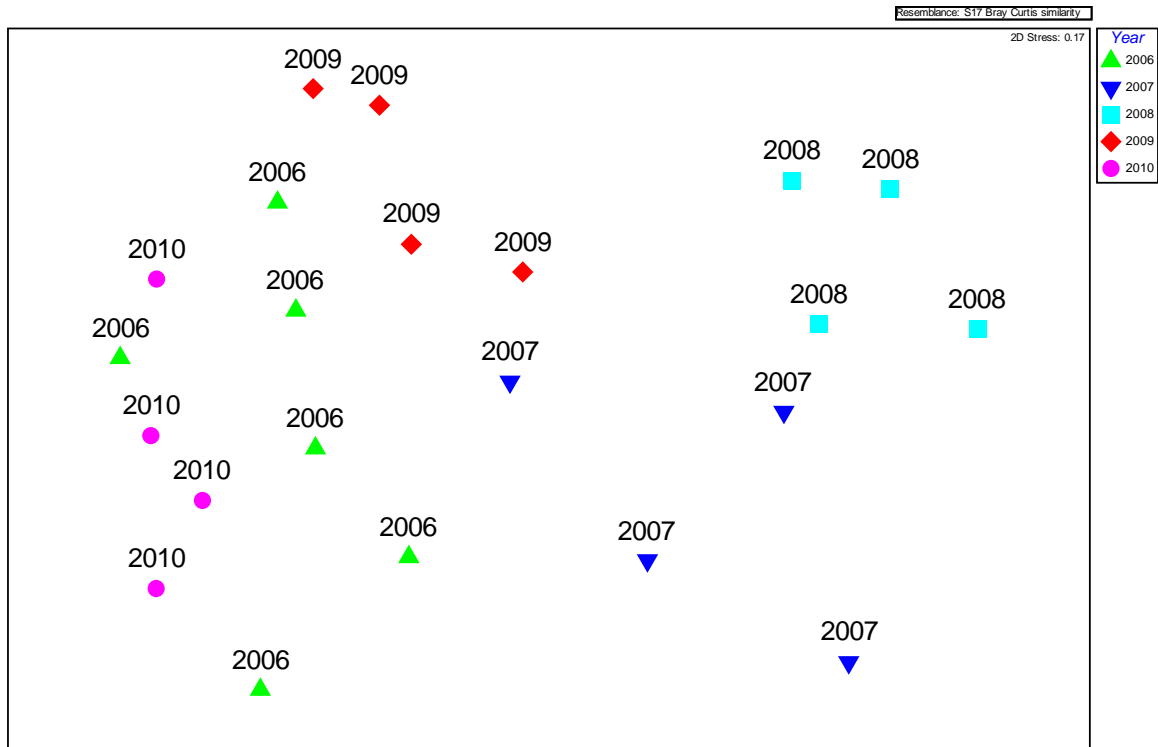


Figure 4.5: Ordination of the multispecies fish assemblage in Cleveland Bay from 2006 – 2010. Multi-dimensional Scaling was undertaken using the Bray Curtis similarity index. Different years are colour coded and represented by different symbol types. A symbol type within a year shows the trawl samples on separate days of sampling. To see which symbols relate to BRD +/- see Figure 4.6.

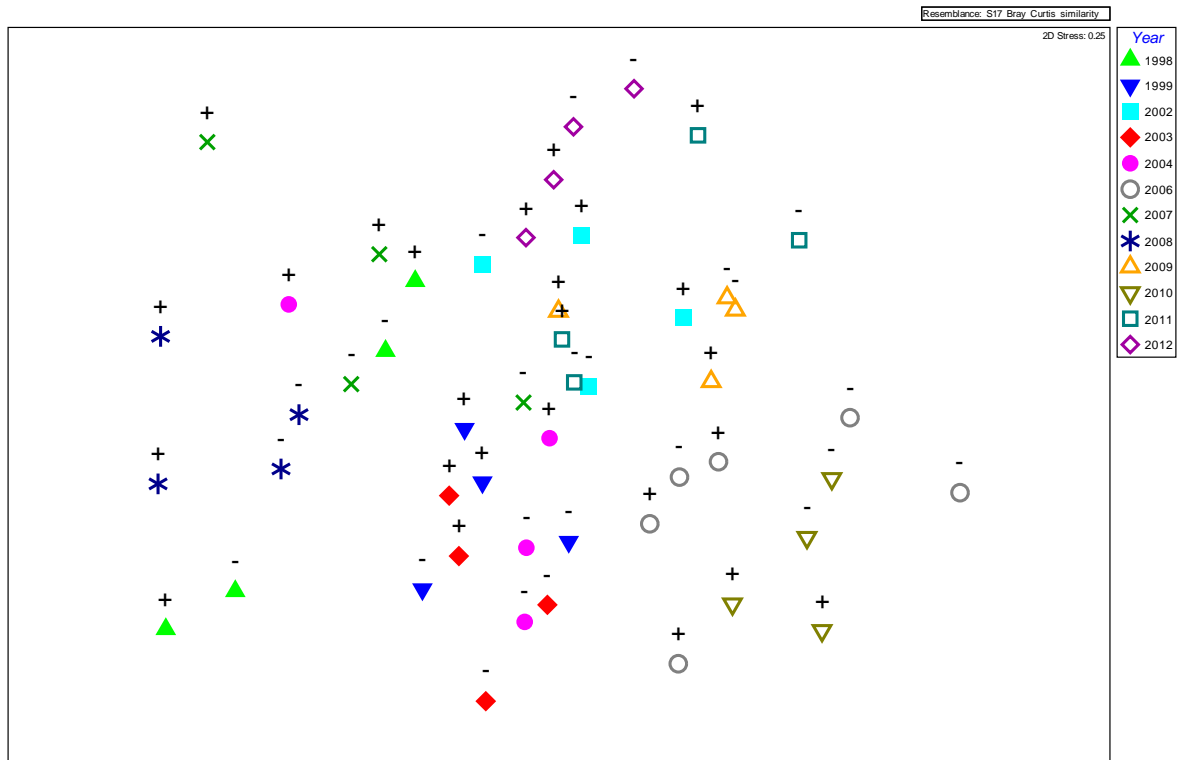


Figure 4.6: Ordination of the multispecies fish assemblage in Cleveland Bay comparing BRD (+ symbols) and Control (- symbols) net catches. Multi-dimensional Scaling was undertaken using the Bray Curtis similarity index. Different years are colour coded and represented by different symbol types. A symbol type within a year shows the trawl samples with and without a BRD on separate days of sampling.

Species	Group +	Group -	Av.Diss	Diss/SD	Contrib%	Cum.%
	Av.Abund	Av.Abund				
<i>Scomberomorus commerson</i>	0.32	0.68	0.52	1.13	1.5	1.5
<i>Rastrelliger kanagurta</i>	0.32	0.6	0.49	1.06	1.43	2.92
<i>Anodontosomachacunda</i>	0.44	0.64	0.47	1.02	1.38	4.3
<i>Alectis indicus</i>	0.64	0.44	0.47	1.02	1.38	5.68
<i>Paraplaygusiabilineata</i>	0.64	0.44	0.47	1.02	1.37	7.05
<i>Scomberoides tol</i>	0.48	0.8	0.47	1.01	1.37	8.42
<i>Chirocentrus dorab</i>	0.44	0.6	0.47	1.01	1.36	9.78
<i>Chelonodon patoca</i>	0.64	0.48	0.47	1	1.35	11.13
<i>Lutjanus malabaricus</i>	0.56	0.44	0.46	1	1.35	12.48
<i>Johnius amblycephalus</i>	0.68	0.48	0.46	1	1.35	13.83
<i>Trixiophichtys weberi</i>	0.48	0.6	0.46	1	1.35	15.18
<i>Pelates quadrilineatus</i>	0.52	0.36	0.46	1	1.35	16.52
<i>Nematalosa come</i>	0.36	0.52	0.46	1	1.34	17.87
<i>Arius thalassinus</i>	0.56	0.48	0.46	0.99	1.34	19.2
<i>Gerres subfasciatus</i>	0.48	0.56	0.46	1	1.34	20.54
<i>Gazza aachlamys</i>	0.48	0.52	0.46	0.99	1.33	21.87
<i>Sphaerena putamae</i>	0.52	0.6	0.46	0.98	1.33	23.2
<i>Pelatessexlineatus</i>	0.48	0.32	0.45	0.97	1.32	24.52
<i>Leiognathus moretoniensis</i>	0.56	0.56	0.45	0.98	1.32	25.84

Table 4.2: Species identified by Simper analysis that accounted for the cumulative top 25% of species differences between the BRD equipped net and the control net. The average dissimilarity between BRD and control nets over the 12 sampling years was 34.42% (this included all 160 species).

Discussion:

Studies have documented long-term changes in the species composition and community structure of temperate and some tropical fish assemblages and attributed the changes to environmental variations and anthropogenic disturbances such as fishing pressure (Daan et al., 2005; Feary et al., 2007; Genner et al., 2004; Henderson et al., 2011; Lison de Loma et al., 2009; Piet and Rijnsdorp, 1998; Rogers and Ellis, 2000; Stobart et al., 2009; Villéger et al., 2010). The present study suggests that there have been no such long-term changes in species composition of the fish assemblage in Cleveland Bay over the period 1998-2012. Over the 15 year study period the species composition of the fish assemblage did not display any systematic, directional shift over time. The present study however only included presence/absence data and thus should be interpreted conservatively. Souda (1997) reached a similar conclusion about this Cleveland Bay trawl fish assemblage, but his study was only of 3 years duration. The findings of the present study could only have been obtained by collecting a long-term (decadal-scale) data set and using consistent sampling methods over this time.

Inter-annual changes in the species composition (in terms of presence and absence of species) of the fish assemblage did occur in Cleveland Bay, at least as detected from a sampling scheme restricted to March each year. These inter-annual changes in species composition of the fish assemblage, if examined over relatively short-time scales of 3 years, could be interpreted as directional shifts in community composition. For example, examining the data collected from just the period 2006-2008 (Figure 4.3) one could conclude that there was a systematic shift in species composition in multivariate space. However, adding the 2009 and 2010 data reverses what appeared to be a systematic directional change in species composition in multivariate space (Figure 4.5). Thus, the species composition of the fish

assemblage appears to oscillate somewhat haphazardly within a large “bubble” of multivariate space, rather than show any systematic, directional shift of species composition in multivariate space over time.

Since many environmental and biological factors affect the species composition of fish assemblages, changes in composition, especially permanent, long-term changes, can only be interpreted correctly if long-term monitoring, using consistent sampling is implemented. As shown in the previous chapter, many mechanisms such as lags in recruitment might operate over a period of several years and hence changes in species composition of fish assemblages over short time periods should be interpreted with caution. One possible interpretation of this study is that no major anthropogenic or environmental disturbances have affected the species composition of the fish assemblage in Cleveland Bay in the last 15 years. Another would be that the fish assemblage in Cleveland Bay is relatively resilient to disturbances or environmental changes. It is possible that Cleveland Bay and the surrounding coast line contain such a large variety of complex habitats (extensive soft-sediment areas, seagrass beds, inshore reefs, mangrove lined coastlines, extensive wetlands), that the fish assemblage is relatively resilient to disturbances due to a buffering effect of these extensive and varied habitat types. Other factors potentially contributing to this stable fish assemblage are the high productivity and the relative predictability of physical conditions in Cleveland Bay (Sondita, 1997). Cleveland Bay is highly productive because a range of river systems discharge into the bay. During the summer wet season, high loads of nutrients are discharged into the bay. The topography of the bay (bordered on the south by Cape Cleveland, to the north by Magnetic Island) makes it relatively sheltered from ocean currents and wave action, limiting nutrient export out of the bay. Cleveland Bay is also a relatively pristine area in terms of commercial fishing effort. Because the fish community in the bay is dominated by small demersal species (for example Robertson and Duke (1990) reported that 97% of the demersal fish community

in the bay is made up of juvenile fishes) not targeted by any fishery in the region, fishing impacts on the species composition of the fish assemblage in the bay is negligible. This study only obtained presence absence data so it was not possible to tell if the relative abundance of species shifted over the last 15 years. Based on the results obtained in Chapter 3, it could be suggested that the community structure of the fish assemblage, while variable among years, has remained stable over the last 15 years.

