Developing fishery-independent surveys for the adaptive management of NSW's estuarine fisheries

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This project was a co-operative study between the Australian Fisheries Research and Development Corporation (FRDC), the Wild Fisheries Program of the NSW Department of Primary Industries, the Centre for Research on Ecological Impacts of Coastal Cities at The University of Sydney. We thank members of the Scientific Advisory Committee for their advice and support throughout the project. Thanks are also extended to Dr Matt Broadhurst for his invaluable contribution towards many aspects of the research. Alan Bodycote, John Brown, Gary Howard are thanked for providing technical assistance, advice and use of their vessels. We are also very grateful to the staff that assisted with fieldwork namely Paul Lokys, Lachlan Barnes, Damien Young, Ben Kendall and James McLeod. We also thank: Dr Steve Kennelly for his advice and support throughout the project; Dr Will Macbeth for assistance with selectivity analyses; Darren Reynolds for reviewing an earlier draft; and the numerous other staff at NSW DPI that provided assistance along the way.

NON-TECHNICAL SUMMARY

2002/059	Developing fishery-independent surveys for the adaptive management of NSW's estuarine fisheries

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OBJECTIVES:

1. Develop scientific sampling tools to catch the widest possible size range and diversity of fish species in NSW's estuaries.
2. Use the gears developed in objective 1 to do pilot studies to determine the most cost-effective, optimal number of replicates, sites, locations and habitats to be sampled in and among estuaries.
3. Use the results from objectives 1 and 2 to design the optimal sampling regime that will become the long-term, large-scale survey of the fish populations in NSW estuaries.

NON TECHNICAL SUMMARY:

Outcomes Achieved

Using an experimental strategy, this project successfully developed methods to use multi-mesh gill nets and a beam trawl as tools for sampling a wide size range and diversity of fish and crustaceans in estuaries of NSW. Experiments also measured spatial and temporal variation of estuarine fauna sampled with these gears and provided variances for cost-benefit analyses. Owing to the variable and dynamic nature of these fauna, however, the project has identified numerous challenges that require the development of novel analytical approaches (which are underway), before a large-scale and long-term survey can be designed and optimised across multiple species. Additional research is also needed to compare decision-making incorporating fishery-independent methodologies against that using data from fishery-dependent sources. This will identify the most reliable, robust and cost-effective sampling programmes required to improve the sustainability of NSW's estuarine fisheries resources.

Estuaries in NSW support a wide diversity of fish and invertebrates, which are subject to commercial and recreational fishing and a range of other human-induced pressures (e.g., habitat degradation, pollution). At present, monitoring of these aquatic resources relies entirely on data from fishery-dependent sources. This includes information provided by commercial fishers about how long they spend fishing and the quantities of each species that are retained. Other data, such as the size-distributions and ages of a few species, are obtained from commercial landings by researchers at ports and markets. Periodic surveys of recreational fishers and sampling of their catches are also done for a small number of estuaries in NSW.

The limitations of using fishery-dependent data for the assessment and management of fisheries resources, particularly in accordance with principles of ecologically-sustainable development (ESD), are well-known. For example, these data: (i) provide no information on species that are

discarded or not caught (e.g., undersize fish and species of no commercial value); (ii) often are not available in estuaries that are closed to fishing or where fishers' choose not to fish; and (iii) are biased and imprecise because of the selectivity of fishing gears, reporting errors and the varying practices and levels of skill across many different fishers. So, the future use of fishery-independent surveys has been advocated to provide a potentially improved quality of information on the status of estuarine fisheries resources in NSW.

Data collected independently of commercial and recreational fisheries using standardised, properly-designed research surveys, are generally less susceptible to the limitations of fishery-dependent data. Sampling gears and surveys are not, however, always properly designed and so can provide data that are also probably biased, imprecise, and costly. The preferred approach for developing reliable and cost-effective surveys is an experimental one, which uses pilot studies to test the design and deployment of sampling gears. This led to the present study, which was to do the necessary pilot work to develop scientific tools and designs of sampling for large-scale and long-term survey of fisheries resources in estuaries of NSW.

Based on key literature on surveys of fisheries resources and experimental design, we developed and used a general experimental strategy to meet the specific objectives of the project. Briefly, the strategy involved: (i) identifying suitable gear to sample target species; (ii) testing configurations of gear and sampling practices to ensure that samples are optimal, representative and cost-efficient; (iii) understanding spatial and temporal variability; and (iv) cost-benefit analyses to determine optimal levels of replication.

We followed the above strategy for two types of gears: multi-mesh gill nets and a beam trawl. These gears sampled a wide size range and diversity of fish and invertebrates (> 60 species), including many species of commercial and recreational importance (Objective 1). Experiments included a comparison of the utility and efficiency of multi-mesh gill and trammel nets. Gill nets were superior, so we tested the effects of using different lengths of net, soak and setting times. Experiments with the beam trawl tested the effects of factors such as the size of mesh in the body and codend, tow duration and diel period (day vs night). A key result of this work was that replicate gill nets and trawls required only short periods of deployment (compared to commercial practices), which had benefits for improving replication and potentially reducing the number of organisms that were harmed or killed.

After determining an optimal configuration and sampling protocol for each gear, we used the gill nets and beam trawl to measure spatial and temporal variation of estuarine fauna over a range of scales (e.g., zones, sites, days, weeks, months, seasons, etc). Such experiments identify appropriate scales of replication for future studies and allow numbers of replicates to be optimised using standard cost-benefit analyses (Objective 2). In our study, individual species and assemblages were extremely patchy at the smallest spatial scale examined (i.e., among replicates) – a pattern that was consistent across gears, depths and estuaries. There were, however, no consistent patterns of variation among the different temporal scales (e.g., days, weeks, months, seasons) for either method of sampling. The only consistent pattern across different species and gears was that variation among replicates (i.e., small-scale spatial variation) was greater than variation at any temporal scale.

The patchiness of many species among samples often precluded sensible univariate cost-benefit analyses. Although cost-benefit analysis was also done on assemblages of species, it may not be appropriate because it was uncertain how different taxa contribute to an optimal design, or whether it was suitable for all species in an assemblage. Further, current analytical procedures cannot determine (or optimise) the power of sampling for assemblages of species. So, before a large-scale and long-term survey can be designed (Objective 3) new analytical techniques are needed to attempt to develop cost-benefit analyses for multiple species. These new procedures are currently

being developed because, prior to this project, there were no data available to address these problems for assemblages of fish in NSW. Further research is needed to test how to use fishery-independent data in making managerial decisions, the cost and benefits of using such data and whether better outcomes are achieved. This will identify the most appropriate sampling programmes for improving the management of fisheries resources and biodiversity in estuaries of NSW.

KEYWORDS: fishery-independent surveys, estuaries, fish, multi-mesh gill net, beam trawl, cost-benefit analysis, New South Wales

1. BACKGROUND

The management of most commercial and recreational fisheries in NSW estuaries is currently undergoing significant change through a variety of initiatives including the creation of several Recreational Fishing Havens, Marine Parks and Reserves, the buy-out of some commercial fishing effort and the development of Environmental Assessments and Fisheries Management Strategies. In addition to these initiatives, the population density of NSW continues to increase and there have never been greater pressures on our estuaries from pollution, land management practices, etc. Throughout all this change, there is a continuing need to maintain our aquatic resources for current and future generations of commercial and recreational fishers as well as to assess the success of those management initiatives that are designed to ensure the sustainability of those resources.

At present, the assessment of NSW's estuarine commercial and recreational fish stocks relies heavily on fishery-dependent information, primarily the catch and effort information supplied by the commercial sector, augmented with some biological sampling of commercial landings for a few key species. These data have minimal utility for fisheries management because: (i) no information is obtained where no fishing takes place (whether that is due to legislative closures and/or fishers' individual choices); (ii) no information is obtained on the critical undersize portion of the exploited populations (which includes the next generation of targeted fish); (iii) no information is obtained on other species that are considered not worthy of retention in any place/time (for whatever reason); (iv) such information is often inaccurately or imprecisely recorded; and (v) the data obtained has little scientific rigour because commercial and recreational fishing is mostly done in biased, non-independent ways – i.e., the execution of any "replicate" fishing attempt DEPENDS on the results of previous attempts.

