Integrating fishery-independent and -dependent data for improved sustainability of fisheries resources and other aspects of biodiversity

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In submitting this report, the researcher has agreed to FRDC publishing this material in its edited form.

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The final reporting stage of the project was novated to the Sydney Institute of Marine Science (SIMS) following the closure of the Cronulla Fisheries Research Centre of Excellence. The novation occurred in April 2014 and Prof Charles Gray (WildFish Research) was contracted by SIMS to finalise the reporting of the project.

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This project was novated to the Sydney Institute of Marine Science in April 2014. The final analyses, interpretations and subsequent reporting of the project were dependent on the data provided by NSW DPI to SIMS at the time of the novation. SIMS and WildFish Research take no responsibility for any errors or incompleteness of the data provided to them for this report and subsequent publications.

NSW DPI did not provide all the project information to SIMS or the contractor with the novation of the project, specifically:

(i) Data that contributed to Experiment 2 - the recreational creel surveys and accompanying fishery-independent surveys,
(ii) Data that contributed to Experiment 3 - developing and testing sampling gears and the comparisons of sampling strategies for the giant mud crab, and
(iii) The costs associated with each experiment.

Furthermore, the otoliths for several secondary species collected as part of various components of the project were lost (prior to being aged) during the closure of the Cronulla Fisheries Research Centre of Excellence, thus limiting comparisons of age compositions to key species in most experiments.

These circumstances did not impact on the project outcomes or recommendations and the overall objectives of the project were successfully achieved. A separate report detailing the recreational creel surveys (Ochwada et al. 2014) has been published, but rigorous comparisons of the fishery-independent and –dependent data are required. Partial reporting of the mud crab experiments are contained within this report (Appendix 5) and NSW DPI staff have drafted separate manuscripts for journal publications concerning this work.

Executive Summary

Overview:

This project was the first to examine the utility of using fishery-independent sampling strategies compared to traditional fishery-dependent sampling strategies for assessing the aquatic biodiversity and fisheries resources in estuaries of New South Wales (NSW), Australia. This was done using a series of novel field-based experiments across several estuaries that tested specific hypotheses concerning each project objective. This research was required to determine the most suitable and reliable sampling strategies to provide the necessary data for sustainable 'ecosystem-based' management of the estuarine fisheries resources of NSW. Ecosystem-based management requires data across all components of the environment, not just harvested species, so that effects on biodiversity can be examined. This project was originally a partnership between the Fisheries Research and Development Corporation and NSW Department of Primary Industries. The project results have applicability to other systems and jurisdictions in Australia and elsewhere.

Background:

Agencies that manage fisheries resources typically use fishery-dependent sampling strategies to obtain data on fish populations for species and fisheries performance assessments. This approach typically involves the surveying of commercial and recreational catches for length and age compositions, and fisher's self-reporting catch and effort information. Such sampling strategies have limitations and are often biased due to the behaviour of fishers.

An alternate strategy involves collecting relative abundance and population demographic data of species using research surveys that are independent of a fishery. Such so called 'fishery-independent sampling' often have fewer limitations and can be used to: (1) assess changes in abundances and structures of harvested and non-harvested populations of fish and invertebrates; (2) investigate patterns of biodiversity; (3) monitor aquatic resources where fishing has been modified as part of management (e.g. recreational fishing havens and marine parks) and (4) assess environmental impacts on fish populations.

There still exists considerable debate, however, about the relative value of fishery-independent and fishery-dependent data for assessing fisheries resources and biodiversity (particularly in estuaries and for small-scale, multi-species fisheries). This is because there have been few comparisons of the two types of data collection strategies. Such comparisons could be used to identify future large-scale and long-term sampling strategies that will provide the most cost-effective and reliable data for the sustainable management of the fisheries resources and aquatic biodiversity of estuaries in NSW.

Aims and objectives:

This project aimed to provide robust and practical demonstrations of the roles of fishery-independent and fishery-dependent data for the assessment and management of the estuarine fisheries resources and biodiversity in NSW. The specific project objectives were:

1. Evaluate the effectiveness of a standardised fishery-independent sampling strategy compared with sources of fishery-dependent data (e.g. data from commercial and recreational fisheries) for assessing fisheries resources and biodiversity

2. Investigate the extent to which fishery-independent data reduce uncertainty in the management of estuarine fisheries resources and lead to decisions that are more reliable and robust

3. Examine the values of fishery-independent sampling for use across estuaries with different management regimes (e.g. estuaries open and closed to commercial and recreational fishing; marine parks) and for assessing the impacts of immediate environmental perturbations (e.g. floods, pollution) and those in the future (e.g. impacts of climatic change on the dynamics of populations of fish and diversity of fish assemblages)

Methodology:

A series of field-based experiments were done to test specific project objectives. A fishery-independent sampling strategy that incorporated two sampling gears (multimesh gillnet and beam trawl) was compared against traditional fishery-dependent strategies of sampling commercial fishers catches (port-based sampling at fishing cooperatives) and recreational anglers catches (creel survey). Data on catch-per-unit-of-effort, length- and age-based demographics of key species were compared across strategies.

The standardised fishery-independent sampling strategy was further used to compare fish populations and assemblages across estuaries with different management arrangements, the responses of fish to a flood event and spatial and temporal differences in the biodiversity of assemblages sampled using each gear type. These experiments examined the value of the fishery-independent sampling strategy for aquatic biodiversity and fisheries resource assessments.

Results/key findings:

This study successfully tested a standardised fishery-independent sampling strategy against typical fishery-dependent strategies for assessing aquatic biodiversity and fisheries resources across estuaries. In doing so it identified strengths and weaknesses in each of the sampling strategies.

The fishery-independent strategy sampled a greater diversity of species and was more superior to quantitatively assess spatial and temporal changes in aquatic biodiversity in a standardised framework compared to the fishery-dependent strategies. This was primarily due to the fishery-dependent samples being limited to harvested species, whereas the fishery-independent gears sampled the same harvested species as well as a plethora of non-harvested species. The most complete picture of assemblages and populations of fish required the combined use of the multimesh gillnet and beam trawl.

Compared to the fishery-dependent sampling, the fishery-independent sampling strategy provided greater representations of the length- and age-based population characteristics of each key harvested species examined. This was because the fishery-dependent samples were limited to legal length fish, which also limited the use of these data for determination of growth and mean length-at-age. Nevertheless, differences in estimates of mortality and exploitation based on the fishery-independent and -dependent samples were inconsistent among species. Further investigations are needed to determine if differences in mortality estimates would affect assessment outcomes for each species.

The sampling of commercial and recreational fishers retained catches can provide very good information on the species, length and age compositions of catches. But, this is only available in estuaries where each type of fishing occurs and where fishers willingly cooperate with sampling. The commercial fisher-reported logbooks have restricted value for assessment purposes at present due to inconsistencies in reported fishing effort and the privacy regulations that restrict usage of data for analyses across some spatial and temporal scales.

The demonstrable benefits of the standardised fishery-independent sampling strategy also included its ability to detect differences in the demographic characteristics of fish species across estuaries of different management arrangements (estuaries open or closed to commercial fishing) and the provision of reliable data on the responses of species to an environmental perturbation (flood event). The latter data reduced uncertainty and made for decisive management actions that were of benefit to the commercial, recreational and indigenous fishers, stakeholder groups and the local community.

The combined results across experiments provide overwhelming evidence of the value and utility of a standardised fishery-independent sampling strategy compared to typical fishery-dependent strategies in assessing estuarine fisheries resources and biodiversity. The strength of the fishery-independent strategy is that it provides a consistent sampling frame that can be used across all types of estuaries regardless of fisheries management arrangements. This is not available using the fishery-dependent strategies.

A standardised fishery-independent sampling strategy may provide the only consistent framework to deliver robust and reliable data essential for assessing and managing the fisheries resources, fisheries management performance and aquatic biodiversity across the breadth of estuaries of NSW.

Implications for relevant stakeholders:

This study identified the utility and value of a standardised fishery-independent sampling strategy for assessing the biodiversity and fisheries resources across estuaries of NSW. The results reported here have applicability to other systems and species in other management jurisdictions in Australia and elsewhere. Incorporation of a fishery-independent sampling strategy into the assessment of a natural flood event proved beneficial to the NSW Government, commercial, recreational and indigenous fishers, associated stakeholders and the local community. Implementation of a broad-scale standardised fishery-independent sampling and assessment strategy could have synergistic benefits across management agencies and stakeholder groups associated with the management of aquatic biodiversity, fisheries resources and fisheries of NSW.

Recommendations:

The agencies that oversee the management of the marine estate of NSW need to consider the across government synergies and benefits of implementing a broad-scale standardised fishery-independent sampling strategy across estuaries for provision of robust and reliable data for the sustainable management of the states aquatic biodiversity and natural fisheries resources.

Keywords: Fishery-independent sampling; Fishery-dependent sampling; Survey; Monitoring; Age-based demographic analyses; Resource assessment; Biodiversity; Management; Conservation; Estuary; Fish; Australia.

1. Introduction

1.1. Background and Need

The Fisheries Research and Development Corporation, in partnership with the NSW Department of Primary Industries and the Centre for Research on Ecological Impacts of Coastal Cities at the University of Sydney, made substantial investment in the development of fishery-independent surveys of fisheries resources in estuaries of NSW. A previous FRDC Project (2002/059) produced a strategy for developing fishery-independent sampling tools and surveys (Rotherham et al. 2007, 2008). Using this strategy, two types of sampling gears (multimesh gillnet and a beam trawl) were evaluated over a hierarchy of spatial and temporal scales (Gray et al. 2009; Rotherham et al. 2011). The project provided sampling designs for standardised surveys of populations and communities of estuarine fish in NSW.

Data from fishery-independent research surveys can be used to: (1) assess changes in abundances and structures of harvested and non-harvested populations of fish and invertebrates; (2) investigate patterns of biodiversity; (3) monitor aquatic resources where fishing has been modified as part of management (e.g. recreational fishing havens and marine parks) and (4) assess environmental impacts on fish populations. Nevertheless, there still exists considerable debate in Australia and elsewhere about the relative value of fishery-independent and fishery-dependent data for assessing fisheries resources and biodiversity (particularly for small-scale, multi-species fisheries). This is because there have been few comparisons of the two types of data and their relative costs and benefits.

The primary benefit of fishery-independent surveys is that the data are potentially less prone to biases and imprecision than is typical of fishery-dependent data. However, are fishery-independent data any better than information available from fishery-dependent sources? Do such alternate data actually improve decision-making? Is there a sufficient marginal benefit to that expenditure? These questions can only be answered by doing research that considers the managerial interpretation of the alternative sources of data. Therefore, before commencing large-scale and long-term surveys, the crucial logical step is to compare the fishery-independent sampling strategy with data from both commercial and recreational fisheries. This involves sampling using fishery-independent surveys across a number of estuaries, open or closed to commercial and recreational fishing and over time scales of 1-3 years, plus additional sampling from fishery-dependent sources (e.g. ports and fishing co-operatives, creel surveys, etc.). Such a program could be used identify future large-scale and long-term sampling strategies that will provide the most cost-effective and reliable data for the sustainable management of the fisheries resources and aquatic biodiversity of estuaries in NSW.

1.2. Aims

This specific project aimed to provide robust and practical demonstrations of the roles of fishery-independent and fishery-dependent data for the assessment and management of the estuarine fisheries resources and biodiversity in NSW. This was done using an experimental approach to test the relative value of the different sources of data. The fishery-independent sampling tools developed in FRDC Project 2002/059 were implemented across a number of estuaries with different management regimes (i.e. open and closed to commercial/recreational fishing). Fishery-dependent data from commercial and recreational fisheries (i.e. fisher-reported catch and effort data, port monitoring of landings, creel surveys of recreational fishing) were also collected simultaneously across estuaries. The utility of each type of data was therefore tested over equivalent spatial and temporal scales. This provided a scientific basis for determining the most appropriate mix of fishery-independent and fishery–dependent data for the improved sustainability of the fisheries resources and biodiversity in estuaries of NSW.

2. Objectives

The objectives of the project as agreed in the original contract with the NSW Department of Primary Industries were:

1. Evaluate the effectiveness of a standardised fishery-independent sampling strategy compared with sources of fishery-dependent data (e.g. data from commercial and recreational fisheries) for assessing fisheries resources and biodiversity

2. Investigate the extent to which fishery-independent data reduce uncertainty in the management of estuarine fisheries resources and lead to decisions that are more reliable and robust

3. Examine the values of fishery-independent sampling for use across estuaries with different management regimes (e.g. estuaries open and closed to commercial and recreational fishing; marine parks) and for assessing the impacts of immediate environmental perturbations (e.g. floods, pollution) and those in the future (e.g. impacts of climatic change) on the dynamics of populations of fish and diversity of fish assemblages

3. Methods

3.1. General overview

A series of field-based experiments were done to test specific project objectives. These included:

1. Comparison of fishery-independent and fishery-dependent (commercial) sampling strategies to assess harvested fish species (Objective 1, 2, 3)
2. Comparison of fishery-independent and fishery-dependent (recreational) sampling strategies to assess harvested fish species and aquatic biodiversity (Objective 1, 2, 3)
3. Development of a fishery-independent sampling strategy for the giant mud crab and comparison with fishery-dependent sampling of commercial catches (Objective 1, 2)
4. Utility of fishery-independent sampling to assess the fisheries resources in estuaries with different management regimes (Objective 2, 3)
5. Utility of fishery-independent sampling to assess aquatic biodiversity across estuaries (Objective 3)
6. Utility of fishery-independent sampling to reduce uncertainty in management decisions (Objective 2, 3)

The specific experiments were done in several estuaries across NSW, including the Richmond, Clarence, Bellinger, Kalang and Macleay Rivers, Wallis Lake, Lake Macquarie, Tuggerah Lake, St Georges Basin and Wallaga Lake.

3.2. Fishery-independent sampling gears

Two previously developed sampling gears were used in the fishery-independent sampling strategy in each experiment (excluding the crab trap); a multimesh gillnet and a beam trawl. Each multimesh gillnet consisted of seven 20-m panels of different sizes of stretched mesh (36, 44, 54, 63, 76, 89 and 102 mm) connected together in a random order (Gray et al. 2005, 2009). Adjacent panels were separated by 5 m of rope to minimise potential effects of the smaller-mesh panels ‘leading’ larger fish to adjacent panels. The 36- and 44-mm panels were made from monofilament netting with twine diameter of 0.15 mm; the remaining panels had multifilament netting with a twine diameter of 0.4 mm. All panels were 2 m deep and had a hanging ratio of 0.5. Nets set for one hour at night upon dusk and retrieved one hour later (Rotherham et al. 2006, 2011).

The beam trawl was 3-m wide and configured with 41-mm diamond-shaped mesh in the body and 20-mm mesh hung on the bar (i.e. square-shaped) in the codend (Rotherham et al. 2008a,b). The net was towed for 5 minutes (Rotherham et al. 2008a).

A brief rationale and description of each experiment is summarised below. Full descriptions of the methodologies, analyses, results and outcomes are provided in the accompanying appendices.

3.3. Experiment 1: Comparison of fishery-independent and fishery-dependent (commercial) sampling strategies to assess harvested fish species (Appendix 3)

Length- and age-based information and estimates of catch-per-unit-effort (CPUE) data are vital to many fisheries resource assessments and form the basis for determining the exploitation status of harvested fish species in NSW.

This experiment tested for differences in indices of CPUE and length- and age-based demographics of five harvested fish species sampled using the multimesh gillnet and beam trawl as part of the fishery-independent sampling strategy and the port-based sampling of commercial fishers retained catches as part of the NSW DPI Resource Assessment and Monitoring Program. Sampling using each strategy was done in the Clarence River and Wallis Lake between February and July in 2008, 2009 and 2010. In each estuary, standardised fishery-independent sampling was stratified across two periods each year, whilst the sampling at cooperatives was stratified temporally. Length and age samples were collected for each species each year. Standard population demographic indices used in fisheries resource assessment procedures, including the length and age compositions of samples, length-at-age and growth, rates of mortality and exploitation, and catch-per-unit-of-effort data were compared for each species between the two sampling strategies. This was done for five key fish species: bream (*Acanthopagrus australis*), dusky flathead (*Platycephalus fuscus*), luderick (*Girella tricuspidata*), sea mullet (*Mugil cephalus*) and sand whiting (*Sillago ciliata*).

3.4. Experiment 2: Comparison of fishery-independent and fishery-dependent (recreational) sampling strategies to assess harvested fish species and aquatic biodiversity (Appendix 4)

This experiment tested for differences in aquatic biodiversity and the length compositions and temporal trends in CPUE of key fish species sampled using the fishery-independent sampling strategy compared to a fishery-dependent survey of recreational anglers.

This work was done over 3 months in 2011 in two recreational fishing havens, Lake Macquarie and St Georges Basin. In each estuary, standardised fishery-independent sampling using the multimesh gillnet and beam trawl was stratified across two six week periods, whilst the concurrent creel survey comprised two contact methods (access-point and roving) to collect data on recreational harvests and fishing effort (Ochwada-Doyle et al. 2014). Each organism obtained in samples from both strategies was identified and key species measured for length.

The diversity of species and the length compositions and trends in CPUE of key species in samples from each strategy were compared.

3.5. Experiment 3: Development of a fishery-independent sampling strategy for the giant mud crab and comparison with fishery-dependent sampling of commercial catches (Appendix 5)

This particular case study had several phases. The first phase developed and tested appropriate trap-based gears and methodologies for sampling estuarine populations of the giant mud crab *Scylla serrata*. Several trap designs that incorporated different numbers of entrances and configurations were tested in two estuaries. The second phase tested the optimal trap design over a hierarchy of spatial and temporal scales to determine levels of replication required for future broader scale sampling surveys of populations across estuaries. The results from these experiments were then used to develop an optimal strategy to sample populations of giant mud crabs across estuaries and throughout time. The third phase involved using the optimal strategy in another two estuaries so that comparisons of the numbers and sizes of crabs and their rates of capture could be compared to fishery-dependent sampling strategies, including the mandatory commercial catch-return data, data collected by commercial fishers sampling their own catches and from fisheries observers who accompanied fishers during normal fishing operations. The results from the third phase of sampling are reported here. Specifically, relative abundance (CPUE) data and derived population demographic parameters of the giant mud crab attained from the fishery-independent and -dependent sources of sampling were compared.

3.6. Experiment 4: Utility of fishery-independent sampling to assess the fisheries resources in estuaries with different management regimes (Appendix 6)

The objective of this experiment was to examine whether the relative abundances and demographic characteristics of key harvested fish species differed between estuaries that had different management regimes; namely recreationally only fished estuaries (i.e. recreational fishing havens) compared to recreationally and commercially fished estuaries.

Fishery-independent sampling using the multimesh gillnet and beam trawl was done in a standard and stratified manner across two recreational fishing havens (Lake Macquarie and St Georges Basin) and two non-havens (Wallis and Tuggerah Lakes) over 3 years (2008-2010). Differences in the relative abundances (CPUE) and length compositions of abundant species across estuary types were tested for each sampling gear. For five species that were also aged, further analyses tested for differences in age compositions and rates of mortality and exploitation between the different estuary types. This experiment documented the utility of using standardised fishery-independent research sampling gears to assess fish populations and their potential responses to different management regimes.

3.7. Experiment 5: Utility of fishery-independent sampling to assess aquatic biodiversity across estuaries (Appendix 7)

The monitoring, evaluation and reporting of spatial and temporal changes in biodiversity due to natural and anthropogenic impacts is mandated by resource management agencies worldwide. This experiment specifically examined whether the multimesh gillnet and the beam trawl sampled different components of assemblages and populations of aquatic fauna in estuaries to determine their potential suitability for use in aquatic biodiversity assessments across estuaries in NSW.

Standardised fishery-independent surveys using the multimesh gillnet and beam trawl were done across five estuaries over two years. For each sampling gear, information on the numbers and diversity of organisms and the length compositions of specific species of fish were examined in each estuary. Multivariate analyses were used to test whether the sampled assemblages differed according to gear type and whether these were consistent across estuaries, within and between years. The results are discussed in terms of using either gear as a sampling tool and developing strategies for assessing estuarine aquatic diversity.

3.8. Experiment 6: Utility of fishery-independent sampling to reduce uncertainty in management decisions (Appendix 8)

This experiment was done immediately following a major natural flood event in the Richmond River in January 2008, which killed many thousands of fish and invertebrates. Fishing to commercial harvesters and recreational anglers in the river was closed subsequent to the demonstrated recovery of fish populations. The sampling of fish populations in this experiment was used to determine when the river would be reopened to fishing.

Approximately six weeks after the flood, fishery-independent sampling using the multimesh gillnet as well as fisher-dependent sampling was done in different sections of the Richmond River over a period of one week. Standardized fishery-independent sampling was also done concurrently in other estuaries in NSW to benchmark the catch rates and population structures of key species between the Richmond River and other estuaries not affected by the flood. The data were provided to management and stakeholders and used to determine the eventual reopening of the river to fishing. Comparisons of the research sampling strategy used here and subsequent impacts on management responses and decisions were made to a similar flood event that occurred in 2001, when no fishery-independent data were available to management.

4. Results

Specific details of each experiment are provided in the accompanying appendices. A general summary of the major results of each experiment is provided here.

4.1. Experiment 1: Comparison of fishery-independent and fishery-dependent (commercial) sampling strategies to assess harvested fish species (Appendix 3)

Except for *M. cephalus*, a greater number of length and age samples were obtained for each species across both estuaries in the fishery-dependent sampling than in the fishery-independent sampling. Each year it took between 10 and 25 days (1 person, 3-5 hours per day) to do the fishery-dependent sampling in each estuary, compared to the 4 to 6 nights it took to do the gillnet and 2 nights (2 persons, 6-8 hours each night) to do the beam trawl sampling in each estuary. Across both estuaries, greater numbers of each species except *A. australis* were sampled in the gillnet than in the beam trawl. Comparisons between the two sampling strategies for all species except *A. australis* were therefore limited to the gillnet.

Indices of CPUE could not be compared between the two sampling strategies due to the lack of effort data available for commercial fishing. Whilst this was primarily a response to recent changes in reporting requirements, it could potentially be a long-term issue clouding fisheries assessments. In comparison, significant spatial and temporal differences were detected in the CPUE data for each species from the standardised fishery-independent gillnet sampling. This demonstrates how such data could potentially be used as an index of relative abundance and CPUE of harvested species across estuaries and throughout time for resource assessments.

The length- and age-based demographic components of each species differed between the two sampling strategies. The fishery-dependent data were restricted to legal length fish, whereas the fishery-independent gillnets sampled sub-legal as well as legal length individuals of all five species. Across both estuaries the mean length and age of samples of *A. australis*, *G. tricuspidata*, *P. fuscus* and *M. cephalus* were greater in the fishery-dependent than in the fishery-independent samples. This was also evident for *S. ciliata* in the Clarence River but not in Wallis Lake. The fishery-independent samples contained young individuals and the mean age of each species in the fishery-independent samples was generally 1-3 years less than the fishery-dependent samples.

The samples obtained from the fishery-dependent and fishery-independent sampling strategies impacted estimates of mean length-at-age and growth of each species. Realistic growth curves could not be generated for any species using the fishery-dependent data, whereas this was possible for all species using the fishery-independent data. Across both estuaries, mean length-at-age estimates were generally greater in the fishery-dependent samples compared to the FI samples for each species. This was most likely due to the fishery-dependent strategy oversampling the fastest growing individuals of each species as they reached legal length and entered the fishery. Whilst these over-estimates were mostly restricted to the younger age classes (< 4 years) of each species, it nevertheless affected all ages of *A. australis*. The greater mean length-at-age observed in the older age classes of some species in the fishery-dependent samples may have been due to the lowered proportions of occurrence of larger individuals in the fishery-independent samples.

Differences between sampling strategies in estimates of total and fishing mortality and rates of exploitation were inconsistent across species, and estuaries for some species. Across both estuaries, estimates of mortality and exploitation derived from the fishery-independent sampling were lower than those from the fishery-dependent sampling for *P. fuscus*, whereas the opposite was evident for *M. cephalus* and *G. tricuspidata*. No consistent patterns were evident for *A. australis* and *S. ciliata*. Such differences alone would probably not impact assessment outcomes.

This experiment demonstrated some strengths and weaknesses in each of the fishery-dependent and -independent sampling strategies examined. The fishery-dependent sampling is only possible for commercially fished estuaries, commercial catch and effort data is inconsistent and privacy laws may impact on the availability and reporting of data. The gillnet was more suitable than the beam trawl for sampling key harvested fish species for resource assessment purposes (i.e. relative abundance and demographic characteristics of populations). Nevertheless, a greater sampling effort than used here (concentrated on the multimesh gillnet) would be required in any future fishery-independent sampling program for fisheries assessments. A fishery-independent sampling strategy may provide the only comparative standardized assessment across estuaries and throughout time of the relative abundance (CPUE) and length- and age-based demographic parameters of the harvested and non-harvested estuarine fisheries resources of NSW, and in other areas that experiencing similar management scenarios.

4.2. Experiment 2: Comparison of fishery-independent and fishery-dependent (recreational) sampling strategies to assess harvested fish species and aquatic biodiversity (Appendix 4)

Across both estuaries, a total of 73 and 45 species were sampled in the fishery-independent and -dependent sampling strategies, respectively. A greater number of species were sampled in the gillnet and beam trawl than in the creel survey in Lake Macquarie, but this was not observed in St Georges Basin. Samples from the creel survey were limited to species retained by recreational anglers, whereas the fishery-independent samples contained most of these species plus several non-harvested species, particularly in Lake Macquarie.

Across both estuaries, greater total numbers of species and individuals were sampled using the gillnet than the beam trawl, and similarly in the boat-based catches than the shore-based catches.

The length compositions of samples of key species (e.g. *A. australis*, *P. saltatrix*, *S. ciliata* and *P. fuscus*) in the creel survey were primarily restricted to individuals of legal length, whereas sub-legal and legal length individuals of these species occurred in samples obtained from the fishery-independent strategy. The mean length of *A. australis* and *P. saltatrix* was less in the fishery-independent compared to the fishery-dependent samples. This was not the case for *S. ciliata* and *P. fuscus*. Compared to the fishery-dependent samples, the fishery-independent samples appeared to under-sample *A. australis* greater than 30 cm FL. The under-sampling of large individuals was not evident for the other key species examined.

4.3. Experiment 3: Development of a fishery-independent sampling strategy for the giant mud crab and comparison with fishery-dependent sampling of commercial catches (Appendix 5)

The effectiveness of two sources of fishery-dependent (observer-based and fisher-logbook) and -independent sampling strategies in providing data for assessing and monitoring populations of *S. serrata* were evaluated. In the Macleay River, catch rates from commercial fishing (i.e. fisher-logbook and observer survey) of male and female *S. serrata* were greater than those obtained in the fishery-independent samples. Except for the mean number of males in the Richmond River, reported logbook catch rates were greater than the observer-based and fishery-independent samples across both rivers.

Patterns of relative abundance, size-structure and sex ratio of trap catches were influenced by sampling strategy, estuary and areas within the estuary sampled. The magnitude and pattern of variation between standardised catch per unit effort (CPUE) data from the observer-based and fishery-independent sampling differed significantly between rivers, sexes and sizes of crabs. Estimates of the level of precision of CPUE also varied between rivers, sampling strategy and sexes. For example, estimates of mean CPUE of total and male *S. serrata* were generally more precise in the Richmond River than the Macleay River. But for female *S. serrata*; estimates of mean CPUE were more precise in the Macleay River than the Richmond River. In general though, the CVs for both the fisher-logbook and observer data were generally more precise than estimates of CPUE from the independent surveys.

Differences in the size-structure of catches between sampling strategies were not consistent between estuaries. For example, in the Richmond River, larger proportions of undersize male *S. serrata* were reported from observer-based and fisher-logbook surveys, whereas in the Macleay River larger proportions of undersized *S. serrata* were sampled in the independent survey.

The results suggest that relying on only one sampling strategy to assess the status of *S. serrata* populations in this region may not be sensible. Management objectives of the fishery are currently being redefined and these will determine the most appropriate sampling strategies to assess populations of *S. serrata* in the future.

4.4. Experiment 4: Utility of fishery-independent sampling strategy to assess the fisheries resources in estuaries with different management regimes (Appendix 6)

The majority of the 14 key species examined, except *A. australis*, *S. maculata*, *G. subfasciatus*, *R. sarba* and *P. auratus*, were caught in greater total numbers in the gillnet compared to the beam trawl across most estuaries. Both *A. australis* and *G. subfasciatus* were caught in high numbers in both gear types, whereas the mugilids, *M. cephalus*, *L. argentea*, *P. georgii* and *M. elongatus* were caught exclusively in the gillnet.

There was a significant effect of estuary type (haven v non-haven) on the numbers of *A. australis*, *R. sarba* and *G. subfasciatus* sampled in the beam trawl and *G. subfasciatus* sampled in the gillnet. The three species sampled in the beam trawl occurred in significantly greater numbers in the non-havens, whereas the opposite was evident for *G. subfasciatus* sampled in the gillnet. The relative abundance of several species fluctuated significantly among individual estuaries and sampling periods, but there was no significant influence of estuary type on their abundances.

The mean lengths and ages of *A. australis*, *P. fuscus*, *G. tricuspidata*, *M. cephalus* and *S. ciliata* sampled in the gillnet were generally greater in the havens than the non-havens. Depending on the species, the mean ages of sampled populations were generally 1 to 2 years greater in the havens than the non-havens. The mean lengths of *L. argentea* and *P. saltatrix* sampled in the gillnet and *G. subfasciatus* sampled in the gillnet and beam trawl were also generally greater in the havens than non-havens. In contrast to all other species, the mean length of *S. maculata* was greater in the non-havens than in the havens.

The catch curve based estimates of total mortality were lower in the havens than the non-havens for *A. australis*, *P. fuscus*, *S. ciliata* and *M. cephalus* but not *G. tricuspidata*, which displayed no consistent differences between estuary type. The determined estimates of fishing mortality and exploitation rate were also lower for *A. australis*, *P. fuscus* and *M. cephalus* in the havens than the non-havens.

The data from this experiment demonstrate that the gillnet is a more suitable sampling gear than the beam trawl for providing demographic data of the general suite of key harvested fish species in estuaries of southeastern Australia.

The data provide correlative evidence that the removal of commercial fishing in the recreational fishing havens has provided greater levels of protection from exploitation to the harvested species of fish. This has facilitated more individuals to grow and reach greater lengths prior to harvesting, allowing for greater proportions of larger and older fish in populations in the havens compared to the non-havens. Despite this, mortality and exploitation levels of the key species were still relatively high with fishing mortality being greater than natural mortality in most cases. Thus, the continued assessment of fish populations in the recreational fishing havens is warranted for their sustainable management.

This experiment demonstrated the power and utility of the fishery-independent sampling strategy to identify differences in the relative abundances and population characteristics of harvested species of fish in estuaries subjected to different management regimes. Such sampling could provide the necessary data required for long-term assessments of harvested and non-harvested species across estuaries and be pivotal in assessing the responses of fish to changes in management arrangements.

4.5. Experiment 5: Utility of fishery-independent sampling strategy to assess aquatic biodiversity across estuaries (Appendix 7)

The gillnet and beam trawl sampled a wide diversity of taxa, with over 116 species sampled across the five estuaries. A greater total number of organisms were sampled in the multimesh gillnet (83,529 individuals) compared to the beam trawl (61,808 individuals). Both gears sampled similar numbers of species, with more than 88 taxa sampled in the gillnet and 86 taxa in the beam trawl. Fifty-eight taxa were common to both gear types, with 30 and 28 taxa sampled exclusively in the gillnet and trawl, respectively.

Multivariate analyses revealed the gillnet and beam trawl sampled different aspects of the aquatic assemblages, which were evident across most estuaries and sample times. Several abundant taxa were important in distinguishing differences in assemblages sampled with either gear type. In general, the gillnet caught a wider length range of several species of fish of economic importance, many of which were mostly absent in catches from the beam trawl. These species included, *Liza argentea*, *Mugil cephalus*, *Girella tricuspidata*, *Pomatomus saltatrix* and *Herklotsichthys castelnaui*. By comparison, the beam trawl was more effective in sampling several small species of fish, such as *Pelates sexlineatus*, *Pseudorhombus arsius*, and *Ambassidae* spp., the juveniles of some larger-growing species, such as *Rhabdosargus sarba* and *Acanthopagrus australis*, and penaeid prawns. Many less abundant species were caught predominately or more frequently occurred in either the trawl (e.g. *Monacanthus chinensis* and *Sillago maculata*) or gillnet (e.g. *Platycephalus fuscus* and *Trachurus novaezelandiae*), but only in one or two estuaries. Several other species (e.g. *Portunus pelagicus*, *Dasyatis fluviorum*, *Pagrus auratus*) were important distinguishing species but whether or not they were caught more frequently in the trawl or gillnet depended on the particular estuary.

Differences between gear types in the length compositions of several important fish species were evident. In general, the beam trawl sampled larger proportions of smaller-sized fish, whereas the gillnet sampled greater proportions of larger-sized fish. Such patterns were not always consistent across all estuaries; for example, the trawl sampled larger proportions of small (approx. <16 cm) *A. australis* in the Clarence River, Wallis Lake and Tuggerah Lake, but this was not evident in Lake Macquarie and St Georges Basin.

These results identified the utility of the gillnet and beam trawl in sampling a wide diversity of different sized estuarine fauna. Both gear types were selective, sampling different aspects of the aquatic assemblages. Thus, neither gear type used in isolation will provide a global panacea to sample entire assemblages and species. Rather, as found elsewhere, incorporating different gear types into sampling strategies will enhance surveys and assessments and provide a greater overall picture of estuarine faunal biodiversity.

4.6. Experiment 6: Utility of fishery-independent sampling strategy to reduce uncertainty in management decisions (Appendix 8)

The fishery-independent and fisher-dependent sampling in the Richmond River identified that economically important species such as mud crab (*Scylla serrata*), sea mullet (*Mugil cephalus*), yellowfin bream (*Acanthopagrus australis*), dusky flathead (*Platycephalus fuscus*) and sand whiting (*Sillago ciliata*) were present in large numbers six weeks after the flood event occurred. The fishery-independent sampling found no significant differences in the total numbers and species of fish and crustaceans among the lower three sections of the main river channel. The fishery-independent sampling program further identified that catch rates of several species of fish, such as sea mullet and yellowfin bream, in the Richmond River were of a similar magnitude to those in other NSW estuaries sampled over the same period. This sampling identified that populations had recovered quickly subsequent to the flood event.

Following assessment of the data from the fishery-independent and fisher-dependent sampling, management decided that the population levels of fish in the Richmond River were sufficiently sustainable to support the recommencement of recreational and commercial fishing. Hence, the Richmond River was reopened to normal fishing activities after a closure of approximately six weeks. This compared favourably to the five-month fishing closure following a comparable flood event in 2001, when no comparative fishery-independent data were available. There were no patterns in the data to justify a lengthier closure this time around. The availability of data from the fishery-independent sampling reduced uncertainty and made the decision by management to reopen the river following the 2008 flood considerably easier and more decisive than in the previous situation. The availability of the fishery-independent data therefore proved beneficial to managers and fishers alike, as well as the local community, recovering after a natural disaster.

5. Discussion

5.1. Objective 1. Evaluate the effectiveness of a standardised fishery-independent sampling strategy compared with sources of fishery-dependent data (e.g. data from commercial and recreational fisheries) for assessing fisheries resources and biodiversity

This study successfully tested a standardised fishery-independent sampling strategy against typical fishery-dependent strategies of sampling the retained catches of commercial fishers (fishing cooperative sampling, observer-based sampling) and recreational anglers (creel survey) for assessing aquatic biodiversity and fisheries resources across estuaries. Experiments 1, 2, 3 and 4 fulfilled this objective.

Each of these experiments demonstrated some strengths and weaknesses in each of the fishery-independent and -dependent sampling strategies for assessing fisheries resources and aquatic biodiversity. The multimesh gillnet was better suited than the beam trawl to sample a broader suite of key harvested species for resource assessment purposes. Nevertheless, at least a doubling of the sampling effort used in this study (4-6 nights per estuary) of the gillnet would be required to provide sufficient length- and age-based samples for key species on an annual basis for robust species resources assessments.

The fishery-independent sampling strategy contained far greater diversity of species and was more superior to assess spatio-temporal changes in aquatic diversity compared to the fishery-dependent strategies. This was primarily due to the fishery-dependent samples only containing harvested species retained by fishers. This severely restricted the utility of the fishery-dependent sampling as a tool for monitoring and assessing aquatic biodiversity. In contrast, the fishery-independent gears sampled the same harvested species as well as a plethora of other non-harvested species. Further, the fishery-independent sampling provided a consistent framework for comparisons of samples across space and time.

Both the multimesh gillnet and the beam trawl sampled a diverse fauna (totalling 106 species across five estuaries). The two gears, however, sampled different components of the aquatic assemblages and biodiversity in an estuary. Thus, the most complete picture of assemblages and populations of fish required sampling with both methods. Sampling with additional methods, such as small-mesh seines (e.g. Gray et al. 1996) could provide an even more comprehensive picture of estuarine fish assemblages for biodiversity assessments (Rotherham et al. 2012).

The sampling of commercial fishers retained catches at fishing cooperatives can provide a good platform and access to ample samples for length and age compositions and subsequent mortality analyses. But, this is only available in estuaries where commercial fishing occurs and where the fishers and the cooperatives willingly allow such sampling to take place. This has proven a problem particularly when there has been a change in management regulations that negatively impacts fishery participants. Moreover, if the privacy laws (in NSW) as they currently exist and applied to catch records are expanded to include the length compositions of fishers catches, then this type of sampling could potentially become redundant in the future.

The fisher-reported logbooks have very limited value for assessment purposes at present due to inconsistencies in reported fishing effort and the privacy regulations that restrict usage of catch data for analyses across some spatial and temporal scales. This could prove a greater problem in the future if the numbers of commercial fishers operating within an estuary are further reduced. Moreover, the potential CPUE data from such a system is limited to estuaries that are open to commercial fishing.

Creel surveys provide very good data on harvests, effort and potentially CPUE of recreational angling that can be used across all types of estuaries where angling takes place. However, like the sampling of commercial catches, the catch data is limited to retained harvested species only, which restricts their use for biodiversity assessments. The truncation of length information close to the MLL for some species may also limit their use for determination of population demographics, similar to samples from commercial catches (see below). Creel surveys also are logistically difficult to do and often costly, as they usually require many staff to survey a range of access points and to undertake roving surveys across extended periods of time.

The fishery-independent and -independent strategies sampled different components of populations; the fishery-dependent samples were limited to legal length fish of harvested species whereas the fishery-independent samples included sub-legal and legal length individuals of harvested as well as non-harvested species. The fishery-independent sampling strategy therefore provided greater representations of the length- and age-based population characteristics of each species. Moreover, the fishery-independent samples contained far greater diversity of species than the fishery-dependent samples, demonstrating their overall superiority for biodiversity assessments. Further, the fishery-independent sampling provided a consistent framework for comparisons of samples across space and time. This was not possible with the fishery-dependent sampling strategies.

For all species, realistic growth curves could not be generated using the fishery-dependent samples alone because of the truncation of sampled lengths at the MLL. There was no such restriction on the fishery-independent samples, thus allowing growth curves to be generated for each species. The estimated mean length-at-age of each species was generally greater in the fishery-dependent compared to the fishery-independent samples. This was particularly evident in the younger age classes and was probably due to the fishery-dependent strategy over-sampling the fastest growing individuals as they attained legal length and entered the fishery. However, for some species this extended to older age classes and may have been due in part to the fishery-independent gillnets under-sampling the largest individuals in populations. This was particularly evident for *A. australis*; larger length individuals were common in samples of retained commercial and recreational catches but not in the fishery-independent samples.

Differences in estimates of mortality and exploitation based on the fishery-dependent and -independent samples were inconsistent among species. We could not determine here which estimates were most representative of the true status of populations. Further investigations are needed to untangle this dilemma and to determine if such differences would affect assessment outcomes for each species.

The combined results from these experiments provide overwhelming evidence of the value and utility of the fishery-independent sampling strategy compared to typical fishery-dependent strategies in assessing estuarine fisheries resources and biodiversity. The strength of the fishery-independent strategy is that it provides a consistent sampling frame that can be used across all types of estuaries regardless of fisheries management arrangements. This is not available using the fishery-dependent strategies.