The species composition of the fish assemblage in Cleveland Bay varied inter-annually. The Multi-dimensional scaling plots shows that species compositional changes do occur on a year to year basis (with the caveat that sampling was restricted to March each year, and that findings are only based on presence/absence of species) with some years showing some degree of differentiation of species composition. Species composition appeared more distinct in some years (e.g. 2008, 2012). This could have been due to the presence of some rare species and the absence of some common species. An influx of uncommon species and the absence of normally common species are the most likely explanation for the degree of differentiation for these two years. This could be due to random migratory events in or out of the bay in these years, annual variations in recruitment success driven by environmental factors, or changes in timing of recruitment depending on wet season conditions and timing of freshwater flow. Recruitment success has been shown to be one of the main drivers of assemblage structures and is dependent on reproductive output, oceanographic conditions, olfactory cues, availability of habitat suitable for settlement of post-larvae and juveniles, food availability, intra- and inter-specific competition and predation rates (Caley, 1995; Caley et al., 1996; Feary et al., 2007; Hay, 2009; Moyano and Hernandez-Leon, 2011; Planque et al., 2007; Trape et al., 2009).

Annual changes of bay conditions driven by rainfall or cyclones (three tropical cyclones affected the Townsville region during the 15 year study) would have had direct effects on

recruitment timing and success in the Cleveland Bay fish assemblage (Lewis et al., 2011; Lin and Shao, 1999; Loneragan and Bunn, 1999; Russ and McCook, 1999; Vinagre et al., 2007). This however is most important for species that use the bay and adjacent mangrove systems and wetlands as nursery areas, such as for example some leiognathids (Cabanban, 1991; Sondita, 1997). The majority of species that are found in the bay however will use the extensive coastal ecosystem mosaic at some stage during their ontogenetic development (Sondita, 1997). For many species recruitment success into the trawl sampling is likely to only be detected one or two years after larval settlement as suggested in Chapter 3.

Even though the MDS did not show any significant differences in species composition of the fish assemblage between the BRD and the Control net, the main dissimilarities seem to be driven by strong-swimming demersal and semi-pelagic species. This result is consistent with the objective of the Jones-Davis BRD to reduce the catch of strong swimming fish (see Chapter 2). Four of the six species accounting for most of the dissimilarity in species composition between the BRD and the control net were carangids and scombrids, strong swimming fish (Table 4.2). Note that this study was based on presence absence of species, limiting the resolution of the study. Thus a single individual of a particular species in each net would cause no difference between nets in that particular year.

Benthic soft bottom fish communities are highly dynamic and affected by many processes, both natural and anthropogenic (Sondita, 1997). Many studies reporting changes in the species composition of fish assemblages are conducted over relatively short periods of time. Such studies may fail to detect changes or falsely interpret changes in the short-term to be major assemblage shifts maintained in the long-term. Many studies reporting shifts in species composition of fish assemblages are short-term. Results of such studies need to be interpreted with caution, unless clear reasons for change are detected. The present study may suggest that fish assemblage structure can change between years but remain stable over many years. The

former is almost to be expected due to the fact that environmental conditions and fish recruitment characteristically vary inter-annually. The latter, systematic shifts in the species composition of fish assemblages over time, versus stability in the longer-term, are much more difficult to differentiate.

Some studies have compared catch data taken over a short period well-back in the past, to modern catch data, also taken over a short period (e.g. Harris and Poiner, 1991). This again could potentially confound interpretation of results, as inter-annual changes are not taken into account and shifts in community composition of catch could be due to inter-annual, as opposed to long-term change. This shows just how important long-term monitoring programs, using a consistent methodology, are if one aims to document temporal changes in fish assemblage structure over time reliably.

Collecting more appropriate data would have been preferable to document variations in the fish assemblage structure in this study. Weight and abundance measures for all species caught would have allowed measurement of changes in relative abundance over time. Such data would have allowed more informed statements about fish community structure (e.g. shifts in relative abundance and trophic structure). Furthermore, data collected at other times of the year than just March would have strengthened this study, and buffered the study against confounding sampling variation caused by short-term (a few days) sampling with inter-annual variations in the species composition of the fish assemblage. In addition, collection of environmental parameters such as temperature, salinity, cloud cover, turbidity, wave height and bottom composition for all trawls would have allowed better inferences about what drivers affected catch composition. These types of data can sometimes be obtained post-hoc from monitoring stations maintained by institutions such as the Australian Institute of Marine Science (AIMS) or the Great Barrier Reef Marine Park Authority (GRMPA), however these stations are often situated well-away from the area of sampling and

may provide inaccurate estimates of the environmental conditions experienced during trawling.

The findings of this study suggest that the species composition of the fish assemblage in Cleveland Bay, whilst consistent in species composition in the long-term, varies inter-annually. It is difficult to specify what causes these inter-annual variations. The results support the findings of a previous study (Sondita, 1997) that suggested that the species composition of the fish assemblage in Cleveland Bay was relatively stable over a 3 year period. The study highlights the importance of long term, consistent, fishery-independent monitoring programs, and could be used to help guide future research that would allow more accurate long-term predictions of variation of and change in the species composition of tropical trawl-fish assemblages.

Chapter 5

General Discussion

The non-selective nature of many fishing gears is a major issue in fisheries. The catch and subsequent mortality of organisms that are not targeted by a fishery is both wasteful and can have negative consequences for marine communities and ecosystems (FAO, 2012). Landed bycatch that is not used for human consumption or other commercial purposes is a waste of protein, especially in developing countries. Reducing bycatch and discards in fisheries is important to the maintenance of healthy marine ecosystems (FAO, 2012).

The first data chapter of this thesis investigated the effectiveness of a trawl BRD in a tropical fish assemblage over a 15 year period. The BRD reduced fish bycatch while maintaining the catch rate of prawns. The Jones-Davis BRD was designed to reduce capture of strong swimming fish in the Gulf of Mexico. It aimed to reduce the catch of commercially important species such as tropical snapper and mackerel. The BRD effect was consistent over time since there was no significant BRD and year interaction, despite frequent year effects on catch rates. While the year effect was confounded with the lunar effect in this study, the conclusion that there was consistent lack of year to year variation in BRD performance holds true. While the BRD was effective at reducing catch rates of most of the strong swimming, semi-pelagic fish, it was ineffective at reducing the catch rates of small, weaker-swimming, demersal and benthic fish species like leiognathids and *Saurida* spp.. The main component of fish assemblages on soft substrata in tropical shallow bays in the Indo-WestPacific are leiognathids, ambassids (glassfish), engraulids (anchovies) and clupeids (herrings) (Cabanban, 1991; Coles et al., 1993; Sondita, 1997). The similarities in size and swimming ability of the target species (penaeid prawns) of many tropical trawl fisheries and the majority

of fish that comprise the tropical fish assemblages on soft substrata in the Indo-West Pacific makes it very challenging to separate the target and no-target organisms in the BRD net (Brewer et al., 1998; Broadhurst, 2000; Glass and Wardle, 1995; Tsukamoto et al., 2008). BRDs to date have not generally been effective in reducing the capture rates of these small benthic and demersal fish while retaining the catch rate of prawns (Brewer et al., 1998; Brewer et al., 2008; Broadhurst et al., 2012a; Eayrs, 2007; Kennelly, 1999; Van der Geest, 2000).

The key point is that over 75% of the fish bycatch in this study was not removed from the trawl net by the Jones-Davis BRD. The results from this study are not atypical of similar attempts to use overseas-designed BRDs to reduce trawl bycatch in the tropical Indo-West Pacific (e.g. Brewer et al 1998). Whilst it is generally agreed that BRDs designed specifically to reduce fish bycatch in trawl fisheries in the tropical Indo-West Pacific are needed, to date, the technical difficulties of separating small weak-swimming fish from small-weak-swimming prawns in a trawl net have proven too difficult. The results of the present study are, in fact, comparable to most others carried out in the tropical Indo-West Pacific region (Brewer et al., 1998; Brewer et al., 2008; Broadhurst et al., 2012a; Heales et al., 2008; Robins-Troeger et al., 1995; Robins and McGilvray, 1999; Van der Geest, 2000).

Most other BRDs that have been tested in the Northern Prawn Fishery reduce fish bycatch by around 20-40% (Brewer et al., 1998; Courtney et al., 2006; Heales et al., 2008; Robins et al., 1999). However, the higher end of this reduction range can result in reduced catch of the target species (penaeid prawns) (Brewer et al., 1998; Robins et al., 1999). The majority of BRDs tested in the Australian prawn trawling industry were designs from overseas, and often these designs were not specifically made to reduce fish bycatch in the tropics (Brewer et al, 1998). The only BRD designed in Australia (the AusTED BRD) was designed for the prawn trawling industry in temperate regions. This BRD is also not very effective in the tropics

(Brewer et al., 1998). The most common designs currently used in Australian prawn trawl fisheries are the Nordmore grid (originally designed in Sweden) and the radial escape section (which is a simplified Jones-Davis BRD) (Broadhurst and Kennelly, 2011). These BRDs are effective in temperate estuarine waters (73% and 35% reduction in fish bycatch, respectively). However, the same limited effectiveness of such designs in reducing total fish bycatch highlighted by the present study still occurs in tropical waters. Fish bycatch reduction remains a very big challenge for the tropical prawn trawling industry, and the demersal trawl fishery as a whole. The majority of tropical marine fish assemblages on soft substrata in shallow waters are dominated by small benthic and demersal fish species. Such fish likely have a very important role in tropical coastal ecosystems. The consequences of removing large quantities of these fish, on the functioning of inshore coastal ecosystems, are poorly understood. This study highlights the importance of developing new or improved BRD designs aimed specifically at reducing the catch of small benthic and demersal finfish species in tropical regions. In addition, development of techniques to sort and separate target and non-target catch on board trawl vessels quickly (e.g. hopper tanks), particularly on the smaller trawlers used on the Queensland east coast, should be encouraged.

The second data chapter of this thesis investigated the effect of environmental factors on inter-annual variations in trawl catch rates in a tropical fish assemblage in Cleveland Bay over 15 years. Environmental factors investigated included rainfall, tidal state and moon phase. Rainfall two years prior to sampling affected total fish catch rates significantly ($p < 0.05$; $R^2 = 0.316$), again with catch rates enhanced by rainfall. The two year lag effect may be due to a two year lag in recruitment of many fish to the fishery. This delayed recruitment into the fishery could be explained by migration of sub-adults out of the estuaries and mangrove forests into the near shore habitats where trawling occurred, combined with fish growing to a catchable size.

It is also worth noting that the 2-year lag effect of rainfall on total fish catch in trawls detected in this study could only have been detected by recording total fish catch. The only reason this was done, was that the study was a fishery-independent, scientific study. Prawn-trawl fisheries in Queensland have rarely measured total fish catch, at least not over extended sampling periods. Commercial prawn fisheries in particular would not have measured total fish catch (i.e. bycatch) in most normal situations, since their focus was on prawn catch, with fish catch considered a nuisance and discarded. This simple point again highlights the value of long-term, fishery independent sampling.