Clearly, we need a better way to collect information on our estuarine fisheries resources if we are to manage them sustainably. We also need a better way to assess the success (or failure) of management initiatives that are designed to improve them in adaptive management frameworks (i.e., frameworks that are sufficiently responsive and flexible to respond appropriately to quality scientific information). To do this, we must have data on the relative abundances of fish in estuaries and demographic information (i.e., length, sex, and age composition, reproduction and recruitment dynamics) on the various life-history phases of these populations, including those stages prior to capture. We also need this information in places and times where fishing does and doesn't always occur (i.e., inside closed areas as well as open areas, in places/times where fishers choose not to fish as well as places/times where they do, etc.). This type of data can only be gathered via standardized, fishery-independent surveys.

2. NEED

The above Background explains why it is necessary to develop a standardized fishery-independent sampling strategy to provide estimates of relative abundances and demographies of populations of fish in the estuaries of NSW which will be used in conjunction with existing and any new sources of fishery-dependent data (from commercial and recreational fisheries). Before these surveys can be implemented, however, it is necessary to do several pieces of very important research.

First, the correct sampling tools and methods need to be developed. Whilst we acknowledge that commercial and scientific fishing gears are available, these have been designed to capture very specific species and sizes of species. We need to modify these and other gears to develop new techniques that will sample wider size ranges and diversities of fish than is the case for commercial and recreational fisheries. Specifically, we need to determine the best suite of gears to use to catch as wide a size and species range of fishes as possible in as many different habitats as possible.

Second, once the best tools have been developed, appropriate spatial and temporal scales of sampling and units of replication need to be determined so that an ongoing survey design based on a rigorous sampling protocol can be implemented for the decades to come.

3. OBJECTIVES

1. Develop scientific sampling tools to catch the widest possible size range and diversity of fish species in NSW's estuaries.
2. Use the gears developed in objective 1 to do pilot studies to determine the most cost-effective, optimal number of replicates, sites, locations and habitats to be sampled in and among estuaries.
3. Use the results from objectives 1 and 2 to design the optimal sampling regime that will become the long-term, large-scale survey of the fish populations in NSW estuaries.

4. METHODS

4.1. Study area

Experiments were done in a number of different estuaries in NSW, spanning a distance of almost 700 km along the coast (Fig. 1). More specific details (or links to key references) of each estuary are provided in the relevant manuscripts and summaries in the appendices.

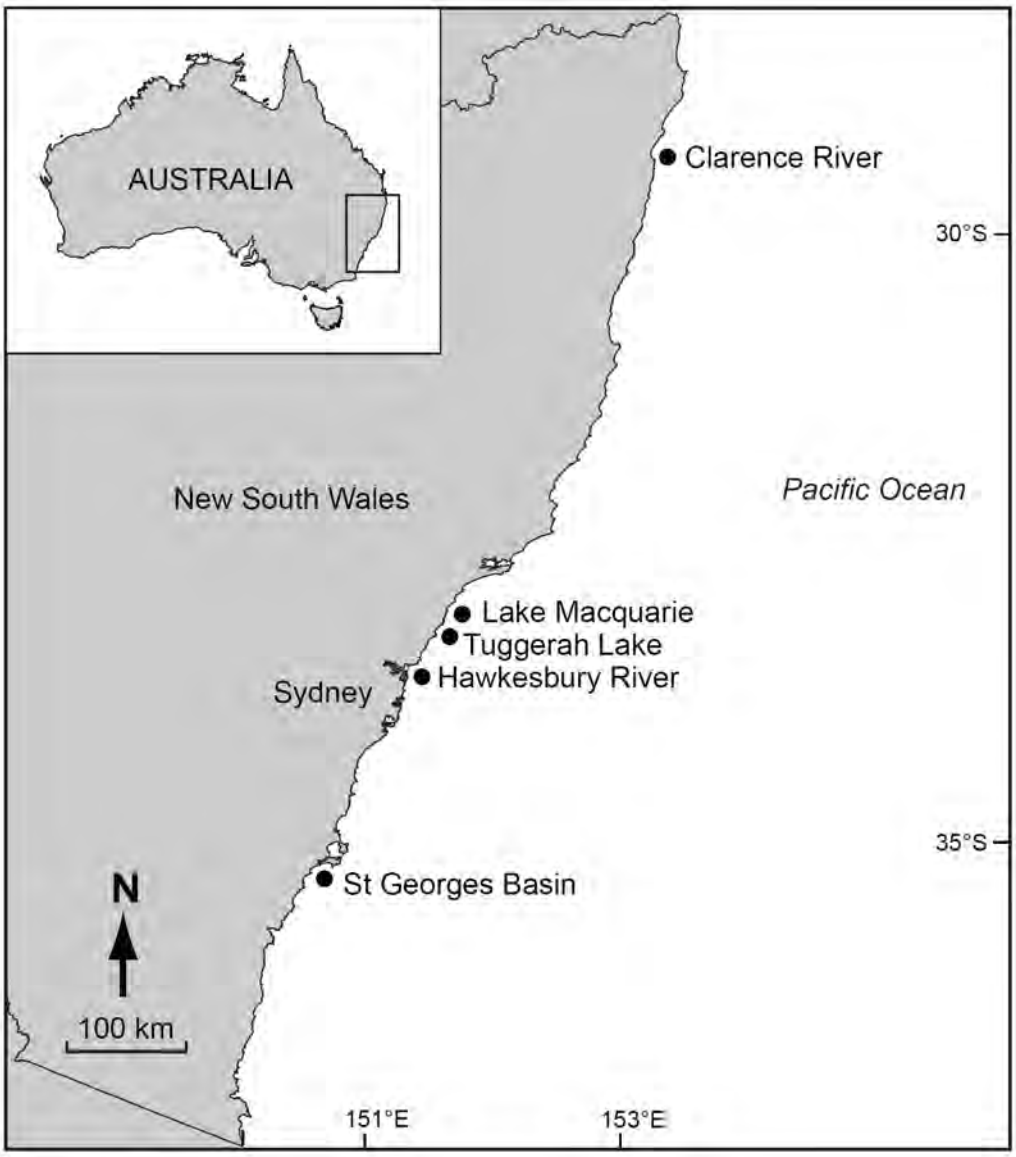

Figure 1. Map of the NSW coast showing the locations of the estuaries where experiments were done.

4.2. An experimental strategy

This project is underpinned by an experimental approach that involves testing of hypotheses. The importance of doing pilot experiments to develop more reliable and cost-effective sampling tools and designs of sampling is generally well-known (Andrew and Mapstone, 1987; Underwood, 1997). Many experimental studies have: (i) compared methods of sampling fish or invertebrates (e.g., Guest et al., 2003; Olin and Malinen, 2003); (ii) tested the effects of biotic and abiotic factors on the performance of sampling gear (e.g., Acosta, 1994; Misund et al., 1999; Petrakis et al., 2001); or (iii) measured spatial and temporal variation in numbers of organisms across hierarchical scales (e.g., Morrisey et al., 1992a, b). There are, however, few examples of necessary pilot work being done before commencing large scale and long term fishery-independent surveys (e.g., Kennelly, 1989; Kennelly et al., 1993). Often, surveys are done using commercial fishing gears and inappropriate designs of sampling, which may provide data that are biased, inaccurate, imprecise and costly. Thus, the hypothesis that fishery-independent sampling provides more reliable and robust information than data from fishery-dependent sources, remains largely untested.

We reviewed: (i) previous fishery-independent studies that have used pilot experiments to develop and optimise methods and designs; and (ii) key literature on surveys of fisheries resources (e.g., Gunderson, 1993) and design and analyses of ecological experiments (e.g., Andrew and Mapstone, 1987; Underwood, 1997). Many surveys did not use a consistent approach to designing sampling gears or surveys. Therefore, we brought the elements together into a general strategy for this type of preliminary work, following an experimental approach advocated in previous studies in NSW (e.g., Kennelly, 1989; Kennelly et al., 1993). The strategy involves: (i) identifying suitable gear to sample target species; (ii) testing configurations of gear and sampling practices to ensure that samples are optimal, representative and cost-efficient; (iii) understanding spatial and temporal variability; and (iv) cost-benefit analyses to determine optimal levels of replication (Fig. 2). More specific details of the strategy are in Rotherham et al. 2007 (Appendix 4). The logical sequence of the strategy addresses the objectives of this project.