5.2. Objective 2. Investigate the extent to which fishery-independent data reduce uncertainty in the management of estuarine fisheries resources and lead to decisions that are more reliable and robust

Experiment 6 alone achieved this objective by specifically demonstrating how incorporation of a fishery-independent sampling strategy reduced uncertainty, leading to a more reliable and robust decision to reopen the Richmond River following a severe flood event. The inclusion of the fishery-independent data meant that the river was reopened to fishing much faster (6 weeks) than it would have without such data, as in a previous flood event (5 months). The fishery-independent data therefore proved beneficial to the NSW Government (i.e. NSW Minister for Fisheries, fisheries and conservation management agencies), the commercial and recreational fishing industries and associated businesses, as well as the local community that rely on fishing in the river.

Each of the other experiments demonstrated the value of the fishery-independent sampling strategy to deliver quantitative and standardised data that can be used to compare and assess the fisheries resources and aquatic biodiversity in a consistent framework across estuaries and throughout time. The logical benefits of such a sampling strategy include more reliable data leading to reduced uncertainty in management decisions.

5.3. Objective 3. Examine the values of fishery-independent sampling for use across estuaries with different management regimes (e.g. estuaries open and closed to commercial and recreational fishing; marine parks) and for assessing the impacts of immediate environmental perturbations (e.g. floods, pollution) and those in the future (e.g. impacts of climatic change) on the dynamics of populations of fish and diversity of fish assemblages

This objective was achieved across all experiments; the collective results providing conclusive evidence that a standardised fishery-independent sampling strategy is a very powerful tool for quantitatively assessing the effects of alternate management regimes and environmental perturbations on the diversity of fish assemblages and the demographic characteristics and relative abundances of key fish species in a consistent manner across a hierarchy of temporal and spatial scales.

Specifically, Experiment 4 identified differences in the demographic characteristics and relative abundances of key harvested species of fish across estuaries with different management regimes (i.e. recreational fishing havens and non-havens). This experiment detected positive longitudinal responses of fish populations to changes in a fishery management regulation (i.e. cessation of commercial fishing). This experiment alone provides a strong example of the value and utility of the fishery-independent sampling strategy to fisheries managers and stakeholder groups. Tests of impacts of management decisions should be an integral part of fisheries management. The fishery-independent sampling strategy could be used to investigate the potential effects of other types of management initiatives on populations and assemblages of fish within and across estuaries.

The study also demonstrated the ability of the fishery-independent sampling strategy to detect impacts on, and responses of, assemblages and populations of aquatic fauna to an environmental perturbation (Experiment 6). This same sampling strategy also successfully detected temporal, spatial and habitat-related differences among estuarine ichthyofauna (Gray et al. 2011; Rotherham et al. 2012). These studies provide collective examples of the capability of the fishery-independent sampling strategy to detect spatial and temporal changes in biodiversity in a consistent and quantitative framework. This sampling strategy implemented across a broad range of estuaries could prove very valuable to monitoring, evaluating and reporting on the aquatic biodiversity and on population and assemblage characteristics of estuarine fauna in NSW.

In summary, a standardised fishery-independent sampling program may provide the only reliable and rigorous strategy to assess environmental and anthropogenic perturbations on the relative abundance (CPUE) and length- and age-based demographic parameters of harvested and non-harvested species and of aquatic biodiversity across estuaries throughout NSW. This may also be the case for other systems and species in other management jurisdictions in Australia and elsewhere.

6. Conclusions

This project successfully tested a standardised fishery-independent sampling strategy against fishery-dependent sampling strategies typically used by fisheries agencies to assess fish populations, fisheries performance and aquatic biodiversity (Objective 1). In doing this research, the study identified strengths and weaknesses of each of the sampling strategies. This study also demonstrated how data from a fishery-independent sampling program reduced uncertainty and made for a more robust and reliable assessment and decision concerning the management of a fishery following a natural perturbation (Objective 2). This research project successfully identified how a fishery-independent sampling strategy can be used to assess: (1) fisheries resources and aquatic biodiversity across estuaries that have different management arrangements, and (2) the impacts on, and responses of, aquatic populations and assemblages to environmental perturbations and management arrangements (Objective 3).

This study has provided compelling evidence that the tested fishery-independent sampling strategy has several advantages over typical fishery-dependent sampling strategies to assess fisheries resources and aquatic biodiversity. A standardised fishery-independent sampling strategy may provide the only consistent framework to deliver robust and reliable data essential for assessing and managing the aquatic biodiversity and fisheries resources across the breadth of estuaries of NSW. The agencies mandated with managing the marine estate of NSW need to consider in detail the synergistic benefits of implementing a broad-scale fishery-independent sampling program across estuaries for delivering the successful monitoring, assessment and improved sustainable management of the states aquatic biodiversity and fisheries resources.

7. Implications

This study has identified the utility and value of a standardised fishery-independent sampling strategy for assessing the biodiversity and fisheries resources across estuaries of NSW. The results reported here have applicability to other systems and species in other management jurisdictions in Australia and elsewhere. Incorporation of a standardised fishery-independent sampling strategy into the assessment of a natural flood event proved beneficial to NSW Government agencies, commercial, recreational and indigenous fishers and associated stakeholders. Implementation of a broad-scale standardised fishery-independent sampling and assessment strategy could have synergistic benefits across agencies mandated to report on aquatic biodiversity, fisheries resources and fisheries in NSW. This could lead to a more united response of government to managing the marine estate of NSW.

8. Recommendations

The agencies that oversee the assessment, management and conservation of the marine estate of NSW need to consider the synergies and multiple benefits of implementing a broad-scale standardised fishery-independent sampling strategy across estuaries for provision of robust and reliable data for the sustainable management of the states aquatic biodiversity and natural fisheries resources. This needs to consider the costs of a combined strategy compared to the potentially isolated and disjointed strategies confined to single agencies. The experiments reported here provide compelling evidence of the utility and potential value to management and the various marine estate stakeholders of such a strategy. Moreover, such agencies need to explicitly define their management objectives and information needs so that sampling and assessment strategies are appropriately designed to provide the defined deliverables.

Further development

NSW DPI has the species assessment data from the different sampling strategies reported here. Potential differences in assessment outcomes from the alternate data sources need to be tested in the NSW DPI resource assessment system. This could not be done here due to the closure of the Cronulla Fisheries Research Centre of Excellence and associated disruptions to the timetabling of resource assessment workshops. Nevertheless, NSW DPI shall investigate the value of each sampling strategy and will incorporate the most suitable strategy in future monitoring and assessment programs.

9. Extension and Adoption

The Project Advisory Committee that included government, fisheries and conservation representatives was kept abreast of developments throughout the study. Numerous briefs and presentations to industry management committees occurred during the study and further reporting will be provided to various stakeholder and interest groups via scientific publications, industry magazines and newsletters and presentations.

Project coverage

This could not be determined as no associated records were provided with the novation of the project.

10. Project Materials Developed

Scientific papers:

1. Rotherham D, Johnson DD, Kesby CL, Gray CA (2012) Sampling estuarine fish and invertebrates with a beam trawl provides a different picture of populations and assemblages than multi-mesh gillnets. *Fisheries Research* 123-124, 49-55. http://dx.doi.org/10.1016/j.fishres.2011.11.019

2. Rotherham D, Chapman MG, Underwood AJ, Gray CA, Johnson DD (2011) Untangling spatial and temporal variation in abundances of estuarine fish sampled with multi-mesh gillnets. *Marine Ecology Progress Series* 435, 183-195. http://dx.doi.org/10.3354/meps09204

3. Rotherham D, Macbeth WG, Kennelly SJ, Gray CA (2011) Reducing uncertainty in the assessment and management of fisheries resources following an environmental impact. *ICES Journal of Marine Science* 68, 1726-1733. http://dx.doi.org/10.1093/icesjms/frs079

4. Young CL, Rotherham D, Johnson DD, Gray CA (2013) Small-scale variation in reproduction and abundance of greentail prawn, *Metapenaeus bennettae* Racek and Dall, 1965. *Journal of Crustacean Biology* 33, 651-659. http://dx.doi.org/:10.1163/1937240X-00002172

5. Johnson DD, Rotherham D, Gray CA (submitted manuscript) Effects of design and deployment of net-covered pots on sampling populations of giant mud crab *Scylla serrata*. *Fisheries Research*

6. Johnson DD, Gray CA, Mcleod JR, (submitted manuscript) Comparison of commercial catch data and research surveys for assessing populations of mud crab (*Scylla serrata*). *Fisheries Research*

Conference Proceedings:

1. Rotherham D, Macbeth WG, Kennelly SJ, Gray CA (2010) Integrating different sources of data to reduce uncertainty in the assessment and management of fisheries resources following an environmental impact. Oral presentation by D. Rotherham at the 2010 'Fishery-dependent information: making the most of fisheries information', conference, Galway, Ireland, 23-26 August 2010.

2. Rotherham D, Johnson DD, Kesby CL, Gray CA (2010). Sampling estuarine fish and invertebrates with a beam trawl provides a different (not 'better') picture of populations and assemblages than multi-mesh gillnets. Oral presentation by D. Rotherham at the 2010 'Fish Sampling with Active Methods' conference, Ceské Budejovice, Czech Republic 8 September to 11 September 2010.

3. Rotherham D (2010) Surveys of estuarine fisheries resources in NSW: who, what, why, where, when and how? Oral presentation given at the 'National Estuaries Network Science Day', Australian Museum, 1 December 2010.

Media:

Media outputs could not be determined as no associated records were provided with the novation of the project.

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Ochwada-Doyle FA, McLeod J, Barrett G, Clarke G, Gray CA (2014) Assessment of recreational fishing in three recreational fishing havens in New South Wales. Fisheries Final Report Series No. 139. NSW Department of Primary Industries.

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Rotherham D, Macbeth WG, Kennelly SJ, Gray CA (2011) Reducing uncertainty in the assessment and management of fisheries resources following an environmental impact. ICES Journal of Marine Science 68, 1726-1733.

Rotherham D, Underwood AJ, Chapman MG, Gray CA (2007) A strategy for developing scientific sampling tools for fishery-independent surveys of estuarine fish in New South Wales, Australia. ICES Journal of Marine Science 64, 1512-1516.

12. Appendices

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Appendix 1. List of researchers, project staff and consultants

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Appendix 2. Intellectual property

There is no intellectual property arising from this report.

Appendix 3. Experiment 1: Comparison of fishery-independent and fishery-dependent (commercial) sampling strategies to assess harvested fish species

Introduction

The assessment of exploited fish populations and the appropriateness of fisheries management strategies often requires information of temporal and spatial changes in levels of harvests, fishing effort and length- and age-based demographic characteristics of harvested populations (Ricker 1975; Hilborn and Walters 1992; Lai et al. 1996; Patterson et al. 2001). Fisher-supplied logbooks that report catch and effort along with fishery-dependent sampling of the lengths and ages of landed commercial catches is a standard method in obtaining this important information (Doubleday and Rivard 1983; Marriott et al. 2010). This strategy has historically been used in assessing harvested fish species in NSW (Gray et al. 2002, 2010; Silberschneider et al. 2009; Stewart and Hughes 2009, 2010). Such data often have limitations, however, due to biases associated with gear selectivity, individual fisher behaviour such as targeting practices, market forces and the general non-randomness of fishing (Rotherham et al. 2007).

An alternate strategy involves the fishery-independent sampling of wild populations in a standardized and appropriately stratified manner data, which is often free of the biases associated with fishery-dependent data (Gunderson 1993; Pennington and Stromme 1998). Nevertheless, such sampling strategies can have their own limitations if they are not adequately designed (Rotherham et al. 2007). There still exists considerable debate about the relative value of fishery-independent and -dependent data for assessing fisheries resources and biodiversity, particularly for small-scale, low-value, multi-species fisheries like the NSW Estuary General Fishery. This is because there have been few comparisons of the two types of data and their relative costs and benefits.

Here we test for differences in the data collected in a typical fishery-dependent strategy of sampling the retained catches of commercial fishers and a novel fishery-independent sampling strategy. This specific experiment tested the hypothesis that there would be differences in the length and age compositions, estimated growth, mean length-at-age and mortality parameters between samples derived from the fishery-dependent and -independent sampling strategies. We further examined whether indices of catch-per-unit-of-effort (CPUE) differed between fisher-dependent logbook data and the fishery-independent sampling. We did this for five key species of harvested fish across two estuaries over three years. This experiment contributed specifically to Project Objective 1.

Materials and Methods

Study estuaries and species

The fishery-independent and fishery-dependent sampling took place between January and July in 2008, 2009 and 2010 in the Clarence River and Wallis Lake. Both estuaries support large multi-species, multi-method commercial fisheries as well as indigenous and recreational fisheries.

The five study species were *Acanthopagrus australis* (Sparidae), *Platycephalus fuscus* (Platycephalidae), *Girella tricuspidata* (Girellidae), *Mugil cephalus* (Mugilidae) and *Sillago ciliata* (Sillaginidae). Each species is consistently ranked among the top 10 species of fish commercially harvested in estuaries in NSW and collectively these species account for over 70% of the total landed catch (weight) taken in the multispecies, multi-method Estuary General Fishery. Each species is also harvested commercially in coastal waters either as part of the Ocean Haul, Ocean Trap & Line and the Ocean Prawn & Fish Trawl Fisheries. All species are important to indigenous fishers and, except for *M. cephalus*, are also key recreational species taken in considerable quantities (Rowling et al. 2010).

Fishery-independent sampling

Each estuary was sampled with the multimesh gillnet and beam trawl in each of two periods (January–March/April and April/May–July) in 2008, 2009 and 2010. The two sampling periods were approximately 3 months long and separated by at least 4 weeks in each estuary. Previous experiments demonstrated that variation in abundances of fish at temporal scales within 3-month periods (e.g. days, weeks and months) was small compared to variation between 3-month periods and at small spatial scales (Rotherham et al., 2011). Further, the order in which each estuary was sampled within each period in each year, and the order in which each gear type was used, was selected at random.

Each multi-mesh gillnet consisted of seven 20-m panels of different sizes of stretched mesh (36, 44, 54, 63, 76, 89 and 102 mm) connected together in a random order (see Gray et al. 2009). Adjacent panels were separated by 5 m of rope to minimize potential effects of the smaller-mesh panels 'leading' larger fish to adjacent panels. The 36- and 44-mm panels were made from monofilament netting with twine diameter of 0.15 mm; the remaining panels had multifilament netting with a twine diameter of 0.4 mm. All panels were 2 m deep and had a hanging ratio of 0.5.

In each estuary, four randomly selected sites separated by one to several kilometers were sampled with the gillnet (see Gray et al. 2009). Two depth strata, shallow (<2 m) and deep (4–6 m) were sampled in CR but only the shallow stratum was sampled in WL due to it being very shallow (Rotherham et al. 2011). In each sampling period in each year, three replicates of the multimesh gillnet (each separated by 50–100 m) were sampled at each depth at each site. It took one night to sample a site in CR, whereas two sites could be sampled each night in WL. The particular week in which sampling commenced within each sampling period and estuary, was selected at random. On each night of sampling, each gillnet was bottom-set at dusk, soaked for 1 h and then retrieved (Rotherham et al. 2006). Catches in each net were sorted by species and counted and individuals of the five study species were kept for processing in the laboratory.

The beam trawl was 3-m wide and configured with 41-mm diamond-shaped mesh in the body and 20-mm mesh hung on the bar (i.e. square-shaped) in the codend (Rotherham et al. 2008a). In each sampling period in each year, nine non-overlapping replicates of the beam trawl (5 min duration at speeds of about 1.2 m s^{-1}) were sampled at night at each of four randomly selected sites (separated by one to several km) within each estuary (Rotherham et al. 2008b). Sampling sites were over relatively flat, predominantly unvegetated sediment interspersed with (or adjacent to) patches of seagrass in waters 1 to 6 m deep. It took two consecutive nights to sample all four sites within an estuary and the two nights of sampling were selected at random from within each sampling period. After each replicate tow was completed, the contents of the net were emptied onto a tray and sorted by species and individuals of the five study species were counted and kept for processing in the laboratory.

All specimens of the five study species were measured for fork length (FL), except for *P. fuscus*, which was measured for total length (TL) (0.5 cm below), and most had their sagittal otoliths extracted for age analyses. The sampling strategy was to collect 200 otoliths of each species from each estuary each year.

Fishery-dependent sampling

Retained commercial catches of each study species were sampled shortly after commercial fishers registered their catches at the fishing cooperative in each port. Whole catches or random sub-samples of whole catches landed on each sampling day were measured (0.5 cm below FL or TL) for length composition. Random subsamples of fish from these catches were measured for length (0.1 cm below FL or TL) and had their sagittal otoliths removed for ageing analyses. Because sampling was restricted to commercial landings, few fish below the minimum legal length (MLL) of each species was examined. The gear-type used to capture sampled catches was obtained from each fisher at the time of sampling. The sampling strategy was to measure up to 1000 lengths and collect 200 otoliths of each species from each estuary each year. In each year, between 10 and 25 days were sampled at each fishing cooperative.

Reported commercial catch and effort information for each species in each estuary were obtained from the mandatory logbook data that commercial fishers supply to the NSW Department of Primary Industries. These data report total monthly catches (kg) and total fishing effort (days fished) for each species in each estuary. Data were separated into two periods in each year to coincide with the fishery-independent sampling. Period 1 included January, February and March and Period 2 included May, June and July.

Catch per unit of effort

Three factor permutational analyses of variance (PERMANOVA; Anderson 2001; Anderson et al. 2008) were used to test if the numbers of each species sampled in the gillnet and beam trawl differed according to the factors: Year (Fixed), Period (nested in Year, Random) and Estuary (Fixed). Because we were only interested in examining the scale of estuary, we did not test for variations in catches among individual sites within each estuary. Each univariate analysis was based on the Euclidean distance measure and Type III (partial) sums of squares were calculated using 9999 unrestricted permutations of the raw data. The mean (±1 SE) catch per unit of effort data (number caught per net per night) for each species from the fishery-independent sampling was determined for each period and estuary and graphically displayed.

The same analytical procedure was to be used to test if the reported commercial catch and catch per-unit-of-effort data of each species differed over the same temporal and spatial scales as the fishery-independent data. This could not be done due to: a) standardized effort data not being available and, b) privacy laws prohibited the analyses and reporting of catch data for each species at the relevant spatial and temporal scales used in the experiment.

Age determination

Counts of completed opaque zones on sectioned sagittal otoliths were used to determine the age of each species of fish. This ageing technique has been validated for all five species (Gray et al. 2000, Gray et al. 2002, Smith and Deguara 2002, Gray et al. 2010, Ochwada-Doyle et al. 2014). Assignment of age and year class was consistent across years and estuaries because all samples were collected within the same 6-month period each year. Further, there was no need to adjust ages based on the month of capture because opaque zone deposition had mostly been completed prior to each years sampling.

Otoliths were embedded in blocks of clear cast polyester resin and three to four transverse serial sections (approx. 0.7 mm) were cut through the otolith core using a diamond saw. The resulting sections were mounted on a microscope slide and polished on a polishing wheel fitted with 500-grit silicon carbide paper till the opaque zones on sections became clear. Sections were viewed using a reflected light against a black background under a compound microscope fitted with a digital camera. The distance between each successive opaque zone was measured using Image J software and digital images of each section were captured. Sections were interpreted without the knowledge of the length or sex of the fish, or the date and location of capture. Ten percent of sections were read twice without the knowledge of the 1st interpretation to obtain a coefficient of variation (CV) of successive reads (Kimura and Lyons 1991).

Length-at-age and growth

Differences between the FD and FI samples in the mean length-at-age of each year class of each species were tested using two-factor permutational analyses of variance (PERMANOVA, Anderson 2001) with the factors Strategy (FIS v FD - Fixed) and Age class (Fixed). Analyses were done separately for each sex and estuary because of differing sample compositions. A one-factor PERMANOVA was done for female *G. tricuspidata*, in the Clarence River as only one age class could be analysed and no analysis could be done for male *P. fuscus* in the Clarence River. For each species, the data for an age class were only included in an analysis if there were more than 5 length measurements. Because of this, comparisons of length-at-age were limited to a restricted number of age classes for each species and estuary (see results). For each species, the mean (±1 SE) length-at-age was calculated separately for each sex and age class in each estuary and graphed.

Length and age compositions

The length and age compositions of the fishery-dependent and fishery-independent samples of each species were determined separately for each gear type in each estuary. Except for *M. cephalus*, this was done for data pooled over the three years of sampling. Inconsistent and sometimes low numbers of samples prevented analyses being done across individual years for the other species. Because all *P. fuscus*, *S. ciliata* and *G. tricuspidata* in the fishery-independent samples were aged, the frequency of counts of each age simply became the age composition sample for each respective species. For *A. australis* and *M. cephalus* collected as part of the fishery-independent sampling, and for all species sampled as part of the fishery-dependent sampling, separate age-length keys were derived for each species for the fishery-dependent and fishery-independent samples for each estuary.

Each species- and estuary-specific age-length key was applied to the corresponding length composition of each FD and FI gear type to determine the age composition of samples.

Mortality and exploitation

The instantaneous rate of total mortality (*Z*) was determined for each method in each estuary using the age-based catch curve method (Ricker, 1975). Separate analyses were done for each gear type in each estuary on age composition data combined across the three years of sampling. The natural logarithm of the proportion of fish in each age class (Nt) was plotted against their corresponding age class (t) and a linear regression was fitted and *Z* determined as the slope of the descending regression. For these analyses, we assumed that the most abundant age class in each sample was fully recruited to the sampled population and that the selectivity, recruitment and growth of fish were constant across years and growth was asymptotic. Standard error (SE) and r^2 values were calculated for each regression.

The instantaneous rate of natural mortality (*M*) was estimated using the method of Hoenig (1983) based on the recent published maximum age of each species in eastern Australia; 22 years for *A. australis* (Gray et al. 2000) and *G. tricuspidata* (Gray et al. 2012), 16 years for *S. ciliata* (Ochwada-Doyle et al. 2014) and *P. fuscus* (Gray and Barnes 2008), and 12 years for *M. cephalus* (Smith and Deguara 2002). Fishing mortality (*F*) was determined by subtracting *M* from each estimate of *Z*, which also provided a corresponding rate of exploitation (*E*) ($E = F/Z$).

Results

Across both estuaries and all three years, greater numbers of all five species were sampled for length composition as part of the fishery-dependent sampling compared to the fishery-independent sampling (except *S. ciliata* in 2008 in Clarence River and *A. australis* in 2008 in Wallis Lake) (Table 1). This was not the case for age samples. Greater numbers of otoliths of *P. fuscus* and *G. tricuspidata* were obtained in the fishery-dependent sampling, but for the other three species the numbers of otoliths collected in each sampling strategy varied according to the estuary and year. The targeted collection of 200 otoliths per-species per-year and estuary for the FD sampling was not achieved in CR, but it was achieved across several years for some species in WL. Adequate numbers of otoliths were collected as part of the fishery-dependent sampling in most years and in both estuaries for all species (except *S. ciliata* in CR). This was also true for *A. australis* and *M. cephalus* in the FI sampling, but low numbers of otoliths were collected for *P. fuscus* and *G. tricuspidata* in both estuaries and *S. ciliata* in WL.

In both estuaries the only species caught in abundant numbers in the beam trawl was *A. australis*, with all other species primarily being caught in the multimesh gillnet (Table 2). The most numeric species sampled in the gillnet was *M. cephalus* and the least was *P. fuscus* in WL and *G. tricuspidata* in CR. Low numbers of *P. fuscus* and *G. tricuspidata* were sampled in the fishery-independent sampling across both estuaries. Except for *A. australis*, all further analyses (except for calculations of mean length-at-age) of the FI data for each species were based on samples obtained from the gillnet sampling. Moreover, for consistency across species the demographic analyses were done on data combined across the three years of sampling because of restricted numbers.

Length-at-age and growth

For each species, the mean length-at-age significantly differed between the FD and FI samples across most age classes examined (PERMANOVA, Table 4; Figs. 7-9). For *A. australis*, mean length-at-age was significantly greater in the FD samples across both sexes for age classes 3 to 10 years in the Clarence River and 3 to 7 years in Wallis Lake. This was also true for 3 to 7 year old *G. tricuspidata* in Wallis Lake. Mean length-at-age was significantly greater in the FD samples for both sexes of *M. cephalus* aged 1 to 4 years, *S. ciliata* aged 2 to 7 years and female *P. fuscus* aged 1 to 5 years. The FD samples did not include young ages (< 2 years) of *A. australis* and *G. tricuspidata*. Across both estuaries, the mean length-at-age of *A. australis* in the FD samples was virtually the same across ages 3 to 10, particularly for males.

Catch and effort

PERMANOVA detected significant differences in the relative abundance according to the factor Estuary for *G. tricuspidata* and *S. ciliata*, Year for *M. cephalus* and Period (Year) for *A. australis*, *P. fuscus* and *G. tricuspidata* sampled in the multimesh gillnet (Table 3). Similar analyses could not be made for the fishery-dependent commercial catch and effort data as outlined in the methods.

Length and age compositions

The lower length range of the FD samples of each species was truncated at the MLL, whereas sub-legal length individuals were prominent in the FI samples (Figs 3 to 8). Across both estuaries the mean length and age of samples of *A. australis*, *G. tricuspidata*, *P. fuscus* and *M. cephalus* (across all 3 years) were greater in the FD than the FI samples. This was also evident for *S. ciliata* in CR but not in WL. The age compositions of each species in the FD samples were generally 1-3 years greater than the FI samples. For example, 3 and 4 year olds dominated the FD age compositions of *P. fuscus*, whereas 1 to 3 year olds dominated the corresponding FI samples (Fig. 3). Similarly, for *M. cephalus* 2 and 3 years olds dominated the FD samples, but 0 to 2 years predominated the FI samples (Fig. 5).

The length and age compositions of sampled commercial catches of *A. australis* and *G. tricuspidata* were fairly consistent among gear types within each estuary. However, for *S. ciliata* a greater proportion of older fish were sampled in catches taken in gillnets compared to beach-seines. For the FI sampling, the beam trawl sampled a greater proportion of smaller and younger *A. australis* than the gillnet.

Mortality

Estimates of *Z*, *F* and *E* derived from the fishery-independent sampling were lower than those from the fishery-dependent for *P. fuscus* across both estuaries, whereas the opposite was evident for *M. cephalus* and *G. tricuspidata* (Table 3). No consistent patterns were evident for *A. australis* and *S. ciliata*. For example, estimated *Z*, *F* and *E* for *A. australis* were lower for samples from the fishery-independent than the fishery-dependent sampling in the Clarence River, but the opposite occurred in Wallis Lake. For *S. ciliata*, the mortality estimates were similar for both sampling strategies.

There was notable variation between methods within each sample type. For example, estimates of *Z*, *F* and *E* of *A. australis* across both estuaries were lower for the multimesh gillnet samples than those derived from beam trawl samples. For the fishery-dependent sampling in both estuaries, the mortality parameters derived for *A. australis* were greater from the gillnet catches than those from the beach-seine and trap samples. Within each estuary, these within sampling strategy differences within were less than those between sampling strategies. Differences between the commercial gillnet and beach-seine derived estimates of mortality for both *G. tricuspidata* and *S. ciliata* were not consistent across estuaries.

Discussion

Sampling

Generally, greater quantities of all five species except *M. cephalus* were sampled for length and age in the FD compared to the FI sampling. It took, however, between 10 and 25 days sampling in each fishing cooperative to obtain these samples, compared to the 4 nights for each of the gillnet and beam trawl sampling in each estuary. Not withstanding this, fishing cooperatives provide a good base to sample the landings (i.e. retained catches) of commercial fishers. This strategy has been successfully used across several cooperatives to obtain length- and age-based information for a range of fish and invertebrates for a number of years in NSW (Gray et al. 2002, 2010; Silberschneider et al. 2009; Stewart and Hughes 2009, 2010).

Despite this study being done in two of the largest producing fishing cooperatives in NSW and the access to numerous commercial catches, the FD sampling target of 200 otoliths for each species per estuary and year was not often reached. There are often logistical problems with doing this sampling. Indeed, this is only available in estuaries where commercial fishing occurs and where the fishers and the cooperatives willingly allow such sampling to take place. This has proven a problem in the past at some cooperatives, particularly when there has been a change in management regulations that negatively impacts fishery participants. Moreover, if the privacy laws in NSW as they currently exist and applied to the commercial catch records are expanded to include actual fishers catches, then this type of sampling could potentially become redundant in the future.

Except for *A. australis*, the multimesh gillnet was more effective at sampling each target species than the beam trawl. The data from this experiment and experiment 4 (Appendix 4) clearly demonstrate it would be better to use the multimesh gillnet than the beam trawl for length- and age-based demographic assessments of harvested (and non-harvested) fish in estuaries. This is also true for obtaining indices of relative abundance (catch per-unit-effort) information for most key species (Appendix 4). Nevertheless, for all species except *M. cephalus*, the samples obtained in each period and year across both estuaries using the multimesh gillnet were relatively low and did not provide adequate age samples for robust assessments for each year of sampling. This limited the value of the data obtained and the strength of the experiment.

Greater sampling effort than used here would be required to obtain sufficient length and age samples on an annual basis using the multimesh gillnet. At a minimum, at least double the current sampling effort would be required. This could be partially achieved by using only one gear type and transferring effort directed to beam trawl to the gillnet. The beam trawl sampling took 2 nights in an estuary, compared to between 2 and 4 nights for the gillnet.

CPUE

The independent sampling strategy as implemented using the multimesh gillnet and beam trawl detected spatial and temporal differences in the relative abundances of harvested and non-harvested species across a hierarchy of scales (see also Appendix 4 and several published papers; Rotherham et al. 2011; Gray et al. 2009). These results clearly demonstrate the potential use of such a sampling strategy to provide rigorous and quantitative standardized catch-per-unit-effort data for a diverse range of fishes within and across estuaries. This could potentially be used as a measure or indicator of the relative abundance of these species for use in aquatic biodiversity and fisheries resource assessments.

In comparison, no standardized CPUE effort data were available for the fisher-supplied logbook data (i.e. fishery-dependent data). This was primarily due to recent changes in reporting requirements. But even so, this has been an issue in this and other fisheries in NSW in the past when reporting requirements and formats have changed. This is a serious issue facing management agencies that seek standardized longitudinal data for resource and fisheries performance assessments. This problem could potentially continue to plague such assessments into the future (particularly if quotas are introduced and reporting requirements again change). Regardless, it will take several years using the current commercial catch returns until a standardized effort unit can be determined and reported for the species harvested in the Estuary General Fishery.

The reporting and analyses of commercial landed catches of each species was also limited. It was not permissible under privacy laws to present the data at the scales of this experiment. It requires eight or more commercial fishers to be actively fishing and reporting catch at the particular spatial and temporal scale of interest for the data to be available. Thus, in some circumstances it is not even possible to report total catch per annum of a particular species (or even all species combined) in an estuary. Nonetheless, general reporting of landed catches across large spatial and temporal scales are mostly possible, But, any further reductions in the commercial fishing fleets within each estuary will further diminish the usefulness of such fisher-supplied data and could potentially be a major concern for reporting of catches and future resource assessments. Thus, there are some severe restrictions on the availability and usefulness of the commercial catch and effort data that potentially prevent it being incorporated into rigorous resource assessments of harvested species.

Implementation of a standardized fishery-independent sampling strategy may be the only option to obtain a continuous time-series of standardized measures of the relative abundance of key harvested (and non-harvested) species within and across estuaries. Such sampling would also provide quantitative information on non-harvested species and be suitable for broader aquatic biodiversity assessments (Appendices 4 and 5; Rotherham et al. 2011). Such value adding to any sampling and assessment strategy may fulfill several aspects of the reporting obligations of management agencies and make such programs more cost-effective. Nevertheless, no sampling strategy will be a panacea for all management needs and circumstances. Surveys need to be specifically designed and tailored to meet the particular questions and requirements of management, which often change through time.

Demographic parameter estimations

In general, a greater proportion of larger and older individuals of each species were sampled in the FD compared to the FI sampling. This was understandable as only legal length fish were sampled. The fishery-independent sampling provided information on the length and age compositions of sub-legal length fish, affording broader representations of the population characteristics of each species. The differences in the data collected using each sampling strategy impacted estimates of mean length-at-age and growth of each species.

The estimated mean length-at-age of each species was generally greater in the FD compared to the FI samples. This was probably due to a combination of the FD over-sampling the fastest growing individuals, and for some species the FI under-sampling the largest individuals, in populations. We hypothesize that the commercial retained catches of each species contained greater proportions of the fastest growing individuals of each age class as they attained legal length and entered the fishery. Whilst this was particularly the case for the younger age classes of each species, it nevertheless extended out to include most age classes of all five species, but particularly for *A. australis.* This species displays variable growth rates, and it is possible that the fishery selects the quickest growing individuals even out to the age of 10 years, potentially providing overestimates of mean length-at-age across all age classes examined. Whilst this was not as pronounced in the other species, it still impacted all estimates of mean length-at-age. This was probably compounded by the FI samples containing fewer of largest length classes of each species, but particularly *A. australis*. An alternate compounding hypothesis is that the FI sampling under-sampled the largest individuals of all species and therefore selectively sampled the slowest growing individuals among populations. However, the length compositions of samples do not support this latter hypothesis for *M. cephalus*, *S. ciliata* and *G. tricuspidata*, as the FI sampled large individuals of each species, albeit in lower proportions than the FD sampling.

Because of the truncation of the FD length-based data at the MLL of each species, no sensible growth curves could be generated for any species using these data alone, demonstrating the inadequacies of such sampling for growth analyses. Moreover, the fastest growing individuals dominated the FD samples, further biasing any potential growth trajectories. Supplementary sampling of the discarded component of commercial catches, by using observers (Gray 2002; Gray and Kennelly 2003) or allowing fishers to retain sub-legal fish, could potentially overcome such shortfalls for growth analyses. But, such sampling could prove logistically difficult and costly. Moreover, if changes in growth or mean length-at-age are incorporated as indicators in fisheries assessments and performance measures, then the FD sampling of landed catches cannot be solely relied on to provide representative data of populations.

The FI sampling provided wider ranges of the lengths and ages of each species, producing different, and presumably more realistic and accurate, estimations of mean length-at-age and growth. A further advantage with the FI sampling is that it provides a consistent sampling frame, thus allowing for more rigorous comparisons of length-at-age relationships across estuaries and throughout time. This is essential for accurately assessing the potential impacts of fishing and environmental perturbations, such as a changing climate, on the population demographics of harvested and non-harvested species.

Differences in estimates of mortality and exploitation based on the FD and FI samples were inconsistent among species. The FD and FI samples produced similar estimates of *Z*, *F* and *E* for *G. tricuspidata* and *S. ciliata*. However, the FD estimates for *M. cephalus* were lower, but for *P. fuscus* greater, than the corresponding FI based estimates, and for *A. australis* it was dependent on the estuary. Further, the FD estimates of *Z*, *F* and *E* for each species based on the different commercial fishing gears were generally within the same range and again inconsistent among species. All sampling and fishing gears have inherent biases and we could not determine here which sampling strategy provided the most representative sample of the true status of populations. Further investigations are needed to untangle this dilemma.

Conclusions

This experiment has demonstrated some strengths and weaknesses in each of the fishery-dependent and -independent sampling strategies examined. The multimesh gillnet was better suited than the beam trawl to sample a broader suite of key harvested species for resource assessment purposes (see also Appendix 4). Nevertheless, greater sampling effort than used here (4 days per estuary) of the gillnet would be required to provide sufficient length- and age-based samples for a range of species on an annual basis for robust assessment purposes. The sampling of fishers retained catches at fishing cooperatives can provide a good platform and access to ample samples for length and age compositions and mortality analyses. But, this is only available in estuaries where commercial fishing occurs and where the fishers and the cooperatives willingly allow such sampling to take place. This has proven a problem in the past at some cooperatives, particularly when there has been a change in management regulations that negatively impacts fishery participants. Moreover, if the privacy laws (in NSW) as they currently exist and applied to catch records are expanded to include actual fishers catches, then this type of sampling could potentially become redundant in the future.

The fisher-reported logbooks have very limited value for assessment purposes at present due to inconsistencies in reported fishing effort and the privacy regulations that restrict usage of data for analyses across some spatial and temporal scales. This could prove a greater problem in the future if the numbers of commercial fishers operating within an estuary are further reduced. Moreover, such data is limited to estuaries that are open to commercial fishing.

A fishery-independent sampling strategy may provide the only comparative standardized assessments of relative abundance (CPUE) and length- and age-based demographic parameters across all estuaries regardless of fishing management arrangement. The utility and success of such a sampling program to detect differences in population demographics of a wide range of harvested species across estuaries has clearly been demonstrated in Appendix 6. Serious consideration of what management requires for assessing fisheries resources and performance, as a fishery-independent sampling program may be the only reliable strategy for assessing the estuarine fisheries resources of NSW, and elsewhere, in the future.

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Table 1. Summary of numbers of length and age samples collected for each species from the fishery-dependent and fishery–independent sampling in each year in the Clarence River and Wallis Lake.

	Length samples				Age samples			
	2008	2009	2010	Total	2008	2009	2010	Total
Clarence River								
Fishery-dependent sampling								
Acanthopagrus australis	778	1136	1005	2919	76	193	120	389
Platycephalus fuscus	334	228	858	1420	83	121	165	369
Girella tricuspidata	182	433	514	1129	45	110	88	243
Mugil cephalus	498	1059	1349	2906	87	155	173	415
Sillago ciliata	74	685	1279	2038	0	0	127	127
Fishery-independent sampling								
Acanthopagrus australis	188	527	175	890	181	179	158	518
Platycephalus fuscus	55	50	56	161	45	48	45	138
Girella tricuspidata	3	61	12	76	3	61	12	76
Mugil cephalus	301	436	1558	2295	225	201	202	628
Sillago ciliata	284	218	248	750	226	201	204	631
Wallis Lake								
Fishery-dependent sampling								
Acanthopagrus australis	0	2216	1273	3489	7	266	157	430
Platycephalus fuscus	1277	1941	1191	4409	250	260	175	685
Girella tricuspidata	179	2124	1616	3919	0	281	163	444
Mugil cephalus	960	2187	2038	5185	120	188	165	473
Sillago ciliata	3047	2239	1338	6624	390	224	170	784
Fishery-independent sampling								
Acanthopagrus australis	340	300	84	724	191	141	84	416
Platycephalus fuscus	44	29	39	112	43	28	39	110
Girella tricuspidata	42	94	42	178	47	94	43	184
Mugil cephalus	203	156	122	481	190	119	122	431
Sillago ciliata	19	54	49	122	18	54	57	129

Table 2. The total number of each study species sampled each year in the multimesh gillnet and beam trawl as part of the fishery-independent sampling in the Clarence River and Wallis Lake.

	Gillnet				Trawl			
	2008	2009	2010	**Total**	2008	2009	2010	**Total**
Clarence River								
Acanthopagrus australis	69	114	66	**249**	119	413	109	**641**
Platycephalus fuscus	48	42	39	**129**	7	8	7	**22**
Girella tricuspidata	3	61	12	**76**	0	0	0	**0**
Mugil cephalus	298	436	1557	**2291**	3	0	1	**4**
Sillago ciliata	279	208	232	**719**	5	10	16	**31**
Wallis Lake								
Acanthopagrus australis	51	21	27	**99**	289	279	57	**625**
Platycephalus fuscus	23	11	28	**62**	21	18	11	**50**
Girella tricuspidata	29	91	42	**162**	13	3	0	**16**
Mugil cephalus	203	158	122	**483**	0	0	0	**0**
Sillago ciliata	16	54	49	**119**	3	0	0	**3**

Table 3. PERMANOVA (univariate analyses) comparing the numbers of each of the five species sampled in the multimesh gillnet across the two estuaries (Clarence River and Wallis Lake), three years (2008, 2009, 2010) and two periods within each year.