Catch rates of prawns (mostly banana prawns, *Fenneropenaeus merguinesis*) were not affected by rainfall in this study. This is somewhat surprising, given that catch rates of this species are so strongly affected by rainfall in the SE Gulf of Carpentaria (Rothlisberg et al., 1986; Staples and Vance, 1985; Vance et al., 1985) and rainfall has a significant effect on catch rates of tiger and endeavour prawns in the East Coast Queensland prawn trawl fishery (Meynecke and Lee 2011). Catch rates of prawns and lizardfish were significantly higher on the full moon ($p < 0.05$). This result is apparently well-known to prawn-trawl fishers on the east Queensland coast (Battersby personal comments, skipper of the James Kirby). The present study appears to be the first to validate this fisher-knowledge with a long-term, empirical study. The effects of environmental drivers like rainfall and moon phase on inter-annual variations in catch rates of both prawns and fish in this tropical fish assemblage could only have been detected by consistent, long-term sampling.

Knowledge of environmental effects on catch rates of fish is an important tool for fisheries management. How fish assemblages change over time and what drives these changes can affect how one understands and exploits these assemblages. Long-term data that explains the dynamics in fish assemblages are very important for fisheries management and the sustainability of the fisheries. This study highlights the importance of consistent long-term

monitoring even though the sampling regime was limited to only 2-3 days per year. Trends such as the 2 year lag effect that rainfall has on the catch rates of fish in Cleveland Bay could not have been detected otherwise. Observations such as the effect of moon phase on catch rates of prawns (*Fenneropenaeus merguienis*) and *Saurida* spp. could not have been detected without a long term, scientific data set.

The year to year variations in abundance of key fish groups in the tropical fish assemblage may also help to explain some of the variation in interannual performance of the BRD tested in this study. However it is acknowledged that interannual variations in catch rate were confounded with lunar phase and thus should be interpreted with caution. Increased biomass of bycatch species in the bay two years after heavy rainfall presumably means that the BRD has a lot more fish bycatch to remove from the trawl nets in such years. Natural changes in abundance of fish such as sciaenids would lead to a proportional increase in larger, stronger swimming fish in the assemblage, which again would alter the effectiveness of the BRD in those particular years. Again, this stresses the importance of sampling BRD effectiveness over the long-term.

Chapter 4 showed that there was no long-term, systematic shift in the species composition of the tropical fish assemblage in Cleveland Bay over the 15 years of study. That is, there was no evidence that the assemblage structure changed from one state to another, despite interannual variations in species composition and environmental conditions. In addition, there was little evidence of differences in species composition of the trawl catch between nets fitted with and without the Jones-Davis BRD. The result of Chapter 4 highlight that long-term monitoring of the species composition of marine communities is vital. Data sets on species composition of fish assemblages collected over short time periods (e.g 2-3 years) might conclude that systematic shifts have occurred in the assemblage structure. The present study demonstrated that monitoring, over long periods (1998-2012) rather than on a scale of a few

years (e.g. 2006-2008), can indicate consistency of species composition in the long-term, in a situation where shorter monitoring may have concluded systematic, directional change in species composition of the fish assemblage.

Improvements to This Study and Future Research

The data for this thesis were collected on almost annual field trips run for an undergraduate subject in marine biology at James Cook University. The sampling was thus usually restricted to the first week of March in each year. The limitation of this sampling design is that it can say nothing about seasonality of BRD effectiveness, trawl catch rates or species composition of the fish assemblage. Sampling over more times of the year clearly would have improved the study. The study only had one Jones-Davis BRD at its disposal. This particular was purchased by Don Battersby, the skipper of the James Kirby in the late 1990s. The purposes of this purchase were to trial the BRD and possibly modify it to suit local conditions. Thus, the study could not trawl with both nets equipped with the Jones-Davis BRD on some days, and compare the catch rates obtained with those on days when the BRD was not fitted to either net. Furthermore, since sampling was restricted to 2-3 days per year, usually the first weekend in March, the chance to vary the placement of the BRD between port and starboard nets among days was very limited each year. Varying placement of the BRD between each net within a sampling time could have improved, but likely not changed, the key conclusions, of the study. The study would also have been improved by increasing the spatial scale of sampling to include deeper parts of Cleveland Bay, and perhaps, other bays. This was not possible. Often a single day of sampling consisted of 3 or 4 short trips with different sets of students. Clearly, restricting the study to inshore areas, close to the harbour was necessary.

The study has pointed to some fruitful lines of research in the future. BRDs designed specifically for tropical Indo-West Pacific conditions have been mentioned above as a major

priority. Greater efforts to identify and quantify environmental drivers of trawl catch rates in tropical Australia are likely to be useful. More studies of the structure of tropical fish assemblages on soft substrata are required. Most importantly, the study suggests the need for long-term consistency of fishery-independent sampling of tropical fish assemblages, coupled with detailed collection of data on environmental and anthropogenic factors that may affect such assemblages.

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Appendix 1:

	Degrees of freedom	F	p
Carangidae			
year	11	3.067	<0.001
BRD	1	82.995	<0.001
year*BRD	11	0.709	0.728
Total Fish			
year	11	6.704	<0.001
BRD	1	14.265	<0.001
year*BRD	11	0.878	0.563
Other Fish			
year	11	16.701	<0.001
BRD	1	38.822	<0.001
year*BRD	11	0.539	0.873
Leiognathidae			
year	11	9.561	<0.001
BRD	1	0.082	0.774
year*BRD	11	0.314	0.981
Prawns			
year	11	56.911	<0.001
BRD	1	0.202	0.653
year*BRD	11	0.153	0.999
Sciaenidae			
year	11	15.582	<0.001
BRD	1	5.447	<0.005
year*BRD	11	0.145	0.999
<i>Saurida</i> sp.			
year	11	12.112	<0.001
BRD	1	0.581	0.447
year*BRD	11	0.318	0.981

Table 1.1: Univariate ANOVAs testing the effects of the BRD and Year of Sampling for all variates. The denominator of all F values is 129.

Appendix 2:

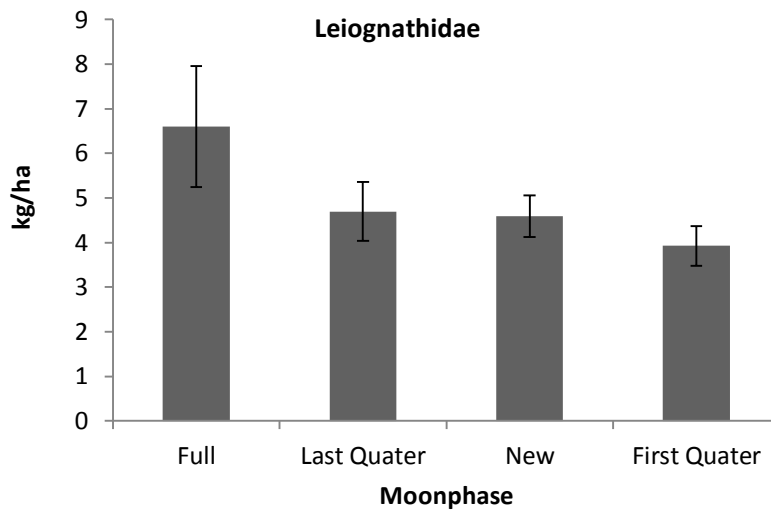


Figure 2.1: Mean (+/- SE) catch rate (kg/ha) of Leiognathidae for all 12 sampling years from 1998-2102 at different phases of the moon.

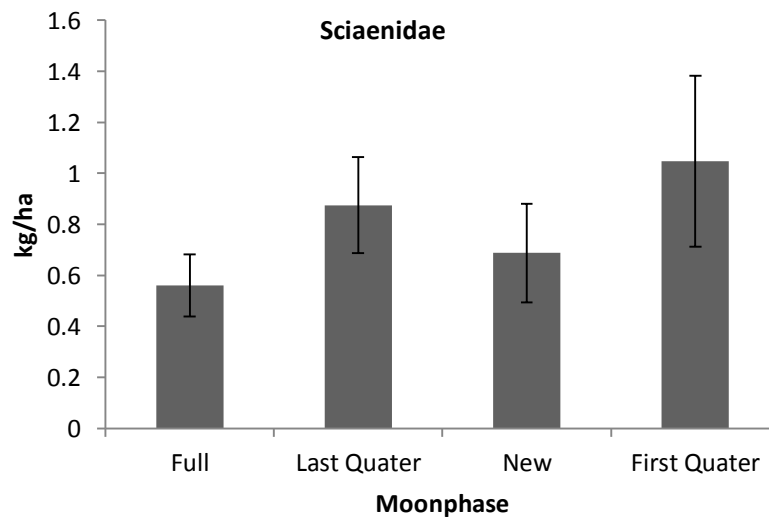


Figure 2.2: Mean (+/- SE) catch rate (kg/ha) of Sciaenidae for all 12 sampling years from 1998-2102 at different phases of the moon.

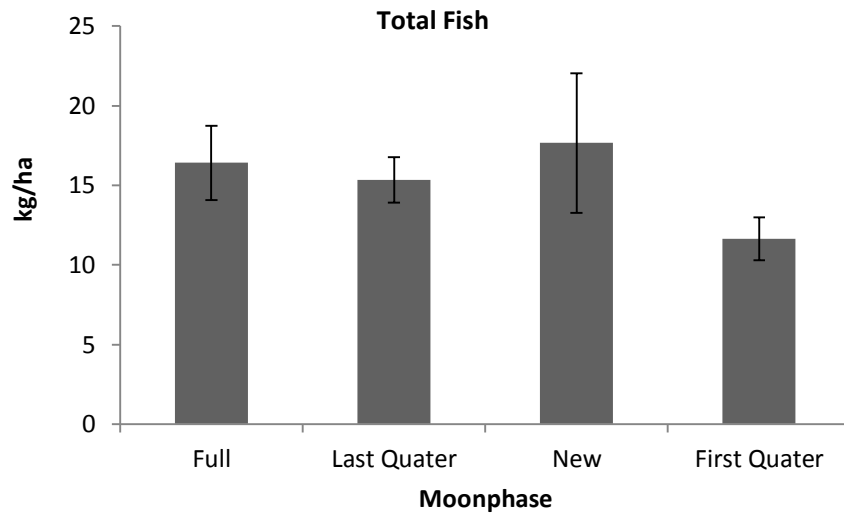


Figure 2.3: Mean (+/- SE) catch rate (kg/ha) of “Total Fish” for all 12 sampling years from 1998-2102 at different phases of the moon.

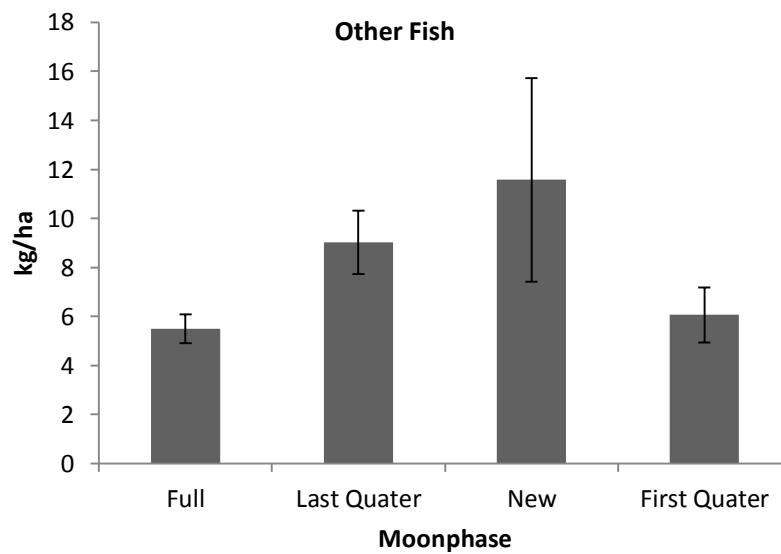


Figure 2.4: Mean (+/- SE) catch rate (kg/ha) of “Other Fish” for all 12 sampling years from 1998-2102 at different phases of the moon.