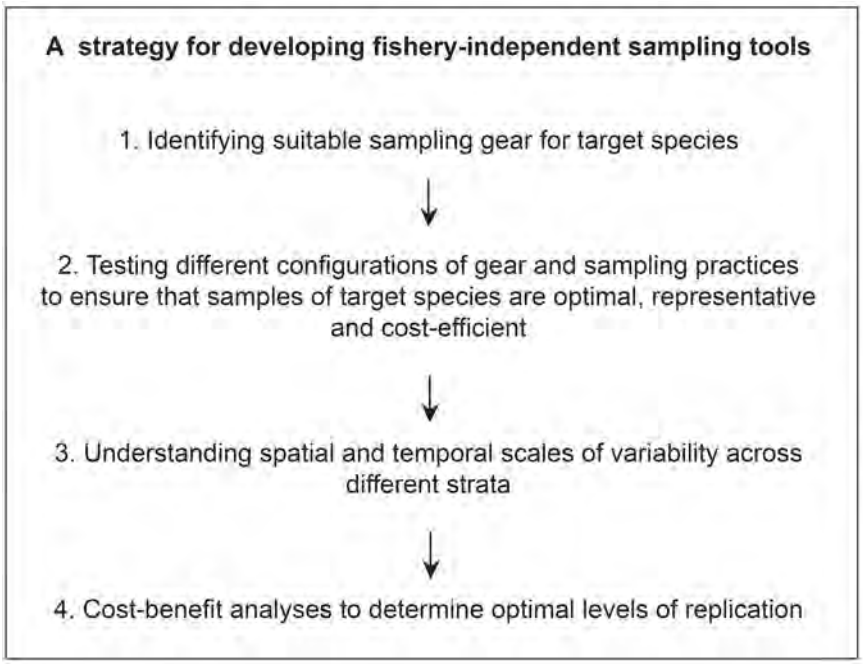

Figure 2. Steps in the strategy used to develop fishery-independent sampling tools.

4.3. Sampling gears

Many methods are used by commercial fishers in estuaries of NSW; the most common are meshing (gill nets), hauling (seining) and trawling (Anon, 2003a, b). These methods catch a diverse range of fish and invertebrates. Nevertheless, using commercial configurations and fishing practices is often inappropriate for scientific sampling (see above). So, we followed the above strategy for two different types of gears: (i) multi-mesh nets; and (ii) trawls.

4.3.1. Multi-mesh nets

In general, multi-mesh nets are either gill nets (single wall of netting) or trammel nets (two large-meshed walls of netting enclosing a loosely-hung centre wall of netting) comprising panels of more than two sizes of mesh (Fig. 3). These sorts of gears are designed to catch a wider range of sizes (and species) of fish than commercial meshing gears (which typically use a single size of mesh). Multi-mesh nets (particularly gill nets) have been used throughout the world for sampling fish and invertebrates (Loneragan et al., 1987; Degerman et al., 1988; Mattson and Mutale, 1992; Acosta, 1997), mainly because they are relatively inexpensive and are easy to deploy and retrieve. Another benefit is that they have minimal impacts on the habitat that is sampled.

Multi-mesh nets are often deployed in gangs with continuous (i.e., no gaps between adjacent panels) or discrete (gaps between adjacent panels) connections. We used discrete connections in all of our nets (Fig. 3) to: (i) reduce non-independence between adjacent nets; and (ii) minimise the potential of small-meshed panels leading fish into larger-mesh panels (Hamley, 1975; Pope et al. 1975).

We used between 3 and 5 sizes of mesh for experiments with multi-mesh nets in Steps 1 and 2 of the strategy (Appendix 6 and 7). We considered this to be sufficient to provide some generality of results for the hypotheses tested. Further, using additional sizes of mesh in these experiments would have restricted levels of replication by imposing logistic constraints. All of our experiments in Step 3 of the strategy, however, used seven sizes of mesh (36, 44, 54, 63, 76, 89, 102 mm, stretched mesh), in a random order, with gaps of 5 m between each panel. The difference between successive sizes of mesh was approximately geometric (i.e., successive sizes of mesh differed from one another by an almost constant ratio). Using a geometric progression of mesh sizes, as opposed to an arithmetic progression (i.e., where each mesh differs from the succeeding mesh by a constant amount), is required so that each size of mesh fishes with similar efficiency (see Regier and Robson, 1966; Pope et al., 1975; Jensen, 1986). We considered this important for testing hypotheses about differences in patterns of abundance over different spatial and temporal scales.

4.3.2. Trawls

Trawls are funnel-shaped nets (Fig. 4) that are towed through the water behind a boat. There are many different types of trawls (von Brandt, 1984). Otter trawls and beam trawls are commonly used in bottom-trawl fisheries and have been widely used as sampling gears in coastal environments worldwide (Stokesbury et al., 1999; West, 2002; Petrakis et al., 2001). We were interested in developing a trawling gear for sampling prawns, crabs and juveniles of key, economically-important species of finfish. Our trawl nets (both the beam trawl and otter trawl) were based on a ‘Florida flyer’ design (Fig. 4). This is widely used in prawn-trawl fisheries in eastern Australia (Hughes, 1972) and is known to retain key species of fish and invertebrates (Liggins and Kennelly, 1996). Other trawl-net designs are used elsewhere (von Brandt, 1984), but we did not test them against the Florida flyer design. Our experiments in steps 1 and 2 of the Strategy involved either manipulating the vertical opening of the nets, the sizes of mesh used in the body and coded, or the time of day or length of time the trawl was towed.

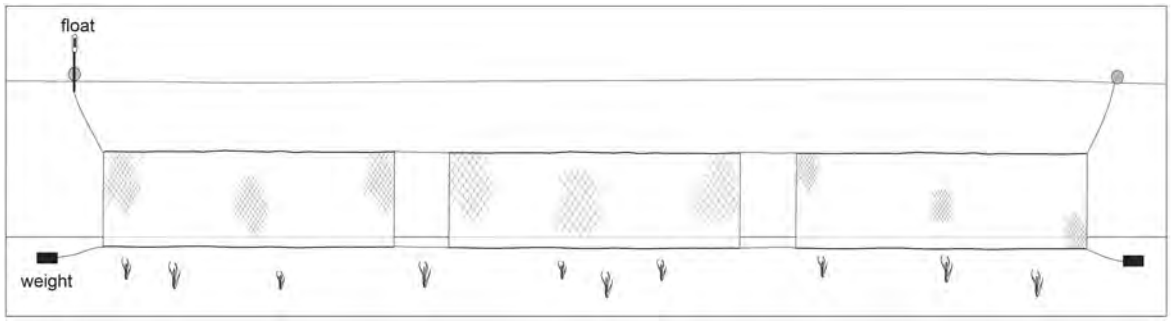

Figure 3. An example of a continuous multi-mesh gill net with discrete connections between adjacent panels of different sizes of mesh.

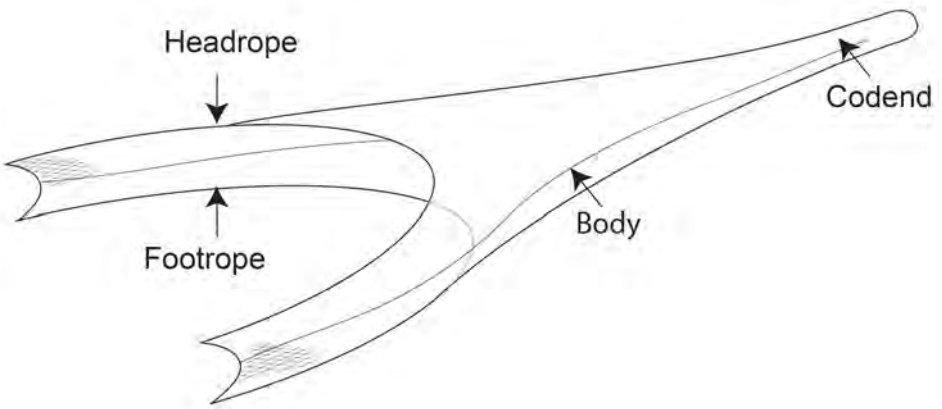

Figure 4. Design of a generic 'Florida flyer" trawl net used in NSW.

4.3.3. Other gears

We intended to follow the above strategy for a third type of gear (i.e., a beach-seine net). Sampling with haul nets is, however, considerably more time-consuming and labour-intensive than with gill nets or trawls, limiting the numbers of replicates that can be sampled in any given period. Our pilot work with gill nets and trawls revealed that among-replicate variability was large (requiring greater replication at this scale). Therefore, we decided not to develop a hauling net because, ultimately, this method would require additional staff (and therefore greater costs) and provide levels of replication that were likely to be inadequate. The Advisory Committee and project investigators also agreed that, before investing in additional sampling gears, it should first be demonstrated that the gears already examined in this project (i.e., the multi-mesh gill nets and beam trawl, see below) actually provide more reliable and cost-effective information than from fishery-dependent sources (see Chapter 8).