Source	df	MS	Pseudo-*F*	*P*(perm)
Acanthopagrus australis				
Estuary	1	5.5579	1.3759	0.3233
Year	2	4.6829	0.1616	0.8655
Period(Year)	3	28.9840	3.4514	0.0175
Estuary x Year	2	18.5160	4.5840	0.1250
Estuary x Period(Year)	3	4.0394	0.4810	0.7072
Residual	204	8.3977		
Total	215			
Platycephalus fuscus				
Estuary	1	0.0208	0.0407	0.8548
Year	2	2.2986	0.6834	0.5406
Period(Year)	3	3.3634	2.4192	0.0649
Estuary x Year	2	2.3819	4.6561	0.1248
Estuary x Period(Year)	3	0.5116	0.3680	0.7800
Residual	204	1.3903		
Total	215			
Girella tricuspidata				
Estuary	1	142.3700	8.8802	0.0168
Year	2	64.7660	1.3858	0.2648
Period(Year)	3	46.7360	5.0436	0.0026
Estuary x Year	2	8.1551	0.5087	0.7177
Estuary x Period(Year)	3	16.0320	1.7302	0.1568
Residual	204	9.2663		
Total	215			
Mugil cephalus				
Estuary	1	4063.9000	1.6100	0.2836
Year	2	2669.1000	4.0359	0.0001
Period(Year)	3	661.3400	0.3834	0.7743
Estuary x Year	2	4044.5000	1.6024	0.2908
Estuary x Period(Year)	3	2524.1000	1.4634	0.2303
Residual	204	1724.9000		
Total	215			
Sillago ciliata				
Estuary	1	533.3300	20.0420	0.0218
Year	2	0.5278	0.0123	1.0000
Period(Year)	3	43.0560	1.0842	0.3630
Estuary x Year	2	40.5280	1.5230	0.3514
Estuary x Period(Year)	3	26.6110	0.6701	0.5658
Residual	204	39.7120		
Total	215			

Table 4. PERMANOVA (univariate analyses) comparing the length-at-age of each sex of the five species sampled across the two sampling strategies (fishery-independent and fishery-dependent) and age class in each estuary. The age ranges used in each analysis are shown.

* = P < 0.05, ** = P < 0.01, *** < 0.001, ns = P > 0.05.

Species	Sex	Estuary	Age Range	Strategy	Age Class	S x AC
M. cephalus	Female	Clarence River	1 to 6	***	***	ns
M. cephalus	Male	Clarence River	1 to 6	***	***	ns
M. cephalus	Female	Wallis Lake	1 to 4	***	***	***
M. cephalus	Male	Wallis Lake	1 to 4	***	***	***
A. australis	Female	Clarence River	3 to 9	***	***	*
A. australis	Male	Clarence River	4 to 10	***	***	*
A. australis	Female	Wallis Lake	3 to 7	***	***	*
A. australis	Male	Wallis Lake	4 to 5	***	ns	ns
P. fuscus	Female	Clarence River	1 to 5	***	***	***
P. fuscus	Male	Clarence River	NA	-	-	-
P. fuscus	Female	Wallis Lake	1 to 4	***	***	***
P. fuscus	Male	Wallis Lake	2 to 3	***	***	ns
G. tricuspidata	Female	Clarence River	5 only	ns	-	-
G. tricuspidata	Male	Clarence River	3 to 5	***	***	**
G. tricuspidata	Female	Wallis Lake	3 to 7	***	***	*
G. tricuspidata	Male	Wallis Lake	3 to 6	***	***	*
S. ciliata	Female	Clarence River	2 to 7	***	***	*
S. ciliata	Male	Clarence River	2 to 7	***	***	ns
S. ciliata	Female	Wallis Lake	2 to 8	***	***	ns
S. ciliata	Male	Wallis Lake	3 to 9	*	***	ns

Table 5. Summary of estimates of rates of mortality and exploitation for each species based on fishery-dependent (FD) and fishery-independent (FI) sampling in the Clarence River and Wallis Lake. Data pooled across 3 years of sampling; total mortality (*Z*) (+ 1SE) derived from catch-curve analyses of data displayed in Figures 1 to 5; natural mortality (*M*) determined using the method of Hoening (1975) and based on the published maximum age attained for each species; fishing mortality (*F*) and exploitation rates (*E*) calculated using estimates of *M* and *Z*.

Estuary	Method	Gear Type	Age range	*Z*	SE	R	*M*	*F*	*E*
Acanthopagrus australis									
CR	FD	Beach-seine	8-18	0.42	0.08	0.77	0.19	0.23	0.55
CR	FD	Gillnet	6-17	0.53	0.09	0.79	0.19	0.34	0.64
CR	FD	Trap	7-18	0.42	0.06	0.84	0.19	0.23	0.54
CR	FI	Multimesh Gillnet	1-11	0.21	0.05	0.66	0.19	0.02	0.11
CR	FI	Beam Trawl	1-13	0.37	0.05	0.83	0.19	0.18	0.49
WL	FD	Beach-seine	7-16	0.38	0.04	0.92	0.19	0.19	0.50
WL	FD	Gillnet	7-20	0.43	0.05	0.88	0.19	0.24	0.55
WL	FD	Trap	7-20	0.39	0.04	0.91	0.19	0.20	0.51
WL	FI	Multimesh Gillnet	3-9	0.55	0.09	0.89	0.19	0.36	0.65
WL	FI	Beam Trawl	0-7	0.80	0.05	0.98	0.19	0.61	0.76
Platycephalus fuscus									
CR	FD	Gillnet	3-7	1.33	0.20	0.94	0.26	1.07	0.80
CR	FI	Multimesh Gillnet	2-6	0.62	0.07	0.97	0.26	0.36	0.58
WL	FD	Gillnet	3-9	1.06	0.21	0.84	0.26	0.80	0.76
WL	FI	Multimesh Gillnet	2-5	0.91	0.13	0.96	0.26	0.65	0.71
Girella tricuspidata									
CR	FD	Beach-seine	4-14	0.46	0.06	0.86	0.19	0.27	0.59
CR	FD	Gillnet	4-13	0.40	0.04	0.91	0.19	0.21	0.52
CR	FI	Multimesh Gillnet	4-12	0.49	0.17	0.61	0.19	0.30	0.61
WL	FD	Beach-seine	5-17	0.35	0.03	0.92	0.19	0.16	0.46
WL	FD	Gillnet	5-17	0.39	0.04	0.90	0.19	0.20	0.52
WL	FI	Multimesh Gillnet	2-12	0.41	0.05	0.89	0.19	0.22	0.54
Mugil cephalus									
CR	FD	Gillnet	2-9	0.71	0.05	0.97	0.35	0.36	0.51
CR	FI	Multimesh Gillnet	1-6	0.96	0.05	0.99	0.35	0.61	0.63
WL	FD	Gillnet	2-7	1.41	0.15	0.96	0.35	1.06	0.75
WL	FI	Multimesh Gillnet	2-6	1.48	0.22	0.94	0.35	1.13	0.76
Sillago ciliata									
CR	FD	Beach-seine	2-10	0.47	0.07	0.87	0.26	0.21	0.45
CR	FD	Gillnet	5-10	0.68	0.04	0.98	0.26	0.42	0.62
CR	FI	Multimesh Gillnet	2-9	0.60	0.05	0.96	0.26	0.34	0.57
WL	FD	Beach-seine	4-14	0.64	0.04	0.96	0.26	0.38	0.59
WL	FD	Gillnet	6-14	0.51	0.05	0.94	0.26	0.25	0.49
WL	FI	Multimesh Gillnet	5-11	0.53	0.09	0.86	0.26	0.27	0.51

Figure 1. Mean (± 1 SE) number of each species caught in each of the two sampling periods in each year using the fishery independent multimesh gillnet sampling and the catch per unit of effort from the commercial logbook data.

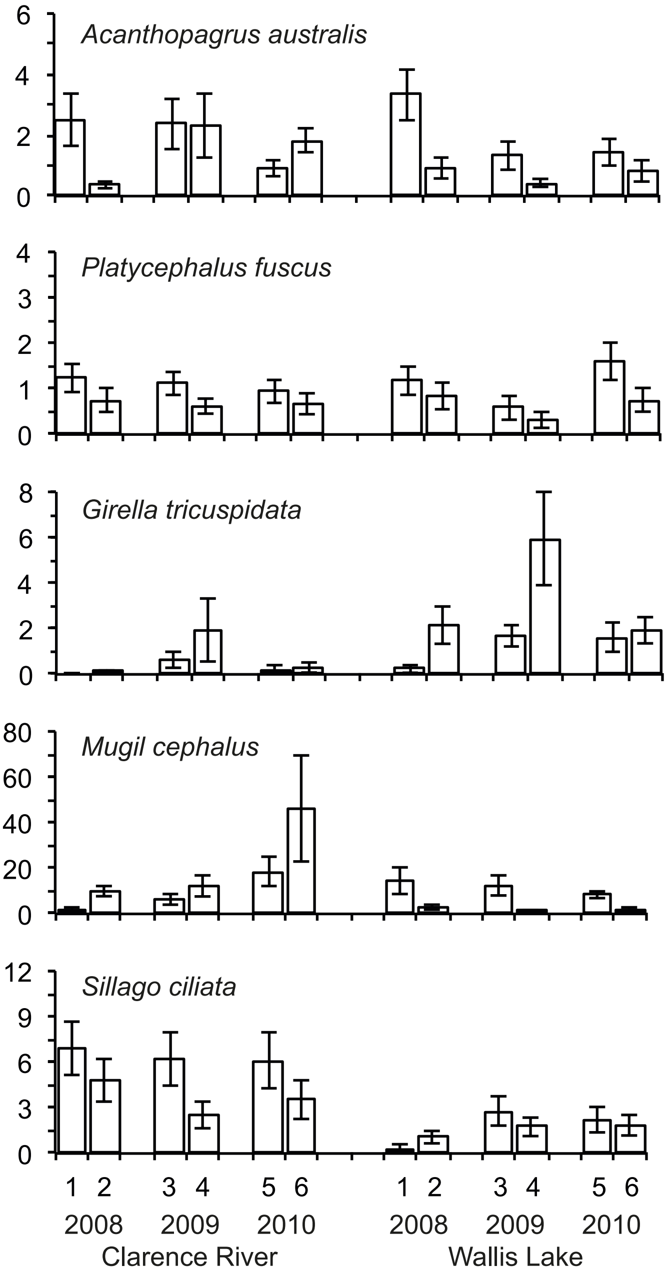


Figure 2. Length and age compositions derived from fishery-dependent and fishery-independent sampling strategies of *Acanthopagrus australis* in the Clarence River and Wallis Lake. Data combined across the three years of sampling; n denotes sample size, ML = mean length (cm), MA = mean age (years).

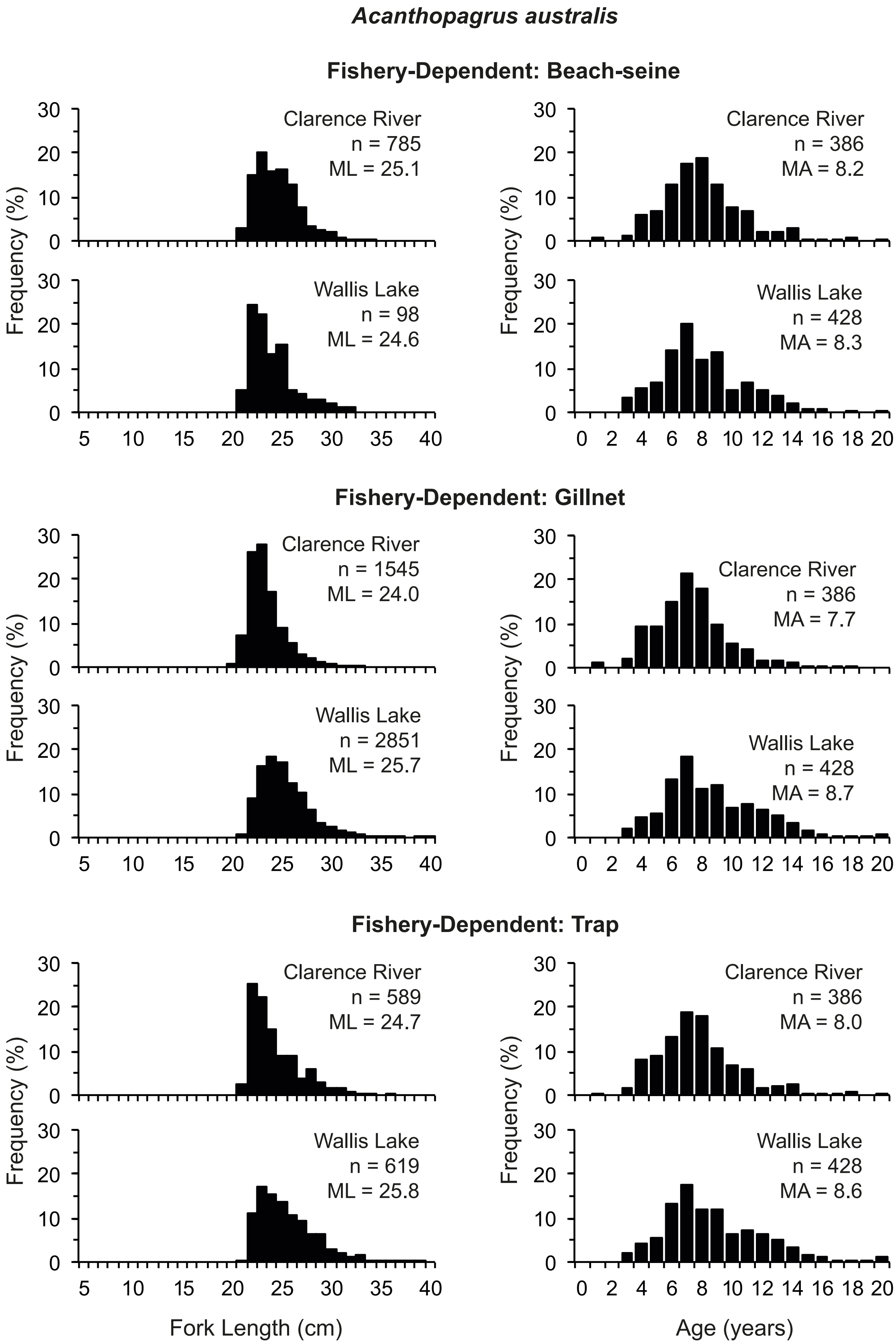


Figure 2. Continued

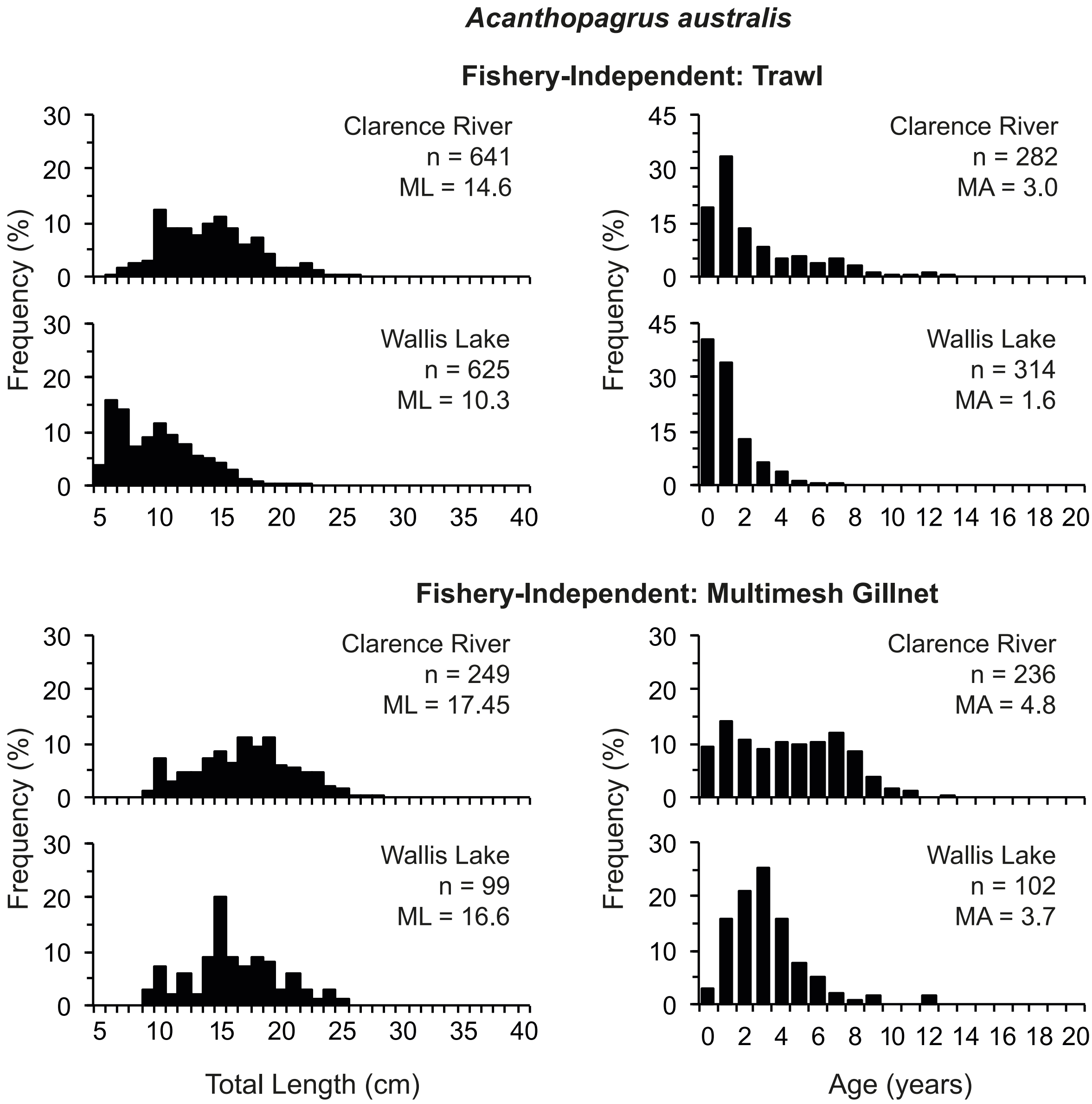


Figure 3. Length and age compositions derived from fishery-dependent and fishery-independent sampling strategies of *Platycephalus fuscus* in the Clarence River and Wallis Lake. Data combined across the three years of sampling; n denotes sample size, ML = mean length (cm), MA = mean age (years).

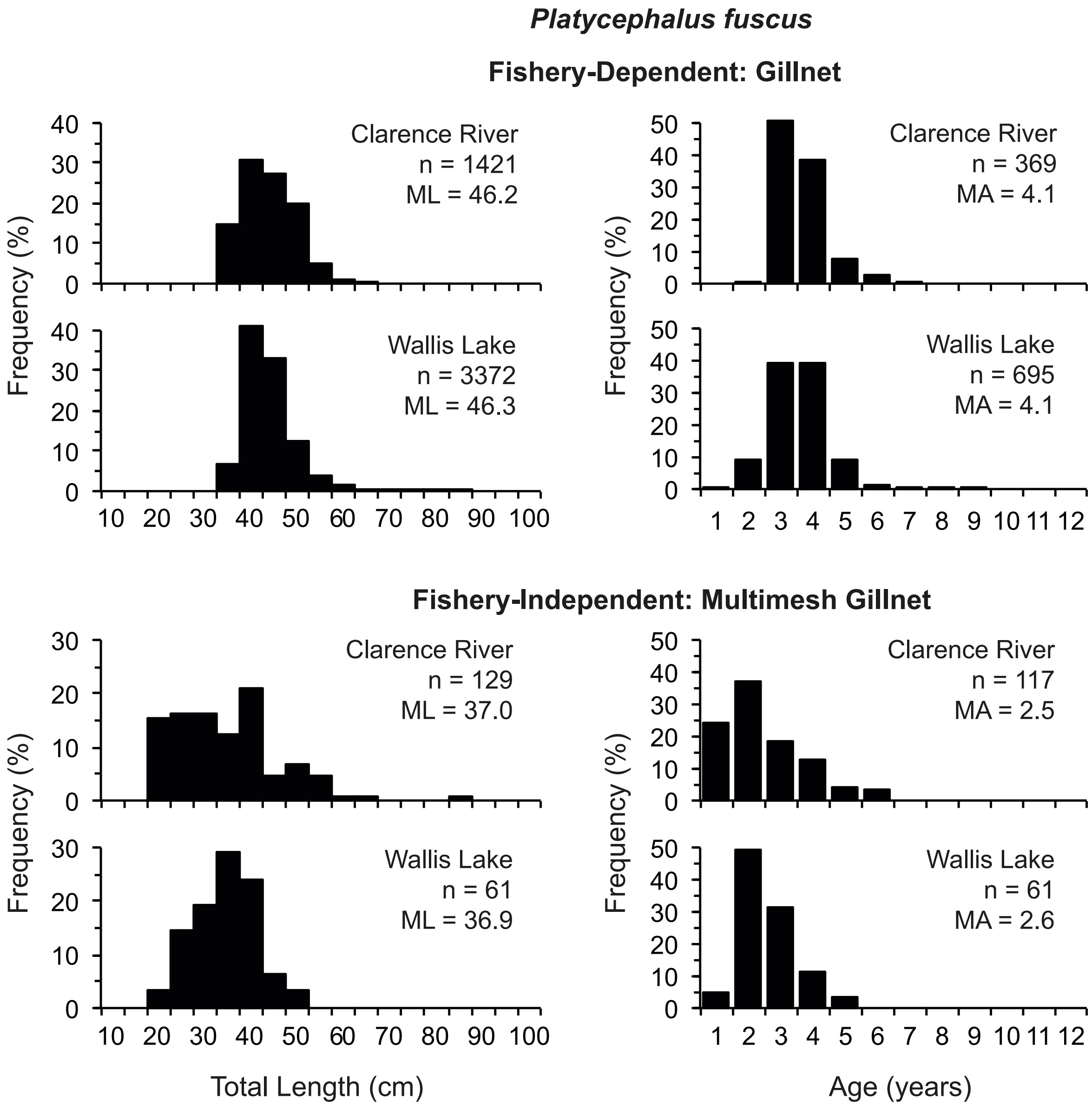


Figure 4. Length and age compositions derived from fishery-dependent and fishery-independent sampling strategies of *Girella tricuspidata* in the Clarence River and Wallis Lake. Data combined across the three years of sampling; n denotes sample size, ML = mean length (cm), MA = mean age (years).

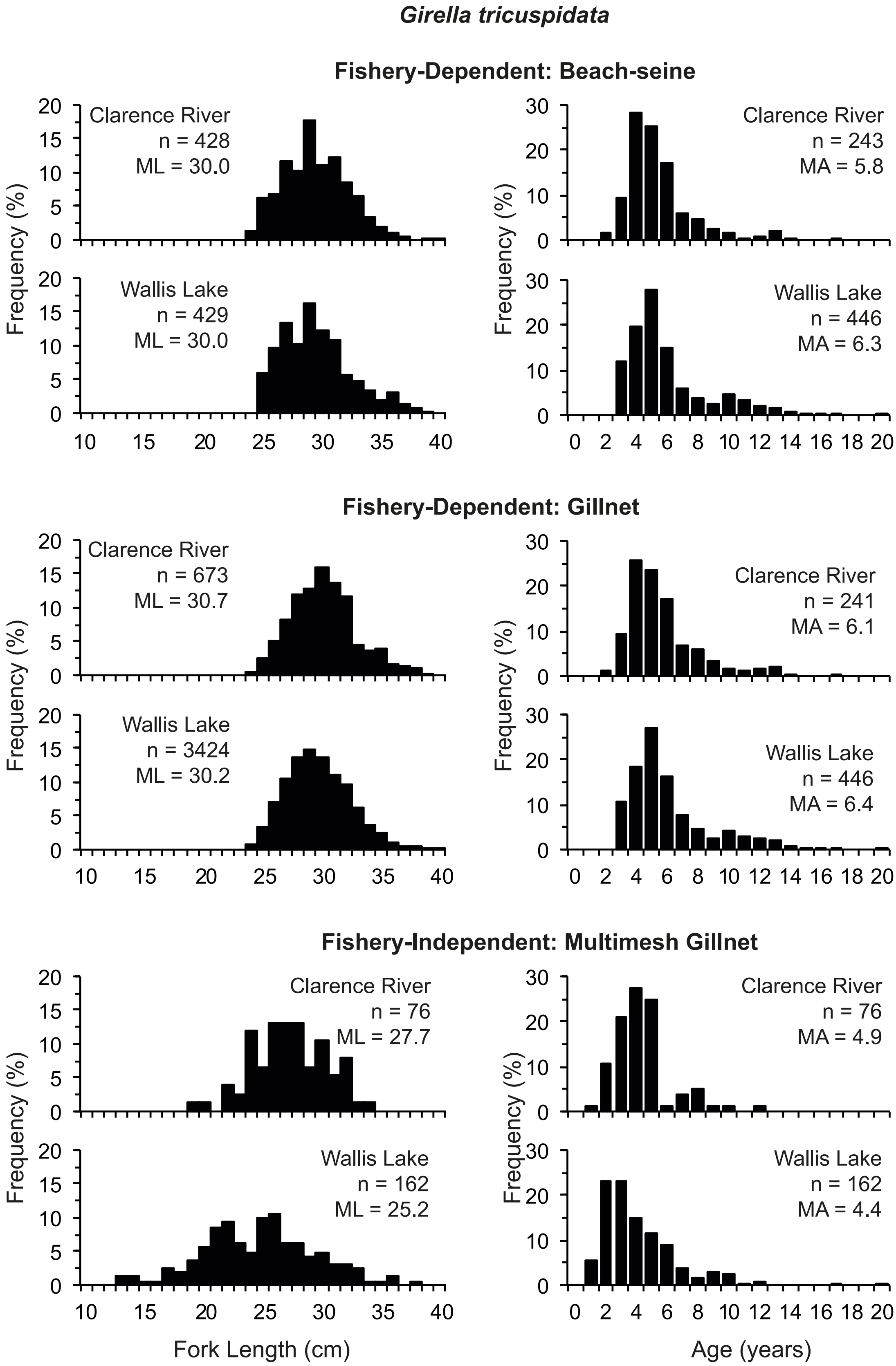


Figure 5. Length and age compositions derived from fishery-dependent and fishery-independent sampling strategies of *Mugil cephalus* in the Clarence River and Wallis Lake. Data combined across the three years of sampling; n denotes sample size, ML = mean length (cm), MA = mean age (years).

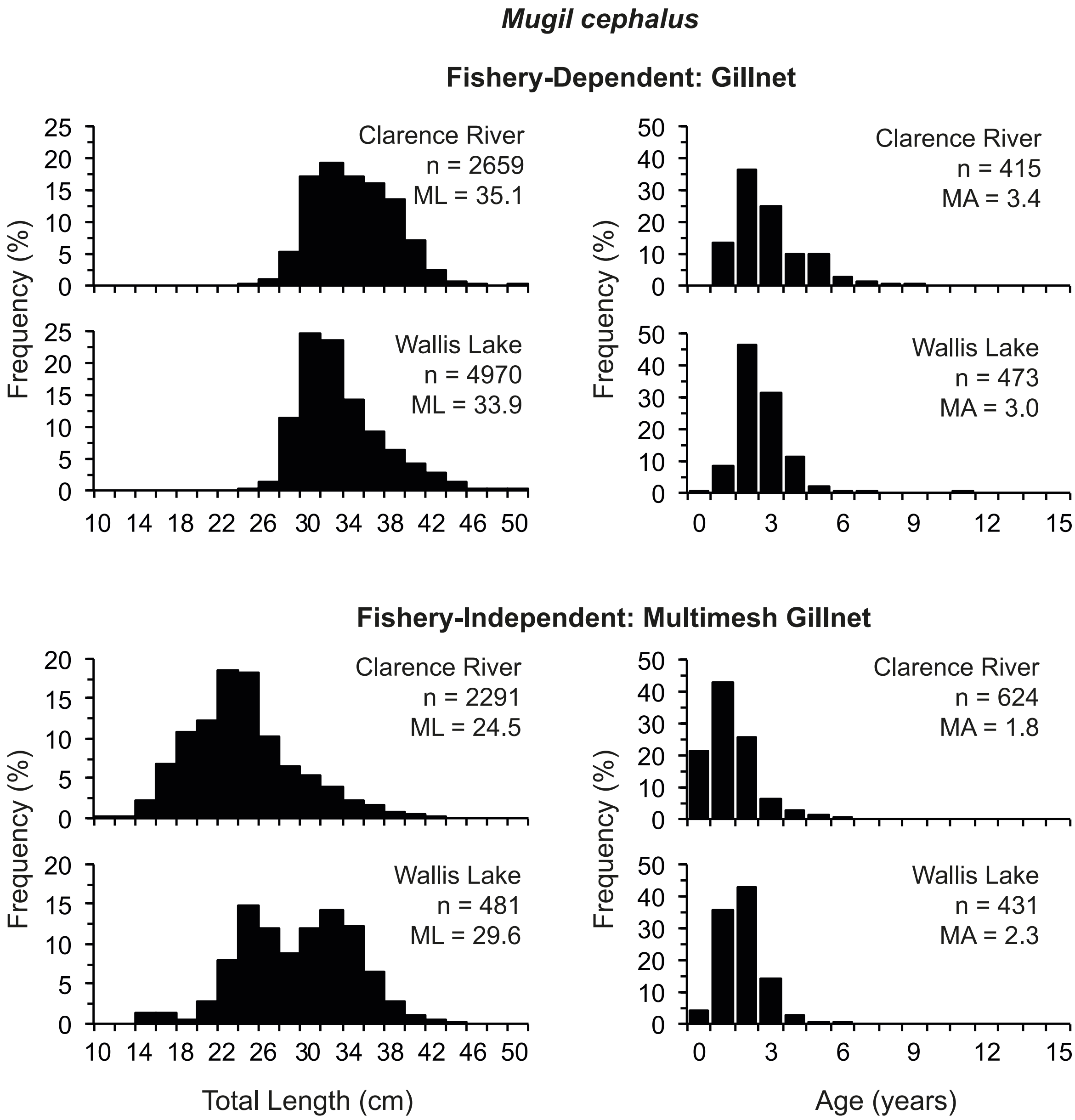


Figure 6. Length and age compositions derived from fishery-dependent and fishery-independent sampling strategies of *Sillago ciliata* in the Clarence River and Wallis Lake. Data combined across the three years of sampling; n denotes sample size, ML = mean length (cm), MA = mean age (years).

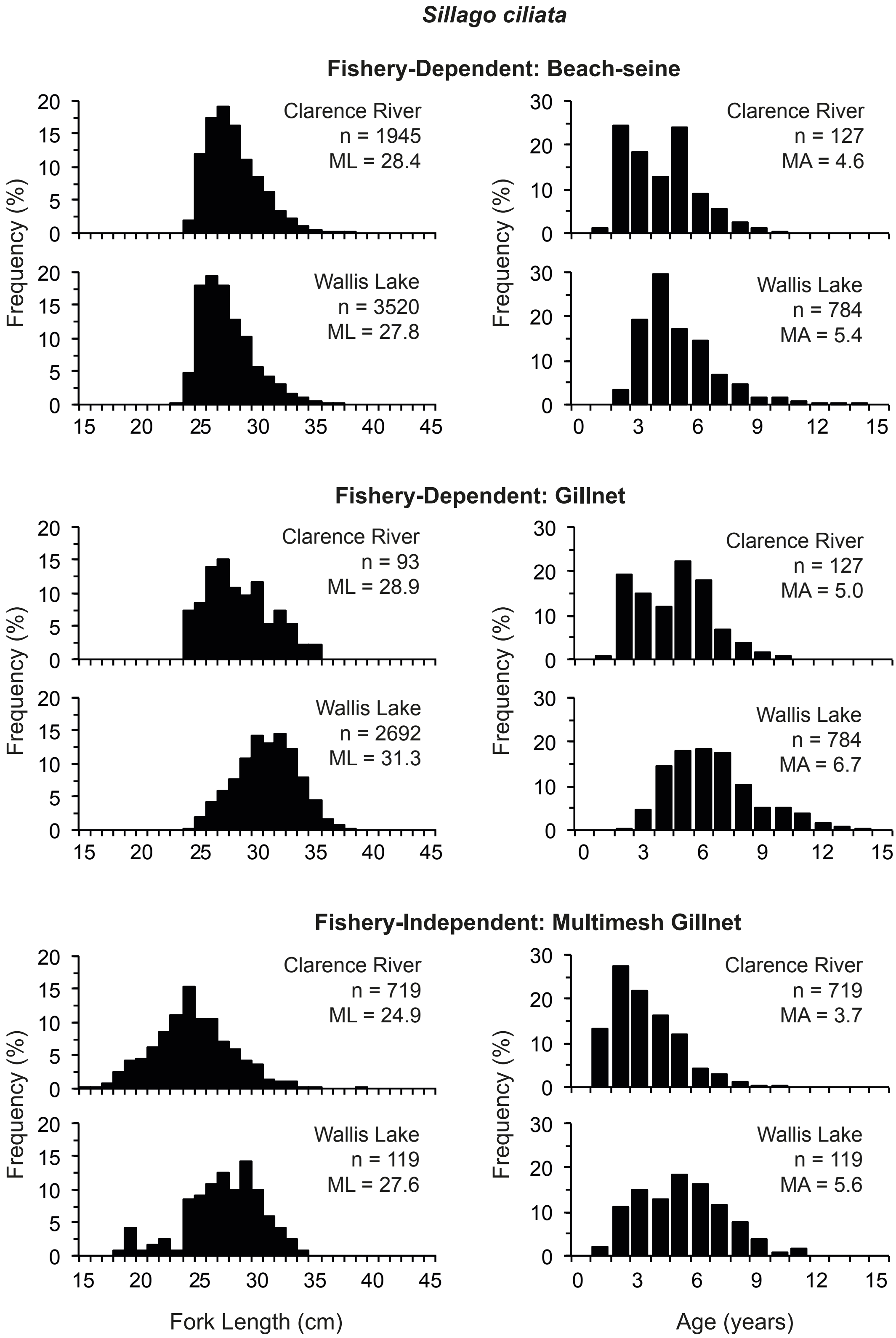


Figure 7. Mean (±1 SE) length-at-age of female and male *Acanthopagrus australis* and *Girella tricuspidata* in the Clarence River and Wallis Lake as determined from the fishery-dependent and fishery-independent sampling strategies. Data combined across the three years of sampling.

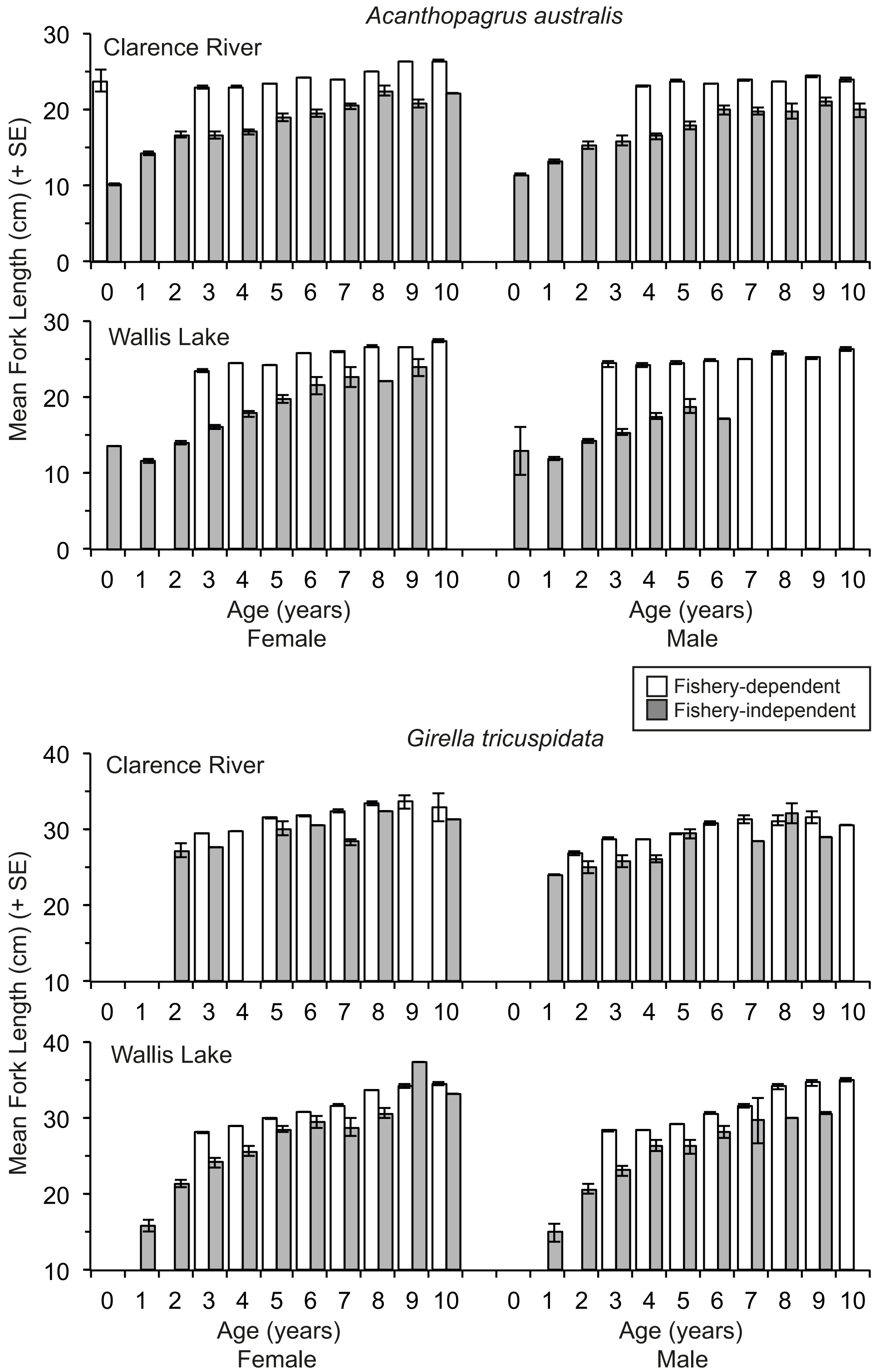
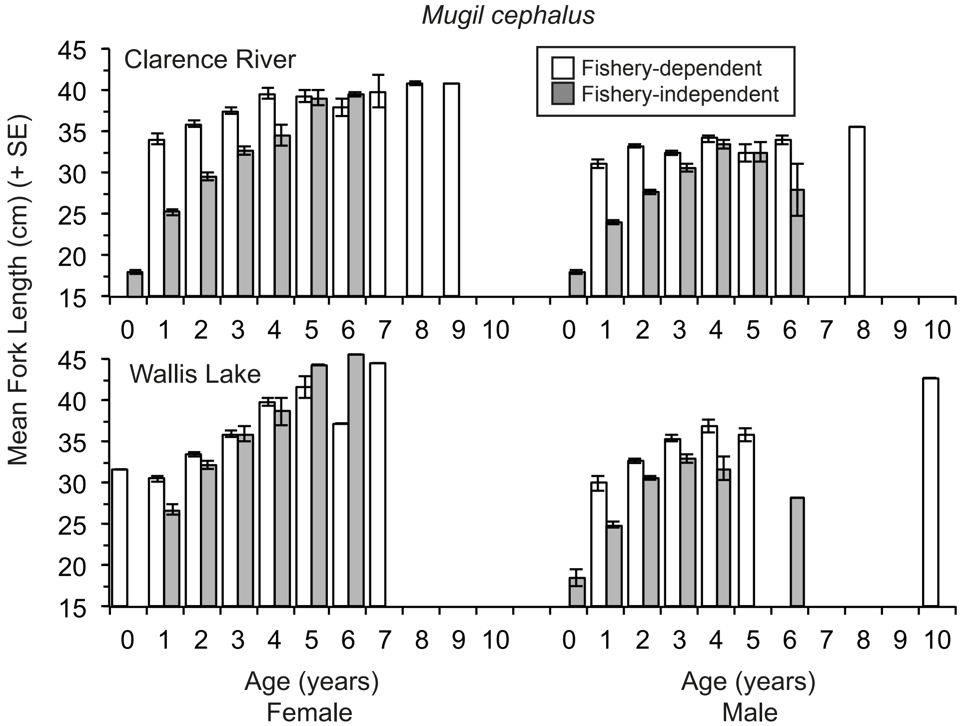


Figure 8. Mean (±1 SE) length-at-age of female and male *Mugil cephalus* and *Sillago ciliata* in the Clarence River and Wallis Lake as determined from the fishery-dependent and fishery-independent sampling strategies. Data combined across the three years of sampling.