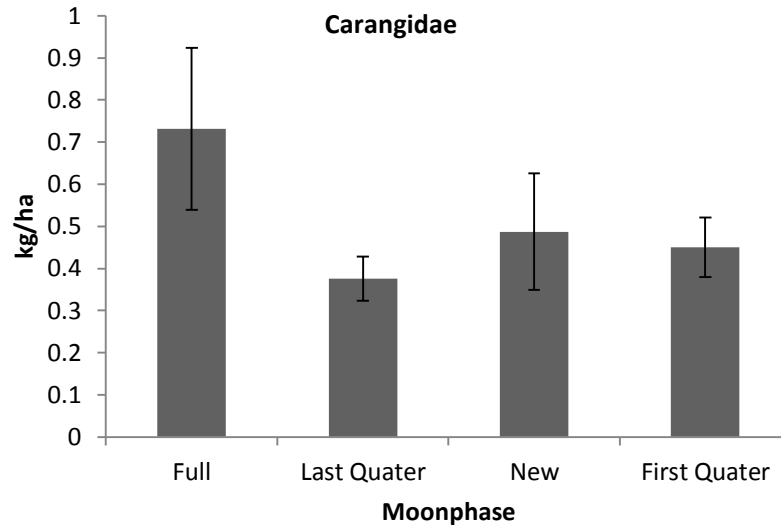


Figure 2.5: Mean (+/- SE) catch rate (kg/ha) of Carangidae for all 12 sampling years from 1998-2102 at different phases of the moon.

Effect	Intercept	p	R2 adj.	F	df	SS	MS error
Two years	3.773	0.033	0.317	6.094	1	112.743	9.395
One Year	7.24	0.532	-0.056	0.42	1	174.145	14.512
Four weeks	5.887	0.178	0.091	2.095	1	150.027	12.502
Three Weeks	8.782	0.923	-0.099	0.01	1	181.274	15.106
Two weeks	8.535	0.791	-0.092	0.074	1	180.121	15.01

Table 2.1: Multiple regression analysis testing the effect of rainfall on the catch rate (kg/ha) of total fish for all treatments.

Effect	Intercept	p	R2 adj.	F	df	SS	MS error
Two years	0.368	0.213	0.066	1.772	10	0.13	0.011
One Year	0.338	0.408	-0.024	0.745	10	0.143	0.012
Four weeks	0.298	0.716	-0.085	0.141	10	0.151	0.013
Three Weeks	0.307	0.54	-0.058	0.402	10	0.148	0.012
Two weeks	0.326	0.249	0.043	1.499	10	0.133	0.011

Table 2.2: Multiple regression analysis testing the effect of rainfall on the catch rate (kg/ha) of Carangidae for all treatments.

Effect	Intercept	p	R2 adj.	F	df	SS	MS error
Two years	-0.143	0.044	0.282	5.328	10	2.277	0.19
One Year	0.22	0.388	-0.017	0.814	10	3.228	0.269
Four weeks	0.368	0.587	-0.066	0.315	10	3.384	0.282
Three Weeks	0.67	0.645	-0.076	0.226	10	3.414	0.284
Two weeks	0.448	0.66	-0.078	0.205	10	3.421	0.285

Table 2.3: Multiple regression analysis testing the effect of rainfall on the catch rate (kg/ha) of Sciaenidae for all treatments.

Effect	Intercept	p	R2 adj.	F	df	SS	MS error
Two years	1.415	0.164	0.103	2.258	10	5.928	0.494
One Year	0.945	0.647	-0.076	0.223	10	7.108	0.592
Four weeks	0.091	0.192	0.08	1.957	10	6.078	0.506
Three Weeks	1.101	0.273	0.031	1.347	10	6.404	0.534
Two weeks	1.25	0.062	0.236	4.394	10	5.049	0.421

Table 2.4: Multiple regression analysis testing the effect of rainfall on the catch rate (kg/ha) of Prawns for all treatments.

Effect	Intercept	p	R2 adj.	F	df	SS	MS error
Two years	3.919	0.095	0.179	3.398	10	8.977	0.748
One Year	3.158	0.641	-0.075	0.231	10	11.756	0.98
Four weeks	2.602	0.718	-0.085	0.138	10	11.864	0.989
Three Weeks	2.925	0.84	-0.095	0.043	10	11.976	0.998
Two weeks	3.346	0.197	0.076	1.91	10	10.099	0.842

Table 2.5: Multiple regression analysis testing the effect of rainfall on the catch rate (kg/ha) of Leiognathidae for all treatments.

Effect	Intercept	p	R2 adj.	F	df	SS	MS error
Two years	0.229	0.784	-0.091	0.079	10	0.244	0.02
One Year	0.238	0.714	-0.085	0.142	10	0.242	0.02
Four weeks	0.036	0.037	0.305	5.82	10	0.155	0.013
Three Weeks	0.202	0.99	-0.1	0	10	0.246	0.02
Two weeks	0.223	0.714	-0.085	0.142	10	0.242	0.02

Table 2.6: Multiple regression analysis testing the effect of rainfall on the catch rate (kg/ha) of *Saurida* spp. for all treatments.

Effect	Intercept	p	R2 adj.	F	df	SS	MS error
Two years	-1.315	0.003	0.551	14.499	10	70.395	5.866
One Year	2.88	0.425	-0.029	0.693	10	161.287	13.441
Four weeks	2.938	0.364	-0.009	0.905	10	158.155	13.18
Three Weeks	4.396	0.741	-0.087	0.116	10	170.488	14.207
Two weeks	3.843	0.456	-0.038	0.601	10	162.694	13.558

Table 2.7: Multiple regression analysis testing the effect of rainfall on the catch rate (kg/ha) of 'Other Fish' for all treatments.

	SS	Degrees of Freedom	MS	F	p
Carangidae					
Intercept	156.1746	1	156.174	254.252	<0.001
Moonphase	1.3984	3	0.466	0.758	0.521
Error	44.8403	73	0.614		
Saurida spp.					
Intercept	8.301	1	8.301	152.031	<0.001
Moonphase	1.247	3	0.415	7.612	<0.001
Error	3.986	73	0.054		
Sciaenidae					
Intercept	1.458	1	1.458	5.992	0.016
Moonphase	0.181	3	0.061	0.248	0.862
Error	17.766	73	0.243		
Leiognathidae					
Intercept	75.961	1	75.961	214.515	<0.001
Moonphase	1.828	3	0.609	1.721	0.171
Error	25.849	73	0.354		
Total Fish					
Intercept	276.081	1	276.081	871.259	<0.001
Moonphase	1.547	3	0.515	1.628	0.191
Error	23.131	73	0.316		
Other Fish					
Intercept	102.262	1	102.262	173.092	<0.001
Moonphase	3.139	3	1.046	1.771	0.161
Error	43.128	73	0.591		

Table 2.8: Univariate test of significance testing the effect of moon phase on catch rate (kg/ha) for all variates.

	SS	Degrees of Freedom	MS	F	p
Prawn					
Intercept	226.042	1	226.042	547.335	<0.001
Year	254.826	11	23.166	56.094	<0.001
Error	99.116	240	0.413		
Total Fish					
Intercept	991.679	1	991.679	2961.565	<0.001
Year	27.644	11	2.513	7.505	<0.001
Error	80.364	240	0.334		
Carangidae					
Intercept	365.684	1	365.684	1015.907	<0.001
Year	13.825	11	1.256	3.492	<0.001
Error	86.391	240	0.36		
Leiognathidae					
Intercept	121.694	1	121.694	208.197	<0.001
Year	65.165	11	5.924	10.135	<0.001
Error	140.282	240	0.584		
Saurida					
Intercept	536.561	1	536.560	1460.613	<0.001
Year	46.102	11	4.191	11.409	<0.001
Error	88.164	240	0.3674		
Sciaenidae					
Intercept	302.291	1	302.291	380.527	<0.001
Year	91.472	11	8.315	10.467	<0.001
Error	190.655	240	0.7941		
Other Fish					
Intercept	347.701	1	347.701	689.453	<0.001
Year	77.778	11	7.071	14.021	<0.001
Error	121.035	240	0.504		

Table 2.9: Univariate test of significance testing the effect of year of sampling on catch rate (kg/ha) for all variates.

2012	0.000018	0.000018	0.000018	0.992752	0.000018	1.000000	0.000018	0.000018	0.037392	1.000000	0.462045	
2011	0.000018	0.000018	0.030929	0.993519	0.000018	0.599493	0.000018	0.000018	0.999272	0.221697		0.462045
2010	0.000018	0.000018	0.000018	0.932232	0.000018	0.999943	0.000018	0.000018	0.009039		0.221697	1.000000
2009	0.000018	0.000018	0.232616	0.650505	0.000035	0.062188	0.000018	0.000018		0.009039	0.999272	0.037392
2008	0.002273	0.000091	0.000018	0.000018	0.000023	0.000018	0.890200		0.000018	0.000018	0.000018	0.000018
2007	0.222890	0.026752	0.000018	0.000018	0.004206	0.000018		0.890200	0.000018	0.000018	0.000018	0.000018
2006	0.000018	0.000018	0.000018	0.998904	0.000018		0.000018	0.000018	0.062188	0.999943	0.599493	1.000000
2004	0.992337	0.999999	0.333995	0.000018		0.000018	0.004206	0.000023	0.000035	0.000018	0.000018	0.000018
2003	0.000018	0.000018	0.000349		0.000018	0.998904	0.000018	0.000018	0.650505	0.932232	0.993519	0.992752
2002	0.010751	0.110255		0.000349	0.333995	0.000018	0.000018	0.000018	0.232616	0.000018	0.030929	0.000018
1999	0.999934		0.110255	0.000018	0.999999	0.000018	0.026752	0.000091	0.000018	0.000018	0.000018	0.000018
1998		0.999934	0.010751	0.000018	0.992337	0.000018	0.222890	0.002273	0.000018	0.000018	0.000018	0.000018
	1998	1999	2002	2003	2004	2006	2007	2008	2009	2010	2011	2012

Table 2.10: Tukey's HSD test comparisons among all years for the Prawn data set

2012	0.997853	1.000000	0.667559	0.081073	0.990560	0.925518	0.827258	0.139882	0.934934	1.000000	0.030742	
2011	0.003075	0.056647	0.976166	0.000018	0.001475	0.000054	0.859488	0.000018	0.714352	0.031082		0.030742
2010	0.997791	1.000000	0.669911	0.080305	0.990351	0.924369	0.829122	0.138686	0.935940		0.031082	1.000000
2009	0.437089	0.939363	0.999987	0.000334	0.314624	0.077912	1.000000	0.000758		0.935940	0.714352	0.934934
2008	0.891733	0.357445	0.000141	1.000000	0.949576	0.945438	0.000245		0.000758	0.138686	0.000018	0.139882
2007	0.285510	0.849545	1.000000	0.000111	0.190832	0.034155		0.000245	1.000000	0.829122	0.859488	0.827258
2006	1.000000	0.986435	0.018669	0.864235	1.000000		0.034155	0.945438	0.077912	0.924369	0.000054	0.925518
2004	1.000000	0.998657	0.116225	0.882776		1.000000	0.190832	0.949576	0.314624	0.990351	0.001475	0.990560
2003	0.792178	0.244771	0.000069		0.882776	0.864235	0.000111	1.000000	0.000334	0.080305	0.000018	0.081073
2002	0.181634	0.708229		0.000069	0.116225	0.018669	1.000000	0.000141	0.999987	0.669911	0.976166	0.667559
1999	0.999794		0.708229	0.244771	0.998657	0.986435	0.849545	0.357445	0.939363	1.000000	0.056647	1.000000
1998		0.999794	0.181634	0.792178	1.000000	1.000000	0.285510	0.891733	0.437089	0.997791	0.003075	0.997853
	1998	1999	2002	2003	2004	2006	2007	2008	2009	2010	2011	2012

Table 2.11: Tukey's HSD test comparisons among all years for the "Total Fish" data set.