In recent times, there has been an increasing use and promotion of non-destructive sampling gears, such as baited remote underwater video stations (BRUVS, e.g., Willis and Babcock, 2000). We also considered using these sorts of gears. Nevertheless, BRUVS (and other visual techniques such as diver surveys) are not commonly used in estuaries owing to the turbidity of water, causing poor visibility. Many taxa in estuaries also follow diel patterns and are more abundant at night, which precludes the use of some visual methods.

A previous study in NSW showed that even in the lower reaches of an estuary, where visibility was relatively clear, BRUVS were only effective for sampling two species of fish (Ianna, 2004). In fact, studies of non-destructive methods, such as BRUVS, often conclude that it is necessary also to sample in concert with other gears (e.g., trawls, Cappo et al., 2004). Until there are significant advances in the technology of non-destructive methods (e.g., Didson SONAR), gears based on commercial fishing methods (as developed here) will continue to play an important role in sampling multi-species assemblages of fish and invertebrates in turbid, estuarine waters. Optimising gears and designs of sampling using the strategy outlined above can, however, reduce the numbers of organisms that are harmed or killed.

4.4. Summary of experiments

The experiments are summarised in Table 1. An initial experiment was done to identify a suitable multi-mesh gear (i.e., gill nets vs. trammel nets). For the trawling method, however, we simply identified otter trawls as being a suitable gear and proceeded to Step 2 of the strategy. After completing some initial experiments on the configuration of the otter trawl, we realised that sampling with this gear was difficult in shallow water. Further, either a commercial-sized vessel would be required to tow the gear in future, or we would have to scale down the gear. The scientific literature also discussed many problems of sampling with otter trawls: (i) catches of fish are often enhanced by the herding effects of the otter boards and rigging; and (ii) the horizontal opening of trawls can be highly variable (Gunderson and Ellis, 1986; Wardle, 1986). So, we decided not to proceed to Steps 3 and 4 of the strategy with the otter trawl. Instead, we decided that a large beam trawl would be a more suitable sampling gear (Step 1) because it overcame the problems of sampling with the otter trawl. We then completed the remaining steps in the strategy for the beam trawl.

In most cases, manuscripts relating to each experiment have been published, or submitted to, peer-reviewed scientific journals. Thus, the specific details of each experiment (i.e., the justification, materials and methods, analyses and results) are provided in these manuscripts, which are attached as appendices. For a small number of experiments, data have been analysed and interpreted, but

manuscripts are still in preparation. In these cases, a summary of each experiment (the hypotheses tested, what was done and the main findings) is also attached as an appendix. The remaining chapters of this report provide a brief summary of the main results of each experiment (Chapter 5), a general discussion (Chapter 6) and implications of the research (Chapters 7, 8, 9, 10).

Table 1. Summary and location of experiments done for each type of gear during the project. TL = Tuggerah Lake; SGB = St Georges Basin; LM = Lake Macquarie; CR = Clarence River; HR = Hawkesbury River. Details of each experiment are found in the relevant appendix.

Experiments	Location	Appendix
Multi-mesh nets		
1. Comparison of multi-mesh gill nets and trammel nets	TL	6
2. Multi-mesh gill nets: effects of soak and setting time and panel length	TL	7
3. Multi-mesh gill nets: spatial variation of fish fauna	SGB, LM	8
4. Multi-mesh gill nets: temporal variation of fish fauna	SGB	9
Trawls		
5. Otter trawl: effects of net height and diel period and tow duration	CR, HR	5
6. Beam trawl: effects of a codend cover and mesh size in the body and codend	CR	10
7. Beam trawl: effects of diel period and tow duration	TL	11
8. Beam trawl: spatial and temporal variation of fish fauna	TL , LM	12

5. RESULTS

Specific details of each experiment are in the relevant appendices. A general summary of the results of each experiment involving multi-mesh nets and the beam trawl (including a brief description of the experiment and hypotheses tested) is also provided below. Results are not summarised below for the preliminary experiments with otter trawls because we decided not to proceed with this gear (but details of the experiments are in Appendix 5).

5.1. Multi-mesh nets

5.1.1. Experiment 1. Comparison of multi-mesh gill nets and trammel nets

The first experiment compared the utility and efficiency of multi-mesh gill nets and trammel nets. We tested the hypothesis that catches of fish would differ between net types and sizes of mesh (38, 54, 70, 90, and 100 mm). Analyses showed no statistically significant differences between the two types of net in compositions and structures of assemblages, abundance, or diversity of catches. The two smallest sizes of mesh (i.e., 38 and 54 mm) captured significantly more fish and species than did the larger sized meshes and were important for capturing sub-adults and juveniles of some species. Greater precision of catch per unit effort, ease of use, and smaller sampling effort made the multimesh gillnet the superior method (see Gray et al., 2005 in Appendix 6).

5.1.2. Experiment 2. Multi-mesh gill nets: effects of soak and setting times and panel lengths

Having identified multi-mesh gill nets as a better sampling gear than trammel nets, experiments were done to determine the most appropriate configuration and soak (i.e., length of time the gear is fished) and setting (i.e., time of night the gear is deployed) times. We tested the hypotheses that catches of fish in multi-mesh gill nets were different among: (1) soak times (1, 3 and 6 hours); (2) setting times (18:00, 22:00 and 3:00) and; (3) panel lengths (20, 50 and 120 m). Univariate and multivariate analyses revealed that 20-m panels soaked for 1 h at any time of the night were optimal (in terms of catch obtained and efficiency of sampling) for sampling populations, assemblages and the sizes of most species caught. The benefits of short soak times and panels include greater replication, smaller costs, and the potential for lower fish mortality (see Rotherham et al., 2006 in Appendix 7).

5.1.3. Experiment 3. Multi-mesh gill nets: spatial variation of fish fauna

We examined patterns of variation of fish fauna sampled with multi-mesh gill nets at several spatial scales at two depths (shallow: < 2 m and deep: 4 – 8 m) in two large coastal lakes (Lake Macquarie and St Georges Basin). The design incorporated spatial scales including randomly-chosen zones (areas separated by 2 – 20 km) within estuaries, randomly-chosen sites separated by at least 1 km within each zone and replicate gill-nets separated by 50 – 100 m (Fig. 2). Abundances of fish were hypothesised to vary at each spatial scale. Depth-related differences were obvious in both lakes, with most species caught in greater numbers or proportions in shallow samples. There were also differences in the sizes of some species of fish caught between depths (e.g., shallow samples had greater proportions of small fish; deep samples had greater proportions of larger fish). Spatial variation was generally greatest at the smallest spatial scale (i.e., among replicate nets taken at a

particular site) and this pattern was consistent across species, depths and lakes. Variation was also generally larger among sites than zones. These results suggest that future sampling need not include the scale of zones, but more effort should be placed on sampling replicate nets and sites in these estuaries (see Gray et al., in press in Appendix 8).

5.1.4. Experiment 4. Multi-mesh gill nets: temporal variation of fish fauna

We examined variation of fish fauna across weeks, months and seasons, again at two depths, in a coastal lake (St Georges Basin). Sampling also included the spatial scale of sites (kms apart). Sites were sampled for two consecutive weeks, in two months in two consecutive 'seasons' (July – August, winter and October – November 2005, spring) to test the hypotheses that abundances of fish vary at each temporal scale. There were no consistent patterns of variation among the different temporal scales. The only consistent pattern across species in both deep and shallow habitats was that variation among replicates (which is spatial variation) was greater than any temporal scale. Thus, temporal variation was small compared to spatial variation. The structure and composition of assemblages differed between depths, with differences largely due to greater numbers and proportions of common species caught in shallow water. Some less-common species were, however, caught mainly at the deeper depth and contributed to differences in the composition of assemblages (see Appendix 9).