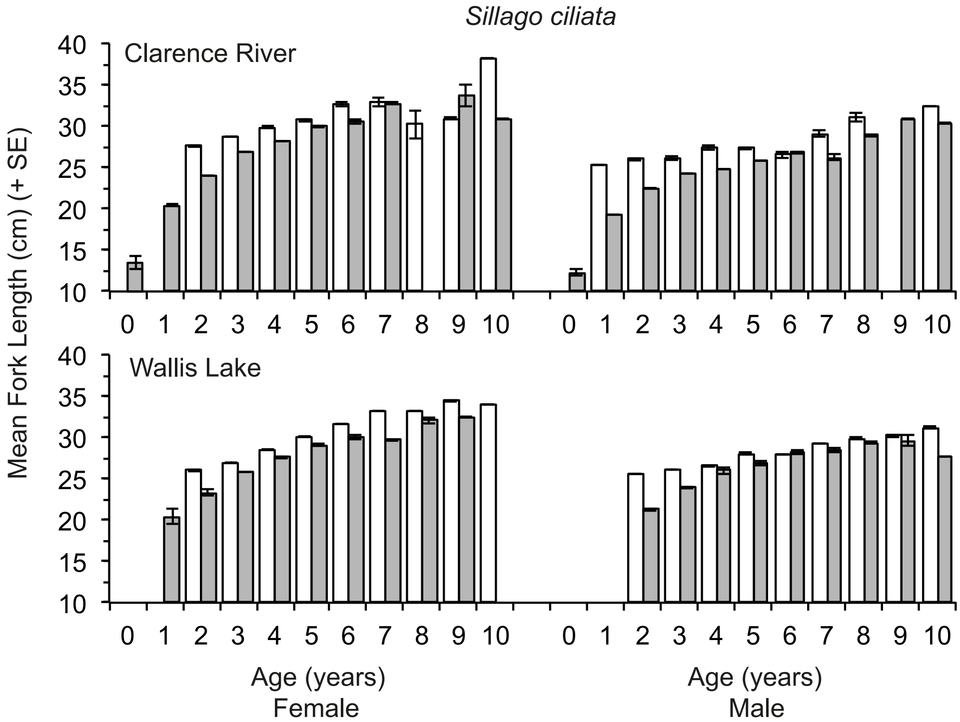
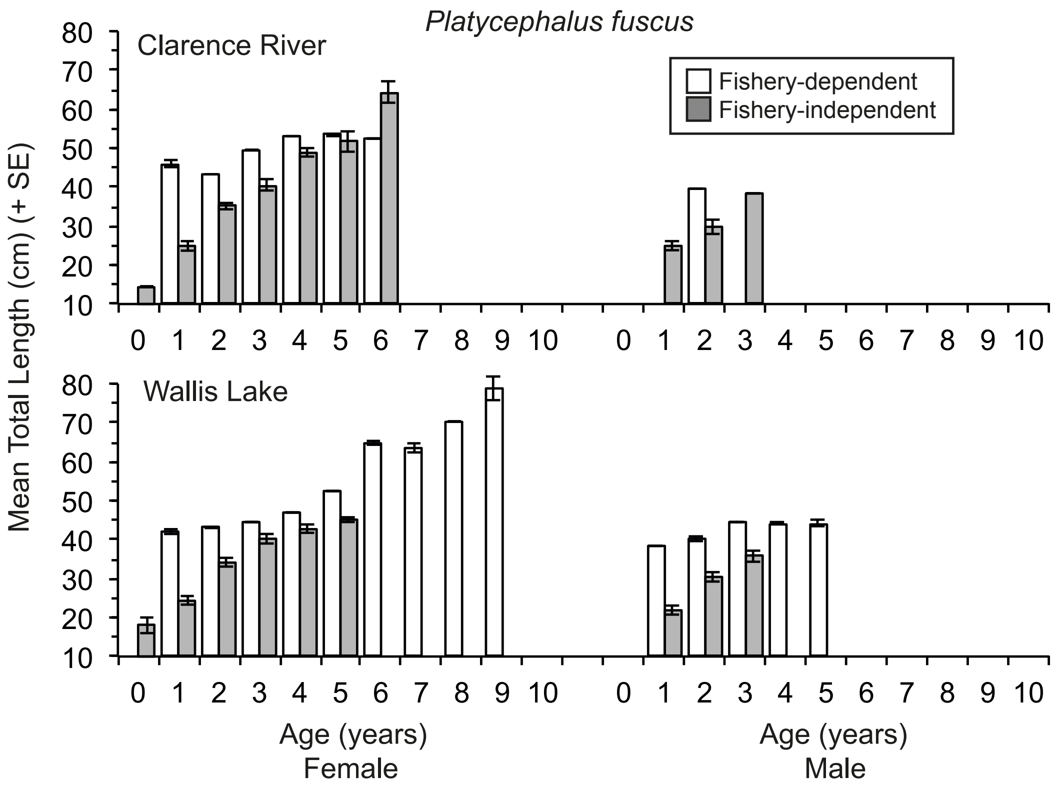


Figure 9. Mean (±1 SE) length-at-age of female and male *Platycephalus fuscus* in the Clarence River and Wallis Lake as determined from the fishery-dependent and fishery-independent sampling strategies. Data combined across the three years of sampling.



Appendix 4. Summary of Experiment 2: Comparison of fishery-independent and fishery-dependent (recreational) sampling strategies to assess harvested fish species and aquatic biodiversity

Introduction

The objective of this experiment was to examine whether the diversity and relative abundance of sampled organisms differed between the fishery-independent sampling strategy and a fishery-dependent strategy of sampling recreational anglers catches using a typical creel survey. So called 'creel surveys' are commonly used to determine harvests, effort and sizes of organisms retained by recreational anglers. These types of studies have been used extensively in NSW.

We hypothesized that the diversity, relative abundances and mean lengths of species would differ between sampling strategies. We tested this by using the fishery-independent sampling strategy to sample fish populations during the same period that a creel survey of recreational anglers catches took place. This was done in two estuaries. This experiment directly contributed to Project Objectives 1 and 3.

Materials and Methods

This experiment was done in two recreational fishing havens, Lake Macquarie and St Georges Basin between February and April 2011. In each estuary, standardised fishery-independent sampling using the multimesh gillnet and beam trawl was stratified across two six-week periods. In each period it took 6 nights to complete the gillnetting and 2 nights to do the beam trawl sampling in each estuary. During this same period, a creel survey that comprised two contact methods (access-point and roving) was used to collect data on recreational harvests and fishing effort of boat- and shore-based anglers (Ochwada-Doyle et al. 2014). A total of 9 week days and 9 weekend days were sampled in each estuary throughout the 3 month period. Each organism obtained in samples from both sampling strategies was identified and key species measured for length.

For this experiment, the creel survey data were restricted to samples of boat- and shore-based anglers taken from the lake (i.e. not the channel or canal). This was done because the fishery-independendnet sampling was done only in the lake of each estuary.

Results

Across both estuaries, a total of 73 and 45 species were sampled in the fishery-independent and -dependent sampling strategies, respectively (Table 1). A greater number of species were sampled in the gillnet and beam trawl than in the creel survey in Lake Macquarie, but this was not observed in St Georges Basin. Samples from the creel survey were limited to species retained by recreational anglers, whereas the fishery-independent samples contained most of these species plus several non-harvested species, particularly in Lake Macquarie.

Across both estuaries, greater total numbers of species and individuals were sampled using the gillnet than the beam trawl, and similarly in the boat-based catches than the shore-based catches.

The length compositions of samples of *A. australis*, *P. saltatrix*, *S. ciliata* and *P. fuscus* in the creel survey were primarily restricted to individuals of legal length, although some sub-legal length individuals were present in catches (Fig. 1 and 2). Both sub-legal and legal length individuals of these harvested species occurred in samples obtained from the fishery-independent strategy (Fig. 1 and 2). The mean length of *A. australis* and *P. saltatrix* was less in the fishery-independent compared to the fishery-dependent samples. This was not the case for *S. ciliata* and *P. fuscus*. Compared to the fishery-dependent samples, the fishery-independent samples appeared to under-sample *A. australis* greater than 30 cm FL. The under-sampling of large individuals was not evident for the other three species examined.

Table 1. The total numbers of each species sampled in the fishery-independent and fishery-dependent sampling strategies in Lake Macquarie and St Georges Basin.

	Lake Macquarie				St Georges Basin			
	Fishery-independent		Fishery-dependent		Fishery-independent		Fishery-dependent	
Species	Gillnet	Trawl	Boat	Shore	Gillnet	Trawl	Boat	Shore
Acanthopagrus australis	174	31	347	52	177	24	584	32
Acanthopagrus butcheri			1				5	
Aldrichetta forsteri			1531	24			9	
Alpheus edwardsi		2						
Aluterus monoceros			1					
Anguilla australis							1	
Ambassis sp.		15				14		
Aptychotrema rostrata								
Apogon angustatus		73						
Aptychotrema rostrata	1				1			
Arenigobius bifrenatus		2						
Arenigobius frenatus		4				2		
Argyrosomus hololepidotus	10		2				1	
Arripis trutta	15		22	2	1			
Atherinomorus vaigiensis	1	20			18	3		
Brachirus nigra	1				3	6		
Chelidonichthys kumu							1	
Cnidoglanis macrocephalus	5				3			
Dasyatis fluviorum	8	3			4			
Dicotylichthys punctulatus	5	3			1	3		
Dinolestes lewini	42			1	1			
Diodontidae sp.					1			
Elops machnata					1			
Eubalichthys mosaicus	1							
Euristhmus lepturus	5	2						
Foetorepus calauropomus		2						
Gerres subfasciatus	335	639			612	72	2	
Girella tricuspidata	339	1	1	5	208	1	3	1
Herklotsichthys castelnaui	875	10		4	86	1		
Hyperlophus vittatus	1	7			9	23		
Hyporhamphus regularis	2	2		2		2	2	14
Leiopotherapon aheneus		3				10		
Liza argentea	601		2		233			5
Macquaria colonorum	34				2			
Macrophthalmus crassipes	2							
Melicertus plebejus		287				272		
Metapenaeus bennettae		845			1	213		
Metapenaeus macleayi		15				23		
Meuschenia freycineti		1	8				8	
Meuschenia trachylepis	10	6	12	1	8	15	25	
Microcanthus strigatus	2	1				1		
Monacanthus chinensis	8	33	2	2	70	95	23	31
Monodactylus argenteus	1				3			
Mugil cephalus	910		4		668		1	
Myliobatis australis	2							
Myxus elongatus	3			6	482		17	2
Nelusetta ayraudi							8	
Neotrygon kuhlii	2							
Notolabrus gymnogenis				2				
Nototodarus gouldi		21				5		
Octopus spp	1		1					
Ocypodidae		1						
Pagrus auratus	14	42	39	5	42	10	71	
Paramugil georgii	273		3	5	17			
Pelates sexlineatus	106	483		17	78	124	2	
Penaeus monodon		34						
Platycephalus bassensis							2	
Platycephalus fuscus	59	4	178	18	83	4	475	25
Plotosus lineatus	2	1						
Platycephalus richardsoni			1					
Pomatomus saltatrix	215	36	84	26	153	3	56	3
Portunus (Portunus) pelagicus	123	52	773	6	49	4	6	1
Portunus (Portunus) sanguinolent	1	3						
Potamalosa richmondia	2							
Priacanthus macracanthus	1	1						
Pseudocaranx dentex	7				54	2		
Pseudocaranx georgianus			4				18	7
Pseudorhombus arsius	4	94	19			32	6	2
Pseudorhombus jenynsii			26	2	1		6	2
Pseudorhombus tenuirastrum			3	1			4	
Rachycentron canadum			1					
Rhabdosargus sarba	27	44	18	2	88	31	15	1
Scobinichthys granulatus			4	2		5	15	
Scorpaena cardinalis			1					
Scylla serrata	4		3	1				
Selenotoca multifasciata	3							
Sepia rozella		14						
Seriola lalandi			2					
Seriolella punctata							1	
Sepiadarium austrinum		29						
Sillago ciliata	77		111		430	4	75	1
Sillago maculata	40	164	73	1	66	322	23	1
Strongylura leiura	3				4			
Tetractenos glaber					2			
Tetractenos hamiltoni	1				7	7		
Trachinocephalus myops	1							
Trachurus novaezelandiae	23	1	1	1	7		1	
Trygonoptera testacea	1				4	3		
Tylosurus crocodilus			1	1			1	
Upeneus tragula	2	11						
Total species	52	42	33	25	39	32	32	15
Total individuals	4385	3042	3279	189	3678	1336	1467	128

Figure 1. Length frequency composition of fishery-independent and fishery-dependent samples of *Sillago ciliata* and *Acanthopagrus australis* in Lake Macquarie and St Georges Basin. n = sample size, ML = mean fork length (cm).

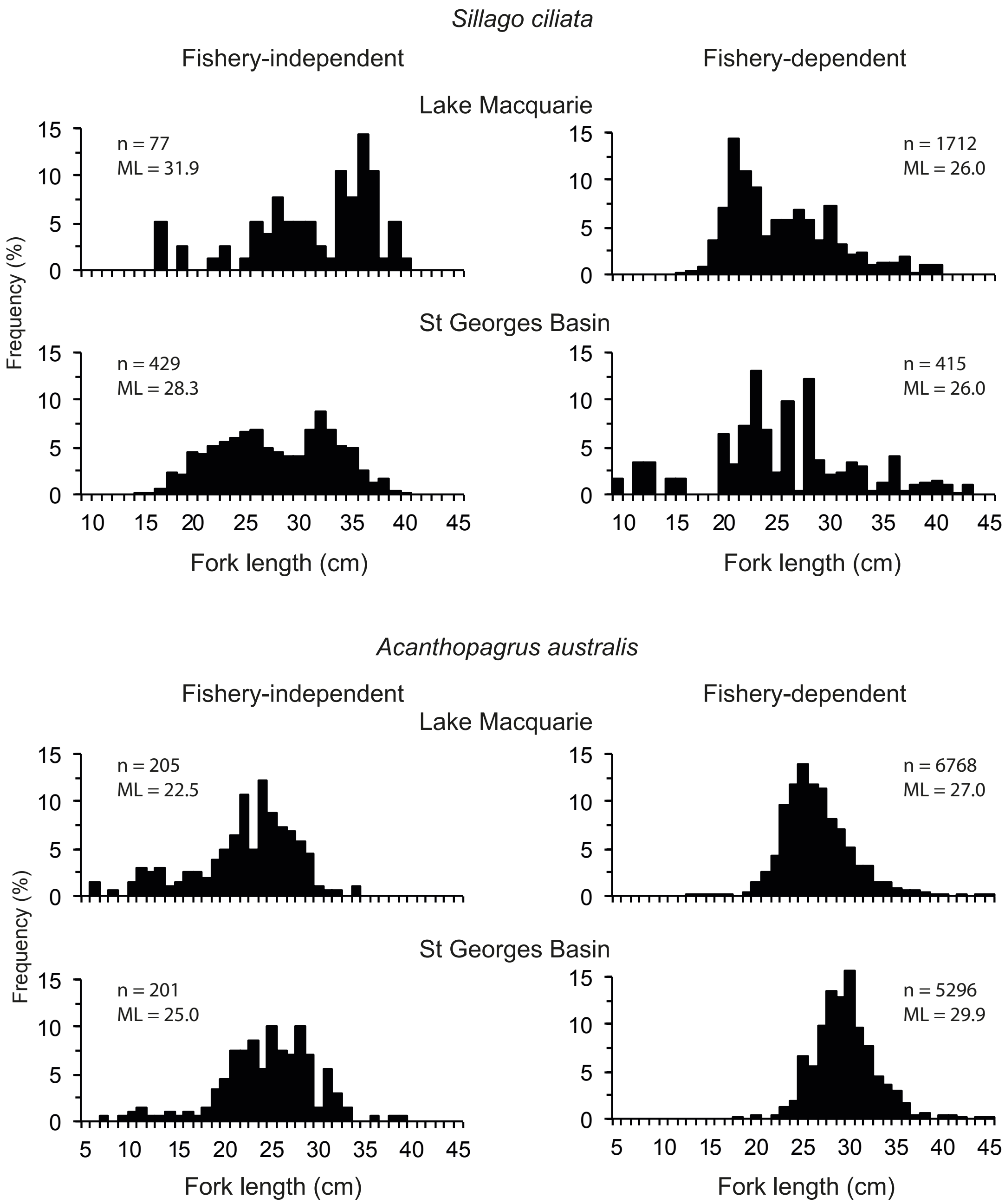
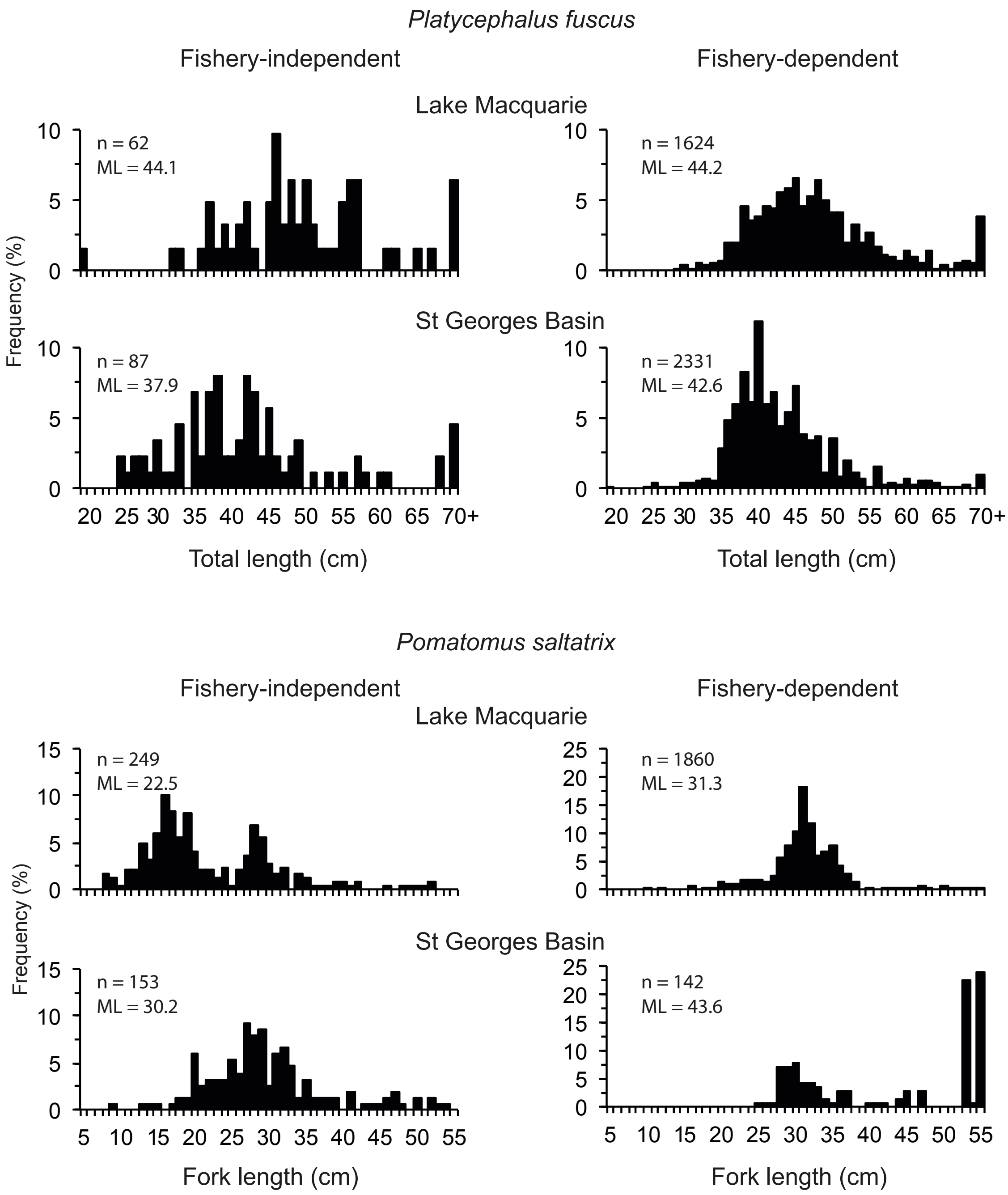


Figure 2. Length frequency composition of fishery-independent and fishery-dependent samples of *Platycephalus fuscus* and *Pomatomus saltatrix* in Lake Macquarie and St Georges Basin. n = sample size, ML = mean length (cm).



Appendix 5. Experiment 3: Development of a fishery-independent sampling strategy for the giant mud crab and comparison with fishery-dependent sampling of commercial catches

Comparison of commercial catch data and research surveys for assessing populations of mud crab (*Scylla serrata*)

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Abstract

The giant mud crab (*Scylla serrata*) is harvested by commercial and recreational fishers in estuaries in northern NSW, Australia. Despite the value of this species to commercial and recreational fishers, the status of *S. serrata* populations in this region remains poorly understood. This study was done to evaluate the effectiveness of two sources of fishery-dependent; observer-based and fisher-logbook and -independent sampling programs in providing data for assessing and monitoring populations of *S. serrata*. The results illustrate that patterns of abundance, size-structure and sex ratio of pot catches are influenced by the source of data, estuary and areas within the estuary sampled. The magnitude and pattern of variation between standardised catch per unit effort (CPUE) data from observer-based and independent surveys differed between rivers, sexes and sizes of crabs. Estimates of the level of precision of CPUE also varied between rivers, survey types and sexes. Differences in the size-structure of catches between survey designs were not consistent between estuaries. In the Richmond River, larger proportions of undersize male *S. serrata* were reported from observer-based and fisher-logbook surveys, while in the Macleay River larger proportions of undersized *S. serrata* were caught in the independent survey. Given our results, relying on only one source of data to assess the status of *S. serrata* populations in this region is not sensible, however, the future management objectives of the fishery and sources of available funding will determine the most appropriate survey type(s) to monitor populations of *S. serrata* in this region.

Introduction

One of the biggest challenges facing the sustainable management of fishery resources worldwide is determining the most cost effective and reliable sources of data for monitoring and assessment (Smith *et al*. 2009, Bentley and Stokes 2013). Traditionally, data collected from commercial fisheries have been used for stock assessments of exploited species (Andrew and Chen 1997, Moffett *et al.* 2011). Yet, the problems of fishery-dependent data for traditional stock assessments and ecosystem-based management are well known (Cotter and Pilling 2007). On the other hand, while fishery-independent research surveys are often advocated for improved assessment and management (Pennington and Stromme 1998, Kennelly and Scandol 2002), they're not without their own problems and limitations. Indeed, the quality and quantity of fishery-independent data are affected by the design of surveys and configuration of sampling gears (Smith and Tremblay 2003, Rotherham *et al*. 2007). Despite this, there have been few attempts to examine the costs and benefits of fishery-dependent- and -independent surveys for assessing and managing fish resources and biodiversity (Fox and Starr 1996, Scheirer *et al.* 2004).

Giant mud crabs, *Scylla serrata* (Forsskal 1775), are found throughout the Indo-west Pacific from Africa to south-east Asia and Australia, in a range of habitats from the intertidal zone to waters 80 m deep (Fratini *et al.* 2002). In Australia, *S. serrata* are distributed throughout northern Australia from central New South Wales (NSW) west to Exmouth Gulf, Western Australia (Meynecke *et al*. 2012). In NSW, significant quantities of *S. serrata* are harvested from estuarine waters by commercial and recreational fishers using baited pots. Recreational catches of *S. serrata* are large and can often exceed commercial landings (Rowling *et al*. 2010). Despite the value of this species to commercial (e.g. the commercial catch in NSW is valued at ~$1.8 million at first point of sale, NSW DPI unpublished data) and recreational anglers, the status of the resource remains undefined. This is largely due to a lack of appropriate monitoring programmes. In fact, monitoring has been limited to observing trends in commercial CPUE together with some haphazard measuring of landed catches at ports. Thus, the value of using data from different sources for assessing populations of mud crabs and making appropriate management decisions is unknown.

Sampling of catches by scientific observers on-board commercial fishing vessels is generally considered to be a more reliable way of obtaining fishery-dependent data on retained and discarded catches compared to port-based sampling and fisher-logbooks (Alverson *et al.* 1994, Kennelly 1995). Indeed, observer surveys have been used effectively to gather information for the management of fisheries in Australia (Gray *et al*. 2004, 2005) and around the world (Benoit and Allard 2009).

Nevertheless, on-board sampling programs are often costly, have operational risks and are problematic due to a shortage of space and the reluctance of fishers to take on an observer (Starr 2010).

An alternative to placing observers on board commercial fishing vessels involves fishers collecting their own data. Sampling done by commercial fishers has the potential to provide representative data on catches provided that sampling is appropriately stratified and replicated in space and time (Starr 2010). Indeed, volunteer sampling by commercial fishers has proven to be a cost-effective way of collecting high-quality data from low value fisheries (Starr and Vignaux 1997). The value of these sorts of fishery-dependent sampling programs can, however, be severely limited if co-operation by commercial operators is on a voluntary basis (Macbeth *et al*. 2009).

Spatial and temporal closures of fishing grounds, output controls (e.g. individual transferable quotas, total allowable catches and trip limits), restrictions on the size and number of pots, and differences in the design and deployment of gear among fishers potentially can lead to data that are biased, inaccurate and imprecise (Scheirer *et al.* 2004). These well-known problems of fishery-dependent data means that fishery-independent surveys are often required to corroborate stock assessments (Sigler and Lunsford 2001; Kennelly and Scandol 2002). Fishery-independent data are often preferred over fishery-dependent data because: (i) sampling is randomized rather then being concentrated where catches or economic yields are maximised; (ii) sampling gear and methodologies remain consistent over time; (iii) data can be collected on other species not usually retained in commercial and recreational fisheries; (iv) potentially, they provide more representative data on the size range of populations, rather than the commercially exploited component; and (v) there is no reliance on fishers reporting catches and effort accurately (Rotherham *et al*. 2007). Despite these benefits, fishery-independent surveys can be prohibitively costly and are often limited by the scale of sampling (Cotter and Pilling 2007). What's more, fishery-independent surveys that use inappropriate sampling gears and strategies can also provide data that are biased, inaccurate and imprecise (Rotherham *et al*. 2007).

The aim of this study was to evaluate the effectiveness of fishery-dependent and fishery-independent sampling programs in providing data for assessing and managing populations of *S. serrata* in NSW. This was done by comparing data collected from a standardised fishery-independent survey with two sources of fishery-dependent data: (1) an on-board observer survey of commercial catches; and (ii) fisher-logbook data. We ran the experiment across two estuaries during the main 4-month summer peak in reported commercial fishing activity (i.e. effort and production) for mud crabs in eastern

Australia. Results of the study are used in recommending an appropriate source of data for improved assessment and management of mud crabs in the region.

Materials and methods

Study locations and mud crab fishery

This study was done in the Richmond River (RR) (28º52'S, 153º34"E) and the Macleay River (MR) (30º52'S, 153º01'E), in New South Wales (NSW, south-eastern Australia). Both rivers are large, complex, barrier estuaries with single tidal openings to the sea constricted by wave-deposited beach sand and flood-tidal deltas. The water surface area is 19 km^2 in RR and 18 km^2 in MR (Roy *et al.* 2001).

Both the RR and MR support valuable commercial fisheries for *S. serrata* with 10 yr average landings of 4.8 t and 9.8 t , respectively (NSW DPI unpublished data). The commercial pot fishery for *S. serrata* in NSW is managed by estuary-specific spatial closures, gear restrictions and a minimum legal size for retained crabs (i.e. Carapace Length (CL) ≥85 mm). For example, pots must not exceed 1.2 m in length; 1 m in width and 0.5 m in depth; or if circular, 1.6 m in diameter at the top or bottom of the pot; have a mesh size > 50 mm; and have not more than four entrance funnels, none of which can be on the top of the pot.

Fishers operating in the RR and MR are permitted to set a maximum of 10 pots per day to catch *S. serrata*. Pots are typically set for 24 to 48 h, after which they are lifted, emptied of catch, rebaited, and re-set on the same or nearby grounds. All ovigerous females and sub-legal sized *S. serrata* along with all other bycatch must be immediately released. Recreational fishers are permitted to use one pot and five hoop nets to capture a maximum of five legal-sized (i.e. CL ≥85 mm) individuals of *S. serrata* per person.

Sampling design and methods

Observer survey

In each river, a scientific observer accompanied a single commercial fisher on four randomly selected fishing trips per month from February to May 2012. For each individual pot lift, the number and size

(CL measured as the distance from the notch between the most protruding frontal teeth to the centre of the posterior margin of the carapace) of retained and discarded male and female crabs were recorded by the observer.

Fisher-logbook

Two commercial fishers in each estuary were selected to incorporate a catch sampling logbook into their daily fishing operations. Each participating fisher labelled three pots with a uniquely marked scientific research tag (i.e. pot 1, 2 and 3). Fishers were instructed to intersperse these pots among their other pots and operate them in the usual manner, i.e. pots should have been deployed in areas normally fished and not placed in areas expected to catch atypical numbers or sizes of crabs. For five fishing days per week from February to May 2012 (total of 80 days per fisher), fishers were required to record the sex and size (CL) of all *S. serrata* (legal and undersize) caught in research trap number one. If research trap one was lost, or, if the fisher suspected that it had been tampered with, the catch from research trap two or three was measured. Therefore, the designated pot represented 10% of each fishers daily fishing effort and 7 -10% of their weekly effort.

Fishery-independent survey

The design of sampling in each estuary included two spatial scales: two sites (1 km apart) nested in each of two zones (> 2 km apart). Both commercial and recreational fishing are permitted at each site and zone in MR. In the RR, however, only recreational fishing is permitted in zone 1. Each site was sampled for two consecutive days in each of two randomly selected weeks each month from February to May 2012.

At sunset on each day of sampling, 10 circular, collapsible knotted-net pots fitted with two funnel-shaped entrances were randomly deployed at each site at depths ranging from 0.5 to 4.5 m. Each replicate pot was separated by at least 100 m to ensure independence of fishing (Williams and Hill, 1982); and soaked for 12 h overnight (see Johnson *et al.* submitted). Each replicate pot was baited with approximately 1.2 kg of sea mullet (*Mugil cephalus)*. Approximately 50% of the bait was skewered on a wire and attached to the center of the pot with the remainder placed in a plastic wire-mesh bag (200 x 300 mm – 20 x 20 mm mesh) secured to the base of the pot. For each individual pot lift, the size (CL), sex and stage of maturity (determined by the shape and attachment of the abdominal flap) of all individuals of *S. serrata* were recorded.

Analysis of data

To enable comparisons between fishery-independent and -dependent data, CPUE was standardized to give both sets of data the same scale (amount of variation around the mean). CPUE was standardized following the standard method of subtracting mean CPUE (calculated from the sampling trip CPUE) from each sampling trip CPUE and dividing that number by the standard deviation (calculated from sampling trip CPUES; see Scheirer *et al*. 2004 for details).

Relationships between CPUE from independent (X variable) and observer surveys (Y variable) were examined using regression analysis. The measures of CPUE used in regression analysis were total numbers of crabs as well as numbers of legal and undersize male and female crabs. The comparison of CPUE from independent and observer surveys was done on daily means (n = 16). The coefficient of variation (CV) was used to compare estimates of relative variation in mean CPUE estimates for each survey type and independent survey zone.

Sex ratios were compared between survey types and zones (Independent survey only) using a chi-square goodness of fit test for departure from 1:1 (Zar 1984). Tests were performed on legal (≥ 85 mm CL) and undersize (< 85 mm CL) crabs, separately. Differences in size-frequency distributions between types of survey (pooled across months and between months and zones, RR only) were examined visually and tested using Kolmogorov-Smirnov (K-S) tests (Sokal and Rohlf, 1995).

Results

General

For all three types of data (i.e. independent, observer and logbook), larger numbers of *S. serrata* were caught in the RR (n = 2,693) than MR (n = 1,235). By contrast, there were larger proportions of legal-sized crabs caught in MR than RR for all but one survey (i.e. independent survey in Zone 1 of RR, Table 1). This pattern was consistent not only for the total numbers of *S. serrata*, but also for numbers of males and females (Table 1).

In the MR, commercial catch rates (i.e. fisher-logbook and observer survey) of male and female *S. serrata* were larger than the independent surveys. In fact, except for the mean number of males in RR, the logbook catch rates were larger than the observer- and fishery-independent surveys across both rivers.

CPUE of *S. serrata* and precision (CV) of means

Regression analysis of daily CPUE data from the fishery-independent and observer surveys revealed no significant regression relationship ($p > 0.05$) between any of the six measures of CPUE tested in the MR (Table 2). In contrast, mean daily catch rates of total crabs and undersize male crabs from the RR independent and observer surveys showed significant regression relationship ($p < 0.05$). Standardised monthly CPUE of all variables from the independent and observer surveys for the RR and MR are shown in Figure 2 and Figure 3, respectively. The magnitude and pattern of variation between CPUE of the two survey designs differed between rivers and measures of CPUE. For example, legal female CPUE showed less variation from the mean than legal male CPUE in the RR (Fig. 2). While variation above or below the mean was consistent between the two survey designs for total male, total and undersize female crabs in the RR (Fig. 2b, e, g). However, no such patterns were observed for the MR for any of the six measures of CPUE (Fig. 3a-f).

Estimates of the level of precision of mean CPUE for total, male and female *S. serrata* varied between rivers, survey types and sexes (Table 3). For example, estimates of mean CPUE of total and male *S. serrata* were generally more precise (smaller CVs, Table 3) in RR than MR. But for female *S. serrata*; estimates of mean CPUE were more precise in MR than RR. Further, CVs for fisher-logbook and observer data were generally smaller (more precise) than estimates of CPUE from the independent surveys (Table 3).

Size-frequency distributions of *S. serrata*

KS- tests detected significant differences in size-frequency distributions of male and female *S. serrata* (pooled across months) between fishery-independent and –dependent data in both rivers ($p < 0.01$, Fig. 4a- f, Table 4). In the RR, larger proportions of undersize male *S. serrata* were caught in the commercial pots (observer and logbook data). But this pattern was not consistent in the MR; with larger proportions of undersized *S. serrata* (males and females) caught in the research pots. What’s more, no significant differences ($p > 0.05$) were detected for the size frequencies of female *S. serrata* between independent surveys and fisher logbooks in the MR (Table 4).

In the RR, the mean sizes (CL) of male (90.94 ± 0.50 mm) and female *S. serrata* (92.49 ± 0.71 mm) were greater in the independent survey than observer survey (male; 85.85 ± 0.51 mm; female; 87.16 ± 0.16 mm). By contrast, the opposite pattern was observed for mean sizes of male and female *S. serrata* in the MR.

Sex ratios of *S. serrata*

In the RR, chi-square tests revealed significant differences in sex ratio for both legal and undersized *S. serrata* caught in the independent and observer surveys. In all cases, larger proportions of male crabs were caught than female crabs. For the fisher-logbook data, however, there were no significant differences in sex ratio for either legal or undersized *S. serrata*. In the MR, the only significant difference detected in sex ratio was for legal-sized *S. serrata* in Zone 2 of the independent survey. In this case, a larger proportion of female *S. serrata* was observed.

Discussion

Our findings illustrate that when it comes to monitoring populations of *S. serrata*, different sources of data show different patterns of abundance. Although catch rates in commercial pots (observer and fisher-logbook) were generally larger and more precise than in the research pots, this was an expected result given that commercial operators fish in a targeted manner, rather than simply deploying traps at random. Nevertheless, our results highlight the danger of relying on commercial catch data alone for monitoring patterns of abundance of *S. serrata* in NSW. Indeed, hyperstability of commercial CPUE has been observed prior to the collapse of many fisheries around the world (Rose and Kulka 1999, Harley *et al*. 2001).

Another important result was that although size compositions of male and female *S. serrata* were significantly different between independent and observer surveys in both rivers, differences were inconsistent between MR and RR. This result may be explained by differences in pot designs and the sizes of mesh used in each type of survey and estuary. For example, commercial fishers in the MR predominately use pots constructed of 50 x 50 mm square-shaped welded-mesh wire or 50 mm hexagonal wire attached to a solid D-shaped frame. By comparison, commercial fishers in the RR predominately use round net-covered pots similar to those used in the independent survey.

Alternatively, a lack of consistent differences in size-frequency distributions of male and female *S. serrata* among survey types could be due to differences in the structure of the populations sampled. The mean sizes of male and female *S. serrata* in Zone 1 of the independent survey in RR were larger than in Zone 2 and the observer and logbook surveys. Further, legal-sized male *S. serrata* comprised more than 70% of the total male catch in Zone 1 (compared to less than 50% in zone 2). Similarly, legal-sized female *S. serrata* comprised more than 75% of the total catch of females in Zone 1, while in Zone 2, < 45% of females measured in each type of survey type were larger than the minimum legal size of 85 mm CL.

The skewed size distributions of male and female *S. serrata* observed in Zone 1 of the RR may be the result of different levels of fishing pressure as no commercial fishing is permitted. Differences in the mean size of *S. serrata* and other crustaceans have been reported in areas subject to different levels of exploitation, including marine reserves (Kelly *et al*. 2000, Rowe 2002, Pillans *et al*. 2005). For example, males of *S. serrata* caught in marine reserves in Queensland were 10% larger than those caught in adjacent fished areas, while the average size of females of *S. serrata* protected from capture in all areas did not differ between reserve and non-reserve sites (Pillans *et al.* 2005).

Our results indicate that sex ratios of *S. serrata* caught in baited pots are influenced not only by the estuary, but also areas within the estuary sampled. Larger proportions of male crabs were recorded for all survey types and zones in the RR. By contrast, greater proportions of females were caught in independent survey Zone 2 and observer surveys in the MR. Nevertheless, where sex-related differences in catch rates of male and female crabs were reported, results of both independent and observer surveys were similar in both estuaries. What's more, the sex ratio of catches from fisher-logbooks showed no significant departure from 1:1. The observed difference in sex ratio of catches between survey designs in the RR may be a result of complex interactions between abiotic factors (e.g. environmental conditions) and biological processes (e.g. reproductive condition, feeding rhythms, intraspecific competition and attraction) (Williams and Hill 1982, Skinner and Hill 1986). For example, Skinner and Hill (1986) reported that reproductive behaviour has a strong influence on the catch composition of spanner crabs (*Ranina ranina*). The collection of catch-per-pot data daily (i.e. fisher-collected data) may reduce the effects of biological (i.e. reproductive condition) and environmental (i.e. temperature) factors on the sex composition of pot catches, that are directly affected by emergence and feeding activity (Skinner and Hill 1986). This implies that the sex ratio of catches from previous studies that chose to sample on single or consecutive days seasonally (Pillans *et al*. 2005) or at predetermined tidal cycles (Moser *et al* . 2005, Walton *et al*. 2006) may be potentially confounded by variation at smaller time –scales. Further experiments are necessary to examine the effects of biological and environmental factors on sex ratio and catch rates of *S. serrata* in baited pots.

Legal and undersize males of *S. serrata* dominated catches in the independent survey in Zone 1 of the RR, with twice as many male crabs caught. This male-biased sex ratio in Zone 1 (a recreational fishing haven) was not expected. Differences in the sex ratio of catches between zones and estuaries may be a result of a combination of factors including: (i) migration of female crabs out of the estuary to spawn (Hill 1975); (ii) increased habitat protection in areas partially protected from fishing pressure (Klein *et al*. 2013); (iii) level of exploitation (Pillans *et al*. 2005, Kleczkowski *et al*. 2008); (iv) differences in catchability between sexes and sizes of crab (Williams and Hill 1982); and (v) differences in environmental variables such as temperature and salinity (Potter *et al*. 1983, Rugolo *et al*. 1998). Such boundaries in distribution may be temporarily flexible, and dependent on seasonal rainfall patterns and temperature fluctuations (Rugolo *et al.* 1998).

The commercial harvest of *S. serrata* in NSW is currently managed using a series of input controls that are assessed on CPUE data reported by commercial fishers. Given the differences in patterns of abundance observed among the different sources of data in our study, relying on only once source of abundance data is not sensible. Ultimately, however, the choice of survey type(s) to assess the status of *S serrata* populations will depend on the future management objectives of the fishery and sources of available funding. To ensure the sustainable harvest of marine resources, there is a need to develop appropriate indicators, methodologies, and decision making rules for most fisheries (O'Neill *et al.* 2010). While our findings indicate that observer, fisher- logbook or independent surveys all provide a similar picture in monitoring the structure of populations of *S. serrata*, fishery-independent surveys may be required in estuaries closed to commercial fishing, such as recreational fishing havens (i.e. Bellinger and Hastings Rivers, Deep Creek) and marine parks (i.e. Wooli and Corindi Rivers) to fully assess the status of NSW *S. serrata* resource.