2012	0.849928	0.972162	0.999998	0.999996	1.000000	0.195694	0.976934	0.989982	0.988130	0.790920	0.999892	
2011	0.455431	0.723461	0.992123	1.000000	0.996858	0.035882	0.705102	0.999999	0.770705	0.996880		0.999892
2010	0.033656	0.106963	0.519060	0.985701	0.658589	0.000204	0.067760	0.999985	0.091754		0.996880	0.790920
2009	0.999961	1.000000	0.999946	0.878633	0.999937	0.936583	1.000000	0.434365		0.091754	0.770705	0.988130
2008	0.196767	0.416939	0.909554	0.999974	0.952210	0.005795	0.364524		0.434365	0.999985	0.999999	0.989982
2007	0.999992	1.000000	0.999786	0.829805	0.999774	0.962520		0.364524	1.000000	0.067760	0.705102	0.976934
2006	0.999945	0.995703	0.596299	0.068471	0.657588		0.962520	0.005795	0.936583	0.000204	0.035882	0.195694
2004	0.987031	0.999363	1.000000	0.999482		0.657588	0.999774	0.952210	0.999937	0.658589	0.996858	1.000000
2003	0.590103	0.833723	0.998461		0.999482	0.068471	0.829805	0.999974	0.878633	0.985701	1.000000	0.999996
2002	0.985632	0.999387		0.998461	1.000000	0.596299	0.999786	0.909554	0.999946	0.519060	0.992123	0.999998
1999	1.000000		0.999387	0.833723	0.999363	0.995703	1.000000	0.416939	1.000000	0.106963	0.723461	0.972162
1998		1.000000	0.985632	0.590103	0.987031	0.999945	0.999992	0.196767	0.999961	0.033656	0.455431	0.849928
	1998	1999	2002	2003	2004	2006	2007	2008	2009	2010	2011	2012

Table 2.12: Tukey's HSD test comparisons among all years for the Carangidae data set.

2012	0.077968	0.000979	0.013898	0.000018	0.000018	0.000018	0.000018	0.000018	0.000138	0.000018	0.000018	
2011	0.204683	0.911782	0.280442	0.999999	0.999991	1.000000	0.948009	0.999819	0.800777	0.957914		0.000018
2010	0.001667	0.109114	0.002048	0.760846	0.999846	0.867095	1.000000	0.514062	0.030768		0.957914	0.000018
2009	0.993209	1.000000	0.999285	0.970318	0.471652	0.809643	0.026458	0.996320		0.030768	0.800777	0.000138
2008	0.656233	0.999217	0.781199	1.000000	0.980226	0.999954	0.482085		0.996320	0.514062	0.999819	0.000018
2007	0.001407	0.097727	0.001711	0.733101	0.999742	0.844588		0.482085	0.026458	1.000000	0.948009	0.000018
2006	0.190868	0.925955	0.261455	1.000000	0.999860		0.844588	0.999954	0.809643	0.867095	1.000000	0.000018
2004	0.073192	0.659169	0.103808	0.997946		0.999860	0.999742	0.980226	0.471652	0.999846	0.999991	0.000018
2003	0.461102	0.991083	0.586490		0.997946	1.000000	0.733101	1.000000	0.970318	0.760846	0.999999	0.000018
2002	1.000000	0.999464		0.586490	0.103808	0.261455	0.001711	0.781199	0.999285	0.002048	0.280442	0.013898
1999	0.995100		0.999464	0.991083	0.659169	0.925955	0.097727	0.999217	1.000000	0.109114	0.911782	0.000979
1998		0.995100	1.000000	0.461102	0.073192	0.190868	0.001407	0.656233	0.993209	0.001667	0.204683	0.077968
	1998	1999	2002	2003	2004	2006	2007	2008	2009	2010	2011	2012

Table 2.13: Tukey's HSD test comparisons among all years for the Leionathidae data set.

2012	0.047417	0.000018	0.000018	0.963087	0.001131	0.004405	0.000018	0.032483	0.000421	0.001688	0.007537	
2011	1.000000	0.503525	0.137253	0.469676	0.999948	1.000000	0.000035	0.999999	0.999998	1.000000		0.007537
2010	0.999999	0.561550	0.155379	0.284588	0.999995	1.000000	0.000030	0.999967	1.000000		1.000000	0.001688
2009	0.999863	0.758019	0.302498	0.143986	1.000000	0.999943	0.000083	0.998607		1.000000	0.999998	0.000421
2008	1.000000	0.224069	0.034241	0.762848	0.994812	1.000000	0.000018		0.998607	0.999967	0.999999	0.032483
2007	0.000037	0.367610	0.632499	0.000018	0.002375	0.000018		0.000018	0.000083	0.000030	0.000035	0.000018
2006	1.000000	0.307905	0.048964	0.462427	0.999420		0.000018	1.000000	0.999943	1.000000	1.000000	0.004405
2004	0.999056	0.939191	0.645303	0.158613		0.999420	0.002375	0.994812	1.000000	0.999995	0.999948	0.001131
2003	0.763440	0.000562	0.000028		0.158613	0.462427	0.000018	0.762848	0.143986	0.284588	0.469676	0.963087
2002	0.100791	0.999998		0.000028	0.645303	0.048964	0.632499	0.034241	0.302498	0.155379	0.137253	0.000018
1999	0.395190		0.999998	0.000562	0.939191	0.307905	0.367610	0.224069	0.758019	0.561550	0.503525	0.000018
1998		0.395190	0.100791	0.763440	0.999056	1.000000	0.000037	1.000000	0.999863	0.999999	1.000000	0.047417
	1998	1999	2002	2003	2004	2006	2007	2008	2009	2010	2011	2012

Table 2.14: Tukey's HSD test comparisons among all years for the *Saurida* spp. data set.

2012	0.332654	0.999996	0.945822	0.000018	0.014333	0.004457	0.429753	0.697925	0.568252	0.000018	0.997962	
2011	0.934231	1.000000	0.430289	0.000090	0.272299	0.205081	0.983081	0.998590	0.995333	0.000023		0.997962
2010	0.030984	0.000021	0.000018	1.000000	0.485765	0.151675	0.001218	0.000640	0.000526		0.000023	0.000018
2009	0.999991	0.958982	0.019504	0.003867	0.870027	0.862361	1.000000	1.000000		0.000526	0.995333	0.568252
2008	0.999966	0.980648	0.039581	0.004205	0.846202	0.838426	1.000000		1.000000	0.000640	0.998590	0.697925
2007	1.000000	0.911259	0.010137	0.007859	0.933620	0.936241		1.000000	1.000000	0.001218	0.983081	0.429753
2006	0.998648	0.113968	0.000025	0.364106	1.000000		0.936241	0.838426	0.862361	0.151675	0.205081	0.004457
2004	0.996983	0.159460	0.000095	0.716742		1.000000	0.933620	0.846202	0.870027	0.485765	0.272299	0.014333
2003	0.093453	0.000058	0.000018		0.716742	0.364106	0.007859	0.004205	0.003867	1.000000	0.000090	0.000018
2002	0.009528	0.796839		0.000018	0.000095	0.000025	0.010137	0.039581	0.019504	0.000018	0.430289	0.945822
1999	0.808920		0.796839	0.000058	0.159460	0.113968	0.911259	0.980648	0.958982	0.000021	1.000000	0.999996
1998		0.808920	0.009528	0.093453	0.996983	0.998648	1.000000	0.999966	0.999991	0.030984	0.934231	0.332654
	1998	1999	2002	2003	2004	2006	2007	2008	2009	2010	2011	2012

Table 2.15: Tukey's HSD test comparisons among all years for the Sciaenidae data set.

2012	0.008802	0.100911	0.991633	0.000018	0.018150	0.000437	0.268226	0.000018	1.000000	0.171631	0.295298	
2011	0.000018	0.000027	0.972345	0.000018	0.000018	0.000018	0.000041	0.000018	0.150594	0.000026		0.295298
2010	0.984502	0.999997	0.006612	0.122300	0.996380	0.911369	1.000000	0.021403	0.336872		0.000026	0.171631
2009	0.024108	0.204421	0.950553	0.000023	0.046118	0.001818	0.472884	0.000018		0.336872	0.150594	1.000000
2008	0.722051	0.232962	0.000018	0.999997	0.585012	0.633320	0.010764		0.000018	0.021403	0.000018	0.000018
2007	0.957476	0.999941	0.013437	0.072347	0.986449	0.818316		0.010764	0.472884	1.000000	0.000041	0.268226
2006	1.000000	0.999120	0.000020	0.936821	1.000000		0.818316	0.633320	0.001818	0.911369	0.000018	0.000437
2004	1.000000	0.999998	0.000450	0.889279		1.000000	0.986449	0.585012	0.046118	0.996380	0.000018	0.018150
2003	0.950915	0.563372	0.000018		0.889279	0.936821	0.072347	0.999997	0.000023	0.122300	0.000018	0.000018
2002	0.000190	0.004288		0.000018	0.000450	0.000020	0.013437	0.000018	0.950553	0.006612	0.972345	0.991633
1999	0.999941		0.004288	0.563372	0.999998	0.999120	0.999941	0.232962	0.204421	0.999997	0.000027	0.100911
1998		0.999941	0.000190	0.950915	1.000000	1.000000	0.957476	0.722051	0.024108	0.984502	0.000018	0.008802
	1998	1999	2002	2003	2004	2006	2007	2008	2009	2010	2011	2012

Table 2.16: Tukey's HSD test comparisons among all years for the "Other Fish" data set.

Appendix 3:

Species	Group Other	Group				
	Years	1998	Av.Diss	Diss/SD	Contrib%	Cum.%
	Av.Abund	Av.Abund				
<i>Torquigener pallimaculatus</i>	0	0.75	0.7	1.69	1.91	1.91
<i>Pomadasys kakaan</i>	0.78	0.25	0.64	1.31	1.74	3.66
<i>Sphyaerena putamae</i>	0.61	0	0.58	1.22	1.58	5.23
<i>Trixiphichtys weberi</i>	0.59	0	0.57	1.17	1.55	6.78
<i>Anodontosoma chacunda</i>	0.59	0	0.57	1.17	1.55	8.33
<i>Apogon poecilopterus</i>	0.35	0.75	0.55	1.15	1.51	9.84
<i>Johnius amblycephalus</i>	0.61	0.25	0.55	1.09	1.5	11.34
<i>Pelates quadrilineatus</i>	0.41	0.75	0.54	1.07	1.47	12.81
<i>Leiognathus equulus</i>	1	0.5	0.53	0.99	1.46	14.27
<i>Leiognathus decorus</i>	0.91	0.5	0.53	0.99	1.44	15.71
<i>Paraplaygusia bilineata</i>	0.57	0.25	0.52	1.05	1.43	17.14
<i>Lactarius lactarius</i>	0.87	0.5	0.52	0.99	1.43	18.56
<i>Arius thallassinus</i>	0.48	1	0.52	1.03	1.41	19.98
<i>Austronibae oedogenys</i>	0.74	0.5	0.51	0.98	1.39	21.37
<i>Chirocentrus dorab</i>	0.54	0.25	0.51	1.03	1.38	22.75
<i>Dussumieria elopsoides</i>	0.72	0.5	0.51	0.99	1.38	24.14
<i>Gerres subfasciatus</i>	0.54	0.25	0.5	1.03	1.37	25.51

Table 3.1: Species identified by Simper as those that cumulatively explained the top 25%

dissimilarity in species composition between 1998 and all other years combined. Average dissimilarity between 1998 and all other years was 36.58% (this includes all 160 species).