5.2. Beam trawl

5.2.1. Experiment 1: Effects of a codend cover and sizes of mesh in the body and codend

The hypotheses tested were that catches of fish and crustaceans in the beam trawl would be affected by a codend cover and the sizes of mesh in the body (26 vs 41 mm) and codend (20 vs 29 mm). The cover had no obvious effects on the catches retained in the codend. Similarly, in comparisons between trawl bodies made from 26- and 41-mm diamond-shaped mesh, there were no differences in the structures of assemblages of fish, nor the mean numbers entering the codends. The results also showed that, for most finfish, there were no differences in catches between codends made from 20- and 29-mm mesh hung on the bar (i.e., square-shaped mesh). In contrast, mesh size in the codend was important for the size selectivity of school prawns (*Metapenaeus macleayi*); with smaller carapace lengths at 50 % retention in the 20-mm codend. We concluded that using 41-mm mesh in the body and 20-mm square mesh in the codend of the beam trawl was appropriate for further sampling with this gear (see Rotherham et al., 2008a in Appendix 10).

5.2.2. Experiment 2: Effects of diel period and tow duration

The hypotheses tested were that diel period (day and night) and tow duration (5, 10 and 20 min) would affect samples of estuarine fauna caught in the beam trawl. Mean catch rates (numbers of fish caught 5 per min) were significantly larger at night for the total numbers of individuals and abundant, economically-important species of fish and invertebrates. Greater proportions of larger fish were also caught at night for some species, but not across all tow durations. Multivariate analyses detected dissimilarities in the composition and structure of assemblages between diel periods, which were associated with species caught predominantly, or in larger proportions, at night. Short tows (5 min) were more efficient than longer tows (10 or 20 min) for sampling the diversity of species. There were, however, no clear or consistent patterns relating to the effect of tow duration on the catch rates of other variables, the size ranges of abundant species, or the structure and composition of assemblages. In future, more cost-effective and reliable information

concerning these taxa would be achieved by sampling with the beam trawl at night using tow durations of 5 min (see Rotherham et al. 2008b in Appendix 11).

5.2.3. Experiment 3: Spatial and temporal variation of fauna in the beam trawl

The hypotheses tested were that: (i) spatial variation would be greater among replicates than sites; (ii) abundances and assemblages of fish and crustaceans would vary from night to night, week to week, month to month and season to season; and (iii) patterns of spatial and temporal variation would be consistent between estuaries. In Tuggerah Lake, spatial variation was generally greater among replicate trawls than among sites. There were, however, no general patterns of variation in Lake Macquarie; variation was greater among sites for some species and among replicates for others. Similarly, patterns of variation among different time-scales were not consistent between estuaries. In Tuggerah Lake, there was generally more variation among nights and weeks for most species. By comparison, in Lake Macquarie, variation was generally greater among seasons. The only consistent pattern was that spatial variation among replicates was generally greater than for any temporal scale examined (as for multi-mesh gill nets) (see Appendix 12).

6. DISCUSSION

Assessing the status of fisheries resources and the success of managerial strategies designed to ensure their sustainability (i.e., adaptive management) requires quality scientific information. Often, however, data are available only from fishery-dependent sources, which are generally biased and imprecise. Fishery-independent surveys may overcome some of these problems. Nevertheless, similar biases and imprecision are also relevant to fishery-independent data if they are collected using poorly-developed gears and designs of sampling. Further, despite fishery-independent surveys being perceived as prohibitively expensive, it is surprising that few studies have actually examined the costs (or indeed the benefits) of such data (e.g., Kennelly, 1989; Kennelly et al. 1993).

Much has been written about the importance of doing pilot experiments to develop more reliable and cost-effective research surveys (e.g., Andrew and Mapstone, 1987; Underwood, 1997). Yet, there are few comprehensive or consistent examples of the necessary pilot work being done prior to the commencement of fishery-independent surveys – i.e., using a series of pilot studies to test hypotheses about the design *and* spatial and temporal deployment of sampling gears (e.g., Kennelly, 1989; Kennelly and Craig, 1989). Unfortunately, pilot work is often considered an unaffordable luxury, despite the potential long-term benefits of more robust and reliable data and savings of time and money. So, the process of 'monitoring' fisheries resources frequently proceeds in *ad hoc* and hypothesis-free ways, with little or no concern for the accuracy, precision or cost-effectiveness of the data.

The current project has completed the initial stages of the necessary pilot work to develop fishery-independent surveys for the adaptive management of NSW's estuarine fisheries. The research has used an experimental strategy to meet its objectives, which draws on key literature about surveys of fisheries resources (e.g., Gunderson, 1993) and the design and analyses of ecological experiments (e.g., Green, 1979; Andrew and Mapstone, 1987; Underwood, 1997).

Objective 1

The first objective of the project was to develop scientific sampling tools to catch the widest possible size-range and diversity of fish species. We have developed both static (multi-mesh gill nets) and towed (beam trawl) sampling gears. Together, these gears have sampled more than 60 species of fish and invertebrates, including most taxa of key importance to local commercial fisheries. For example, the gears were effective at sampling 13 of the 19 (68%) most important species identified in the Estuary General Fishery Management Strategy (Table 2); 7 of the 10 (70%) primary species and 6 of the 9 (67%) secondary species. The gears also sampled key recreational species (e.g., Tailor) and many taxa with little or no economic value, but which may be ecologically important (see species lists included in appendices 6 to 12).

Key commercially-important species that were not generally caught by our gears included Mud crab, River eels, Pipis, Mulloway, Cockles and Beachworms (Table 2). Some of these species were not caught because they are associated with ocean beaches (e.g., pipis and beachworms). Other species (e.g., mud crabs) generally require the use of traps, which were not considered in this study (reasons for this are explained in section 4.3.3). Sampling of River eels also requires traps and other methods, which have been developed elsewhere (Silberschneider et al., 2001; Pease, 2004). Mulloway were not frequently caught in the beam trawl or multi-mesh gill nets because we did not test these gears in estuaries where the species is generally abundant (e.g., deeper riverine estuaries). We did, however, catch Mulloway in an otter trawl that was used in deeper areas of the

Hawkesbury River and Clarence River (which are large riverine estuaries). Thus, we would expect the beam trawl and gill nets to catch Mulloway if future sampling was done in this type of estuary.

Although our gears caught fish and crustaceans across a wide range of sizes (in many cases from juveniles to adults), they were not developed for the purpose of sampling immediate post-settlement fishes, or other small fauna, commonly associated with seagrass beds in the region. Much research has already been done on sampling assemblages of seagrass fauna in NSW (e.g., Ferrell and Bell, 1991; Gray et al., 1996) and appropriate methodologies, particularly for post-larval recruits of economically-important species, have already been developed (Gray et al., 2000).

Table 2. Species of primary and key secondary importance to the Estuary General Fishery (adapted from Table 11 in Anon, 2003a) and whether they were generally sampled by the gill net or beam trawl developed in this project.

Common name	Scientific name	Primary or key secondary	Sampled by the gill net or beam trawl
Sea mullet	*Mugil cephalus*	Primary	Yes
Luderick	*Girella tricuspidata*	Primary	Yes
Yellowfin bream	*Acanthopagrus australis*	Primary	Yes
School prawn	*Metapenaeus macleayi*	Primary	Yes
Dusky flathead	*Platycephalus fuscus*	Primary	Yes
Eastern king prawn	*Melicertus plebejus*	Primary	Yes
Sand whiting	*Sillago ciliata*	Primary	Yes
Mud crab	*Scylla serrata*	Primary	No
River eels	*Anguilla* spp.	Primary	No
Pipis	*Donax deltoides*	Primary	No
Blue-swimmer crab	*Portunus pelagicus*	Key secondary	Yes
Greasyback prawn	*Metapenaeus bennettae*	Key secondary	Yes
Mulloway	*Argyrosomus japonicus*	Key secondary	No
Cockles	various	Key secondary	No
Beachworms	various	Key secondary	No
River garfish	*Hyporhamphus regularis*	Key secondary	Yes
Silver biddy	*Gerres subfasciatus*	Key secondary	Yes
Flat-tail mullet	*Liza argentea*	Key secondary	Yes
Trumpeter whiting	*Sillago maculata*	Key secondary	Yes

The second objective of the project was to use the gears developed in objective 1 to do pilot studies to determine the most cost-effective, optimal number of replicates, sites, locations and habitats to be sampled in and among estuaries. This was done using experiments incorporating hierarchical designs of sampling and nested analyses of variance (see Appendices 8, 9 and 12). These types of experiments provide information about variation at different scales and estimates of variances for cost-benefit analyses.