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Table 1: Summary of numbers (n) and proportion of legal sized (≥ 85mm CL) total, male and female crabs caught by survey type and zone (Independent survey only).

	Total		Male		Female	
	n	% legal	n	% legal	n	% legal
Richmond River						
Independent survey						
Zone 1	760	75.92	516	76.55	244	74.59
Zone 2	490	46.33	295	48.14	195	43.59
Observer	1040	45.96	603	46.77	437	44.86
Fisher-logbook	403	47.89	211	51.18	192	44.27
Macleay River						
Independent survey						
Zone 1	276	51.09	150	52.76	126	49.21
Zone 2	356	66.86	156	62.82	200	66.86
Observer	497	68.21	233	68.67	264	67.80
Fisher-logbook	106	70.75	49	73.47	57	68.42

Table 2. Independent survey versus observer survey daily CPUE regression results

		Male			Female		
	Total crabs	Total	Legal	Undersize	Total	Legal	Undersize
Richmond River							
Slope	0.627	0.421	0.702	0.657	0.506	-0.271	0.504
P value	0.025	0.075	0.096	0.037	0.108	0.591	0.264
Adjusted r^2	0.262	0.152	0.127	0.223	0.115	-0.049	0.023
N (number of days)	16	16	16	16	16	16	16
Macleay River							
Slope	-0.052	0.004	0.011	-0.058	-0.004	0.033	-0.034
P value	0.435	0.971	0.959	0.648	0.931	0.829	0.615
Adjusted r^2	0.054	-0.071	-0.071	-0.055	-0.071	-0.068	-0.052
N (number of days)	16	16	16	16	16	16	16

Table 3: Summary of catch statistics for Richmond River and Macleay River. CPUE is number of crabs per pot per night (n = 16) and s.d. is standard deviation. The coefficient of variation (CV = 100 x s.d./mean) represents the relative variation in mean CPUE for each survey type.

	Total			Male			Female		
	Mean CPUE	s.d.	CV	Mean CPUE	s.d.	CV	Mean CPUE	s.d.	CV
Richmond River									
Independent survey									
Combined	2.36	1.63	68.94	1.5	1.31	87.33	0.86	1.07	124.39
Zone 1	2.77	1.73	62.30	1.89	1.42	75.36	0.88	1.06	120.51
Zone 2	1.94	1.41	72.35	1.11	1.06	95.10	0.83	1.07	128.86
Observer survey	2.00	1.42	70.57	1.16	1.06	90.76	0.84	1.11	131.71
Fisher-Logbook	2.30	1.61	70.06	1.21	1.09	90.37	1.10	1.30	118.35
Macleay River									
Independent survey									
Combined	0.99	1.05	106.74	0.47	0.70	147.48	0.51	0.76	149.78
Zone 1	0.86	1.00	115.75	0.47	0.69	148.06	0.39	0.70	176.68
Zone 2	1.11	1.09	98.36	0.49	0.72	147.06	0.63	0.81	129.51
Observer survey	1.53	1.37	89.05	0.72	0.93	129.80	0.81	0.98	120.02
Fisher-Logbook	1.74	1.36	78.56	0.80	1.01	126.18	0.93	0.95	101.29

Table 4: Results of Kolomogorov-Smirnov (K-S) tests examining differences on size-frequency distributions of male and female *S. serrata* caught in the RR and MR between survey types pooled across months (* $p<0.05$; ** $p<0.01$).

	Male	Female
Richmond River		
Independent v Observer	**	**
Independent v fisher-logbook	**	ns
Observer v fisher-logbook	ns	**
Macleay River		
Independent v Observer	**	**
Independent v fisher-logbook	ns	ns
Observer v fisher-logbook	ns	ns

Table 5: Results of chi-square goodness of fit test for departure from 1:1 and sex ratio (M:F) for legal and undersize crabs for independent survey, observer survey and fisher-collected data. (P<0.05*, P< 0.01**, P<0.001***)

		Sex ratio (M:F)	Chi-Square	P
Richmond River				
Independent survey				
Zone 1	legal	2.17:1	78.63	***
	undersize	1.95:1	19.02	***
Zone 2	legal	1.67:1	14.31	***
	undersize	1.39:1	7.03	***
Observer	legal	1.4:1	15.47	***
	undersize	1.3:1	11.39	***
Fisher-collected	legal	1.3:1	2.97	ns
	undersize	0.95:1	0.12	ns
Macleay River				
Independent survey				
Zone 1	legal	1.27:1	2.05	ns
	undersize	1.11:1	0.36	ns
Zone 2	legal	0.73:1	5.59	*
	undersize	0.88:1	0.52	ns
Observer	legal	0.89:1	0.30	ns
	undersize	0.86:1	0.34	ns
Fisher-collected	legal	0.92:1	0.12	ns
	undersize	0.72:1	0.81	ns

Figure 1. Mean number (+ 1 SE) of total, legal and undersize *S. serrata* caught per pot lift in the RR; (a) male, (b) female and MR; (c) male and (d) female from independent survey (zone 1 and 2), observer survey and fisher-logbooks.

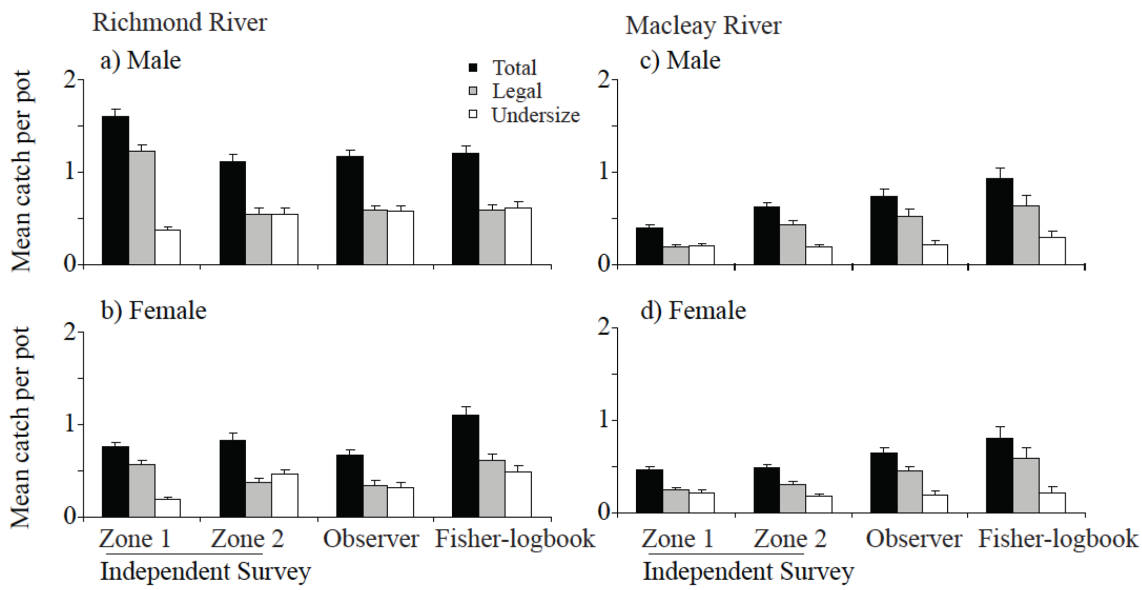


Figure 2. Standardised number of total (a), male; (b) total (c) legal, (d) undersize and female; (e) total, (f) legal and (g) undersize *S. serrata* per pot lift in the RR independent and observer surveys.

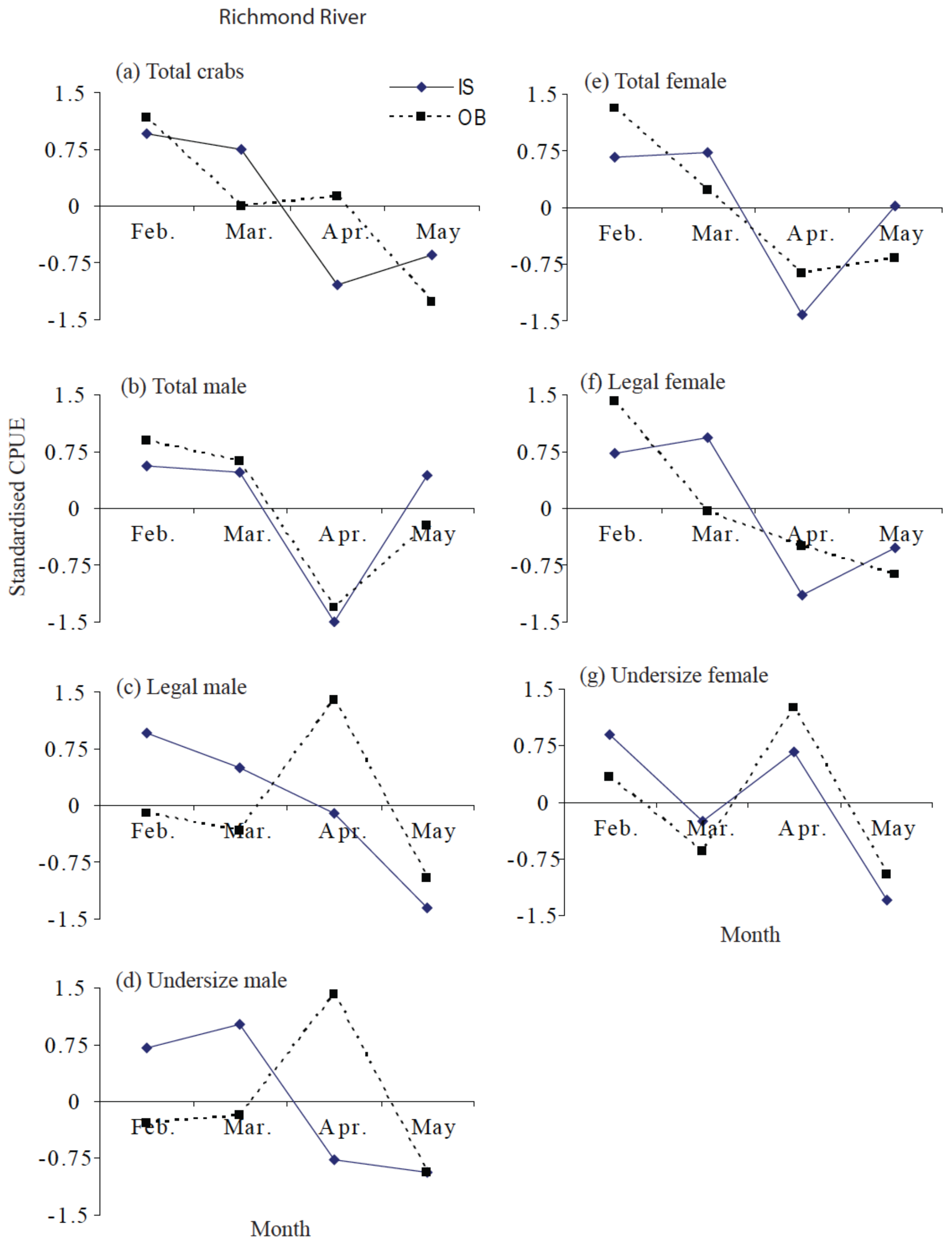


Figure 3. Standardised number of total (a), male; (b) total (c) legal, (d) undersize and female; (e) total, (f) legal and (g) undersize *S. serrata* per pot lift in the MR independent and observer surveys.

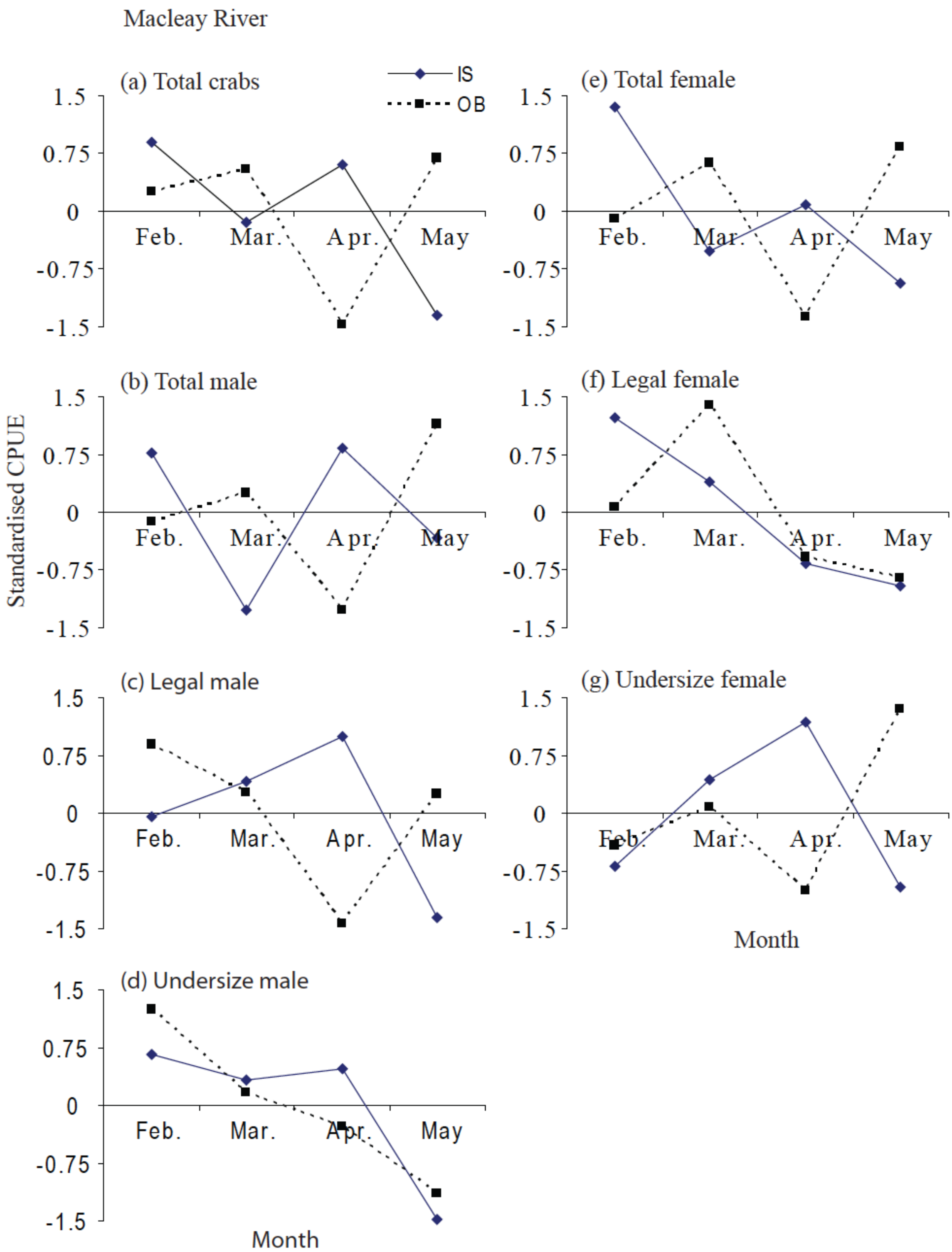
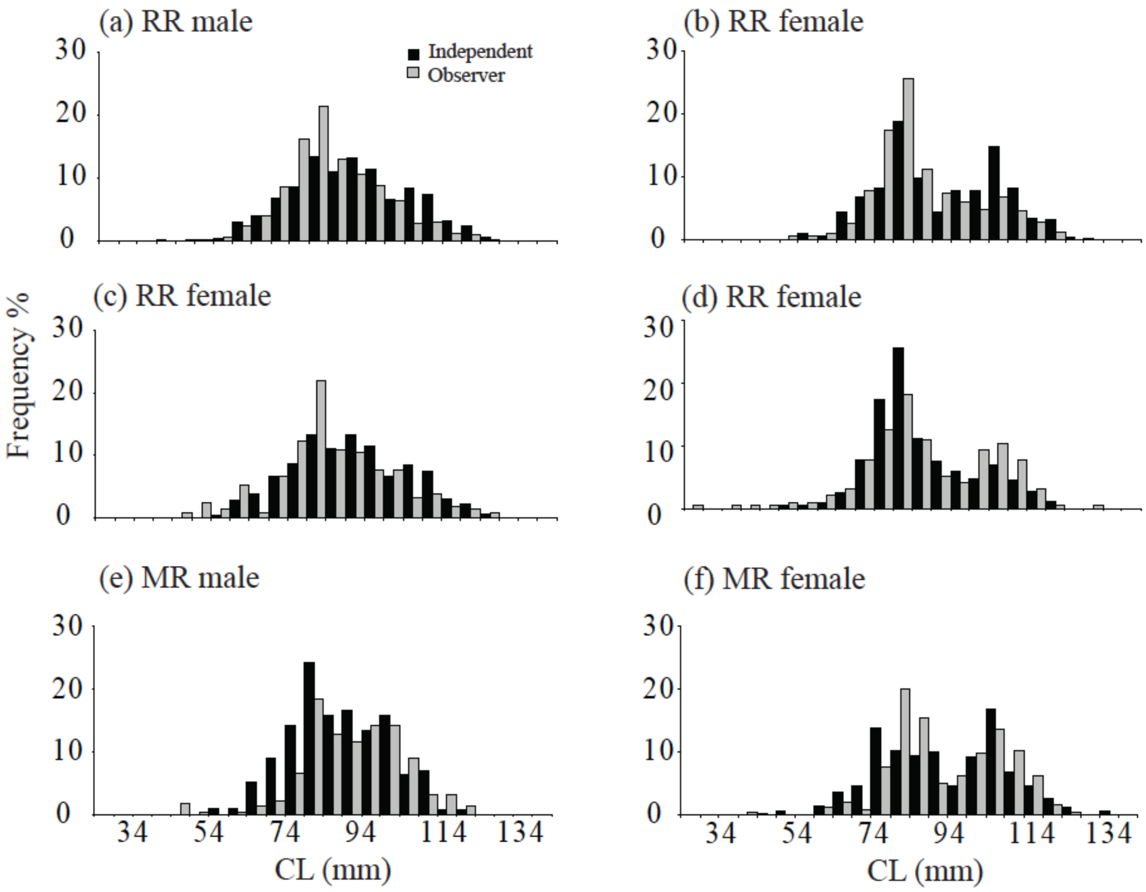


Figure 4. Size-frequency distribution of male; (a) independent and observer survey, female; (b) independent and observer surveys, (c) independent and fisher-logbook, (d) observer and fisher-logbook from the RR and male; (e) and female (f) *S. serrata* from independent and observer survey in the MR, pooled across days, weeks and months.



Appendix 6. Experiment 4: Utility of fishery-independent sampling strategy to assess the fisheries resources in estuaries with different management regimes

Introduction

The objective of this experiment was to examine whether the relative abundances and demographic characteristics of key fish species differed between estuaries that have different management regimes; namely recreationally only fished estuaries (i.e. recreational fishing havens) compared to recreationally and commercially fished estuaries. We hypothesized that the relative abundances and mean lengths and ages of harvested species would be greater, and that their rates of total and fishing mortality would be lower, in havens than non-havens due to the potential reduced fishing pressures. We specifically tested these hypotheses by using the fishery-independent sampling strategy to sample fish populations in two havens and two non-havens. This experiment directly contributed to Project Objectives 2 and 3.

Materials and Methods

Study estuaries

The two recreational fishing havens (hereafter termed havens) were Lake Macquarie (LM) and St Georges Basin (SGB) whereas the two non-havens were Wallis Lake (WL) and Tuggerah Lake (TL). These estuaries are permanently open, shallow (mean depth < 5 m), micro-tidal barrier estuaries (i.e. coastal lagoons, Roy et al. 2001) that have minimal riverine input and constricted entrances to the sea. The typical tidal range in the entrance channel of each lagoon is ~1.5 m, whereas within the central basin of each lagoon it is ~ 0.10 to 0.30 m. Typically, the shallow margins (50 – 100 m width) of the lagoons are covered by a mosaic of vegetated (predominantly *Zostera capricorni*) and bare substrata, whereas the deeper (> 2 m) central basins consist mostly of bare sand and mud substrata.

Sampling

Each estuary was sampled with the multimesh gillnet and beam trawl in each of two periods (January–March/April and April/May–July) in 2008, 2009 and 2010. The two sampling periods were approximately 3 months long and separated by at least 4 weeks in each estuary. The order in which each estuary was sampled and the dates that sampling began (within each period in each year) for each gear type was selected at random. Previous experiments demonstrated that variation in abundances of fish at temporal scales within 3-month periods (e.g. days, weeks and months) was small compared to variation between 3-month periods and at small spatial scales (Rotherham et al. 2011).

Each multimesh gillnet consisted of seven 20-m panels of different sizes of stretched mesh (36, 44, 54, 63, 76, 89 and 102 mm) connected together in a random order (Gray et al. 2009). Adjacent panels were separated by 5 m of rope to minimize potential effects of the smaller-mesh panels ‘leading’ larger fish to adjacent panels. The 36- and 44-mm panels were made from monofilament netting with twine diameter of 0.15 mm; the remaining panels had multifilament netting with a twine diameter of 0.4 mm. All panels were 2 m deep and had a hanging ratio of 0.5. In each estuary, there were four randomly selected sites separated by one to several kilometers (Gray et al., 2009). Each gillnet was set in depths < 2 m and in each sampling period in each year, three replicate gillnets (each separated by 50–100 m) were sampled at each site. It took one night to sample a site in LM and SGB, whereas two sites were sampled each night in WL and TL. The particular week in which sampling commenced within each estuary and sampling period, was selected at random. On each night of sampling, each gillnet was bottom-set at dusk, soaked for 1 h and then retrieved (Rotherham et al., 2006). Catches in each net were identified and counted and specific species of fish were measured for either fork (FL) or total length (TL) to the nearest 0.5 cm.

The beam trawl was 3-m wide and configured with 41-mm diamond-shaped mesh in the body and 20-mm mesh hung on the bar (i.e. square-shaped) in the codend (Rotherham et al. 2008a). In each sampling period in each year, nine non-overlapping replicates of the beam trawl (5 min duration at speeds of about 1.2 m s^{-1}) were sampled at night at each of four sites (separated by one to several km) within each estuary (Rotherham et al. 2008a,b). Sampling sites were randomly selected over relatively flat, predominantly unvegetated sediment interspersed with (or adjacent to) patches of seagrass in waters 1 to 6 m deep. It took two consecutive nights to sample all four sites within an estuary and the two nights of sampling were selected at random from within each sampling period. After each replicate tow was completed, the contents of the net were emptied onto a tray and sorted by species, counted and the lengths of key species of fish measured.

Specimens of several key species, including *Acanthopagrus australis* (Sparidae), *Platycephalus fuscus* (Platycephalidae), *Girella tricuspidata* (Girellidae), *Mugil cephalus* (Mugilidae) and *Sillago ciliata* (Sillaginidae), *Gerres subfasciatus* (Gerreidae), *Liza argentea* (Mugilidae) and *Pomatomus saltatrix* (Pomatomidae) were kept for age determination and subsequent demographic analyses. All captured individuals of each of these species were kept unless more than 100 individuals of a species were captured and then a random subsample of these fish were kept for processing. The sagittal otoliths of each fish were removed and information on their length and sex was obtained in the laboratory.

Relative abundance

Four factor permutational analyses of variance (PERMANOVA; Anderson 2001; Anderson et al. 2008) were used to test if the numbers of key species captured in each gear type differed according to the factors: Estuary type (i.e. haven vs non-haven; Fixed) and Estuary (nested in estuary type; Random), Year (Fixed) and Period (nested in Year, Random). Each univariate analysis was based on the Euclidean distance measure and Type III (partial) sums of squares were calculated using 9999 unrestricted permutations of the raw data. Because we were only interested in examining the scale of estuary, we did not test for variations among individual sites within each estuary and so the data from each replicate gillnet or beam trawl from all sites within each estuary were included as replicates for the factor estuary in each analysis.

Age determination

Counts of completed opaque zones on sectioned sagittal otoliths were used to determine the age of each species of fish. This ageing technique has been validated for *A. australis*, *P. fuscus*, *S. ciliata*, *S. maculata*, *G. tricuspidata*, *L. argentea*, *G. subfasciatus* and *M. cephalus* (Gray et al. 2000, Gray et al. 2002, Smith and Deguara 2002, Kendall and Gray 2009a,b, Gray et al. 2010, Ochwada-Doyle et al. 2014). Otoliths were embedded in blocks of clear cast polyester resin and three to four transverse serial sections (approx. 0.7 mm) were cut through the otolith core using a diamond saw. The resulting sections were mounted on a microscope slide and polished on a polishing wheel fitted with 500-grit silicon carbide paper till the opaque zones on sections became clear. Sections were viewed using a reflected light against a black background under a compound microscope fitted with a digital camera. The distance between each successive opaque zone was measured using Image J software and digital images of each section were captured. Sections were interpreted without the knowledge of the length or sex of the fish, or the date and location of capture. Ten percent of sections were read twice without the knowledge of the 1st interpretation to obtain a coefficient of variation (CV) of successive reads (Kimura and Lyons 1991).

Length and age compositions

For each estuary, the length and age compositions of samples of each species pooled across all sampling years were determined separately for the gillnet and beam trawl. Because all sampled *P. fuscus*, *S. ciliata* and *G. tricuspidata* were aged, the entire sample for each gear type in each estuary simply became the age composition. Since a subsample of *A. australis* and *M. cephalus* were aged, a separate age-length key for each method and each estuary was applied to the corresponding length composition data to determine the age composition of samples.

Mortality and exploitation

The instantaneous rate of total mortality (*Z*) was determined for each species in each estuary using the age-based catch curve method (Ricker, 1975). This was done for age compositions of gillnet catches combined across the 3 years. The beam trawl data was not used as this gear primarily sampled juvenile fish. The natural logarithm of the proportion of fish in each age class (Nt) was plotted against their corresponding age class (t) and a linear regression was fitted and *Z* determined as the slope of the descending regression. For these analyses, we assumed that the most abundant age class in each sample was fully recruited to the sampled population. Standard error (SE) and R^2 values were calculated for each regression. These analyses assumed that the selectivity, recruitment and growth of fish were constant across years and growth was asymptotic.

The instantaneous rate of natural mortality (*M*) was estimated using the method of Hoenig (1983) based on the published (previous 10 years) maximum age of each species; 22 years for *A. australis* (Gray et al 2000) and *G. tricuspidata* (Gray et al 2010), 16 years for *S. ciliata* (Ochwada-Doyle et al. 2014) and *P. fuscus* (Gray and Barnes 2008), and 12 years for *M. cephalus* (Smith and Deguara 2002). Fishing mortality (*F*) was determined by subtracting *M* from each estimate of *Z*, which also provided a corresponding rate of exploitation (*E*) ($E = F/Z$).

Results

The majority of the key species examined, except *A. australis*, *S. maculata*, *G. subfasciatus*, *R. sarba* and *P. auratus* were caught in greater total numbers in the gillnet compared to the beam trawl across most estuaries (Table 1). Both *A. australis* and *G. subfasciatus* were caught in high numbers in both gear types. The mugilids, *M. cephalus*, *L. argentea*, *P. georgii* and *M. elongatus* were caught exclusively in the gillnet, but there was a contrasting pattern of capture for the sillaginids, with *S. ciliata* primarily being captured in the gillnet and *S. maculata* in the beam trawl. The total numbers of each species captured varied greatly among estuaries, with some species, including *P. georgii* and *M. elongatus*, being caught primarily in only one or two estuaries.

Relative abundance

There was a significant effect of estuary type on the numbers of *A. australis*, *R. sarba* and *G. subfasciatus* sampled in the beam trawl (Table 2, Fig 1) and *G. subfasciatus* sampled in the gillnet (Table 2, Fig 2). The three species sampled in the beam trawl occurred in significantly greater numbers in the non-havens, whereas the opposite was evident for *G. subfasciatus* sampled in the gillnet. The relative abundance of several species fluctuated significantly among individual estuaries and sampling periods (significant high order interactions in PERMANOVA), but there was no significant influence of estuary type (i.e. haven or non-haven) (Table 2). Most factors returned non-significant effects for most species.

Length and age compositions

The mean lengths and ages of *A. australis*, *P. fuscus*, *G. tricuspidata*, *M. cephalus* and *S. ciliata* captured in the gillnet were generally greater in the havens than the non-havens (Figs. 3 to 7). This was due to greater proportions of larger and older fish being captured, with the mean age of each species being generally more than one year greater in the havens than in the non-havens. This was also evident for *A. australis* captured in the beam trawl (Fig. 8). The mean lengths of *L. argentea*, *P. saltatrix* sampled in the gillnet and *G. subfasciatus* sampled in the gillnet and beam trawl were also generally greater in the havens than non-havens (Figs. 9 and 10), but this was not evident for *R. sarba* or *S. maculata* sampled in the beam trawl (Fig. 11). In contrast to all other species, the mean length of *S. maculata* was greater in the non-havens than in the havens (Fig. 11).

Mortality and exploitation

The catch curve based estimates of total mortality were lower in the havens than the non-havens for *A. australis*, *P. fuscus*, *S. ciliata* and *M. cephalus* but not *G. tricuspidata*, which displayed no consistent differences between estuary type (Table 3). The determined estimates of *F* and *E* were also lower for *A. australis*, *P. fuscus* and *M. cephalus* in the havens than the non-havens. Estimates of *Z* for *S. ciliata* in the two havens were lower than that of natural mortality (*M*), which provided non-sensible (negative) estimates of *F* and *E*. The age range used for determining each catch curve differed among estuaries for each species (Table 3).

Discussion

Sampling

Compared to the beam trawl, the multimesh gillnet generally caught a greater number and wider length range of the key harvested species examined. The data from this experiment and that reported in Appendix 1 clearly demonstrate that for resource assessment purposes the gillnet is a more suitable sampling gear than the beam trawl for providing demographic data of the general suite of key harvested fish species in estuaries of southeastern Australia.

Nevertheless, the beam trawl was most effective at sampling fish that do not attain large lengths such as *G. subfasciatus* and the small juveniles of some key species, such as the Sparidae *A. australis*, *R. sarba* and *P. auratus*. It did, however, sample a wider length range of *S. maculata* than the gillnet. The beam trawl could potentially be useful in providing indices of recruitment of these key species prior to them entering the fishery. Trawl-based surveys are often used to provide such information (Gunderson 1993), which can be used in setting future catch levels in a fishery. Before embarking on any such pre-recruit survey, however, much other vital information such as the identification of the most suitable habitats and times of the year to do sampling for each species of interest are required for determining optimal sampling designs (Rotherham et al. 2008a).

Annual samples of most species taken in both gears were relatively low for population assessment purposes so these data had to be pooled across the 3 years for each estuary. This limited the robustness of the experiment, as one of the original objectives was to investigate the temporal consistency of comparisons of populations within and between estuary types, similar to that previously done across habitats (Gray et al. 2011). Greater sampling effort would be required to obtain sufficient samples to provide robust data for annual assessments of the key species examined here. This could potentially be achieved by using only one gear type and simply transferring the effort directed to the beam trawl to the gillnet. This could effectively double sampling effort of the gillnet as the beam trawl sampling took 2 nights in an estuary. Regardless of the shortfall in providing replicated annual comparisons, the data identified clear differences in demographic characteristics of populations in the two havens and non-havens.

Relative abundances

Several studies have reported greater abundances of fish and other organisms in fully protected areas compared to areas that are open to various forms of fishing (Edgar and Stuart-Smith 2009; Lester et al. 2009), but fewer such trends have been observed between partially and fully fished areas (Sala et al. 2012). Our findings generally concur with the latter as no global pattern across all sampling periods and both sampling gears of greater abundances in the havens was evident for the suite of harvested species examined here. In fact only one species, *G. subfasciatus*, which is not recreationally fished displayed greater abundances in the havens and this was only in the gillnet samples. In contrast, this species along with *A. australis* and *R. sarba* displayed greater abundances in the non-havens in the beam trawl samples. Elsewhere, different species and even life history stages of a species have been found to respond to exploitation changes in different ways. Nevertheless, the six to eight year time period between the initial change in management and this particular study should have been sufficient for any abundance-related responses of fish to be manifest.

The relative abundances of most species examined fluctuated greatly between individual estuaries independent of their management type. Such spatial and temporal fluctuations in fish abundance are in general accordance with observed natural variability and patchiness of fish populations in estuaries and could be due to a plethora of interacting biotic and abiotic factors that were not investigated here (Jones and West 2005; Miller and Skilleter 2006; Gray et al. 2009;

Rotherham et al. 2011). Importantly, unless the effect-sizes of different management regimes are large, such inherent variability in fish abundances will inevitably make it difficult to detect clear differences in abundances of fish (especially mobile species) among fully and partially fished estuaries. Indeed, such variability was probably a major contributing factor to why few differences in the abundances of harvested fish have been detected between partially fished and fully fished zones on coastal reefs (Di Franco et al. 2009; Sala et al. 2012; Curley et al. 2013). Recreationally only fished estuaries may therefore provide little or no detectable benefits in terms of increased abundances of harvested organisms compared to fully (recreationally and commercially) fished estuaries.

Habitat and/or the selectivity of each gear type may have influenced some patterns in fish abundance observed here. For example, greater mean numbers of *A. australis* and *G. subfasciatus* were captured in the beam trawl in the non-havens than in the havens, but no such pattern was evident for these species in the gillnet samples. In fact, the opposite pattern was evident for *G. subfasciatus* sampled using the gillnet with more caught in the havens than the non-havens. Further, *G. subfasciatus* were least abundant in the beam trawl samples in Tuggerah Lake as very few were caught, whereas they were taken in large numbers and displayed the greatest mean abundance in three periods in this particular lake in the gillnet. A smaller length class of *G. subfasciatus* was sampled in the beam trawl compared to the gillnet, suggesting that the contrasting patterns of abundance were driven by the selectivity characteristics of each gear or that the population structure of this species differed between sampling areas and times. Sampling with only one gear type could therefore potentially confound comparisons across estuary management types. This concurs with the outcomes from other studies that caution the use of only one gear type to examine spatial and temporal patterns of occurrence and abundance of aquatic organisms (Watson et al. 2005; Eros et al. 2009; Olin et al. 2009; Rotherham et al. 2012).

Population structure and mortality

The length and age compositions of the majority of the harvested species examined were influenced by estuary management type, with greater proportions of larger and older individuals observed in the havens compared to the non-havens. This is in general accordance with observations of larger fishes inhabiting protected compared to exploited areas elsewhere (Lester and Halpern 2008; Edgar and Stuart-Smith 2009; Lester et al. 2009: Russ and Alcala 2010), but contrasts the situation of no observable differences in the lengths of harvested fish in partially fished and fully fished zones on coastal reefs (Di Franco et al. 2009; Curley et al. 2013). It could be argued that the removal in 2002 of all commercial fishing in the two havens has had localized positive effects on these fish populations by allowing a greater proportion of individuals to survive longer, grow and attain larger lengths and for populations in these estuaries to be comprised of more older individuals. These estuaries had been commercially fished with gillnets, beach-seines and traps for over 100 years and in the five years prior to 2002, an average of approximately 25.9 and 8.8 tonnes of fish (all species combined) were commercially harvested per annum in LM and SGB respectively, with catches primarily containing the key species examined here (Gray 2002, Gray and Kennelly 2003, Gray et al. 2003).

Despite the clear differences in population structure of several harvested fish species between the two estuary types, we do not have comparable demographic data across these estuaries 'before' the management changes occurred in 2002 and so these patterns may have existed even before the estuaries became havens. As an alternative hypothesis, it is plausible that the observed differences among estuaries in the length and age compositions of populations, and in the concomitant rates of mortality, existed before 2002 and were due to intrinsic estuary-specific differences in demographic processes of these fish populations (Gray et al. 2012; Gray 2008). Some fishery-dependent data exists which show differences in the length and age compositions and rates of mortality for some key species across some of these estuaries prior to them being declared havens (Gray et al. 2000, 2002, 2010). However, any comparisons among estuaries and time are potentially confounded due to differences in the types and mesh sizes of the commercial fishing gears and in the choices of either retaining or discarding some species, such as *G. tricuspidata* and *G. subfasciatus*, by individual fishers and commercial fishers collectively across estuaries.

Regardless of this, surveys of recreational anglers catches previously identified that the mean lengths of the retained catches of some key species, including *A. australis*, *P. fuscus*, *S. ciliata* and *P. saltatrix*, increased in two estuaries (Lake Macquarie and Tuross Lake) from before (1999-2000) to soon after their inception as havens (2003-2004) (Steffe et al. 2005a,b). However, no control (non-haven) estuaries were concurrently examined in those particular studies so it was not known whether such changes were specific to the havens examined or were more general across a number of estuaries (havens and non-havens) throughout eastern Australia at that time. Regardless, these results taken collectively with the results from the current study add strong support to the hypothesis that recreationally only fished estuaries provide greater protection from exploitation to fish populations than estuaries that have combined recreational and commercial fishing, allowing for greater proportions of fish to attain larger lengths. The data in Steffe et al. (2005a,b) indicate that such beneficial effects occurred relatively fast, within 1.5 to 2.5 years after the closure of commercial fishing, for some species. We acknowledge that there can be considerable intra- and inter-specific variability in the response times of fish populations following management intervention depending on the life history and population characteristics of organisms (Lester et al. 2009; Russ and Alcala 2010). Our data indicate that the six to eight year period since the recreational fishing havens were implemented has been sufficient for changes to be manifest in the length and age compositions of a range of harvested species observed here.

The data presented here clearly demonstrate there are strong differences in some aspects of the demographic characteristics of harvested populations of fish between haven and non-haven estuaries. Whether such differences perpetuate as the pressures and stressors on these estuaries intensify with increasing urbanization needs to be evaluated. Further, even though the levels of total and fishing mortality were lower for some species in the havens than in the non-havens, they were still relatively high with F being greater than M in most cases. Thus, the continued assessment of fish populations in the recreational fishing havens is warranted for their sustainable management.

Conclusions

The removal of commercial fishing in the declared recreational fishing havens has changed the allocation of the harvested resources among fishing sectors with the proportion of fish previously taken by the commercial sector now solely available to the recreational sector. This study has provided correlative evidence that this change in management has led to a reduction in overall fishing pressure and fishing mortality on populations of key harvested fish species within these estuaries. Consequently, this has facilitated more fish to grow before they are harvested, allowing for greater proportions of larger and older fish in populations in the recreationally only fished estuaries compared to estuaries in which commercial fishing is still permitted.

To date, the 'freed' commercial allocation of the fisheries resources in these estuaries may not have been fully absorbed by the recreational sector (Steffe et al. 2005a). This situation may change with the burgeoning recreational fishing population and with a potentially greater recreational fishing effort directed to the havens due to the perception that they are better places to fish due to the absence of commercial fishers (similar to partial-take MPAs - Denny and Babcock 2004; Westera et al. 2003). Consequently, the identified localized benefits to the harvested fish populations and to recreational anglers of the havens to date could easily be fished away and consumed by an increasing recreational fishing sector.

Although we have demonstrated differences in the demographics of several harvested fish species in two recreational fishing havens, it needs to be determined if such effects are manifest in similar havens in other estuaries and in estuaries that have partial havens, where only sections of estuaries are classified as recreational fishing only. It is possible that the responses of fish populations to havens in different types of estuaries (e.g. coastal rivers as opposed to coastal lakes) and afforded different levels of protection could be dissimilar to those observed here. Moreover, the potential 'spill-over' effects and broader-scale conservation and sustainability benefits of the 22 declared estuarine recreational fishing havens in NSW to these harvested fish species, as well as any cascading effects to aquatic ecosystems in general, remain to be determined. For example, the observed larger lengths and ages of fish in havens could enhance the total population reproductive outputs of these species and provide recruitment subsidies to other places. Most species examined here are relatively migratory and are known to move between estuaries (Gray et al. 2012; Gray and Barnes 2014). Nevertheless, the levels of connectivity among estuaries require further determination to ascertain more global benefits of havens on fish populations (Curley et al. 2013).

Given the projected expansion of the human and recreational fishing population in coastal areas in general, conflicts over resource allocation will probably intensify and there will inevitably be increased pressure to provide more recreational fishing havens. It is therefore imperative that the broader scale impacts on fish populations of havens continue to be assessed.

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Table 1. Summary of numbers of key commercial, recreational and indigenous fish species sampled each year in the multimesh gillnet and the beam trawl in each Recreational Fishing Haven and non-haven estuary.