Species	Group	Group	Av.Diss	Diss/SD	Contrib%	Cum.%
	Other	1999				
	Years					
	Av.Abund	Av.Abund				
<i>Mugil cephalus</i>	0.16	1	0.76	2.28	2.29	2.29
<i>Parastomatus niger</i>	0.02	0.75	0.65	1.66	1.97	4.26
<i>Herklotsichthys lippa</i>	0.31	1	0.61	1.47	1.86	6.12
<i>Scomberomorus queenslandicus</i>	0.78	0.25	0.57	1.32	1.72	7.84
<i>Nematalose come</i>	0.4	1	0.54	1.21	1.63	9.47
<i>Apolectus niger</i>	0.36	0.75	0.51	1.15	1.54	11.01
<i>Centriscus scutatus</i>	0.4	0.75	0.49	1.1	1.48	12.49
<i>Alectis indicus</i>	0.58	0.25	0.48	1.07	1.45	13.93
<i>Rastrelliger kanagurta</i>	0.44	0.75	0.47	1.05	1.43	15.36
<i>Paraplaygusia bilineata</i>	0.56	0.25	0.47	1.05	1.41	16.77
<i>Lutjanus malabaricus</i>	0.53	0	0.47	1.06	1.41	18.18
<i>Trixiphichtys weberi</i>	0.56	0.25	0.46	1.05	1.4	19.58
<i>Thryssa mystax</i>	0.13	0.5	0.46	0.99	1.38	20.96
<i>Chirocentrus dorab</i>	0.53	0.25	0.45	1.02	1.37	22.33
<i>Scomberomorus commerson</i>	0.49	0.75	0.45	1	1.37	23.7
<i>Sphaerena putamae</i>	0.51	1	0.45	0.97	1.36	25.06

Table 3.2: Species identified by Simper as those who cumulatively explained the most 25%

dissimilarity in species composition between 1999 and all other years combined. Average dissimilarity between 1999 and all other years was 33.13% (this includes all 160 species).

Species	Group	Group	Av.Diss	Diss/S D	Contrib%	Cum. %
	Other Years	2002				
	Av.Abund	Av.Abund				
<i>Sillago maculata</i>	0.15	1	0.78	2.3	2.33	2.33
<i>Sillago analis</i>	0.11	0.75	0.63	1.49	1.88	4.21
<i>Triacanthus biaculeatus</i>	0.76	0.25	0.58	1.29	1.73	5.94
<i>Herklotsichthys koningsbergen</i>	0.63	0	0.57	1.29	1.72	7.65
<i>Upeneus sundaicus</i>	0.35	0.75	0.53	1.15	1.59	9.25
<i>Leiohnathus blochii</i>	0.39	0.75	0.51	1.11	1.52	10.77
<i>Johnius amblycephalus</i>	0.61	0.25	0.5	1.11	1.51	12.28
<i>Chirocentrus dorab</i>	0.48	1	0.48	1.03	1.45	13.73
<i>Anodontosoma chacunda</i>	0.57	0.25	0.48	1.06	1.45	15.18
<i>Scomberomorus commerson</i>	0.48	0.75	0.47	1.01	1.41	16.59
<i>Gazza achlams</i>	0.52	0.25	0.46	1.01	1.39	17.97
<i>Engraulis australis</i>	0.17	0.5	0.46	0.99	1.37	19.35
<i>Paraplaygusia bilineata</i>	0.5	1	0.46	0.99	1.37	20.72
<i>Alectis indicus</i>	0.54	0.5	0.46	0.99	1.37	22.09
<i>Pelates quadrilineatus</i>	0.43	0.5	0.46	0.99	1.37	23.45
<i>Gerres subfasciatus</i>	0.52	0.5	0.46	0.99	1.37	24.82
<i>Lutjanus malabaricus</i>	0.5	0.5	0.46	0.99	1.37	26.18

Table 3.3: Species identified by Simper as those that cumulatively explained 25%

dissimilarity in species composition between 2002 and all other years combined. Average dissimilarity between 2002 and all other years was 33.33% (this includes all 160 species).

Species	Group Other	Group 2003					
	Years	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
<i>Sphyaerena flavicauda</i>		0.11	0.75	0.6	1.5	1.77	1.77
<i>Stolephorus nelsonii</i>		0.13	0.75	0.59	1.46	1.76	3.53
<i>Sphyaerena fosteri</i>		0.37	1	0.55	1.29	1.63	5.16
<i>Pomadasys kakaan</i>		0.78	0.25	0.55	1.33	1.62	6.78
<i>Apogon quadrifasciatus</i>		0.37	1	0.54	1.29	1.61	8.39
<i>Nematalose come</i>		0.39	1	0.53	1.24	1.57	9.96
<i>Pelates quadrilineatus</i>		0.39	1	0.52	1.24	1.55	11.5
<i>Rastrelliger kanagurta</i>		0.41	1	0.51	1.18	1.5	13.01
<i>Herklotsichthys lippa</i>		0.33	0.75	0.51	1.18	1.5	14.51
<i>Apogon poecilopterus</i>		0.35	0.75	0.49	1.16	1.46	15.97
<i>Leiognathus moretoniensis</i>		0.59	0.25	0.46	1.08	1.37	17.34
<i>Gerres subfasciatus</i>		0.48	1	0.46	1.04	1.37	18.71
<i>Paraplaygusia bilineata</i>		0.57	0.25	0.46	1.06	1.35	20.07
<i>Otilithes ruber</i>		0.89	0.5	0.44	0.99	1.3	21.36
<i>Gazza achlamys</i>		0.52	0.25	0.44	1.01	1.3	22.66
<i>Lactarius lactarius</i>		0.87	0.5	0.44	0.99	1.3	23.95
<i>Apogon semilineatus</i>		0.02	0.5	0.44	0.99	1.29	25.25

Table 3.4: Species identified by Simper as those who cumulatively explained the most 25%

dissimilarity in species composition between 2003 and all other years combined. Average dissimilarity between 2003 and all other years was 33.82% (this includes all 160 species).

Species	Group Other Years	Group 2004	Av.Diss	Diss/SD	Contrib%	Cum.%
	Av.Abund	Av.Abund				
<i>Apogon poecilopterus</i>	0.33	1	0.61	1.41	1.8	1.8
<i>Lutjanus russelli</i>	0.22	0.75	0.57	1.32	1.68	3.48
<i>Sphaerena fosteri</i>	0.39	0.75	0.51	1.1	1.48	4.96
<i>Chirocentrus dorab</i>	0.57	0	0.5	1.13	1.47	6.44
<i>Johnius amblycephalus</i>	0.61	0.25	0.5	1.1	1.47	7.91
<i>Nematalose come</i>	0.41	0.75	0.5	1.08	1.47	9.37
<i>Rastrelliger kanagurta</i>	0.43	0.75	0.49	1.05	1.43	10.81
<i>Paraplagusia bilineata</i>	0.57	0.25	0.48	1.05	1.42	12.23
<i>Alectis indicus</i>	0.57	0.25	0.48	1.05	1.42	13.65
<i>Eleutheronema tetradactylum</i>	0.83	0.5	0.47	0.99	1.39	15.04
<i>Gazza achlamys</i>	0.52	0.25	0.46	1.01	1.35	16.39
<i>Anodontosoma chacunda</i>	0.5	1	0.46	0.99	1.35	17.74
<i>Trixiphichtys weberi</i>	0.5	1	0.46	0.99	1.35	19.09
<i>Scomberomorus commerson</i>	0.52	0.25	0.46	1.01	1.34	20.44
<i>Chelonodon patoca</i>	0.57	0.5	0.46	0.99	1.34	21.78
<i>Fistularia commersonii</i>	0.04	0.5	0.46	0.99	1.34	23.12
<i>Triacanthus biaculeatus</i>	0.74	0.5	0.46	0.99	1.34	24.46
<i>Arius thalassinus</i>	0.52	0.5	0.45	0.99	1.33	25.79

Table 3.5: Species identified by Simper as those who cumulatively explained the most 25%

dissimilarity in species composition between 2004 and all other years combined. Average dissimilarity between 2004 and all other years was 34.11% (this includes all 160 species).

Species	Group Other	Group 2006		Diss/SD	Contrib%	Cum.%
	Years	Av.Abund	Av.Diss			
<i>Caranx ignobilis</i>	0.25	1	0.68	1.7	1.91	1.91
<i>Scomberomorus semifasciatus</i>	0.16	0.83	0.65	1.61	1.82	3.73
<i>Austronibae oedogenys</i>	0.8	0.17	0.61	1.5	1.73	5.47
<i>Leiognathus leuciscus</i>	0.25	0.83	0.6	1.4	1.68	7.15
<i>Trichiurus lepturus</i>	0.95	0.33	0.6	1.35	1.68	8.82
<i>Terapon puta</i>	0.14	0.67	0.55	1.27	1.56	10.38
<i>Sphaerena fosteri</i>	0.36	0.83	0.54	1.19	1.52	11.9
<i>Johnius vogleri</i>	0.82	0.33	0.53	1.23	1.5	13.4
<i>Thryssa setirostris</i>	0.8	0.33	0.53	1.21	1.49	14.89
<i>Gerres subfasciatus</i>	0.45	1	0.5	1.08	1.41	16.31
<i>Johnius amblycephalus</i>	0.61	0.33	0.48	1.07	1.37	17.68
<i>Centriscus scutatus</i>	0.39	0.67	0.48	1.07	1.35	19.03
<i>Apogon quadrifasciatus</i>	0.39	0.67	0.48	1.07	1.35	20.38
<i>Lutjanus malabaricus</i>	0.45	0.83	0.47	1.05	1.34	21.71
<i>Chelonodon patoca</i>	0.59	0.33	0.47	1.05	1.33	23.04
<i>Anodontosoma chacunda</i>	0.57	0.33	0.46	1.04	1.31	24.35
<i>Chirocentrus dorab</i>	0.48	0.83	0.46	1.02	1.31	25.66

Table 3.6: Species identified by Simper as those who cumulatively explained the most 25%

dissimilarity in species composition between 2006 and all other years combined. Average

dissimilarity between 2006 and all other years was 35.49% (this includes all 160 species).

Species	Group Other	Group	Av.Diss	Diss/SD	Contrib%	Cum.%
	Years	2007				
	Av.Abund	Av.Abund				
<i>Lagocephalus lunaris</i>	0.17	1	0.76	2.11	2.2	2.2
<i>Platax teira</i>	0.24	1	0.7	1.74	2.02	4.23
<i>Pomadasys argenteus</i>	0.89	0.25	0.65	1.49	1.89	6.12
<i>Apogon quadrifasciatus</i>	0.37	1	0.58	1.28	1.68	7.8
<i>Pelates quadrilineatus</i>	0.39	1	0.56	1.23	1.61	9.41
<i>Scomberoides tol</i>	0.67	0.25	0.54	1.18	1.57	10.98
<i>Apolectus niger</i>	0.35	0.75	0.52	1.15	1.52	12.5
<i>Pelates sexlineatus</i>	0.37	0.75	0.51	1.12	1.49	13.98
<i>Leiohnathus blochii</i>	0.39	0.75	0.51	1.1	1.47	15.45
<i>Centriscus scutatus</i>	0.39	0.75	0.51	1.1	1.47	16.92
<i>Sphaerena putamae</i>	0.59	0.25	0.49	1.08	1.42	18.34
<i>Arius thalassinus</i>	0.48	1	0.48	1.03	1.39	19.74
<i>Pomadasys kakaan</i>	0.76	0.5	0.47	0.99	1.37	21.11
<i>Dussumieria elopsoides</i>	0.72	0.5	0.47	0.99	1.36	22.47
<i>Gazza achlamys</i>	0.52	0.25	0.47	1.01	1.35	23.82
<i>Eleutheronema tetradactylum</i>	0.83	0.5	0.46	0.99	1.35	25.17

Table 3.7: Species identified by Simper as those who cumulatively explained the most 25%

dissimilarity in species composition between 2007 and all other years combined. Average dissimilarity between 2007 and all other years was 34.53% (this includes all 160 species).