Cost-benefit analysis is an important component of experimental design because it allows allocation of resources (be it money, time or any other variable) in the most cost-effective manner to best capture variability at different scales (Underwood, 1997). For example, it can determine whether more effort should be put into sampling replicate nets, or different sites or zones in an estuary, or whether sampling times need to be separated by days, weeks or months to optimize measurements of temporal changes in species. It incorporates information about costs (in any "currency") and variability in abundances (diversity, biomass or any other measurement) which is collected prior to the analysis in pilot experiments. Such procedures are well-known for univariate measures. Although they have been suggested to be useful for measurement of assemblages of species, it is not clear how the different species in an assemblage contribute to the optimal design, nor how well it will apply to different species in an assemblage. Thus, new procedures are needed to attempt to develop cost-benefit procedures to a multi-species fishery. Problems are aggravated when many species in the assemblage are extremely patchy and overdispersed among samples, leading to many negative components of variation.

Some of these issues are resolved by post-hoc pooling, but robust methods for doing this are not available for multivariate analyses. As an example of the problems, consider analyses of the data from the experiment on spatial variation using gill nets in shallow water in Lake Macquarie (6 nets in each of 2 replicate sites in each of 3 zones; see Appendix 8). Of the 33 species caught, only 10 were considered widespread or abundant enough to give meaningful cost-benefit analysis using current techniques. Of these, 3 produced negative components of variation for sites and 6 for zones, leaving only 1 species in the set of 33 which could yield sensible analyses for all scales. With respect to costs, there are added complications because multiple sites (or zones) cannot easily be sampled on the same night using gill nets because of the length of time nets need to soak, thus always confounding spatial with temporal variation. This problem was overcome in the study using beam trawls because 5-minute trawls were shown to sample the assemblage adequately. Thus, many trawls and sites could be completed on a single night.

Cost-benefit analysis can be optimized by cost, or by minimizing variation around the estimate of the mean of the variable being measured. Although many species could not be analysed for reasons given above, analyses were possible for a number of taxa. Using a realistic cost-structure for sampling, effort could be allocated to, for example, replicate nets or sites (see Appendix 13), but two problems arose. First, the optimal design for any species varied among habitats (e.g., between shallow and deep sites in Lake Macquarie and St Georges Basin) and it varied among species within any one habitat (results of cost-benefit analyses are in Appendix 13).

The second method of optimising this design minimises the SE around the mean. This method is better in this case because: (i) resources are adaptable to the needs of the fishery and (ii) decisions about changing managerial practices will be determined by differences in mean values from time to time. Unfortunately, to minimise these errors to match those suggested as important in plans of management, an impossibly large number of sites would need to be sampled (See tables in Appendix 13).

Thus, in this part of the project alone, numerous problems have been identified that need the development of novel approaches. These include, but are not restricted to the need to: (i) optimise a design across multiple species, which have varying importance to the fishery and which have patterns of abundance that vary among habitats and estuaries; (ii) develop procedures which will weight the relative contributions of different species to develop a method that can be applicable across multiple species to provide a measure for the assemblage; (iii) if possible, reduce variance to get reasonable designs that are logistically possible.

The above has specifically concentrated on the gill-net sampling in Lake Macquarie and St Georges Basin, but similar comments are applicable to the data obtained from the beam trawl sampling, so

will not be repeated here. These novel techniques will be developed over the next year. Although preliminary analyses have been done, the development of these techniques could not be completed earlier. First, they are dependent on the data that have only become available from this project because prior to the fully nested, rigorous designs used in this project, data were not available to allow anyone to address this problem. The identified problems are not due to the way that these data were collected, but reflect the variable and dynamic nature of these assemblages. They thus present a major challenge, but one that needs to be met if cost-effective sampling is to replace more *ad hoc* programmes in the future. The data are now available to develop such procedures, which are underway.

Objective 3

The third objective of the project was to use the results from objectives 1 and 2 to design an optimal sampling regime that will become the long-term, large-scale survey of the fish populations in NSW estuaries. Designing an overall sampling regime that is 'optimal' requires consideration of a number of different elements (which are covered by our experimental strategy – see Appendix 4). These include: (i) optimising the configuration of the sampling gear (i.e., its actual design) and its deployment (i.e., how and when the gear is used) so that *samples* of target species are optimal, representative and cost-efficient; (ii) identifying the relevant spatial and temporal scales that should be included in a survey; and (iii) optimising the *numbers* of replicates to be sampled with the gear at the relevant spatial and temporal scales.

The results of our tool-development work (Objective 1) were used to optimise the configuration and deployment of the multi-mesh gillnet and the beam trawl (in terms of catch obtained and efficiency of sampling). For multi-mesh gillnets, we concluded that an optimal strategy for future sampling would be the use of 20 m panels and a soak time of 1 hour, at any time of the night (see Appendix 7). The benefits of using short panels and soak times in future surveys has been highlighted above and is discussed in Appendix 7. For the beam trawl, the optimal configuration of the net was 41 mm diamond-mesh in the body and 20 mm square-mesh in the codend (Appendix 10). Sampling at night using a tow duration of 5 min was considered optimal for deployment of the beam trawl. The benefits of this strategy have also been highlighted above and in Appendix 11.

Our pilot studies on spatial and temporal variation (Objective 2) identified the relevant scales to be included in a long-term, large-scale survey. A general pattern from these experiments was that variation among replicate gill-nets and trawls was large compared to variation at larger spatial scales (i.e., sites or zones within an estuary). Thus, future designs of sampling should not include the scale of 'zones' and more effort should be placed on sampling replicate nets (or trawls) and sites (see Appendix 8).

A second important pattern (which was consistent between gears and depths; and among species and estuaries) was that temporal variation within a season was small compared to spatial variation). These results suggest that comparisons among different sites or estuaries sampled at different times within a season are unlikely to be confounded by short-term temporal variation i.e., any differences probably represent small-scale *spatial* variation, rather than *temporal* variation (see Appendix 12 and 13). This is advantageous to the design of a large-scale and long-term survey because less effort would be required to sample replicate 'times' within a season.

The small-scale spatial variation observed in our experiments is probably explained by a host of potential models (which were not tested here and remain speculative), including the schooling behaviour of many species of fish and the heterogeneity of the underlying substrate. Stratification of sampling into different habitats (to reduce variances) was extremely difficult for each gear because each replicate invariably covered a large area that often contained a mosaic of bare and vegetated habitats. Further, some species of seagrass (e.g., *Zostera capricorni*) die on a seasonal

basis, further exacerbating reliable stratification. For multi-mesh gill nets, we found the only reliable option was to stratify samples by depth, which subsequently revealed large differences in the abundances and sizes of fish, and the composition of assemblages between shallow (< 2 m) and deep (4 to 8 m) strata. So, future surveys should include both shallow and deep strata to obtain representative samples of fish populations and assemblages.

Since beam trawls can damage vegetated habitats (although less so than otter trawls), sampling with this gear was mostly done in areas of predominantly bare sediment. Apart from the undesirable effect of damaging vegetation, trawling through beds of seagrass also results in the gear retaining large amounts of leaves; potentially affecting proper operation of the trawl and selectivity of its codend. Although different depth strata were sampled with the gill nets, this was not possible with the beam trawl because much of our pilot work was done in shallow (< 3-m deep) coastal lakes. Nevertheless, given the distinct differences in fish fauna between depth strata for gill nets, sampling shallow and deep strata (as candidates for standardising future surveys) is also likely to be required for the beam trawl in estuaries with deeper water (e.g., riverine systems).

We have produced an optimal sampling regime in relation to the design and deployment of our sampling gears (i.e., so that *samples* are optimal, representative and cost-efficent); and identified the relevant spatial and temporal scales of sampling and strata (i.e., deep and shallow habitats) to be included in future surveys (see above). Given the results of our cost-benefit analyses and the problems identified above, however, it is clear that the optimal numbers of *replicates* to be sampled at each of the relevant spatial and temporal scales cannot yet be determined for all species and estuaries in NSW. Solving these problems remains a significant challenge for the future. In the meantime, we have identified an additional, crucial step that must be done *before* a large-scale, long-term fishery-independent survey is implemented in NSW. This step involves testing the hypothesis that fishery-independent data provides better information and leads to better management decisions than fishery-dependent data (see 'Further Development').

Completing this next stage of research is not dependent on a generalized optimal design as relatively powerful experiments can be done to test the above hypothesis. The issue of optimal designs (i.e., in terms of the *numbers* of replicates) is more critical to implementing large-scale and long-term surveys, which need to be powerful *and* cost-effective (because of the need to sample many different estuaries). Once these challenges have been addressed we will be in a strong position to: (i) determine whether or not fishery-independent surveys really are a worthwhile investment for monitoring estuarine fisheries and biodiversity in NSW; and (ii) prescribe a general design that also incorporates optimal numbers of *replicates* at the important scales we have identified using the optimal configurations of gear and sampling practices already developed in this project.