		Multimesh Gillnet				Beam Trawl			
Type	Estuary	2008	2009	2010	Total	2008	2009	2010	Total
Acanthopagrus australis									
Haven	Lake Macquarie	89	83	76	248	182	125	32	339
	St Georges Basin	43	41	41	125	23	13	12	48
Non-Haven	Wallis Lake	51	21	27	99	289	279	66	634
	Tuggerah Lake	88	59	67	214	449	493	168	1110
Girella tricuspidata									
Haven	Lake Macquarie	157	214	198	569	2	1	0	3
	St Georges Basin	102	135	99	336	3	3	3	9
Non-Haven	Wallis Lake	29	91	42	162	13	3	0	16
	Tuggerah Lake	451	332	229	1012	11	8	1	20
Mugil cephalus									
Haven	Lake Macquarie	208	297	303	808	0	0	0	0
	St Georges Basin	134	184	266	584	0	0	0	0
Non-Haven	Wallis Lake	203	158	122	483	0	0	0	0
	Tuggerah Lake	490	513	592	1595	1	1	0	2
Platycephalus fuscus									
Haven	Lake Macquarie	17	40	29	86	9	7	3	19
	St Georges Basin	44	37	28	109	2	4	0	6
Non-Haven	Wallis Lake	24	11	28	63	21	18	11	50
	Tuggerah Lake	81	45	31	157	8	32	7	47
Sillago ciliata									
Haven	Lake Macquarie	40	47	39	126	0	0	0	0
	St Georges Basin	213	202	75	490	3	4	2	9
Non-Haven	Wallis Lake	17	54	49	120	3	0	0	3
	Tuggerah Lake	87	26	35	148	3	81	10	94
Sillago maculata									
Haven	Lake Macquarie	5	3	3	11	504	233	97	834
	St Georges Basin	6	15	16	37	35	41	98	174
Non-Haven	Wallis Lake	0	0	0	0	15	21	14	50
	Tuggerah Lake	0	1	0	1	64	113	36	213
Gerres subfasciatus									
Haven	Lake Macquarie	93	104	119	316	2260	1754	724	4738
	St Georges Basin	204	241	125	570	14	2	8	24
Non-Haven	Wallis Lake	55	47	27	129	2221	4565	1468	8254
	Tuggerah Lake	78	31	38	147	3764	3605	2436	9805
Liza argentea									
Haven	Lake Macquarie	685	165	240	1090	0	0	0	0
	St Georges Basin	108	130	419	657	0	0	0	0
Non-Haven	Wallis Lake	336	277	370	983	0	0	0	0
	Tuggerah Lake	136	46	188	370	0	0	0	0
Paramugil georgii									
Haven	Lake Macquarie	12	16	31	59	0	0	0	0
	St Georges Basin	30	10	1	41	0	0	0	0
Non-Haven	Wallis Lake	219	110	300	629	0	0	0	0
	Tuggerah Lake	2	3	4	9	0	0	0	0
Myxus elongatus									
Haven	Lake Macquarie	7	9	3	19	0	0	0	0
	St Georges Basin	83	143	178	404	0	0	0	0
Non-Haven	Wallis Lake	1	2	9	12	0	0	0	0
	Tuggerah Lake	16	1	3	20	0	0	0	0
Pomatomus saltatrix									
Haven	Lake Macquarie	14	62	40	116	43	36	43	122
	St Georges Basin	37	55	44	136	1	8	5	14
Non-Haven	Wallis Lake	8	1	21	30	2	0	1	3
	Tuggerah Lake	121	108	54	283	33	29	0	62
Rhabdosargus sarba									
Haven	Lake Macquarie	17	22	10	49	103	131	67	301
	St Georges Basin	34	59	12	105	94	21	17	132
Non-Haven	Wallis Lake	6	5	5	16	908	187	63	1158
	Tuggerah Lake	38	12	4	54	503	382	208	1093
Pagrus auratus									
Haven	Lake Macquarie	2	2	3	7	31	31	3	65
	St Georges Basin	4	3	0	7	73	9	7	89
Non-Haven	Wallis Lake	0	0	1	1	10	9	39	58
	Tuggerah Lake	2	2	0	4	30	39	4	73
Herklotsichthys castelnaui									
Haven	Lake Macquarie	296	431	965	1692	18	8	23	49
	St Georges Basin	60	49	86	195	0	0	0	0
Non-Haven	Wallis Lake	62	176	189	427	4	1	2	7
	Tuggerah Lake	790	495	1099	2384	9	4	87	100

Table 2. Summary of the significance of the Pseudo-*F* ratios of PERMANOVAs comparing the numbers of key harvested species across estuaries and time. Separate analyses were done for the beam trawl and multimesh gillnet. Factors in bold are those that, if significant, might indicate an effect of the Haven.

$* = P < 0.05$, $** = P < 0.01$, $*** = P < 0.001$, ns $= P > 0.05$

Beam Trawl		Species			
Source of Variation	df	Aa	Gs	Sm	Rs
Haven	**1**	*	*	**ns**	***
Year	2	***	ns	ns	ns
Estuary (Haven)	2	*	*	***	ns
Period(Year)	3	ns	ns	*	ns
Hvn x Yr	**2**	*	**ns**	**ns**	**ns**
Hvn x Pe(Yr)	**3**	**ns**	**ns**	**ns**	**ns**
Yr x Est(Hvn)	4	ns	ns	***	*
Est(Hvn) x Pe(Yr)	6	**	***	ns	ns
Res	840				
Total	863				

Gillnet		Species										
Source of Variation	df	Aa	Mc	Gs	Pf	La	Gt	Sc	Ps	Hc	Pg	Me
Haven	**1**	**ns**	**ns**	*	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**
Year	2	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
Estuary (Haven)	2	ns	**	*	*	ns	*	**	*	*	*	*
Period(Year)	3	ns	*	ns	ns	ns	ns	ns	ns	ns	ns	ns
Hvn x Yr	**2**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**
Hvn x Pe(Yr)	**3**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**	**ns**
Yr x Est(Hvn)	4	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns
Est(Hvn) x Pe(Yr)	6	***	ns	ns	ns	ns	*	ns	**	**	ns	*
Res	264											
Total	287											

Table 3. Summary of estimates of rates of mortality and exploitation for the five species based on the fishery-independent multimesh gillnet sampling in each Recreational Fishing Haven and non-haven estuary. Data pooled across 3 years of sampling; total mortality (*Z*) (+ 1 SE) derived from catch-curve analyses of data displayed in Figures 4 to 8; natural mortality (*M*) determined using the method of Hoening (1975) and based on the published maximum age attained for each species; fishing mortality (*F*) and exploitation rates (*E*) calculated using estimates of *M* and *Z*.

Type	**Estuary**	**Age range**	***Z* (SE)**	***M***	***F***	***E***
Acanthopagrus australis						
Haven	Lake Macquarie	4-12	0.31 (0.08)	0.19	0.12	0.39
	St Georges Basin	3-12	0.28 (0.05)	0.19	0.09	0.33
Non-Haven	Wallis Lake	3-9	0.55 (0.09)	0.19	0.36	0.65
	Tuggerah Lake	3-10	0.45 (0.05)	0.19	0.26	0.58
Platycephalus fuscus						
Haven	Lake Macquarie	2-10	0.36 (0.09)	0.26	0.10	0.28
	St Georges Basin	5-9	0.66 (0.25)	0.26	0.40	0.61
Non-Haven	Wallis Lake	2-5	0.91 (0.13)	0.26	0.65	0.71
	Tuggerah Lake	2-5	1.06 (0.25)	0.26	0.80	0.75
Girella tricuspidata						
Haven	Lake Macquarie	5-14	0.30 (0.05)	0.19	0.11	0.37
	St Georges Basin	6-13	0.37 (0.04)	0.19	0.18	0.48
Non-Haven	Wallis Lake	2-12	0.39 (0.04)	0.19	0.20	0.51
	Tuggerah Lake	2-16	0.36 (0.05)	0.19	0.17	0.47
Mugil cephalus						
Haven	Lake Macquarie	2-10	0.72 (0.18)	0.35	0.37	0.51
	St Georges Basin	3-11	0.52 (0.07)	0.35	0.17	0.33
Non-Haven	Wallis Lake	3-7	1.48 (0.22)	0.35	1.13	0.76
	Tuggerah Lake	2-5	1.20 (0.32)	0.35	0.85	0.71
Sillago ciliata						
Haven	Lake Macquarie	2-13	0.15 (0.03)	0.26	-0.11	-0.78
	Lake Macquarie	6-13	0.24 (0.05)	0.26	-0.02	-0.08
	St Georges Basin	5-12	0.38 (0.09)	0.26	0.12	0.31
Non-Haven	Wallis Lake	5-11	0.53 (0.09)	0.26	0.27	0.51
	Tuggerah Lake	2-8	0.81 (0.09)	0.26	0.55	0.68

Figure 1. Mean (± 1 SE) number of each species sampled in the bean trawl in each of the six periods across the three years in the two recreational fishing havens and two non-havens.

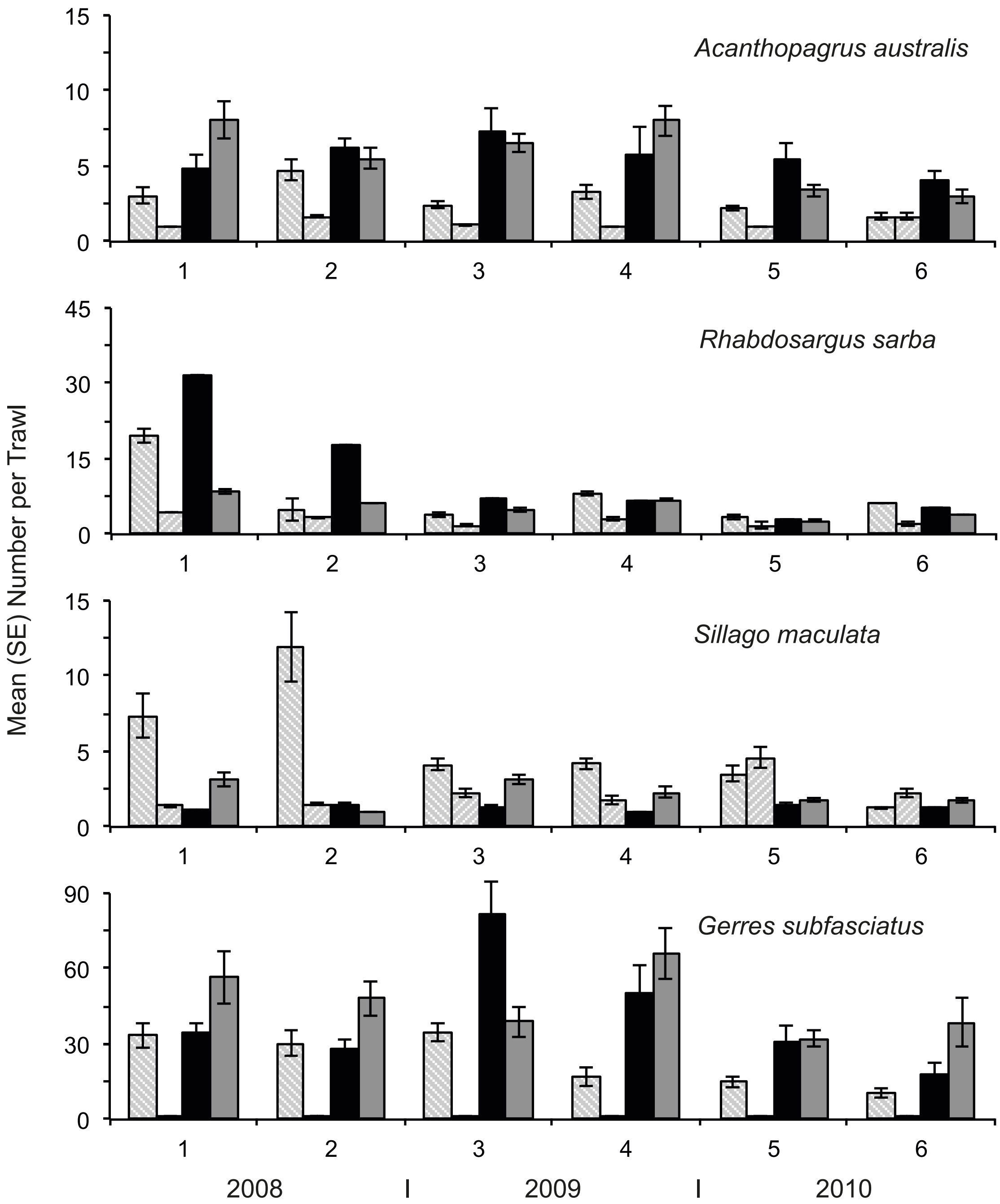


Figure 2. Mean (± 1 SE) number of each species sampled in the multimesh gillnet in each of the six periods across the three years in the two recreational fishing havens and two non-havens.

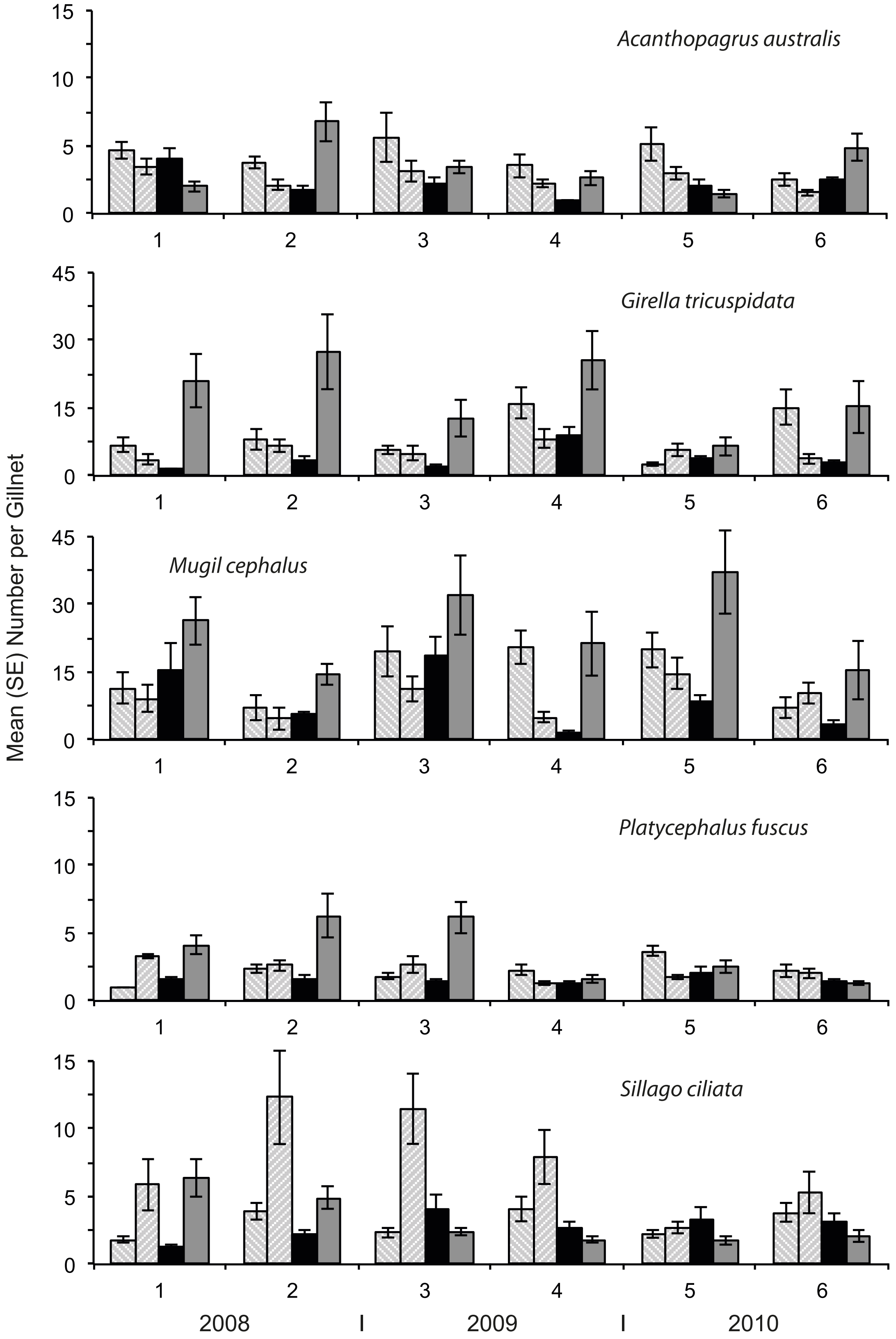


Figure 2. Continued.

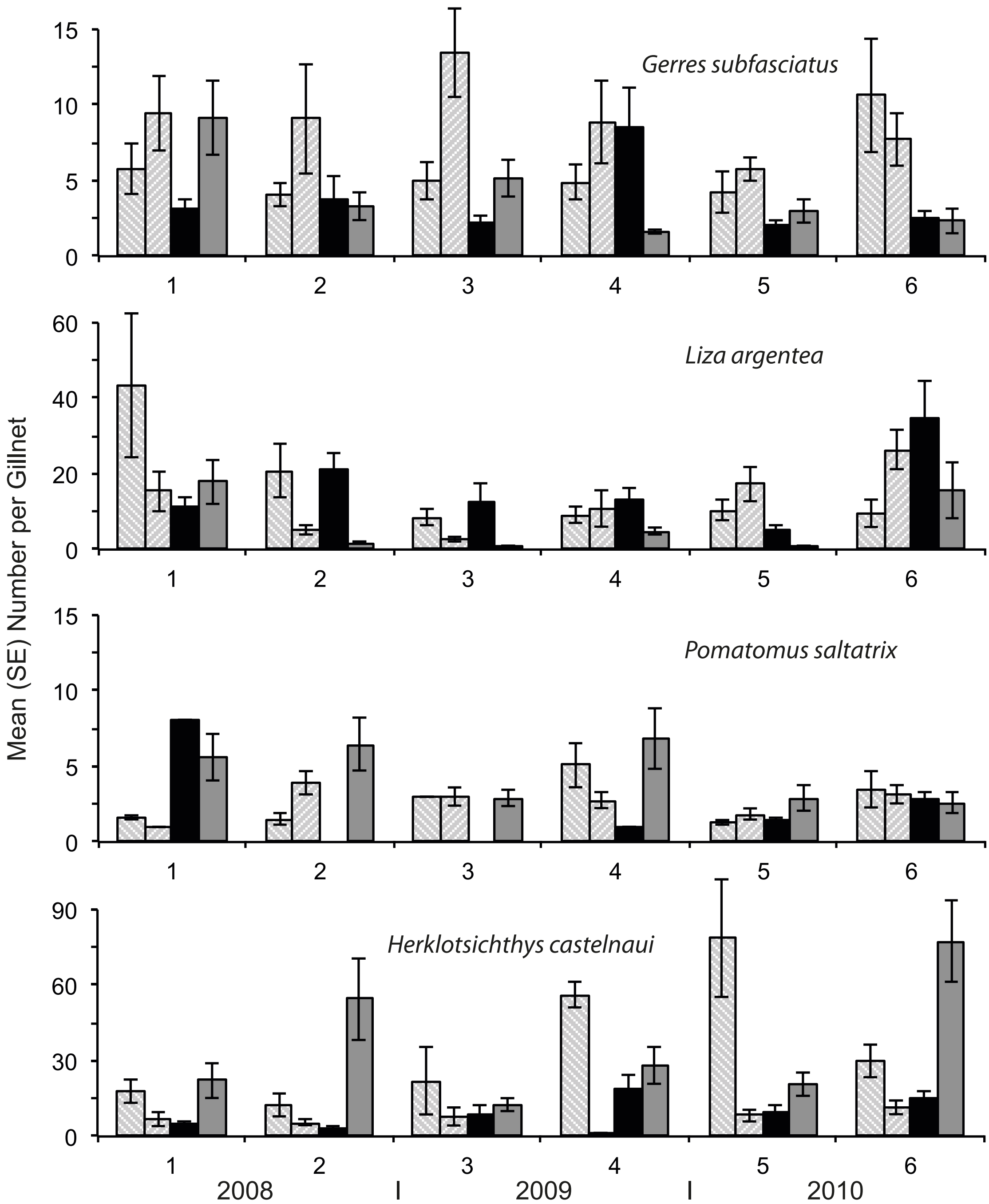


Figure 3. The length and age compositions of *Acanthopagrus australis* sampled in the multimesh gillnet in each recreational fishing haven and each non-haven. Data pooled across the three years of sampling. n = sample size, ML = mean fork length (cm), MA = mean age (years).

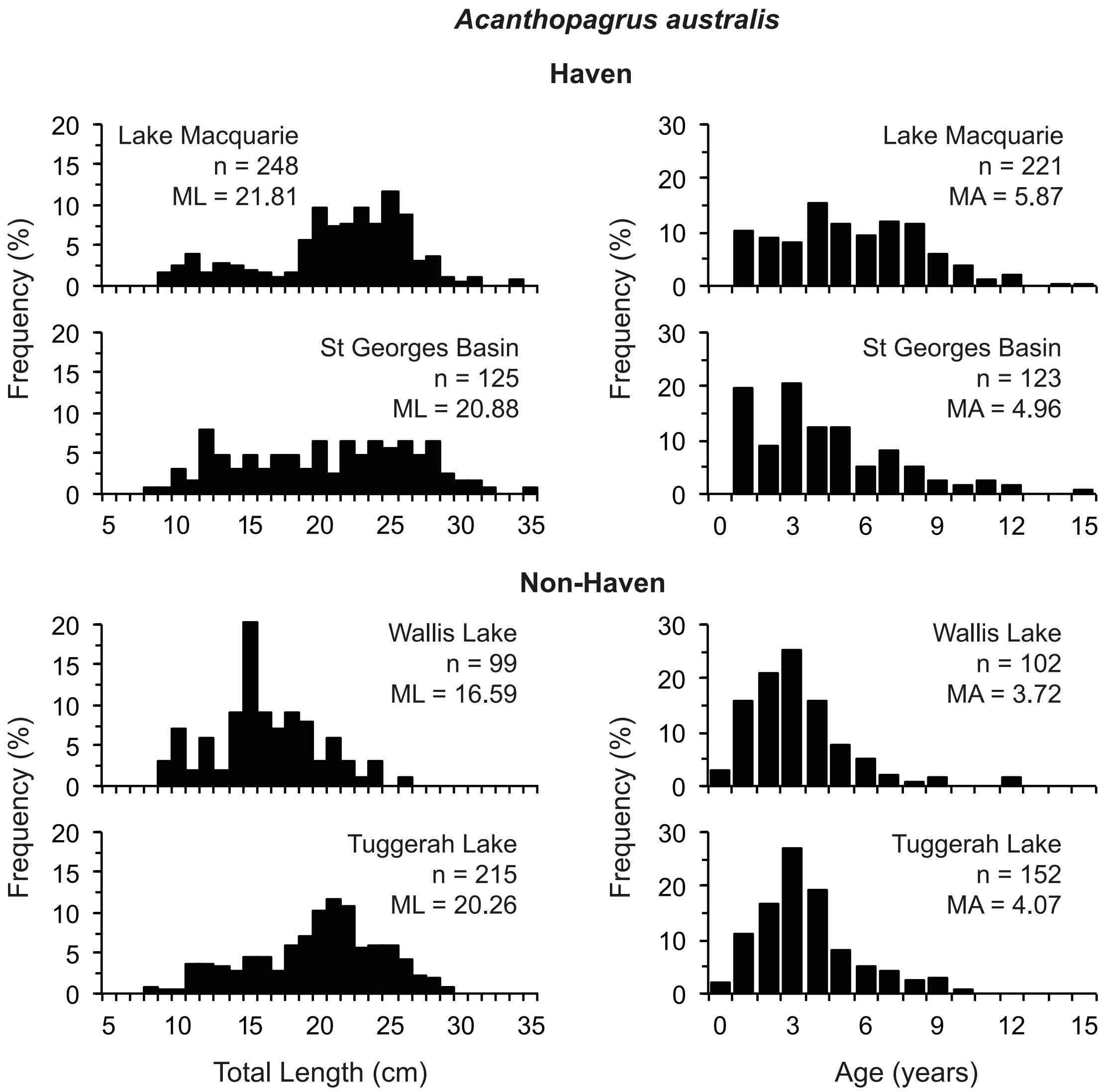


Figure 4. The length and age compositions of *Platycephalus fuscus* sampled in the multimesh gillnet in each recreational fishing haven and each non-haven. Data pooled across the three years of sampling. n = sample size, ML = mean total length (cm), MA = mean age (years).

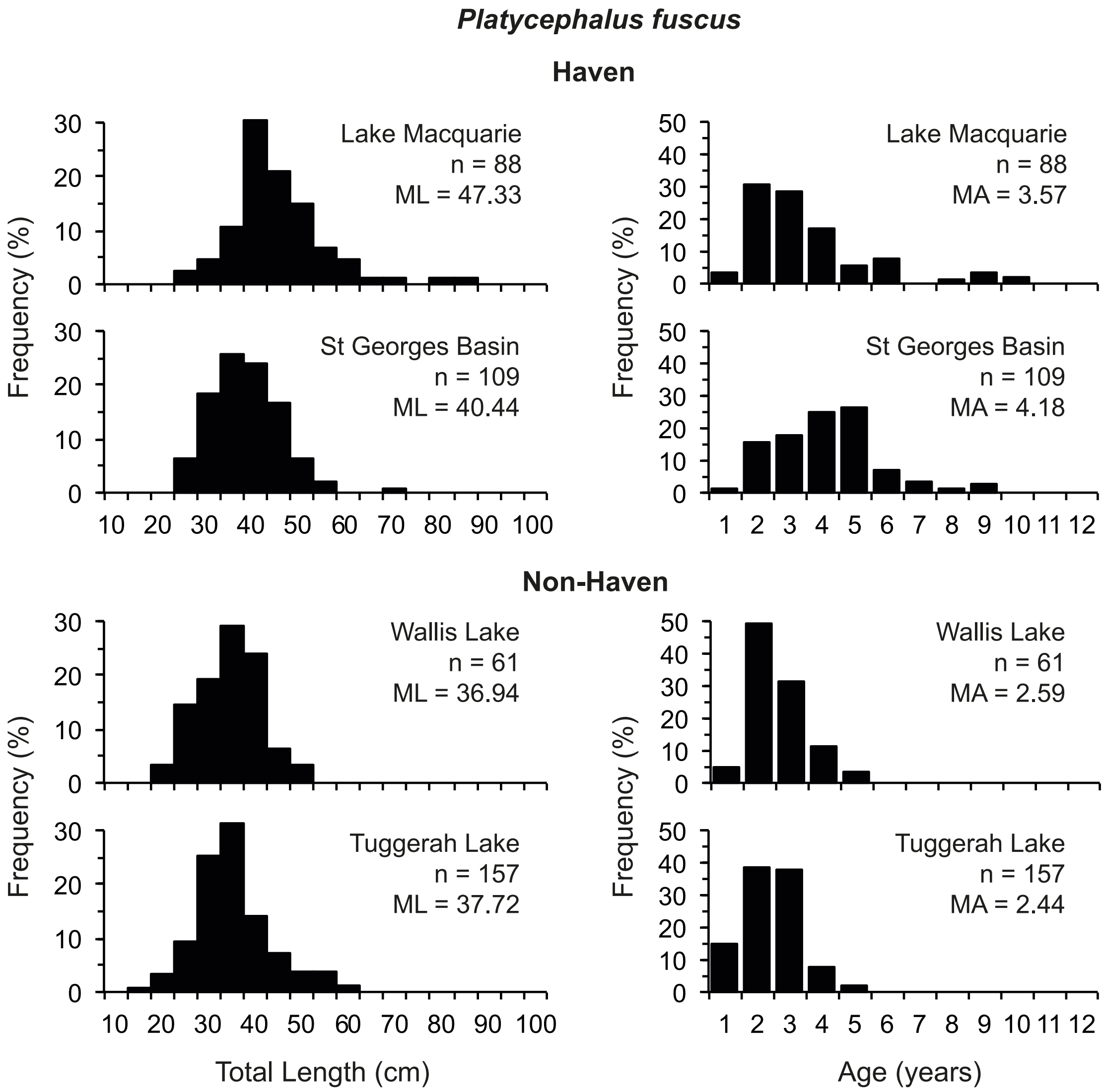


Figure 5. The length and age compositions of *Girella tricuspidata* sampled in the multimesh gillnet in each recreational fishing haven and each non-haven. Data pooled across the three years of sampling. n = sample size, ML = mean fork length (cm), MA = mean age (years).

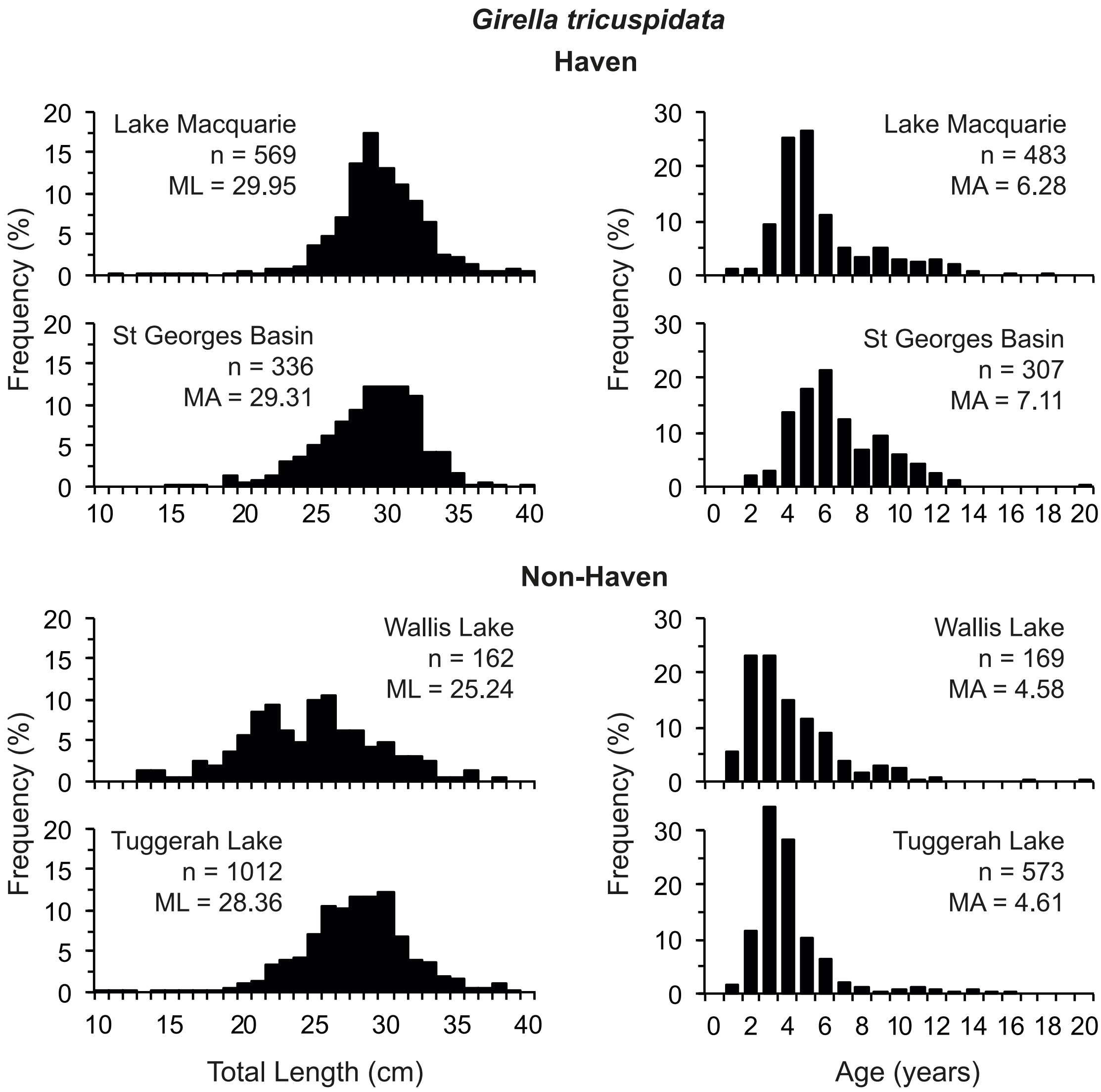


Figure 6. The length and age compositions of *Mugil cephalus* sampled in the multimesh gillnet in each recreational fishing haven and each non-haven. Data pooled across the three years of sampling. n = sample size, ML = mean fork length (cm), MA = mean age (years).

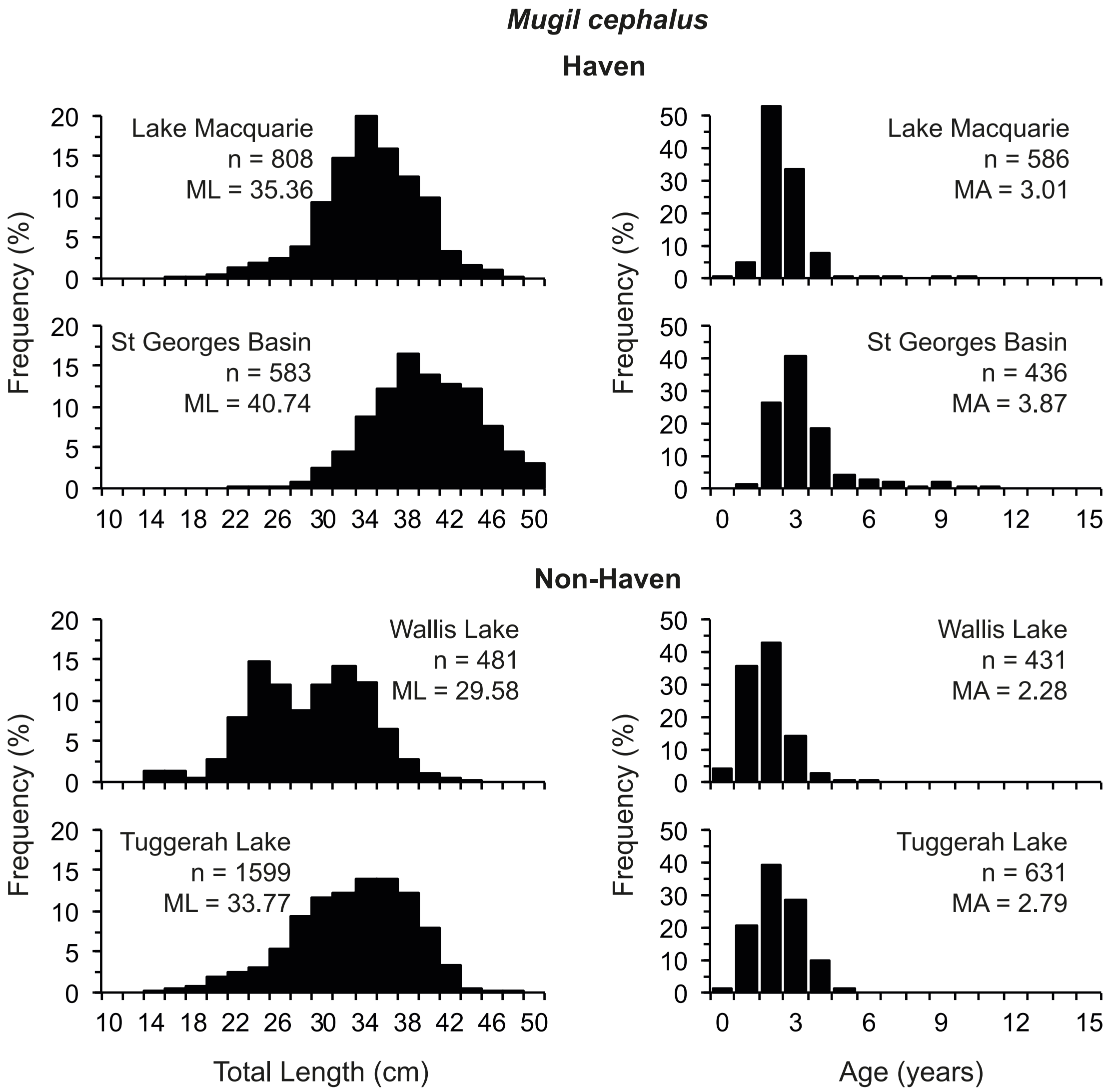


Figure 7. The length and age compositions of *Sillago ciliata* sampled in the multimesh gillnet in each recreational fishing haven and each non-haven. Data pooled across the three years of sampling. n = sample size, ML = mean fork length (cm), MA = mean age (years).

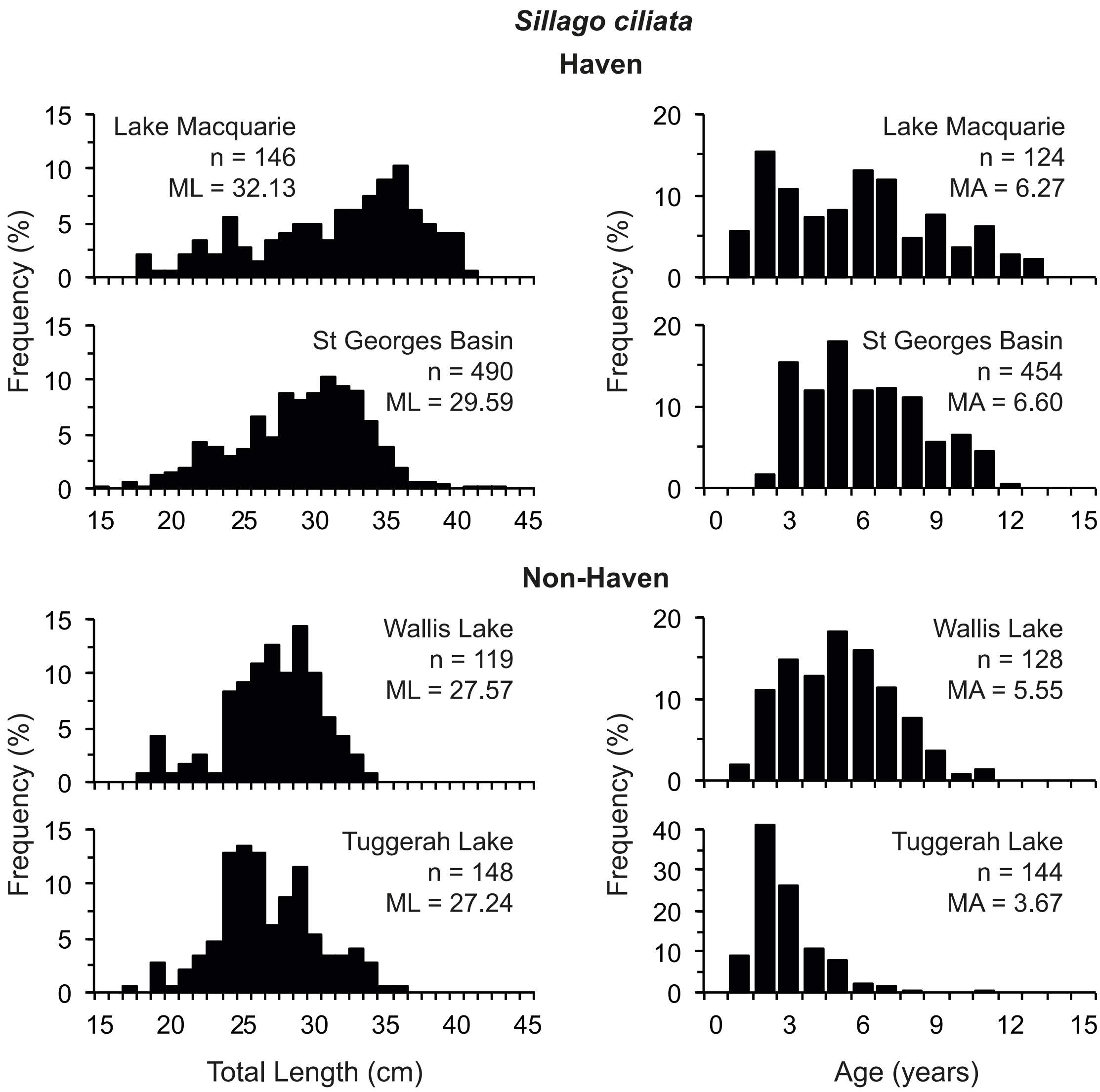


Figure 8. The length and age compositions of *Acanthopagrus australis* sampled in the beam trawl in each recreational fishing haven and each non-haven. Data pooled across the three years of sampling. n = sample size, ML = mean fork length (cm), MA = mean age (years).