Species	Group Other	Group	Av.Diss	Diss/SD	Contrib%	Cum.%
	Years	2008				
	Av.Abund	Av.Abund				
<i>Lactarius lactarius</i>	0.91	0	0.85	3.1	2.29	2.29
<i>Stolephorus indicus</i>	1	0.25	0.7	1.7	1.87	4.16
<i>Thryssa mystax</i>	0.13	0.75	0.66	1.45	1.77	5.93
<i>Apogon poecilopterus</i>	0.33	1	0.64	1.41	1.72	7.65
<i>Scomberomorus queenslandicus</i>	0.78	0.25	0.62	1.32	1.65	9.3
<i>Lutjanus russelli</i>	0.22	0.75	0.6	1.32	1.61	10.91
<i>Triacanthus biaculeatus</i>	0.76	0.25	0.6	1.29	1.6	12.51
<i>Dussumieria elopsoides</i>	0.74	0.25	0.59	1.26	1.57	14.08
<i>Pelates quadrilineatus</i>	0.39	1	0.57	1.23	1.54	15.63
<i>Sphaerena putamae</i>	0.61	0	0.56	1.23	1.49	17.12
<i>Scomberoides tol</i>	0.67	0.25	0.56	1.18	1.49	18.61
<i>Paraplagusia bilineata</i>	0.59	0	0.55	1.18	1.48	20.09
<i>Alectis indicus</i>	0.59	0	0.55	1.18	1.48	21.58
<i>Herklotsichthys lippa</i>	0.33	0.75	0.55	1.18	1.47	23.05
<i>Trixiphichtys weberi</i>	0.59	0	0.55	1.18	1.47	24.52
<i>Anodontosoma chacunda</i>	0.59	0	0.55	1.18	1.46	25.98

Table 3.8: Species identified by Simper as those who cumulatively explained the most 25%

dissimilarity in species composition between 2008 and all other years combined. Average dissimilarity between 2008 and all other years was 37.24% (this includes all 160 species).

Species	Group Other	Group	Av.Diss	Diss/SD	Contrib%	Cum.%
	Years	2009				
	Av.Abund	Av.Abund				
<i>Lagocephalus spadiceus</i>	0.04	1	0.85	4.42	2.52	2.52
<i>Carangoides hedlandensis</i>	0.11	1	0.79	2.79	2.35	4.87
<i>Stolephorus nelsonii</i>	0.13	0.75	0.61	1.46	1.81	6.68
<i>Escualosa thoractata</i>	0.13	0.75	0.61	1.46	1.8	8.48
<i>Scomberoides tol</i>	0.7	0	0.61	1.5	1.8	10.28
<i>Upeneus sundaicus</i>	0.33	1	0.6	1.42	1.79	12.06
<i>Lagocephalus lunaris</i>	0.2	0.75	0.58	1.36	1.71	13.78
<i>Chelonodon patoca</i>	0.61	0	0.54	1.23	1.59	15.36
<i>Caranx ignobilis</i>	0.3	0.75	0.54	1.21	1.59	16.95
<i>Leptobrama mulleri</i>	0.3	0.75	0.54	1.21	1.58	18.53
<i>Sardinella gibbosa</i>	0.35	0.75	0.51	1.16	1.52	20.06
<i>Anodontosoma chacunda</i>	0.59	0	0.51	1.18	1.52	21.57
<i>Johnius amblycephalus</i>	0.61	0.25	0.49	1.11	1.46	23.03
<i>Nematalose come</i>	0.41	0.75	0.49	1.08	1.44	24.47
<i>Chirocentrus dorab</i>	0.48	1	0.47	1.04	1.39	25.87

Table 3.9: Species identified by Simper as those who cumulatively explained the most 25%

dissimilarity in species composition between 2009 and all other years combined. Average dissimilarity between 2009 and all other years was 33.85% (this includes all 160 species).

Species	Group		Av.Diss	Diss/SD	Contrib%	Cum.%
	Other Years	Group 2010				
<i>Johnius vogleri</i>	0.83	0	0.75	2.13	2.04	2.04
<i>Thryssa setirostris</i>	0.8	0	0.72	1.99	1.98	4.02
<i>Leiognathus leuciscus</i>	0.26	1	0.67	1.66	1.83	5.85
<i>Eluetheronema tetradactylum</i>	0.85	0.25	0.6	1.42	1.64	7.49
<i>Sphyaerena fosteri</i>	0.37	1	0.58	1.29	1.58	9.07
<i>Centriscus scutatus</i>	0.37	1	0.58	1.29	1.58	10.66
<i>Scomberomorus munroi</i>	0.24	0.75	0.57	1.29	1.57	12.22
<i>Herklotsichthys koningsbergen</i>	0.63	0	0.57	1.29	1.55	13.77
<i>Paraplagusia bilineata</i>	0.59	0	0.53	1.18	1.44	15.21
<i>Sardinella gibbosa</i>	0.35	0.75	0.53	1.15	1.44	16.65
<i>Upeneus sundaicus</i>	0.35	0.75	0.52	1.15	1.41	18.07
<i>Pelates sexlineatus</i>	0.37	0.75	0.51	1.13	1.4	19.47
<i>Gazza achlamys</i>	0.46	1	0.49	1.08	1.35	20.82
<i>Nematalose come</i>	0.41	0.75	0.49	1.08	1.34	22.16
<i>Chelonodon patoca</i>	0.59	0.25	0.49	1.08	1.33	23.49
<i>Gerres subfasciatus</i>	0.48	1	0.49	1.03	1.33	24.82
<i>Lutjanus malabaricus</i>	0.54	0	0.48	1.08	1.32	26.14

Table 3.10: Species identified by Simper as those who cumulatively explained the most 25%

dissimilarity in species composition between 2010 and all other years combined. Average dissimilarity between 2010 and all other years was 36.61% (this includes all 160 species).

Species	Group	Group				
	Other	Group	Av.Diss	Diss/SD	Contrib%	Cum.%
	Years	2011				
	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
<i>Leiognathus moretoniensis</i>	0.61	0	0.57	1.23	1.66	1.66
<i>Arius thallassinus</i>	0.57	0	0.54	1.13	1.56	3.22
<i>Centriscus scutatus</i>	0.39	0.75	0.53	1.1	1.55	4.77
<i>Herklotsichthys</i>						
<i>koningsbergen</i>	0.61	0.25	0.53	1.1	1.53	6.3
<i>Gerres subfasciatus</i>	0.57	0	0.52	1.13	1.52	7.82
<i>Alectis indicus</i>	0.57	0.25	0.51	1.06	1.47	9.29
<i>Scomberomorus commerson</i>	0.48	0.75	0.49	1.01	1.44	10.73
<i>Trichiurus lepturus</i>	0.91	0.5	0.49	0.99	1.43	12.16
<i>Lutjanus malabaricus</i>	0.48	0.75	0.49	1.01	1.43	13.58
<i>Anodontosoma chacunda</i>	0.5	1	0.49	0.99	1.42	15
<i>Thryssa setirostris</i>	0.76	0.5	0.49	0.99	1.41	16.42
<i>Gazza achlamys</i>	0.52	0.25	0.48	1.01	1.41	17.83
<i>Gerres filamentosus</i>	0.8	0.5	0.48	0.99	1.41	19.23
<i>Carangoides talamporooides</i>	0.22	0.5	0.48	0.99	1.41	20.64
<i>Dussumieria elopsoides</i>	0.72	0.5	0.48	0.99	1.4	22.04
<i>Chirocentrus dorab</i>	0.5	0.75	0.48	0.99	1.4	23.43
<i>Paraplaygusia bilineata</i>	0.5	1	0.48	0.99	1.4	24.83
<i>Stolephorus devisi</i>	0.2	0.5	0.48	0.99	1.39	26.22

Table 3.11: Species identified by Simper as those who cumulatively explained the most 25%

dissimilarity in species composition between 2011 and all other years combined. Average

dissimilarity between 2011 and all other years was 34.4% (this includes all 160 species).

Species	Group Other Years		Group 2012			
	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
<i>Saurida undosquamis</i>	0.91	0	0.88	3.14	2.48	2.48
<i>Sillago sihama</i>	0.98	0.25	0.72	1.66	2.02	4.5
<i>Leptobrama mulleri</i>	0.28	1	0.71	1.57	1.98	6.48
<i>Johnius coitor</i>	0.11	0.75	0.67	1.49	1.89	8.37
<i>Engraulis australis</i>	0.15	0.75	0.65	1.42	1.83	10.21
<i>Gerres filamentosus</i>	0.83	0.25	0.64	1.38	1.79	12
<i>Chelonodon patoca</i>	0.61	0	0.6	1.23	1.67	13.67
<i>Sphaerena putamae</i>	0.61	0	0.58	1.24	1.62	15.29
<i>Arius thallassinus</i>	0.57	0	0.55	1.13	1.54	16.83
<i>Gerres subfasciatus</i>	0.57	0	0.53	1.13	1.5	18.33
<i>Gazza achlamys</i>	0.48	0.75	0.5	1.01	1.41	19.74
<i>Chirocentrus dorab</i>	0.54	0.25	0.5	1.04	1.41	21.15
<i>Saurida micropectoralis</i>	0.96	0.5	0.49	0.99	1.38	22.54
<i>Paraplaygusia bilineata</i>	0.5	1	0.49	0.99	1.38	23.92
<i>Scomberomorus munroi</i>	0.26	0.5	0.49	0.99	1.37	25.29

Table 3.12: Species identified by Simper as those who cumulatively explained the most 25%

dissimilarity in species composition between 2012 and all other years combined. Average dissimilarity between 2012 and all other years was 35.66% (this includes all 160 species).

<i>Alepes djebada</i>	<i>Lagocephalus scleratus</i>	<i>Pseudorhombus argus</i>
<i>Alutera monooceros</i>	<i>Leiognathus stercorarius</i>	<i>Pterois volitans</i>
<i>Argirops spinnifer</i>	<i>Lutjanus argentimaculatus</i>	<i>Rachycentron canadrum</i>
<i>Carangoides humerosus</i>	<i>Lutjanus carponotatus</i>	<i>Sardinella fimbriata</i>
<i>Elates ransonetti</i>	<i>Monacanthus chinensis</i>	<i>Siganus punctatus</i>
<i>Engyrosopon grandisquama</i>	<i>Nemipterus forcosus</i>	<i>Terapon sexlineatus</i>
<i>Epinephelus malabaricus</i>	<i>Plectorhinchus gibbosus</i>	<i>Torquigener hicksi</i>
<i>Gnathanodon speciosus</i>	<i>Psammodicius ocellatus</i>	

Table 3.13: List of species caught on only one day during the 15 year study period

Appendix 4:

List of species identified in trawl samples from 1998 – 2012.