7. BENEFITS

This project has provided a general experimental strategy for developing fishery-independent sampling tools and designs (see Chapter 4). The strategy was presented at a local and at an international conference and published in a peer-reviewed scientific journal (Rotherham et al., 2007), which should benefit researchers interested in developing more reliable and cost-effective surveys of fisheries resources.

Using the strategy outlined above, we have developed and optimised two different types of gears for sampling estuarine fish and invertebrates in NSW (i.e., a multi-mesh gill and a beam trawl). We have also provided new data on spatial and temporal variation of estuarine fauna sampled with these gears. The main benefit of this work is that these sampling gears and data now provide an important basis for further research, which is necessary before any commitments are made to do large-scale and long-term fishery-independent surveys in estuaries of NSW. This includes the need to: (i) develop new procedures for multi-species cost-benefit analysis; and (ii) directly test the benefits of fishery-independent sampling against data from fishery-dependent sources for assessing and managing fisheries resources (see 'Further Development' below). Prior to this project, there were no properly-developed sampling gears or data to investigate these problems.

Significant and long-term benefits to estuarine fisheries in NSW (and indeed Australia) and their stakeholders can only be realised if commitments are made to do the additional research that is now required (see below). This includes identifying the most cost-effective and reliable long-term sampling programmes (i.e., fishery-independent, fishery-dependent, or both) that should be invested in by NSW DPI for assessing and managing fisheries resources, biodiversity and the effects of natural and anthropogenic impacts in estuarine systems. This should also benefit other management agencies and jurisdictions in Australia. First, it will increase awareness about the need to do this type of research in order to allocate limited resources more carefully. Second, it will provide a framework for other agencies to do similar research. Ultimately, the more of this type of work that is done, the more that unreliable and unusable sources of information will be identified and discouraged. Indeed, if commonly-used sources of data are found to be unreliable for providing indicators of abundance and population structure, alternative strategies for assessing fisheries resources and biodiversity will need to be developed.

8. FURTHER DEVELOPMENT

In the short-term, the data from this project will allow the development of new techniques to attempt to overcome the problems of optimising designs of sampling for multiple species (see 'Discussion' above). In March 2007, the Advisory Committee for the project met to discuss specific directions for longer-term development. It was strongly agreed that before commitments are made to do large-scale and long-term surveys (or indeed to develop other types of sampling gears) of estuarine fisheries resources in NSW, research is urgently required to test the benefits of our fishery-independent methodologies against data from fishery-dependent sources. Because funding of future monitoring studies relies on the public purse, with cost-recovery from stakeholders, it is necessary to identify and support programs that will provide the most cost-effective and reliable data.

The primary benefit of fishery-independent surveys is that the data are less prone to biases and imprecision than is realistic for most fishery-dependent data – but how much better are the data than information already (or potentially) available from fishery-dependent sources? Do better data actually improve decision-making? Is there a sufficient marginal benefit to offset the extra expenditure? These questions can only be answered by doing research that also 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 our fishery-independent sampling strategy with data from both commercial and recreational fisheries. We have submitted a proposal to FRDC to do this essential research. The specific objectives of the proposed research are to:

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).

An experimental approach will be used to test the relative value of different sources of data for the assessment and management of estuarine fisheries resources and biodiversity in NSW. Fishery-independent sampling tools developed in the present project would be implemented across a number of estuaries with different management regimes (i.e., open and closed to commercial/recreational fishing). Data from commercial and recreational fisheries would also be collected simultaneously in these estuaries (where available), requiring additional sampling to enhance existing port monitoring and creel surveys. The costs and benefits of each type of data and their managerial response could then be tested over equivalent spatial and temporal scales. This new research will provide a scientific basis for determining the most appropriate mix of fishery-independent and –dependent data for improving the sustainability of fisheries resources and biodiversity in estuaries of NSW.

9. PLANNED OUTCOMES

The key planned outcome for this project was 'a large-scale, long-term data collection regime which will become the primary source of information used in NSW to manage our commercial and recreational estuarine fisheries'. We have successfully developed two types of gears for sampling estuarine fish and crustaceans in the most optimal, reliable and cost-effective way. We have also completed pilot experiments examining spatial and temporal variation of estuarine fauna sampled with these gears and performed cost-benefit analyses. Our research has, however, identified further challenges that need to be addressed before a large-scale and long-term survey can be finalised and implemented. First, populations and assemblages of fish and crustaceans sampled in this project were extremely variable at small spatial scales and among habitats and estuaries, making current univariate and multivariate cost-benefit procedures inappropriate. Thus, novel analytical techniques to optimise designs of sampling across multiple species are needed, and currently in development (see 'Discussion' above).

Second, the planned outcome of this project relies on the hypothesis that fishery-independent surveys provide better data than from fishery-dependent sources – leading to better management of fisheries resources. There are, however, no data to test this hypothesis. Decisions concerning the appropriate mix of fishery-dependent and -independent data should not be based simply on pragmatic considerations and value judgments. Ideally, an experimental approach that directly compares the costs and benefits of each type of data and their management response, is required. This has not yet been done because in many cases appropriate fishery-independent sampling tools and designs have not yet been developed. The sampling tools developed in this project now provide an excellent (and rare) opportunity to test the relative value of different sources of data for the assessment of estuarine fisheries resources, before long-term (and possibly illogical) commitments to any additional sampling programs are made.

10. CONCLUSIONS

This project has successfully developed procedures to use multi-mesh gill nets and a beam trawl as tools for sampling a wide size range and diversity of fish species in estuaries of NSW, in an optimal, reliable and cost-effective way (Objective 1). We also did pilot studies using these gears to identify important spatial and temporal scales of variation across different strata; and performed analyses to determine cost-effective and powerful levels of replication (Objective 2). The variable and dynamic nature of the populations and assemblages of fish and crustaceans measured in this project, however, meant that optimising designs of sampling across multiple species was not straightforward or appropriate using current analytical techniques. Thus, novel procedures (which are currently being developed) are required to analyse these data before optimal levels of replication for a long-term, large-scale survey (Objective 3) can be finalised. Further, we have identified a crucial step that must be completed if fishery-independent surveys are to become a primary source of information to assess the status of estuarine fisheries in NSW. This step involves demonstrating that such surveys actually provide better data and management outcomes, than data currently (or potentially) available solely from commercial and recreational fisheries. There is now an urgent requirement for methods to incorporate fishery-independent data into decision making and the costs and benefits of using such data. The relevant experiments must be done to provide a scientific basis for determining the most appropriate sources of data for improving the sustainability of fisheries resources and biodiversity in estuaries of NSW.

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12. APPENDICES

Appendix 1: Intellectual Property

The intellectual property owned by FRDC as specified in the agreed contract is 66.80%. No specific commercial value was derived in terms of patents and copyrights.

Appendix 2: Staff

Staff that worked on the project using funds from NSW DPI:
Dr Charles Gray
Dr Matt Broadhurst
Dr Steve Kennelly

Staff that worked on the project using funds from FRDC:
Dr Doug Rotherham
Dr Martine Jones
Mr Daniel Johnson
Mr Lachlan Barnes
Mr Paul Lokys
Mr Damien Young
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Staff that worked on the project using funds from the University of Sydney:
Professor Tony Underwood
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Professor Gee Chapman (Centre for Research on Ecological Impacts of Coastal Cities, University of Sydney)
Dr Charles Gray
Dr Greg Jenkins (Primary Industries Victoria)
Professor Neil Loneragan (Murdoch University)
Mr Russell Massey (Industry)
Mr Darren Reynolds (Fisheries Manager, NSW DPI)
Dr Douglas Rotherham
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Appendix 4: Rotherham, D., Underwood, A.J., Chapman, M.G. and Gray, C.A. 2007. A strategy for developing scientific sampling tools for fishery-independent surveys of estuarine fish in New South Wales, Australia. ICES J. Mar. Sci. 64: 1512–1516.

1512

A strategy for developing scientific sampling tools for fishery-independent surveys of estuarine fish in New South Wales, Australia

D. Rotherham, A. J. Underwood, M. G. Chapman, and C. A. Gray

Rotherham, D., Underwood, A. J., Chapman, M. G., and Gray, C. A. 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.