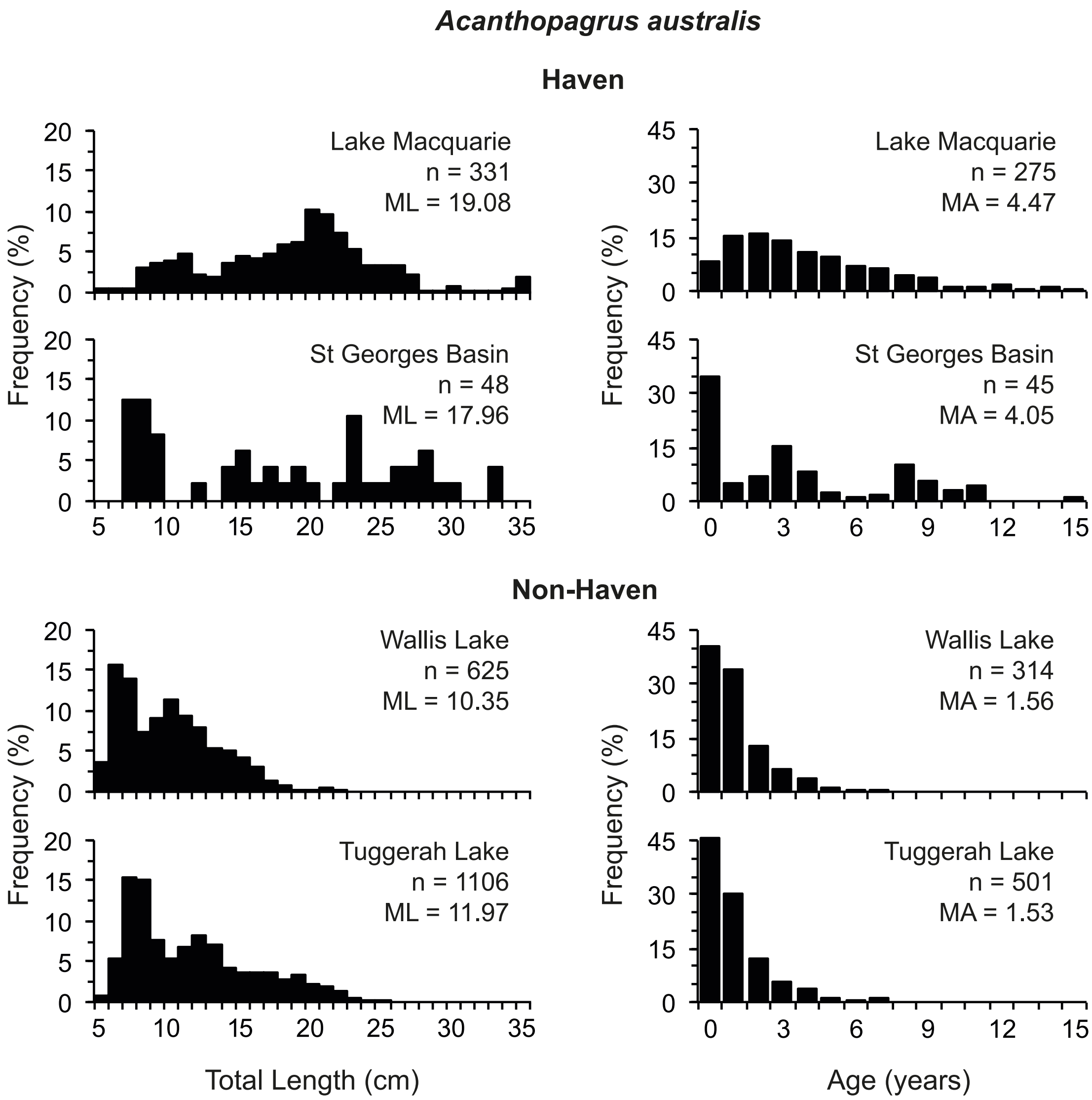


Figure 9. The length compositions of *Gerres subfasciatus* sampled in the multimesh gillnet and the beam trawl in each recreational fishing haven and each non-haven. Data pooled across the three years of sampling. n = sample size, ML = mean fork length (cm).

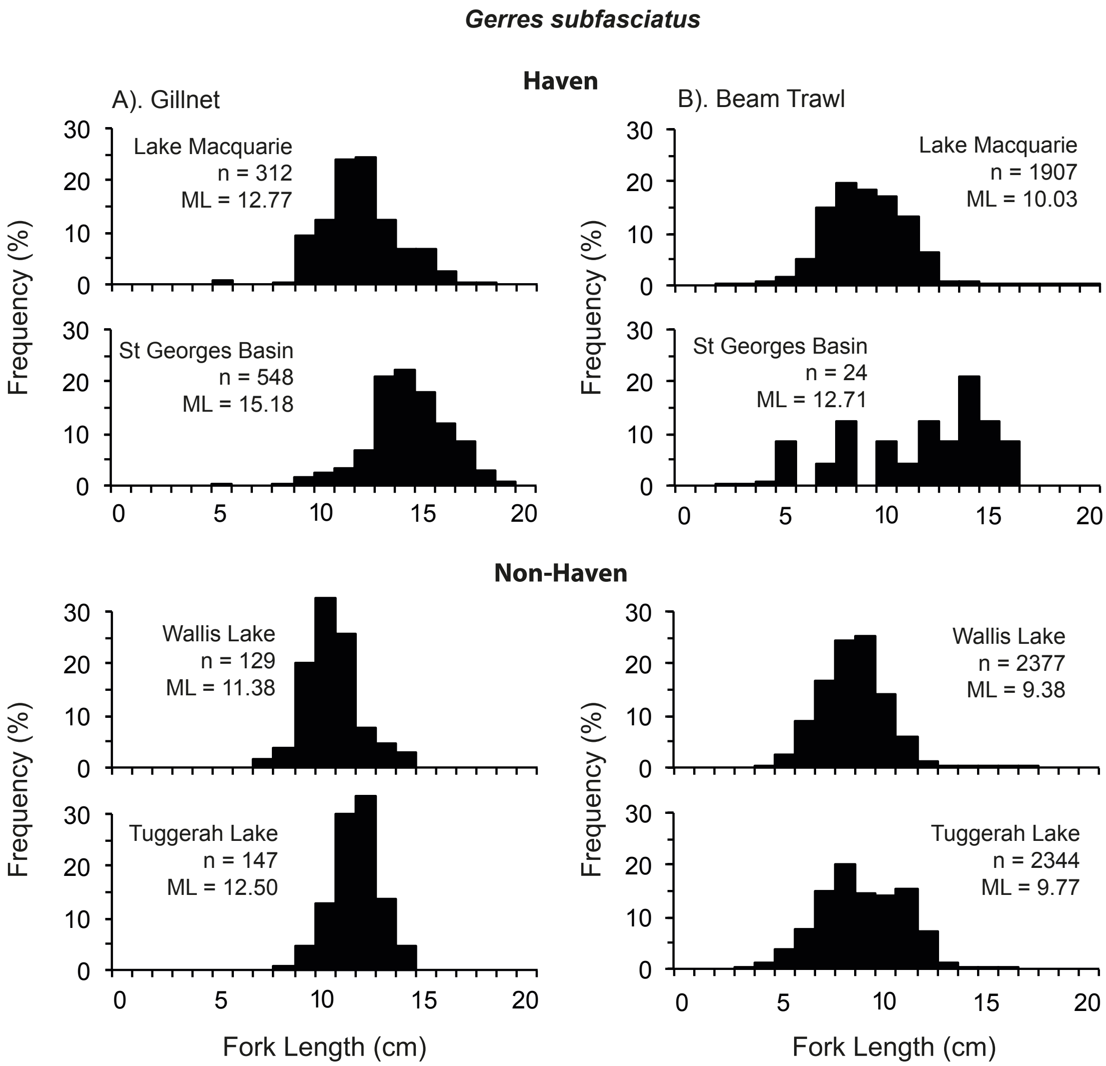


Figure 10. The length compositions of *Liza argentea* and *Pomatomus saltatrix* sampled in the multimesh gillnet in each recreational fishing haven and each non-haven. Data pooled across the three years of sampling. n = sample size, ML = mean fork length (cm).

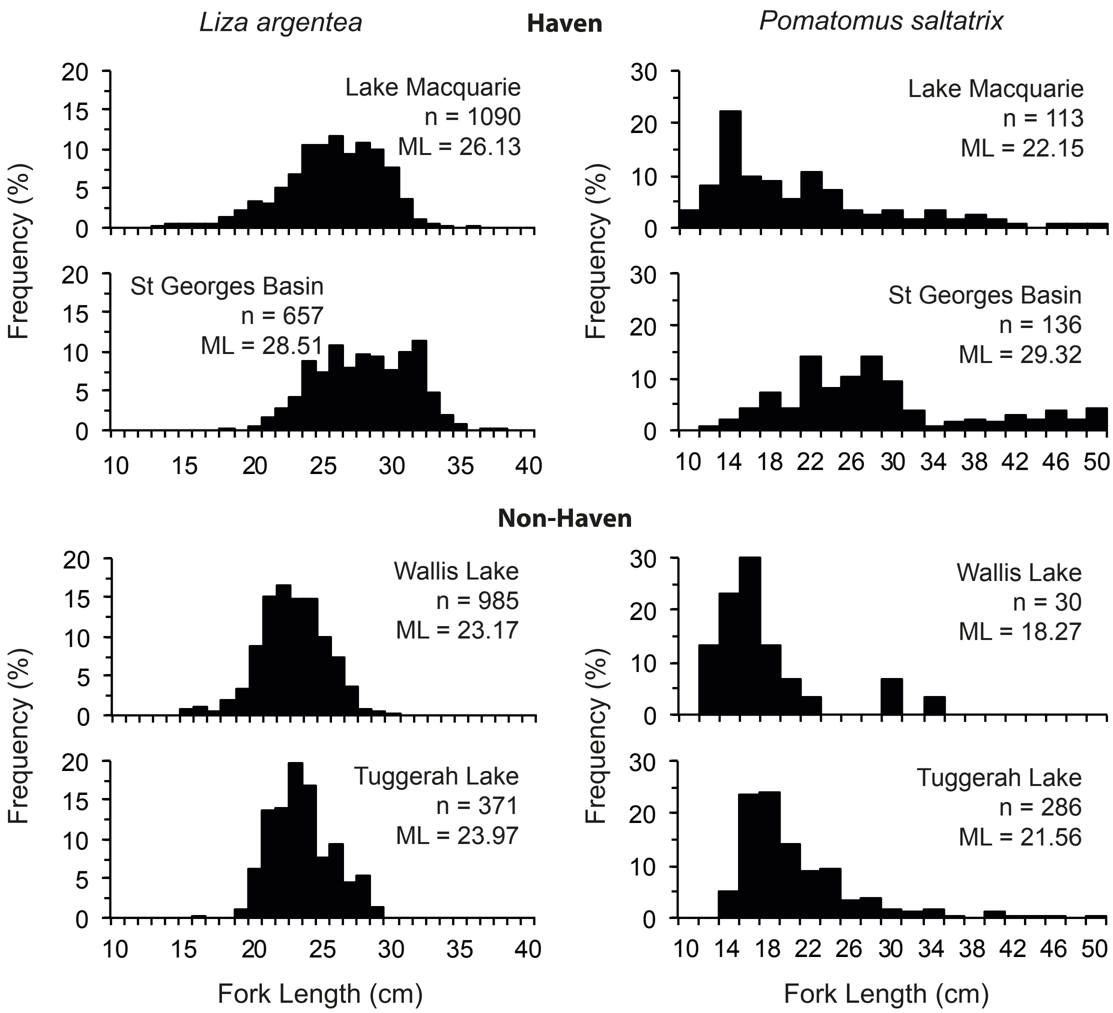
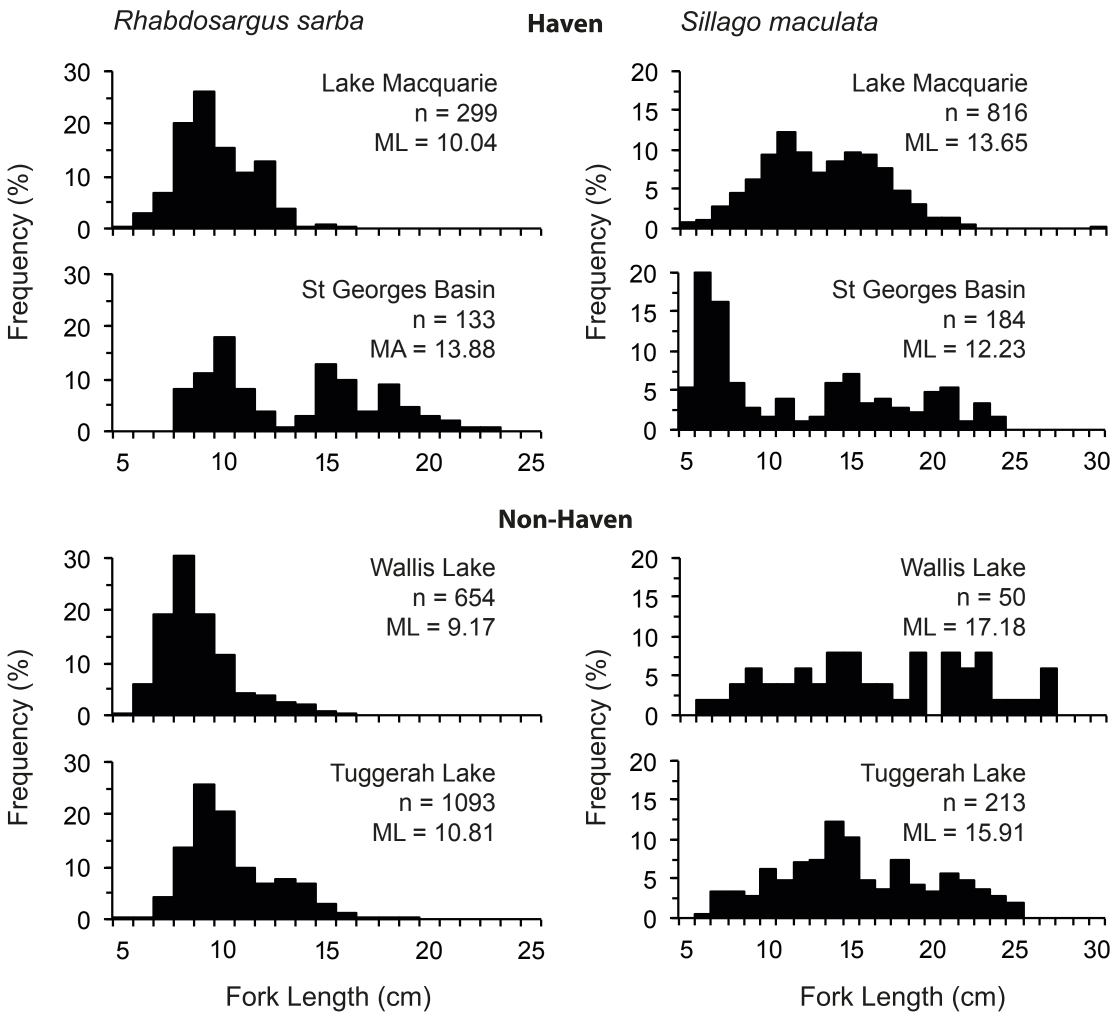


Figure 11. The length compositions of *Rhabdosargus sarba* and *Sillago maculata* sampled in the beam trawl in each recreational fishing haven and each non-haven. Data pooled across the three years of sampling. n = sample size, ML = mean fork length (cm).



Appendix 7. Experiment 5: Utility of fishery-independent sampling strategy to assess aquatic biodiversity across estuaries

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Sampling estuarine fish and invertebrates with a beam trawl provides a different picture of populations and assemblages than multi-mesh gillnets

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ABSTRACT

Although many studies have compared different methods of sampling fish fauna, few have examined differences between active and passive gears over large spatial and temporal scales, which may lead to misleading conclusions about their suitability as sampling tools. Using data from two years of sampling in five estuaries of New South Wales (Australia), we illustrate differences in assemblages and size structures of populations of fish and invertebrates sampled with a beam trawl and multi-mesh gillnets. Multivariate analyses revealed that each method gave a different picture of assemblages of fauna. In general, the beam trawl was more effective than the gillnets in sampling penaeid prawns and several small species of fish. By comparison, the gillnets caught a wide size-range of several fishes of commercial and recreational importance, many of which were mostly absent in catches from the trawl. In some cases, however, differences in assemblages and size-structures of populations between methods depended on the particular estuary or period of time in which sampling was done. These findings not only reinforce the need for pilot studies in identifying suitable sampling gears, but also demonstrate that careful attention must be paid to ensure such studies are replicated over appropriate spatial and temporal scales. Moreover, while sampling with both the trawl and gillnets provided the most comprehensive picture of populations and assemblages, we highlight that the suitability of either sampling method depends on the specific objectives of a study and the particular species (or assemblages of species) of interest.

1. Introduction

A range of active and passive methods, including otter trawls (Smith and Gavaris, 1993; Korsbrekke et al., 2001), beam trawls (Gunderson and Ellis, 1986; Hamer and Jenkins, 2004), seine and gillnets (Degerman et al., 1988; Whitfield et al., 1994), longlines (Simpfendorfer et al., 2002), traps (Kennelly, 1992; Smith and Tremblay, 2003) and underwater video (Willis et al., 2000; Watson et al., 2005), have been used to sample fish and invertebrates in marine, estuarine and freshwater environments. Nevertheless, all of these methods have biases and limitations which may affect the numbers, sizes and types of species that are sampled. Therefore, before commencing any study, it is necessary to identify appropriate methods of sampling (Andrew and Mapstone, 1987).

Ideally, the suitability of different sampling methods should be examined using an experimental approach (Andrew and Mapstone, 1987; Rotherham et al., 2007). It is the case, however, that many studies of fish have been carried out in the absence of any pilot work, with the decision to use one particular type of method perhaps relying on past experience, the results of studies done elsewhere, or common perceptions. For example, trawl nets were initially perceived by Olin and Malinen (2003) and Olin et al. (2009) to be less selective and provide more reliable estimates of species abundance and length distributions than gillnets. Nevertheless, their studies subsequently demonstrated that gillnets and a trawl gave different pictures of the composition of assemblages and size-structures of populations of fish in eutrophic lakes. Indeed, these and many other studies have stressed the need for multiple types of gears to provide the most complete picture of assemblages and size-structures of populations of fish fauna (Olin and Malinen, 2003; Watson et al., 2005; Morrison and Carbines, 2006; Eros et al., 2009; Olin et al., 2009). It remains unclear, however, whether generalisations about sampling with gillnets and trawls are consistent for assemblages of fish or invertebrates in lakes and estuaries in other parts of the world.

Some studies have compared the utility and efficiency of different sampling gears over relatively short periods of time (e.g. within a single week; Olin and Malinen, 2003; Cappo et al., 2004) and in a small number of places (e.g. small part of a single lake or estuary e.g. Guest et al., 2003). Owing to spatial and temporal variability in abundance and size-structure of populations and assemblages, differences between or among sampling gears may depend on the

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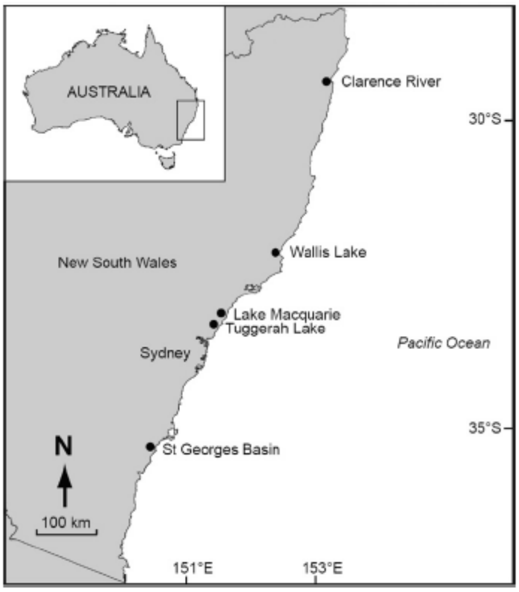

Fig. 1. Locations of estuaries sampled in the study.

spatial and temporal scope of an experiment. Unless the generality of patterns have been examined at different spatial and temporal scales, the suitability of a particular method of sampling may be misleading (Andrew and Mapstone, 1987).

Here, we examine differences in assemblages and size-structures of populations of fish and invertebrates sampled with a beam trawl and multi-mesh gillnets over a two-year period in five estuaries of New South Wales (NSW), Australia. Based on our earlier work (Rotherham et al., 2008a,b; Gray et al., 2009) and results of previous studies done elsewhere (Olin and Malinen, 2003; Olin et al., 2009), we predicted that: (i) the beam trawl and multi-mesh gillnets would sample different assemblages of fish fauna; (ii) for species caught in both gears, the beam trawl and multi-mesh gillnets would catch smaller and larger individuals, respectively; and (iii) patterns would be consistent among estuaries and within and between years.

2. Methods

2.1. Study area

Five estuaries of NSW were sampled with a beam trawl and multi-mesh gillnets over a two-year period as part of a larger study evaluating the status of estuarine fisheries resources. The estuaries sampled included the Clarence River, Wallis Lake, Lake Macquarie, Tuggerah Lake and St. Georges Basin (Fig. 1). All of these estuaries are wave-dominated barrier estuaries with tidal inlets constricted by wave-deposited sand and flood-tidal deltas, but they differ in size and characteristics (see Roy et al., 2001). The Clarence River is a mature, infilled, barrier estuary with a riverine channel which dominates most of the estuarine environment. The remaining estuaries are large, well-mixed, microtidal, coastal lakes. Specific characteristics of each estuary (e.g. water area, catchment area, habitats, etc.) and its catchment have been described elsewhere (Roy et al., 2001).

2.2. Sampling design and methods

2.2.1. Temporal scales

Each estuary was sampled with the beam trawl and gillnets in each of two periods (January–March/April and April/May–July) in each of two years (2008 and 2009). The two sampling periods were approximately 3 months long and separated by at least 4 weeks in each estuary. Results of our previous studies of gillnets (Rotherham et al., 2011) and beam trawls (unpublished data) demonstrated that variation in abundances of fish at temporal scales within 3-month periods (e.g. days, weeks and months) was small compared to variation between 3-month periods and at small spatial scales. Further, in the present study, the order in which each estuary was sampled with each gear (within each period in each year) was selected at random. Thus, we consider that differences between gears are unlikely to be substantially affected by differences related to the particular day, week or month that sampling was done within a 3-month period. Further, sampling at random times within a 3-month period with each gear avoided potential problems of non-independence of data, which may occur if comparisons of gears are done at particular spatial scales simultaneously, or over short periods of time (*sensu* Underwood, 1997).

2.2.2. Beam trawl

The beam trawl was 3-m wide and configured with 41-mm diamond-shaped mesh in the body and 20-mm mesh hung on the bar (i.e. square-shaped) in the codend (see Rotherham et al., 2008a). In each estuary (except for the Clarence River), four sites (separated by 1 km to several km) were randomly selected over relatively flat, predominantly unvegetated sediment interspersed with (or adjacent to) patches of seagrass. In the Clarence River, two sites (selected at random and separated by at least 1 km) were nested within each of two zones (entrance and middle) to account for potential differences in salinity among different sections of the river.

In each sampling period in each year, nine non-overlapping replicates of the beam trawl were sampled at night in each site (Rotherham et al., 2008b). It took two consecutive nights to sample all four sites within an estuary. As explained above, the two nights of sampling were selected at random from within each sampling period. During each night, each replicate trawl was towed for 5 min (see Rotherham et al., 2008b) at speeds of about 1.2 m s^{-1}. The design of trawl sampling was not stratified by depth (i.e. into deep and shallow strata as done for gillnets in some of the estuaries, see below) because there were insufficient trawlable areas available in most of the estuaries sampled to achieve the necessary levels of replication in each strata. Nevertheless, sampling with the trawl was done over a similar range of depths and habitats (predominately unvegetated sediment interspersed with, or adjacent to, patches of seagrass) as sampling done with the gillnets (i.e. from a minimum of about 1 m up to a maximum of about 6 m, see below). After each replicate tow was completed, the contents of the codend were emptied onto a tray and sorted by species. Collection of data included: the total numbers of individuals of each species; and the sizes of economically important finfish (fork length – FL, to the nearest 0.5 cm below).

2.2.3. Multi-mesh gillnets

Each multi-mesh gillnet consisted of seven 20-m panels of different sizes of stretched mesh (36, 44, 54, 63, 76, 89 and 102 mm) connected together in a random order (see Gray et al., 2009). Adjacent panels were separated by 5 m of rope to minimise potential effects of the smaller-mesh panels 'leading' larger fish to adjacent panels. The 36- and 44-mm panels were made from monofilament netting with twine diameter of 0.15 mm; the remaining panels had

Table 1
Results of permutational multivariate analysis of variance (PERMANOVA) testing for differences in assemblages of fish fauna between sampling methods (i.e. beam trawl and gillnets; Method: fixed), years (Year: random) and sampling periods (period: nested within year). Analyses were done on both standardised, abundance data and presence/absence data, separately for (a) Clarence River, (b) Wallis Lake, (c) Lake Macquarie, (d) Tuggerah Lake and (e) St. Georges Basin.

Source	df	Standardised			Presence/absence		
		MS	*F*	*P*	MS	*F*	*P*
(a) Clarence River							
Method – M	1	36191	28.4	***	14001	8.0	*
Year – Y	1	1677	0.8	ns	1159	0.3	ns
Period (Y)	2	2117	0.9	ns	3531	2.8	**
M × Y	3	1276	0.7	ns	1760	1.0	ns
M × P(Y)	24	1905	0.8	ns	1683	1.3	ns
Residual	31	2362			1264		
(b) Wallis Lake							
Method – M	1	40546	17.6		33179	34.0	**
Year – Y	1	2328	0.8	ns	1785	1.3	ns
Period (Y)	2	2951	2.1		1343	2.3	ns
M × Y	3	2302	0.7	ns	975	0.9	ns
M × P(Y)	24	3094	2.2	*	1031	1.8	ns
Residual	31	1421			572		
(c) Lake Macquarie							
Method – M	1	33726	16.6	**	20643	25.0	**
Year – Y	1	1975	1.3	ns	1404	1.6	ns
Period (Y)	2	1512	1.2	ns	885	1.3	ns
M × Y	3	2037	1.5	ns	827	1.1	ns
M × P(Y)	24	1345	1.0	ns	721	1.1	ns
Residual	31	1292					
(d) Tuggerah Lake							
Method – M	1	49714	42.2	***	25237	26.8	**
Year – Y	1	1198	0.8	ns	549	0.7	ns
Period (Y)	2	1562	1.4	ns	741	1.5	ns
M × Y	3	1177	0.7	ns	942	1.5	ns
M × P(Y)	24	1676	1.5	ns	641	1.3	ns
Residual	31	1122			482		
(e) St. Georges Basin							
Method – M	1	37343	10.8		18384	21.7	
Year – Y	1	2949	0.7	ns	1621	0.6	ns
Period (Y)	2	4471	3.5		2563	3.4	
M × Y	3	3454	0.8	ns	849	0.5	ns
M × P(Y)	24	4260	3.3	**	1739	2.3	*
Residual	31	1283			758		

* $P < 0.05$.
** $P < 0.01$.
*** $P < 0.001$

multifilament netting with a twine diameter of 0.4 mm. All panels were 2 m deep and had a hanging ratio (E) of 0.5.

In each estuary, there were four randomly selected sites separated by one to several kilometres (see Gray et al., 2009). Consistent with the design of sampling with the beam trawl (see above), two sites (selected at random and separated by at least 1 km) were nested within each of two zones (entrance and middle) in the Clarence River. The design of sampling in the Clarence River, Lake Macquarie and St Georges Basin, included two habitats: shallow (<2 m) and deep (4–6 m) strata. Wallis Lake and Tuggerah Lake are very shallow (mean depth of about 2 m) and so it was not possible to stratify sampling into shallow and deep strata – sampling was done only in shallow habitats (i.e. <2 m).

In each sampling period in each year, three replicates of the multi-mesh gillnet (each separated by 50–100 m) were sampled in each habitat at each site. It took one night to sample a site. In Wallis Lake and Tuggerah Lake, however, two sites could be sampled within a night because only shallow habitats were available to be sampled. As with the beam trawl, the particular week in which sampling with the gillnets commenced within each sampling period and estuary, was selected at random. On each night of sampling, the gillnets were bottom-set at dusk, soaked for 1 h and then retrieved (Rotherham et al., 2006). Catches in each net were identified and counted; economically important species of fish were measured for FL as above.

2.3. Analysis of data

The hypothesis that the beam trawl and multi-mesh gillnets would sample different assemblages of fish fauna consistently among estuaries and within and between years was tested using permutational multivariate analysis of variance (PERMANOVA; Anderson, 2001; Anderson et al., 2008). For each estuary, data were averaged across replicates to produce a 'centroid' (see Anderson et al., 2008) for each site at each time of sampling, separately for each method. The centroids were calculated in order to produce a balanced design of sampling (i.e. equal numbers of replicates within each level of the factor 'Method'). Thus, 'sites' became the replicate samples in the analyses. For the Clarence River, Lake Macquarie and St. Georges Basin, however, the centroid for the gillnets at each site and time of sampling was also averaged across the deep and shallow habitats.

Bray–Curtis similarity matrices were then calculated on standardised, untransformed data (which controlled for differences in total numbers of fish sampled between the two methods and emphasised abundant species); and separately on presence/absence data (which emphasised composition of assemblages and frequencies of occurrence of species).

PERMANOVAs were done on the standardised and presence/absence data separately for each estuary, using a model that included the factors – method (fixed factor), year (random) and period (nested within year and random). Multivariate patterns of assemblages were visualised using non-metric multidimensional scaling (nMDS; Clarke, 1993). When significant interactions involving the factor Method were detected by PERMANOVA, pair-wise tests between levels of this factor (i.e. beam trawl vs. gillnets) were then done separately for each level of the other factor using a separate run of the PERMANOVA routine. Similarity percentage analysis (SIMPER; Clarke, 1993) was done to identify individual species that contributed to significant differences in assemblages between the two sampling methods (Clarke, 1993).

Differences in size-frequency distributions of abundant, economically important species caught between sampling gears were examined visually and tested using Kolmogorov–Smirnov (K-S) tests (Sokal and Rohlf, 1995). It was necessary to pool data from the shallow and deep gillnets to enable comparisons with the trawl.

3. Results

3.1. General

A total of 145,337 fish from more than 116 species were caught during the study. Three species comprised more than 50% of the total catch: *Metapenaeus macleayi* (~25%), *Gerres subfasciatus* (17%) and *Herklotsichthys castelnaui* (9%). Although more organisms were caught in the gillnets (83,529 individuals) than in the beam trawl (61,808 individuals), both gears caught similar numbers of species (>86 taxa in trawl and >88 taxa in the gillnets). Fifty-eight taxa were common to both methods, with 28 and 30 taxa caught exclusively in the trawl and gillnets, respectively.

3.2. Multivariate analyses

PERMANOVA detected significant differences in assemblages of fish fauna between sampling methods for most estuaries (3/5 analyses of the standardised data; 4/5 analyses of the presence absence data; Table 1). Samples from the trawl clearly separated from samples from the gillnet in nMDS plots (examples are shown in Fig. 2a and b to illustrate the general patterns). For the remaining analyses (i.e. 2/5 analyses of the standardised data and 1/5 analyses of the presence/absence data), PERMANOVA detected significant Method × Period (Y) interactions, which indicated that differences between methods depended on the particular period of time that sampling was done (illustrated in Fig. 2c, for St. Georges Basin, standardised data). Nevertheless, for all analyses with significant Method × Period (Y) interactions, subsequent pair wise tests done separately for each sampling period revealed consistent differences ($P < 0.05$) in assemblages between the trawl and gillnets among each of the four sampling periods. Therefore, the reasons for the significant interactions were unclear.

In addition to differences between methods, PERMANOVA also detected a significant effect of Period for the analysis of the presence/absence data in the Clarence River. Pair-wise tests were not done among levels of this factor because it was random (*sensu* Underwood, 1997).

Several abundant taxa were important in distinguishing between assemblages sampled with the trawl and gillnets for the standardised data (Table 2). Differences were driven by larger proportions of *Liza argentea*, *Mugil cephalus*, *H. castelnaui*, *Girella tricuspidata* and *Pomatomus saltatrix*, which were caught mainly in the gillnets consistently among estuaries; and larger proportions of *Pelates sexlineatus*, *Rhabdosargus sarba*, *M. macleayi* and *Ambassidae* spp., which were caught predominantly in the trawl among the different estuaries. Other species contributed to differences in assemblages between methods, but only in one estuary, e.g. *Metapenaeus bennettae*, *Sillago maculata* and *Apogon angustatus* in the trawl; and *Paramugil georgii*, *Arius graeffei* and *Trachystoma petardi* in the gillnets. Although *G. subfasciatus* and *Acanthopagrus australis* were important species in distinguishing between sampling methods, in some estuaries they were caught in larger proportions in the trawl than gillnets and vice versa.

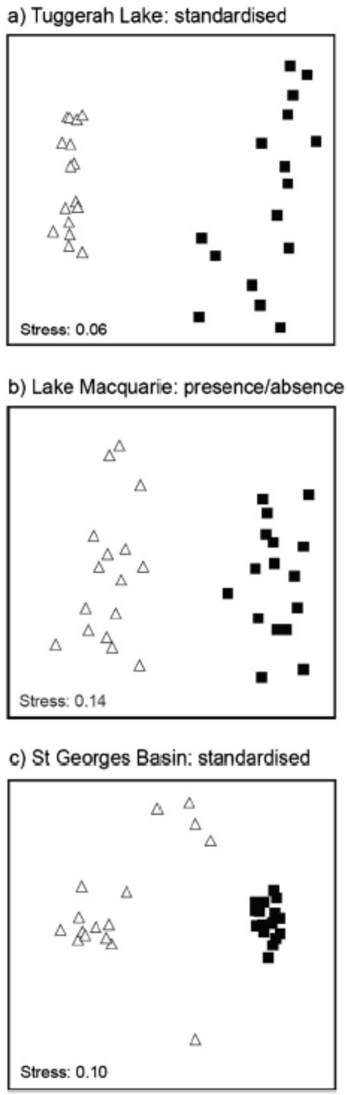

Fig. 2. Two-dimensional nMDS ordinations showing examples of the main differences between assemblages of fish sampled with a beam trawl (open triangles) and multi-mesh gillnets (shaded squares) in (a) Tuggerah Lake, (b) Lake Macquarie and (c) St Georges Basin. Data were averaged across replicates (and deep and shallow habitats for the gillnets) to produce a centroid for each site for each time of sampling. Stress values are shown to indicate the usefulness of the representation of differences among samples. Ordinations with stress values less than 0.2 are considered acceptable (Clarke, 1993).

Table 2
Species contributing more than 2% to the average dissimilarity between assemblages of fish fauna sampled with the beam trawl (T) and multi-mesh gillnets (G) for both standardised and presence/absence data. The number of times a species was important in distinguishing between assemblages (out of the three significant PERMANOVAs for standardised data and four significant PERMANOVAs for the presence/absence data, see Table 1 is shown next to the method in which it was consistently caught (i.e. T or G) in parentheses. 'Inc' denotes species that were not consistently caught in any one particular method among estuaries. Asterisks indicate species that contributed 10% or more to the average dissimilarity. All taxa are finfish unless otherwise indicated; 'p' and 'c' denote prawns and crabs, respectively.

Taxa	Standardised	Presence/absence
Ambassidae spp.	T(2)	T(4)
Pseudorhombus arsius	–	T(4)
Melicertus plebejus[p]	T(1)	T(3)
Metapenaeus macleayi[p]	T(2)*	T(3)
Sepiadarium austrinum	–	T(3)
Metapenaeus bennettae[p]	T(1)*	T(3)
Pelates sexlineatus	T(3)	T(2)
Monacanthus chinensis	–	T(2)
Leiopotherapon aheneus	–	T(2)
Euristhmus lepturus	–	T(2)
Sillago maculata	T(1)	T(2)
Meuschenia trachylepis	–	T(2)
Brachirus nigra	–	T(1)
Penaeus monodon[p]	–	T(1)
Apogon quadrifasciatus	T(1)	T(1)
Notesthes robusta	–	T(1)
Foetorepus calauropomus	–	T(1)
Arenigobius frenatus	–	T(1)
Hyperlophus vittatus	–	T(1)
Portunus pelagicus[c]	–	Inc
Dasyatis fluviorum	–	Inc
Pagrus auratus	–	Inc
Rhabdosargus sarba	T(2)	Inc
Acanthopagrus australis	Inc	–
Gerres subfasciatus	Inc*	–
Liza argentea	G(3)	G(4)
Girella tricuspidata	G(2)	G(4)
Sillago ciliata	G(1)	G(4)
Mugil cephalus	G(3)*	G(4)
Herklotsichthys castelnaui	G(3)*	G(4)
Platycephalus fuscus	–	G(2)
Paramugil georgii	G(1)	G(2)
Arius graeffei	G(1)	G(1)
Trachystoma petardi	G(1)	G(1)
Monodactylus argenteus	–	G(1)
Macquaria colonorum	–	G(1)
Potamalosa richmondia	–	G(1)
Dinolestes lewini	–	G(1)
Belonidae spp.	–	G(1)
Argyrosomus japonicus	–	G(1)
Arrhamphus sclerolepis	–	G(1)
Pomatomus saltatrix	G(2)	G(1)
Dicotylichthys punctulatus	–	G(1)
Trachurus novaezelandiae	–	G(1)

For the presence/absence data, larger numbers of species contributed to differences in assemblages of fish fauna between the trawl and the gillnets, which included many species that were caught relatively infrequently or in small numbers (Table 2). Several species were unique to, or caught more frequently in the trawl (e.g. *Ambassidae* spp., *Pseudorhombus arsius*, *Melicertus plebejus*, *M. macleayi*, *Sepiadarium austrinum* and *M. bennettae*) or gillnets (e.g. *L. argentea*, *G. tricuspidata*, *Sillago ciliata*, *M. cephalus* and *H. castelnaui*) consistently among estuaries. Many other species were caught predominately or more frequently in either the trawl (e.g. *Monacanthus chinensis*, *S. maculata*, *Penaeus monodon* and *Brachirus nigra*) or the gillnets (e.g. *Platycephalus fuscus*, *P. georgii*, *Macquaria colonorum*, *Trachurus novaezelandiae* and *Arrhamphus sclerolepis*), but only in one or two estuaries. Several other species (i.e. *Portunus pelagicus*, *Dasyatis fluviorum*, *Pagrus auratus* and *R. sarba*) were important distinguishing species, but whether or not they were caught more frequently in the trawl or gillnets depended on the particular estuary.

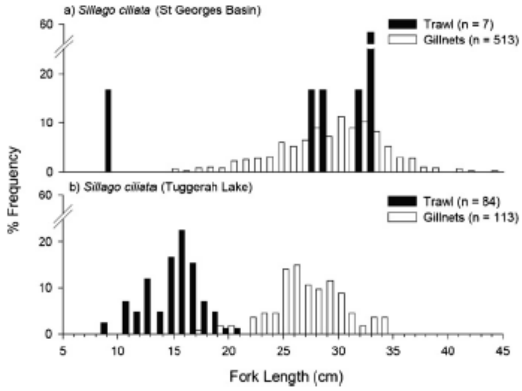

Fig. 3. Size-frequency distributions of *Sillago ciliata* caught between the beam trawl and multi-mesh gillnets in (a) St. Georges Basin and (b) Tuggerah Lake. All data are pooled across sites, periods and years; data from gillnets in St. Georges Basin are pooled across deep and shallow habitats. Kolmogorov–Smirnov tests were not done in a) due to low numbers of fish in the trawl. Size-distributions of *S. ciliata* in Tuggerah Lake (b) were significantly different (K-S test, $D = 0.96$, $P < 0.001$).