Family	Genus	Species
Apogonidae	<i>Apogon</i>	<i>elliotti</i>
Apogonidae	<i>Apogon</i>	<i>semilineatus</i>
Apogonidae	<i>Apogon</i>	<i>septemstriatus</i>
Apogonidae	<i>Apogon</i>	<i>poecilopterus</i>
Apogonidae	<i>Apogon</i>	<i>quadrifasciatus</i>
Apogonidae	<i>Archamia</i>	<i>fucata</i>
Ariidae	<i>Arius</i>	<i>thalassinus</i>
Bothidae	<i>Engyrosopon</i>	<i>grandisquama</i>
Carangidae	<i>Absalom</i>	<i>radiatus</i>
Carangidae	<i>Alectis</i>	<i>ciliaris</i>
Carangidae	<i>Alectis</i>	<i>indicus</i>
Carangidae	<i>Alepes</i>	<i>djebada</i>
Carangidae	<i>Atule</i>	<i>mate</i>
Carangidae	<i>Carangoides</i>	<i>caeruleopinnatus</i>
Carangidae	<i>Carangoides</i>	<i>chrysophrys</i>
Carangidae	<i>Carangoides</i>	<i>hedlandensis</i>
Carangidae	<i>Carangoides</i>	<i>humerosus</i>
Carangidae	<i>Carangoides</i>	<i>malabaricus</i>
Carangidae	<i>Carangoides</i>	<i>talamporoides</i>
Carangidae	<i>Caranx</i>	<i>bucculentus</i>
Carangidae	<i>Caranx</i>	<i>ignobilis</i>
Carangidae	<i>Caranx</i>	<i>para</i>
Carangidae	<i>Gnathanodon</i>	<i>speciosus</i>
Carangidae	<i>Megalapis</i>	<i>cordila</i>
Carangidae	<i>Parastomatus</i>	<i>niger</i>
Carangidae	<i>Scomberoides</i>	<i>tala</i>
Carangidae	<i>Scomberoides</i>	<i>tol</i>
Carangidae	<i>Selaroides</i>	<i>leptolepis</i>
Carangidae	<i>Seriolina</i>	<i>nigrofasciata</i>
Carangidae	<i>Ulna</i>	<i>aurochs</i>
Centriscidae	<i>Centriscus</i>	<i>scutatus</i>
Chaetodontidae	<i>Paracheatodon</i>	<i>ocellatus</i>
Chirocentridae	<i>Chirocentrus</i>	<i>dorab</i>
Clupeidae	<i>Anodontosoma</i>	<i>chacunda</i>
Clupeidae	<i>Dussumieria</i>	<i>elopsoides</i>
Clupeidae	<i>Escualosa</i>	<i>thoractata</i>
Clupeidae	<i>Herklotsichthys</i>	<i>koningsbergen</i>
Clupeidae	<i>Herklotsichthys</i>	<i>lippa</i>
Clupeidae	<i>Nematalose</i>	<i>come</i>
Clupeidae	<i>Pellona</i>	<i>ditchela</i>
Clupeidae	<i>Sardinella</i>	<i>albella</i>
Clupeidae	<i>Sardinella</i>	<i>brachysoma</i>

Clupeidae	<i>Sardinella</i>	<i>fimbriata</i>
Clupeidae	<i>Sardinella</i>	<i>gibbosa</i>
Cynoglossidae	<i>Cynoglossus</i>	<i>macrophthalmus</i>
Cynoglossidae	<i>Paraplaygusia</i>	<i>bilineata</i>
Cynoglossidae	<i>Paraplaygusia</i>	<i>guttata</i>
Engraulidae	<i>Engraulis</i>	<i>australis</i>
Engraulidae	<i>Papuengraulis</i>	<i>micropinna</i>
Engraulidae	<i>Stolephorus</i>	<i>devisi</i>
Engraulidae	<i>Stolephorus</i>	<i>indicus</i>
Engraulidae	<i>Stolephorus</i>	<i>nelsonii</i>
Engraulidae	<i>Thryssa</i>	<i>hamiltonii</i>
Engraulidae	<i>Thryssa</i>	<i>mystax</i>
Engraulidae	<i>Thryssa</i>	<i>setirostris</i>
Ephippidae	<i>Drepane</i>	<i>punctata</i>
Ephippidae	<i>Platax</i>	<i>teira</i>
Fistularidae	<i>Fistularia</i>	<i>commersonii</i>
Formionidae	<i>Apolectus</i>	<i>niger</i>
Gerreidae	<i>Gerres</i>	<i>filamentosus</i>
Gerreidae	<i>Gerres</i>	<i>oyena</i>
Gerreidae	<i>Gerres</i>	<i>subfasciatus</i>
Gerreidae	<i>Pentaprion</i>	<i>longimanus</i>
Haemulidae	<i>Plectorhinchus</i>	<i>gibbosus</i>
Haemulidae	<i>Pomadasys</i>	<i>argenteus</i>
Haemulidae	<i>Pomadasys</i>	<i>kakaan</i>
Haemulidae	<i>Pomadasys</i>	<i>maculatum</i>
Hemiramphidae	<i>Hyporhamphus</i>	<i>affinis</i>
Lactariidae	<i>Lactarius</i>	<i>lactarius</i>
Leiognathidae	<i>Gazza</i>	<i>achlamys</i>
Leiognathidae	<i>Gazza</i>	<i>minuta</i>
Leiognathidae	<i>Leiognathus</i>	<i>aureus</i>
Leiognathidae	<i>Leiognathus</i>	<i>bindus</i>
Leiognathidae	<i>Leiognathus</i>	<i>blochii</i>
Leiognathidae	<i>Leiognathus</i>	<i>decorus</i>
Leiognathidae	<i>Leiognathus</i>	<i>equulus</i>
Leiognathidae	<i>Leiognathus</i>	<i>leuciscus</i>
Leiognathidae	<i>Leiognathus</i>	<i>moretoniensis</i>
Leiognathidae	<i>Leiognathus</i>	<i>smithursi</i>
Leiognathidae	<i>Leiognathus</i>	<i>splendens</i>
Leiognathidae	<i>Leiognathus</i>	<i>stercorarius</i>
Leiognathidae	<i>Secutor</i>	<i>insidiator</i>
Leiognathidae	<i>Secutor</i>	<i>ruconius</i>
Leptobramidae	<i>Leptobrama</i>	<i>mulleri</i>
Lethrinidae	<i>Lethrinus</i>	<i>lentjan</i>
Lethrinidae	<i>Lethrinus</i>	<i>nebulosus</i>
Lethrinidae	<i>Lethrinus</i>	<i>nematacanthus</i>
Lutjanidae	<i>Lutjanus</i>	<i>argentimaculatus</i>

Lutjanidae	<i>Lutjanus</i>	<i>carponotatus</i>
Lutjanidae	<i>Lutjanus</i>	<i>fulviflamma</i>
Lutjanidae	<i>Lutjanus</i>	<i>malabaricus</i>
Lutjanidae	<i>Lutjanus</i>	<i>russelli</i>
Monacanthidae	<i>Alutera</i>	<i>monooceros</i>
Monacanthidae	<i>Monacanthus</i>	<i>chinensis</i>
Mugilidae	<i>Mugil</i>	<i>cephalus</i>
Mullidae	<i>Upeneus</i>	<i>assymmetricus</i>
Mullidae	<i>Upeneus</i>	<i>mollocensis</i>
Mullidae	<i>Upeneus</i>	<i>sulphureus</i>
Mullidae	<i>Upeneus</i>	<i>sundaicus</i>
Nemipteridae	<i>Nemipterus</i>	<i>forcosus</i>
Nemipteridae	<i>Nemipterus</i>	<i>mesoprion</i>
Nemipteridae	<i>Nemipterus</i>	<i>peronii</i>
Paralichthyidae	<i>pseudorhombus</i>	<i>argus</i>
Paralichthyidae	<i>Pseudorhombus</i>	<i>arsius</i>
Paralichthyidae	<i>Pseudorhombus</i>	<i>dupliciocellatus</i>
Platycephalidae	<i>Elates</i>	<i>ransonetti</i>
Platycephalidae	<i>Platycephalus</i>	<i>endrachtensis</i>
Platycephalidae	<i>Suggrundus</i>	<i>isacanthus</i>
Pleuronectidae	<i>Psammodicius</i>	<i>ocellatus</i>
Plotosidae	<i>Euristhmus</i>	<i>lepturus</i>
Plotosidae	<i>Euristhmus</i>	<i>nudiceps</i>
Polynemidae	<i>Eleutheronema</i>	<i>tetradactylum</i>
Polynemidae	<i>Polydactylus</i>	<i>multiradiatus</i>
Psettodidae	<i>Psettodes</i>	<i>erumei</i>
Rachycentridae	<i>Rachycentron</i>	<i>canadrum</i>
Sapridae	<i>Argirops</i>	<i>spinnifer</i>
Scatophagidae	<i>Selenotoca</i>	<i>multifasciata</i>
Sciaenidae	<i>Austronibae</i>	<i>oedogenys</i>
Sciaenidae	<i>Johnius</i>	<i>amblycephalus</i>
Sciaenidae	<i>Johnius</i>	<i>coitor</i>
Sciaenidae	<i>Johnius</i>	<i>vogleri</i>
Sciaenidae	<i>Otilithes</i>	<i>ruber</i>
Scombridae	<i>Rastrelliger</i>	<i>kanagurta</i>
Scombridae	<i>Scomberomorus</i>	<i>commerson</i>
Scombridae	<i>Scomberomorus</i>	<i>munroi</i>
Scombridae	<i>Scomberomorus</i>	<i>queenslandicus</i>
Scombridae	<i>Scomberomorus</i>	<i>semifasciatus</i>
Scorpaenidae	<i>Pterois</i>	<i>volitans</i>
Serranidae	<i>Epinephelus</i>	<i>malabaricus</i>
Serranidae	<i>Epinephelus</i>	<i>sexfasciatus</i>
Siganidae	<i>Siganus</i>	<i>fuscescens</i>
Siganidae	<i>Siganus</i>	<i>punctatus</i>
Sillaginidae	<i>Sillago</i>	<i>analis</i>
Sillaginidae	<i>Sillago</i>	<i>maculata</i>

Sillaginidae	<i>Sillago</i>	<i>sihama</i>
Sphyraenidae	<i>Sphyraerena</i>	<i>flavicauda</i>
Sphyraenidae	<i>Sphyraerena</i>	<i>fosteri</i>
Sphyraenidae	<i>Sphyraerena</i>	<i>obstutata</i>
Sphyraenidae	<i>Sphyraerena</i>	<i>putamae</i>
Synodontidae	<i>Saurida</i>	<i>micropectoralis</i>
Synodontidae	<i>Saurida</i>	<i>undosquamis</i>
Teraponidae	<i>Pelates</i>	<i>quadrilineatus</i>
Teraponidae	<i>Pelates</i>	<i>sexlineatus</i>
Teraponidae	<i>Terapon</i>	<i>jarbua</i>
Teraponidae	<i>Terapon</i>	<i>sexlineatus</i>
Teraponidae	<i>Terapon</i>	<i>puta</i>
Teraponidae	<i>Terapon</i>	<i>theraps</i>
Tetradontidae	<i>Arothron</i>	<i>manillensis</i>
Tetradontidae	<i>Chelonodon</i>	<i>patoca</i>
Tetradontidae	<i>Lagocephalus</i>	<i>lunaris</i>
Tetradontidae	<i>Lagocephalus</i>	<i>scleratus</i>
Tetradontidae	<i>Lagocephalus</i>	<i>spadiceus</i>
Tetradontidae	<i>Torquigener</i>	<i>hicksi</i>
Tetradontidae	<i>Torquigener</i>	<i>pallimaculatus</i>
Tetradontidae	<i>Torquigener</i>	<i>whitleyi</i>
Triacanthidae	<i>Pseudotriacanthus</i>	<i>strigilifer</i>
Triacanthidae	<i>Triacanthus</i>	<i>biaculeatus</i>
Triacanthidae	<i>Triacanthus</i>	<i>nieuhofi</i>
Triacanthidae	<i>Trixiphichtys</i>	<i>weberi</i>
Trichiuridae	<i>Trichiurus</i>	<i>lepturus</i>

Appendix 5:

Year	1998	1999	2002	2004	2007	2008	2009	2011
CPUE (kg) March	133.24	137.72	155.83	39.10	87.83	166.68	287.26	337.02

Table 5.1: Commercial catch data (Grid J21) of *Fenneropenaeus merguensis* in March from 1998 – 2011. (Some monthly catch data could not be included as it was confidential).