3.3. Size-frequency distributions

There were different patterns in the size-frequency distributions of abundant, economically important species of fish between the trawl and the gillnets; and among species and estuaries. Although similar differences were observed between the trawl and the gillnets for *S. ciliata* in the Clarence River, Wallis Lake and St. Georges Basin (illustrated in Fig. 3a for St. Georges Basin), this pattern was not consistent in Tuggerah Lake (Fig. 3b) – similar numbers of *S. ciliata* were caught in both methods in Tuggerah Lake, with larger proportions of smaller (approx. <22 cm) and larger fish (approx. >22 cm) caught in the trawl and the gillnets, respectively.

Differences between the trawl and the gillnets were also observed for size-frequencies of *A. australis* in most estuaries (i.e. Clarence River, Wallis Lake and Tuggerah Lake) – the trawl sampled larger proportions of small fish (about <16 cm) and the gillnets caught greater proportions of large fish (example shown for Tuggerah Lake in Fig. 4a). This pattern was not, however, consistent in the other estuaries. In Lake Macquarie, the trawl caught a wider distribution of *A. australis* than the gillnets, including larger proportions of small (approx. <16 cm) and very large (approx. >34 cm) fish (Fig. 4b). In St. Georges Basin, small and large size-classes of *A. australis* were sampled with the trawl, but overall, the trawl caught relatively small numbers of this species compared to the other estuaries (Fig. 4c).

4. Discussion

As predicted, this study found consistent differences between assemblages of fish and invertebrates sampled with a beam trawl and multi-mesh gillnets. Although similar numbers of species were caught by both gears, differences in assemblages of fauna occurred because many taxa were unique to, or caught more frequently (or in larger proportions among samples) in either the trawl or the gillnets. Thus, each gear gave a different picture of assemblages of fauna, which is consistent with results of previous studies that have examined differences between an otter trawl and gillnets in eutrophic lakes (Olin and Malinen, 2003; Olin et al., 2009).

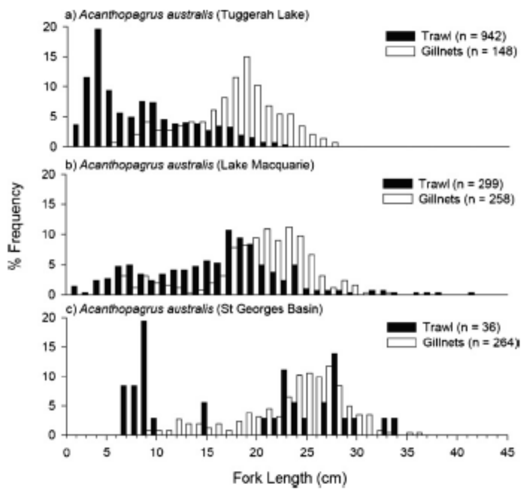

Fig. 4. Size-frequency distributions of *Acanthopagrus australis* caught between the beam trawl and multi-mesh gillnets in (a) Tuggerah Lake (K-S test, $D = 0.84$, $P < 0.001$), (b) Lake Macquarie (K-S test, $D = 0.63$, $P < 0.001$) and (c) St. Georges Basin (K-S test, $D = 0.38$, $P < 0.001$). All data are pooled across sites, periods and years; data from gillnets in Lake Macquarie and St. Georges Basin are pooled across deep and shallow habitats.

In general, the beam trawl was more effective than the gillnets in sampling penaeid prawns (e.g. *M. macleayi*, *M. plebejus and M. bennettae*) and several small species of fish (e.g. *P. sexlineatus*; *Ambassidae* spp. and *S. austrinum*). Although gillnets have been observed to underestimate proportions of small fish in studies elsewhere (Olin and Malinen, 2003; Olin et al., 2009), our study showed that the gillnets caught some small species of fish (e.g. *G. subfasciatus* and *H. castelnaui*), together with a relatively wide size-range of several fishes of commercial and recreational importance (e.g. *M. cephalus*, *L. argentea*, *G. tricuspidata* and *S. ciliata*), which were mostly absent in catches from the trawl.

Differences in assemblages of fauna between the beam trawl and the gillnets may be explained by a combination of factors, such as the selectivity of the gears (Hamley, 1975) and behaviour of organisms. For example, the ability of penaeid prawns to avoid trawls is limited (Broadhurst, 2000). It is also likely that many of the smaller species of fish were unable to avoid being captured in the beam trawl. Conversely, such species were probably not captured in the gillnets owing to slow swimming speeds (Finstad et al., 2000), a lower probability of encountering the gear (Prchalová et al., 2008), or by passing through the smallest sizes of mesh (Hamley, 1975).

Several of the species of fish caught predominately in the gillnets probably avoided capture in the trawl by moving out of the trawl mouth (Wardle, 1986) or the path of the vessel (Ona and Godo, 1990); or by swimming in front of the trawl for the entire duration of the tow (Bethke et al., 1999). Although short tows may affect the catchability of large or fast-swimming fish in trawls (Godo et al., 1990), our earlier pilot work showed that tows longer than 5 min (i.e. 10 and 20 min) did not catch larger numbers, proportions or sizes of many of the species caught mainly in the gillnets here (see Rotherham et al., 2008b). Indeed, in the present study, one species (i.e. *M. cephalus*) was not captured in the trawl even at relatively small sizes (i.e. <20 cm) despite being caught in the gillnets, which suggests that individuals were able to actively avoid the trawl as juveniles and sub adults.

Although sampling with both methods covered a similar range of depths, unlike sampling with the gillnets, sampling with the trawl was not stratified by deep and shallow areas (see above). Differences between the trawl and the gillnets in some of the estuaries (i.e. Clarence River, Lake Macquarie and St. Georges Basin) may, therefore, be confounded by differences in the proportion of deep and shallow areas sampled by the trawl. While our study was not initially designed to examine differences between sampling methods, an ideal comparative experiment would have attempted to sample the same proportions of deep and shallow areas with both gears in each estuary. Notably, however, there were clear differences in assemblages and size-frequencies of fauna between the trawl and the gillnets in Wallis Lake and Tuggerah Lake, where sampling with both methods was done in depths less than 2 m.

While some species were caught in both gears, differences in size-frequencies between the beam trawl and the gillnets depended on the particular estuary sampled. For example, patterns in size-frequencies of *S. ciliata* (Tuggerah Lake) and *A. australis* (Tuggerah Lake, Clarence River and Wallis Lake) were consistent with our hypothesis that smaller fish would be caught in the beam trawl and larger fish in the gillnets. Although there was some evidence of a similar pattern for *A. australis* in other estuaries (i.e. St Georges Basin and Lake Macquarie, Fig. 4), relatively few individuals of *S. ciliata* or *A. australis* were caught in the trawl in St Georges Basin (Figs. 3 and 4); and in Lake Macquarie, the trawl caught some of the largest individuals of *A. australis* observed during the study. These inconsistent results may be explained by some physical or biogenic features of habitat (Thrush et al., 2002) that improved the capture of certain species and sizes of fish. Alternatively, the results might also be explained by differences in the proportion of shallow and deep areas sampled by the trawl in each estuary (see above).

Our study has important implications for the design of pilot studies examining differences between or among sampling methods. First, although assemblages appeared to be different between the gillnets and the beam trawl in St. Georges Basin and Wallis Lake (illustrated in Fig. 2c for St. Georges Basin), significant interactions between the factors 'method' and 'period' were detected by PERMANOVA (Table 1b and c). This means that differences between the gillnets and the trawl depended on the particular period in which sampling was done within a year. The reasons for these significant interactions were unclear, but in St. Georges Basin they may have been related to potential differences in the proportion of deep and shallow areas sampled by the trawl. Nevertheless, this was not a problem in Wallis Lake because sampling with both gears was done in depths less than 2 m (see above). The results highlight the potential for pilot studies done within a small period of time (i.e. a 3-month period) to give misleading conclusions about the generality of differences between sampling methods, over periods of time longer than 3 months.

Second, if a pilot study had been done only in St. Georges Basin, even over a relatively long period of time (in this case, two years), one might conclude that the beam trawl was not effective at sampling small individuals of *S. ciliata*. Results from Tuggerah Lake, however, showed that relatively large proportions of small *S. ciliata* were captured in the beam trawl – they were just not captured (or available to be captured) at the sites sampled in St. Georges Basin. Similarly, conclusions about the effectiveness of the beam trawl and the gillnets for sampling *A. australis* would probably have been different if a pilot experiment had been done in only one of these estuaries. Further, some species of fish were important in distinguishing between sampling methods in only one estuary. The results demonstrate that care must be taken to ensure that pilot studies are replicated over spatial and temporal scales relevant to the aims and hypotheses of a longer-term or larger-scale study.

Equally importantly, our study illustrates that the suitability of the beam trawl and the gillnets as sampling tools depends on the objectives of a study, the species (or assemblages of species) of interest and the hypotheses to be tested. If the focus of a study involved testing hypotheses about penaeid prawns, or perhaps, juveniles of *A. australis*, the beam trawl would be an appropriate method of sampling. If, however, the focus was on species of Mugilidae (e.g. *M. cephalus* and *L. argentea*), then sampling with the gillnets would be required. The most complete picture of assemblages and populations of fish and invertebrates required sampling with both methods.

Our results are consistent with several studies that have recommended a combination of methods for sampling broad components of assemblages of fish (Lincoln-Smith, 1989; Olin and Malinen, 2003; Watson et al., 2005; Morrison and Carbines, 2006; Eros et al., 2009; Olin et al., 2009). Indeed, sampling with additional methods, such as small-mesh seines (e.g. Gray et al., 1996), would be required to provide a more comprehensive picture of assemblages of fauna in estuaries of NSW. Nevertheless, obtaining comprehensive and absolute estimates of the diversity and abundances of assemblages of fish and invertebrates (i.e. how many individuals of each species live in a given volume of a lake at a particular moment in time, after Kubecka et al., 2009) in estuaries of NSW was beyond the scope of our study. The focus of our research was on testing hypotheses about *relative* changes in populations and assemblages of fish and invertebrates. The assemblages sampled in the gears examined here included a diverse range of taxa (>100 spp.), including many species of importance in commercial and recreational fisheries.

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Appendix 8. Experiment 6: Utility of fishery-independent sampling strategy to reduce uncertainty in management decisions

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Reducing uncertainty in the assessment and management of fish resources following an environmental impact

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The limitations of using information from commercial fisheries for assessing and managing resources and ecosystems are well known. Although fishery-independent data may overcome many such limitations, few studies have examined how incorporating data from different sources affects assessments and subsequent management decisions. Here, the value of integrating data from two types of sampling survey to assess the recovery of faunal populations following a severe fish-kill event in the Richmond River (New South Wales, Australia) in 2008 is evaluated. There is occasional large-scale mortality of fish and invertebrates in certain estuaries of eastern Australia following major flood events. In extreme cases, the management response involves closing an entire estuary to all fishing, to facilitate the recolonization and recovery of fish and other fauna. Decisions to resume normal fishing activities have environmental, economic, and social implications. Using lessons learned from a similar fish-kill event in 2001, it is shown how, in 2008, fishery-independent sampling, combined with improved sampling by commercial fishers, reduced uncertainty in decision-making and led to greatly improved socio-economic outcomes for stakeholders. The work highlights the need to examine the value of different sources of information to improve management decisions.

Keywords: adaptive management, decision-making, fishery-dependent data, fishery-independent survey, fish kill, uncertainty.

Introduction

In many parts of the world, the assessment and management of fish resources rely on fishery-dependent information, e.g. landings, catch per unit effort (cpue), and size and age compositions of catches. Nevertheless, because of well-known biases and limitations, such data are often unreliable or inappropriate for either traditional assessments of harvested species, or ecosystem-based approaches to management (Hilborn and Walters, 1992; Harley *et al.*, 2001; Maunder *et al.*, 2006; Cotter *et al.*, 2009). Therefore, assessments and management decisions underpinned by fishery-dependent data are frequently confounded by uncertainty (Sissenwine, 1984; Ludwig *et al.*, 1993; Charles, 1998). This problem has sometimes contributed to declines in fish populations through erroneous recommendations (Sinclair and Murawski, 1997; Maunder *et al.*, 2006), and perhaps more commonly, through the inability or unwillingness of managers and policy-makers to act with precaution in the face of uncertain information and political pressure (i.e. the ratchet effect; see Ludwig *et al.*, 1993).

Strategies for reducing uncertainty and improving decision-making in fishery management often call for the collection of more and better data (Rose and Cowan, 2003; Mace, 2004; Murawski *et al.*, 2008). Fishery-independent data are commonly advocated (Pennington and Strømme, 1998; Maunder *et al.*, 2006), along with improvements in the quality and quantity of fishery-dependent information (National Research Council, 2000). Yet, it is often unclear whether more or better data actually reduce uncertainty and lead to different or improved decisions. Management has failed and fish stocks collapsed despite substantial investments in the collection and integration of both fishery-dependent and fishery-independent data into state-of-the-art assessments (Walters and Maguire, 1996; Myers and Worm, 2003). Therefore, problems of uncertainty are not necessarily resolved by collecting additional data or improving scientific research (Ludwig *et al.*, 1993; Johannes, 1998).

Underwood (1995, 1998) argued that more research was needed on the consequences of uncertainty in environmental management and decision-making. He examined relationships between ecological research and decision-making in response to environmental disturbances, and identified several categories of research that interact in their effects on management decisions (Figure 1). He argued that information in the available pool (category 1 research) sought by managers needing to solve a problem (e.g. declining cpue in a fishery) was often inadequate or uncertain because other types of important research were rarely, if ever, done. These included (i) examining the success or failure of management by treating management decisions as testable hypotheses using manipulative experiments (category 2 research, i.e. adaptive management experiments; Hilborn and Walters, 1981; Walters, 1986), (ii) developing novel, strategic programmes to provide new information when management fails to solve a problem (category 3 research), and (iii) investigating the processes of management and the procedures by which decisions are reached using information already available under category 1 (category 4 research).

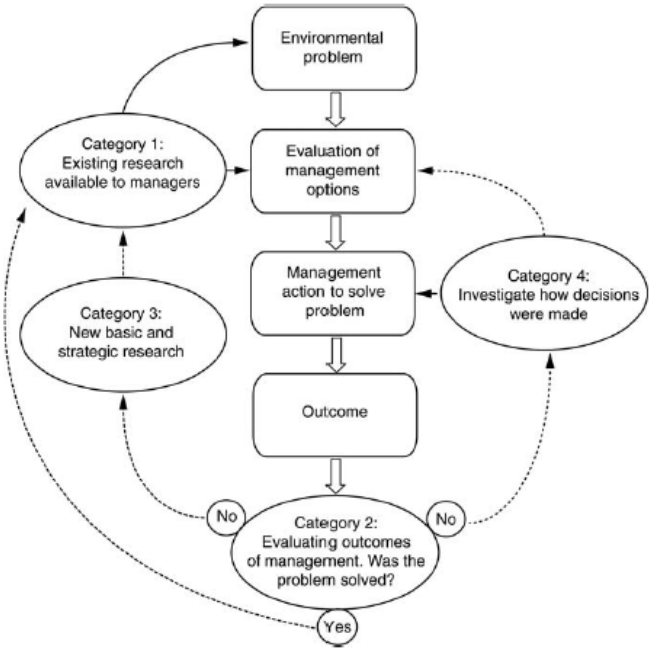

Figure 1. Some relationships between different types of research and the management of an environmental problem as outlined by Underwood (1995, 1998). The four categories of research are: (i) available research sought by managers to solve a problem; (ii) research done to examine the success or failure of management by treating their decisions as testable hypotheses; (iii) novel, strategic research to provide new information when management fails to solve a problem; and (iv) research on the processes of management and the procedures by which decisions are reached using information provided under category 1. Management steps are connected by open arrows, solid arrows show the connections of research to management, and dotted arrows indicate feedbacks and interconnections among the research categories.

By doing more research under categories 2, 3, and 4, Underwood (1995, 1998) argued that the problems of uncertainty might eventually be resolved by providing the necessary information for more effective management of environmental problems. Unfortunately, there seems to have been little progress in these areas of research in either aquatic (Walters, 2007) or terrestrial (Gosselin, 2009) environments. Adaptive management experiments have proven difficult to implement (Walters, 2007), and there are few real-world examples of research into the process of decision-making. Hence, decisions about whether to make the most of existing information, invest in alternative sources of data, or allocate limited resources to management actions rather than the collection of data are often complex and contentious (Kelly and Codling, 2006; Hansen and Jones, 2008a, b; Murawski *et al.*, 2008).

Here, we illustrate how elements of the research categories advocated by Underwood (1995, 1998) improved the assessment and management of a large fish-kill event in the Richmond River, New South Wales (NSW), Australia, in 2008. Using an adaptive approach incorporating lessons learned from a previous fish kill in 2001 (i.e. category 2 research), we show how new strategic research (i.e. category 3) led to an additional source of information (in category 1), so reducing uncertainty in decision-making and leading to better socio-economic outcomes for stakeholders. Our *a posteriori* evaluation of the different sources of data in assessing and managing large fish kills is an important step towards much-needed research into the process of decision-making about environmental issues (i.e. category 4 research), including the sustainability of fishery resources.

Case study: fish kills in the Richmond River, NSW

The Richmond River (28°52′S 153°34′E) is a large river (~170 km long) on the north coast of NSW that flows into a wave-dominated barrier estuary (Roy *et al.*, 2001) that is permanently open to the ocean. The river (surface area ~19 km^2) and its catchment (~6850 km^2) support valuable commercial and recreational fisheries together with agricultural production and tourism; these are of considerable social and economic importance to the region.

In early February 2001, a series of extreme storms in northern NSW caused widespread flooding throughout the Richmond River catchment. About 1 week later, there was a major fish kill in the lower reaches of the river on a scale rarely seen before in Australia (Macbeth *et al.*, 2002; Walsh *et al.*, 2004). The massive mortalities of finfish and invertebrates were attributed to the drainage of anoxic waters from floodplains, creeks, and riparian swamps into the main river channel. It was hypothesized that deoxygenation of slow-draining floodwaters was caused by strong solar radiation, high temperature, and the decay of floodplain and riparian vegetation (Slavich, 2001; Macbeth *et al.*, 2002; Walsh *et al.*, 2004). Although millions of fish and invertebrates were killed (Walsh *et al.*, 2004), it appeared that many organisms had migrated or were flushed from the river during the flood (Macbeth *et al.*, 2002).

Immediately following the fish kill (9 February 2001), the NSW Government closed the Richmond River and adjacent, inshore oceanic waters to all fishing. Hence, the management hypothesis (*sensu* Hilborn and Walters, 1981; Underwood, 1995) was that removing commercial and recreational fishing pressures would enhance the recolonization of these areas by fish and invertebrates when the water quality improved. This hypothesis was not tested experimentally during the post-fish-kill monitoring owing to a lack of resources to achieve appropriate levels of replication. Similarly, it was not possible to carry out an experiment to test for the impact of the fish-kill event using an appropriate before–after–control–impact (BACI) design (Underwood, 1991), because there were insufficient "before" data available for either the Richmond River or other non-impacted (control) estuaries (Macbeth *et al.*, 2002). The decision on when to reopen the commercial fishery relied solely on the monitoring of spatial and temporal patterns in the distribution and abundance of fish and invertebrate populations in the Richmond River after the fish kill (Macbeth *et al.*, 2002).

Given the urgent need to implement a programme to monitor the recovery of the Richmond River, most of the sampling of fish and invertebrates was done by commercial fishers (instructed and supervised by scientific staff) using various commercial fishing gear, including gillnets, seines, and traps (see Macbeth *et al.*, 2002, for detail). Following the closure of the river, post-fish-kill sampling surveys were carried out in mid- and late February 2001. These indicated that water quality was very poor and abundances of fish and crustaceans extremely low (Macbeth *et al.*, 2002), so the fishing closure was extended pending the results of subsequent four-weekly surveys.

Parts of the Richmond River were eventually reopened to limited recreational or commercial fishing activities on 1 July 2001. A four-month survey was then implemented to assess the quality of recreational fishing following the partial lifting of the total closure (Steffe *et al.*, 2007), additional to the four-weekly sampling surveys. Large catches were made during the survey of recreational fishing, along with favourable population assessments from the August 2001 sampling survey, so the river was fully reopened to normal fishing activities on 28 September 2001 (i.e. more than six months after the fish kill).

Lessons learned from the 2001 response to the fish kill

The response to the 2001 fish kill in the Richmond River involved successful collaboration among fishers, scientists, and other stakeholders in implementing a sampling programme to assess and monitor the recovery. Nevertheless, given the unprecedented nature of the fish kill and lack of appropriate data available before the impact, from either the river or any control estuaries, there was considerable uncertainty surrounding decisions about when to reopen the river to normal fishing activities. Anecdotal evidence from commercial and recreational fishers, and the broader community, indicated that the adopted precautionary approach to management had serious socio-economic implications. Many fishers, local businesses, and tourism operators suffered financial hardship as a result of the extended fishing closure. Therefore, there were important lessons in the assessment and management of this environmental impact (*sensu* category 2 research; Underwood, 1995) that could be used to improve environmental and socio-economic outcomes in the event of similar fish kills in future.

The first lesson was that the monitoring programme was too inflexible regarding when and where the sampling took place (Macbeth *et al.*, 2002). As discussed above, the programme could not determine whether populations of fish and crustaceans in the river had recovered to pre-flood levels. Instead, it attempted to address the hypothesis that populations had recovered (or were recovering) to levels that could support the resumption of normal fishing activities. During the initial sampling surveys, however, there was high flood-induced turbidity in the river. As the four-weekly sampling surveys continued (and turbidity decreased), surface-set gillnet catches by day remained low, whereas those using other fishing methods indicated that populations were recovering. The fishers maintained this was because gillnetting was being done in the wrong places at the wrong time of day, given the improved water conditions (i.e. the catches would have been low even if there had been no fish kill). Indeed, extra daylight and night-time gillnetting some months into the programme (at sites selected by commercial fishers) confirmed the presence of large numbers of fish in the river despite poor catches during the regular daytime gillnetting (Macbeth *et al.*, 2002). Therefore, it was recommended that in monitoring future fish-kill events, sampling by commercial fishers should be more flexible in terms of when and where it was done, to improve its relevance to the management questions being posed (Macbeth *et al.*, 2002).

The second lesson was that regular fishery-independent surveys were required in estuaries and rivers of NSW to provide appropriate data on the status of populations of fish and crustaceans in times of relative health (Macbeth *et al.*, 2002). It was recommended that new research be undertaken to develop reliable and robust sampling methodologies and regimes (i.e. category 3 research; Underwood, 1995) to allow appropriate comparisons should a fish kill (or other environmental impact) take place in future. This had potential benefits not only by reducing the uncertainty in decisions for managing impacts such as fish kills, but also in improving the overall assessments and management of estuarine fish resources in NSW (Macbeth *et al.*, 2002).

Using lessons from 2001: assessing and managing a fish kill in 2008

In early January 2008, there was another severe flood-induced fish-kill event in the Richmond River. Processes contributing to the deoxygenation of floodwaters were the same as those implicated in the 2001 fish kill (see above). Further, testimony from fishers, scientists, and managers indicated that the 2008 fish kill was of a similar magnitude to that of 2001, i.e. that millions of organisms were killed. On 18 January, the river was closed to all fishing activities. We consider the 2001 and 2008 fish-kill events to have been comparable because they (i) were at similar times of the year, (ii) were caused by similar processes, and (iii) resulted in large-scale mortalities of fish and invertebrates that forced the closure of commercial and recreational fisheries.

Using lessons from 2001, the recovery assessment in 2008 proceeded quite differently. About one month after the closure, i.e. six weeks after the fish kill, two separate sampling surveys were carried out over a period of ~one week. The first was by commercial fishers and is described here as fisher-dependent sampling, because the fishery *per se* was closed. All commercial fishers operating in the river were contracted to provide boats, sampling equipment, and expertise. During the 2001 event, in contrast, only a few commercial fishers participated in and benefitted financially from the monitoring programme.

Fisher-dependent sampling

Fisher-dependent sampling used local commercial-fishing methods (fish- and prawn-seining, crab- and eel-trapping, and gillnetting) throughout the lower Richmond River (Figure 2) to examine whether catch rates of fish and crustaceans were similar to those when the river was reopened to normal fishing after the fish kill in 2001. For the purposes of spatial and temporal comparisons, the lower Richmond River was partitioned into four distinct sections: Ballina (B), Lower (L), Middle (M), and Upper (U) sections (Figure 2). The various commercial fishing methods were used in some or all these sections (depending on suitability and practicality), with a sampling design similar to that used in 2001 (Table 1). The main differences in 2008 were (i) both surface set and demersal gillnetting were done at night, (ii) fish-seining was included, and (iii) fishers were allowed to apply their knowledge and experience in selecting the sampling sites.

Fishery-independent sampling

The second type of sampling survey was by Government researchers using multimesh gillnets specially designed to catch a wide size range and diversity of fish fauna, including categories not caught in commercial gillnets, e.g. undersized fish (Rotherham *et al.*, 2007; Gray *et al.*, 2009). Although much pilot work had been done on the development of fishery-independent sampling tools and strategies (i.e. category 3 research), a standardized large-scale, long-term survey had not yet been implemented in NSW. Nevertheless, having these tools allowed specific hypotheses to be tested while monitoring the recovery of the river.

Much anecdotal evidence from commercial and recreational fishers suggested that in the days following the 2008 fish kill, larger numbers and a greater diversity of fish were present near the river entrance (i.e. the Ballina section) than at sites farther upstream. Such patterns may indicate some effect of the flood; either fish were displaced from upstream sites nearer to the river entrance or fish entering the river did not move upstream because of unfavourable conditions there (or some other unknown reason). Therefore, the fishery-independent sampling was subsequently designed to test the hypothesis that the diversity and the abundance of fish were different among sections of the river. There were two randomly selected sampling sites in the Richmond River (separated by at least 1 km and considered independent) nested within each of three river sections (B, L, and M, Figure 2). At each site, there were three replicate gillnet samples (separated by 50–100 m and considered independent) in each of two habitats: shallow (<2 m); deep (>4 m). Gillnets were bottom-set at night, soaked for 1 h, then retrieved (Rotherham *et al.*, 2006). It took one night to sample a site, and the order in which sites and habitats were sampled was randomized. Standardized fishery-independent sampling was also done concurrently elsewhere in NSW to benchmark the catch rates of key species between the Richmond River and other estuaries not affected by major flooding. The estuaries were Lake Macquarie and St Georges Basin, which are both large, well-mixed coastal lakes with no distinct salinity gradients. In each estuary, four sites one to several kilometres apart were selected randomly for sampling (Gray *et al.*, 2009), again with three replicate gillnet samples in both shallow and deep habitats.

Summary of the 2008 results

Data collected during the 2008 fisher-dependent survey showed that populations of fish and crustaceans throughout the lower river channel had recovered to levels similar to or exceeding those recorded immediately before the reopening of the commercial fishery in 2001. The survey demonstrated that economically important species such as mud crab (*Scylla serrata*, Figure 3), sea mullet (*Mugil cephalus*), yellowfin bream (*Acanthopagrus australis*), dusky flathead (*Platycephalus fuscus*), and sand whiting (*Sillago ciliata*) were present in large numbers. Hence, the populations in the river had recovered quickly to levels that were considered sufficiently sustainable to support the recommencement of commercial

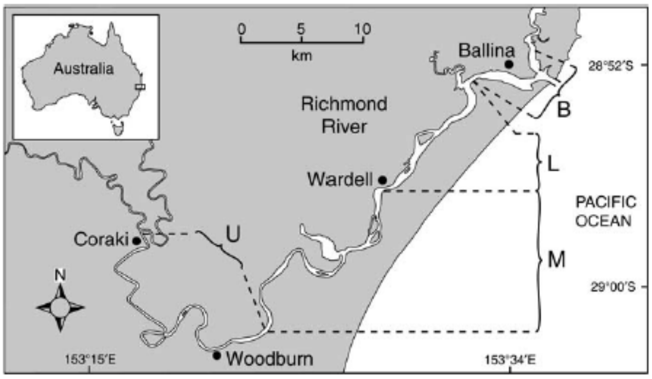

Figure 2. Spatial partitions of the lower Richmond River (NSW, Australia) used for the 2008 post-fish-kill sampling surveys. The four sections are Ballina (B), Lower (L), Middle (M), and Upper (U).

and recreational fishing. In particular, sizable and diverse fish catches resulted from seining in the lower and middle sections. Notably, the most successful haul caught 3 t of finfish, consisting of some 11 000 individuals and 15 different species.

Some other examples also illustrate the swift recovery of the riverine populations. The mean catch of mud crabs per trap-set in the lower section during the February 2008 survey compared favourably with that recorded in September 2001, i.e. immediately before reopening the commercial fishery, and in February 2002, i.e. at the same time of year as the 2008 fisher-dependent survey (Figure 3a). Similarly, the mean weight of mud crabs per trap-set (i.e. the cpue) during the February 2008 fisher-dependent survey was similar to that recorded by commercial fishers in February of each year for the past decade (derived from commercial catch records for the Richmond River; Figure 3b). Finally, the mean number of species caught per prawn-haul during the February 2008 survey was generally similar to that recorded during each of the September 2001 and February 2002 surveys in the upper, middle, and lower sections (Figure 4).

Analyses of data from the fishery-independent survey found no significant differences (ANOVA, $p > 0.05$) in the total numbers and species of fish and crustaceans among the lower three sections of the main river channel (Figure 5a and b). These patterns were consistent for both shallow (<2 m) and deep areas (>4 m). Similar results were observed for most of the key economically important species, e.g. yellowfin bream, dusky flathead, luderick, and silver biddy (*Gerres subfasciatus*). Numbers of sea mullet were, however, significantly greater ($p < 0.05$) in the middle (M) section than downstream (Figure 5c), contradicting the assumption that more fish would be found at the entrance to the river. For several analyses (total numbers of fish, numbers of sand whiting, and numbers of silver biddy), there were significant differences ($p < 0.05$) between sites within different sections of the river (an example is shown for sand whiting in Figure 5d). There was, therefore, generally more variation between sites within a section than among the means of the different sections.

The results of fishery-independent surveys carried out concurrently in other NSW estuaries indicated more species than were sampled in the Richmond River (Figure 6). Nevertheless, the total numbers of fish and the abundances of several economically important species, e.g. sea mullet and yellowfin bream, were similar among the different estuaries (Figure 6).

Table 1. Number of replicate sampling units (i.e. net-sets/hauls/trap-sets) for each of the fisher-dependent sampling methods used in the four sections of the Richmond River during February 2008.

Fisher-dependent sampling method	Section of Richmond River			
	Ballina	Lower	Middle	Upper
Demersal gillnet	3	3	3	3
Surface gillnet	3	3	3	3
Boat-based demersal fish-seine	–	2	1	–
Boat-based demersal prawn-seine	–	6	6	6
Mud-crab trap	–	20	20	–
Eel trap	–	–	20	20

Management outcomes

Following assessment of the data from the two types of sampling survey, the Richmond River was reopened to normal fishing activities after a closure of some six weeks. There were no patterns in the data to justify a lengthier closure. The fisher-dependent sampling confirmed the hypothesis that there were large and diverse fish catches in the river. Moreover, the fishery-independent sampling did not detect any differences in the diversity and abundance of fish among the various sections, and the catch rates of fish in the river were of similar magnitude to those in other NSW estuaries sampled over the same period.

Discussion

The case study reveals that fishery-dependent information is not always sufficient for assessing the status of fish populations or providing advice for management and policy-making (Cotter *et al.*, 2009). In some parts of the world, the quantity of data obtained from commercial fisheries is decreasing (or no longer available) owing to changes such as the creation of marine protected areas (Castilla, 2000; Lubchenco *et al.*, 2003), and zones where only

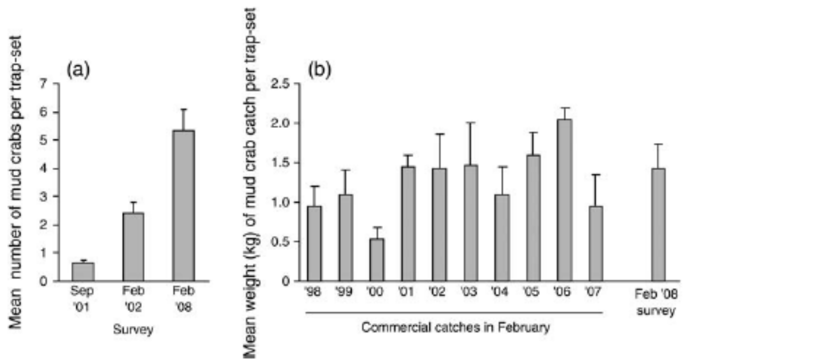

Figure 3. (a) Mean number of mud crabs per overnight trap-set (+s.e.) from fisher-dependent sampling in section L during September 2001, February 2002, and February 2008 ($n = 30$, 26, and 20 trap-sets, respectively); (b) mean weight of mud crabs per overnight trap-set (+s.e.), sections L and M combined. Results from the February 2008 fisher-dependent sampling ($n = 40$ trap-sets) are compared with equivalent data from the commercial fishery during the month of February in each year of 1998–2007 ($n = 2–7$ fishers).

recreational fishing is allowed (Einarsson and Gudbergsson, 2003; Steffe *et al.*, 2005; Lauer *et al.*, 2008). In these cases, making the most of fishery-dependent information is a concept that is becoming less relevant or, as in our example (and in fisheries that have always been data-poor), impractical.

Our example supports Underwood's (1995, 1998) view that the issue of uncertainty may be solved best by providing information that leads to more-effective management of fish resources and their environment, which requires different types of research. Although we were unable formally to test any management hypotheses, our approach was adaptive (*sensu* Walters, 1986), and it incorporated elements of the research categories advocated by Underwood (1995, 1998). This included using lessons learned from the 2001 fish kill in the Richmond River (category 2 research) to establish new research to develop fishery-independent sampling gears and strategies (category 3 research), and providing the tools and information (category 1) needed to improve the assessment and management of a similar fish kill in 2008. The different management outcomes between the two fish-kill events in the Richmond River illustrate the improved effectiveness of assessment and increased confidence in decision-making. In 2008, integrating the results of improved sampling carried out by commercial fishers, together with new data from fishery-independent sampling, led to the river being reopened to normal fishing activities after ~6 weeks. It was more than six months before the river was completely reopened to fishing after the fish kill in 2001.

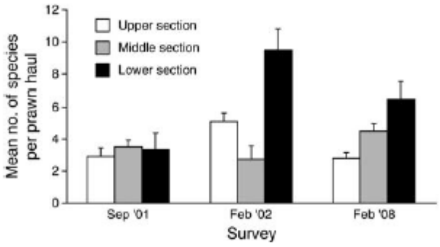

Figure 4. Mean number of species recorded per prawn-haul (+s.e.) during the September 2001, February 2002, and February 2008 fisher-dependent sampling. Results are given for sections U, M, and L ($n = 5-10$ hauls).

Anecdotal evidence from stakeholders indicated that improved assessment and management of the 2008 fish kill also gave better socio-economic outcomes. These included the obvious benefits of a relatively short fishing closure, as well as the hiring of commercial fishers to assist with sampling. Therefore, although the river was reopened much earlier than in 2001, there was no ratchet effect (*sensu* Ludwig *et al.*, 1993), because the commercial fishers had some income from their sampling effort during the closure. A more-precautionary approach was justified in 2001 because that fish kill was unprecedented and there was a lack of reliable data on which to base management decisions (Macbeth *et al.*, 2002).

The long-term effects of the 2008 fish kill and subsequent fishing closure on populations and assemblages of fish in the

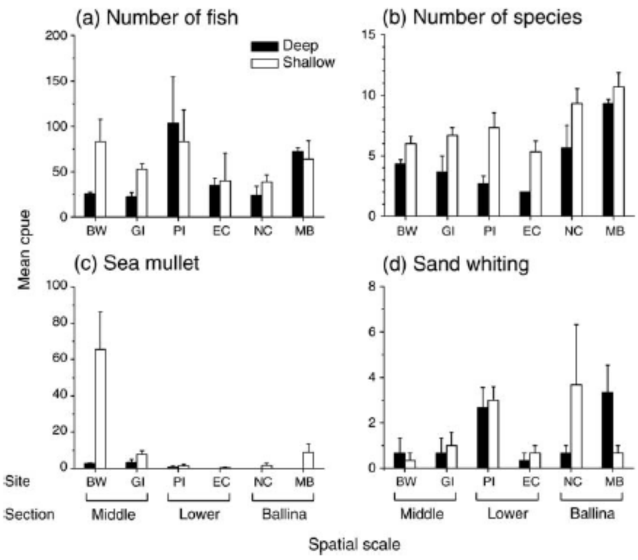

Figure 5. Mean cpue (number caught per multimesh gillnet per hour, +s.e.) of the total numbers of (a) fish, (b) species, (c) sea mullet, and (d) sand whiting caught in deep (>4 m) and shallow (<2 m) habitats at two sites in each of the Middle (BW = Broadwater; GI = Goat Island), Lower (PI = Pimlico Island; EC = Emigrant Creek), and Ballina (NC = North Creek; MB = Mobbs Bay) sections of the Richmond River in February 2008, using multimesh gillnets ($n = 3$ in each depth at each site) in fishery-independent sampling.

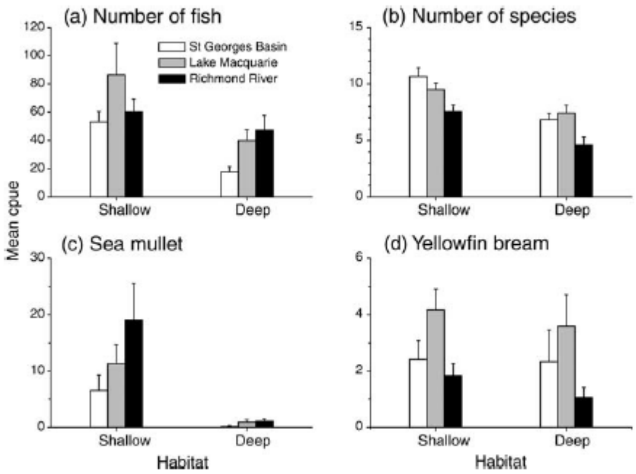

Figure 6. Mean cpue (number caught per multimesh gillnet per hour; +s.e.) of the total numbers of (a) fish, (b) species, (c) sea mullet, and (d) yellowfin bream caught in multimesh gillnets in shallow and deep habitats of St Georges Basin ($n = 12$), Lake Macquarie ($n = 12$), and the Richmond River ($n = 18$) during February 2008 in fishery-independent sampling.

Richmond River remain unknown. Nevertheless, fishery-independent sampling has since continued in the river (and in other estuaries in NSW) to examine its longer term recovery and to address the question of whether its relatively early reopening led to poor environmental outcomes (i.e. additional category 2 research). It is worth noting, however, that reports from commercial and recreational fishers indicate no evidence of reduced stocks of fish in the river since the 2008 reopening.

Although this investigation was not a controlled experiment, it provides an *a posteriori* comparison of the outcomes of management decision-making with and without different sources of data. We consider this to be an important element of category 4 research (Underwood, 1995), i.e. examining how particular starting information contributes to a management decision and consequent actions. Despite the potential for inherent differences between the fish kills in 2001 and 2008 to confound any differences in decision-making, the two events were similar with respect to their timing, magnitude, and initial management response, i.e. closing the river to all fishing. The benefits of integrating fishery-independent data into the assessment and management of short-term environmental impacts, such as the NSW fish kills, should encourage managers to make more use of such data in future. On the other hand, the value of fishery-independent data in routine and long-term assessment and management of estuarine fish resources in NSW, which currently rely on fishery-dependent information, is not yet clear.

Determining the relative value of different types of data in achieving long-term, sustainable management of fish resources and biodiversity requires planned experiments. Novel experimental procedures are needed to test hypotheses about whether different sources of data lead to different management decisions. A similar adaptive, experimental approach has been advocated in evaluating trade-offs in allocating resources between data collection and other actions that could support (or improve) fishery management (Hansen and Jones, 2008a).

Nevertheless, any sort of adaptive-management experiment requires careful attention to its design and sampling procedures (Walters and Holling, 1990; Underwood, 1995). Often, however, data from commercial fisheries are not collected with sufficient rigour to ensure their independence, adequate controls, and replication at relevant spatial and temporal scales (Castilla, 2000). Addressing such problems is perhaps the best way of making the most of available fishery information. This might allow a more experimental approach to the management of fisheries (Holling, 1978; Walters, 1986), a concept that has been strongly advocated (but largely ignored) for more than three decades (Walters, 2007).

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