The limitations of using fishery-dependent data, i.e. from commercial and recreational fisheries to assess harvested stocks of fish and invertebrates, are well known. Increasingly, fishery-independent surveys are used to validate data from fishery-dependent sources and to provide indices of recruitment and broader ecological information about species not normally retained in fishing operations. Any large-scale, long-term, fishery-independent study must develop sampling gear and designs that are standardized, representative, optimal with respect to the quantity and structure of catch, and replicated over relevant spatial and temporal scales. We present a strategy for achieving appropriate sampling designs. This involves: (i) identifying suitable sampling gears for target species; (ii) testing different configurations of gear and sampling practices to ensure that samples are optimal, representative, and cost efficient; (iii) understanding scales of spatial and temporal variability; and (iv) cost–benefit analyses to optimize replication. Examples of this strategy are illustrated, with brief considerations of the values of pilot research in developing fishery-independent sampling.

Keywords: cost–benefit analysis, fishery-independent survey, pilot study, spatial and temporal variation, standardized sampling.

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Introduction

Fishery-independent surveys of fisheries resources play an important role in assessment and management of populations of fish and invertebrates (Gunderson, 1993; Pennington and Stromme, 1998). Research surveys are used to calibrate stock assessment models based on commercial catches (i.e. fishery-dependent data) and to provide empirical, independent checks of populations (Kline, 1996; Pennington and Stromme, 1998). Fishery-independent surveys are increasingly important: (i) to monitor aquatic resources where fishing has been modified as part of management; and (ii) to obtain better scientific assessments consistent with principles of ecologically sustainable development (ESD).

Fishery-independent data are often preferred over fishery-dependent data for monitoring the status of harvested populations because: (i) sampling is randomized rather than being concentrated where populations are (or are thought to be) most abundant; (ii) potentially, they provide more representative data on the entire size range of populations, rather than just retained components; (iii) there is no reliance on fishers reporting their catches and effort accurately; (iv) methodologies remain consistent over time; and (v) data can be collected on species not usually retained in commercial and recreational fisheries. Nevertheless, fishery-independent surveys that use inappropriate sampling gear or poorly designed sampling would also provide biased, inaccurate, and imprecise data.

Developing reliable, robust, and cost-effective sampling requires pilot studies to test specific hypotheses about the design and deployment of sampling gear (Andrew and Mapstone, 1987). Many experimental studies have: (i) compared methods of sampling fish or invertebrates (e.g. Guest *et al.*, 2003; Olin and Malinen, 2003); (ii) tested the effects of biotic and abiotic factors on the performance of sampling gear (e.g. Acosta, 1994; Misund *et al.*, 1999; Petrakis *et al.*, 2001); and (iii) measured spatial and temporal variation in numbers of organisms across hierarchical scales (e.g. Morrisey *et al.*, 1992a, b).

There are few examples, however, of necessary pilot work being done before a large-scale or long-term fishery-independent survey (Kennelly, 1989; Kennelly *et al.*, 1993). We reviewed: (i) previous fishery-independent studies that have used pilot experiments to develop and optimize methods and designs; and (ii) key literature on surveys of fisheries resources (e.g. Gunderson, 1993) and design and analyses of ecological experiments (e.g. Andrew and Mapstone, 1987; Underwood, 1997). Many surveys were not consistent in their approaches to survey design. Therefore, we bring the elements together in a strategy for this type of preliminary work, following the approach advocated by Kennelly *et al.* (1993). The strategy involves: (i) identifying suitable gear to sample target species; (ii) testing configurations of gear and sampling practices to ensure that samples are optimal, representative, and cost efficient; (iii) understanding spatial and temporal

variability; and (iv) cost–benefit analyses to determine optimal levels of replication (Figure 1).

Example of the strategy: developing fishery-independent surveys

Estuaries in New South Wales (NSW), Australia, support commercial, recreational, and indigenous fisheries. Of 100 species landed from these estuaries, fewer than ten account for ~80% of commercial catches. Stocks in NSW are currently assessed using fishery-dependent data, including catch and effort information supplied by the commercial sector, biological sampling of some species from commercial landings (e.g. Gray *et al.*, 2002), and sporadic recreational creel surveys (e.g. Steffe and Chapman, 2003). The limitations of such data have led to increasing use of fishery-independent surveys to provide better information about the ecology and status of fisheries resources in NSW.

A collaboration between the NSW Department of Primary Industries, the Centre for Research on Ecological Impacts of Coastal Cities at the University of Sydney, and the Fisheries Research and Development Corporation (FRDC 2002/059) is developing methods, procedures, and analyses for these surveys, so that more rigorous sampling designs can be implemented. Although commercial and scientific sampling gears are available, they are typically designed to be size-selective and species-specific. Often, such gears need to be modified to sample wider size ranges and greater diversities of fish. Many types of fish are targeted in estuaries in NSW; no single method can effectively estimate relative abundances and population structures for all species. Rather, a complementary suite of mobile (e.g. trawls) and static (e.g. gillnets) gears is required to estimate abundances, lengths, sex and age composition, reproduction, recruitment dynamics, etc.

We consider how the above strategy is being used in fishery-independent surveys of estuarine fish stocks in NSW. Surveys use several methods; we only illustrate one, a multimesh gillnet made of panels with different mesh sizes, designed to catch many species of different sizes and morphologies. Similar pilot work is being done with other methods (e.g. trawls).

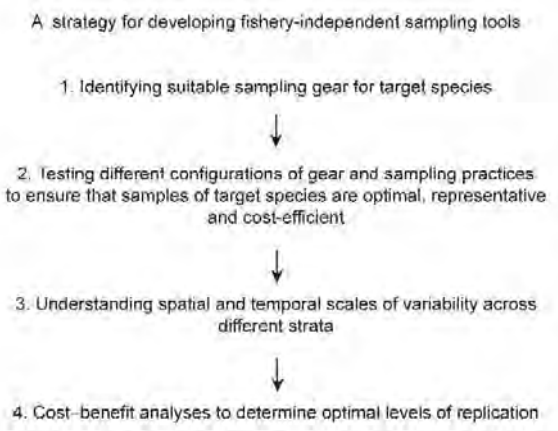

Figure 1. Steps in the strategy used to develop fishery-independent sampling tools.

Step 1: identifying suitable sampling gears for target species

The first step was an experimental comparison of the utility and efficiency of multimesh gillnets and trammelnets. These gears were chosen because they are used widely in local fisheries. Much local knowledge and preliminary experiments not described here demonstrated that night-time sampling was less variable and more representative of size classes and diversity of species of fish than daytime sampling. All sampling described here was done at night. The gears differ: gillnets comprise a single panel of netting; trammelnets have two large-mesh panels enclosing a loosely hung centre panel of smaller mesh.

Replicate multimesh gillnets and trammelnets, each comprising five 30-m long panels of 38-, 54-, 70-, 90-, and 100-mm stretched mesh openings, were used in a NSW barrier estuary to test the hypothesis that catches of fish would differ between net types and mesh sizes (Gray *et al.*, 2005). Analyses showed no statistically significant differences between the two types of net in compositions and structures of assemblages, abundance, or diversity of catches. Greater precision of catch per unit effort, ease of use, and smaller sampling effort made the multimesh gillnet the superior method.

Step 2: testing different configurations of gear and sampling practices

The next step was to determine the most appropriate configuration and period of soak (i.e. length of time the gear is fished) and time of setting (i.e. time of night the gear is deployed). Experiments tested the hypotheses that catches and catch rates of fish were different between soak (1, 3, and 6 h) and setting times (18:00, 22:00, and 03:00), and net lengths (20-, 50-, and 120-m panels).

Univariate and multivariate procedures revealed that 20-m panels soaked for 1 h at any time of the night were optimal (in terms of catch and efficiency of sampling) and the best representative strategy for sampling populations, assemblages, and sizes of fish (details in Rotherham *et al.*, 2006). The benefits of shorter soak times include greater replication, smaller costs, and potentially lower fish mortality (because the catch is processed and released as the gear is retrieved).

Step 3: understanding scales of variability

Spatial and temporal variation in populations and assemblages of estuarine fish were examined using hierarchical sampling designs and nested analysis of variance (e.g. Morrisey *et al.*, 1992a, b; Underwood, 1997). These analyses provide information about variation at different scales and the estimates of variances for cost–benefit analyses (Step 4). Here, one experiment on spatial and one on temporal variation were designed.

Experiment 1: spatial variation

Experiments investigating spatial heterogeneity in the abundance of organisms across hierarchical scales are relatively common (e.g. Green and Hobson, 1970; Kennelly, 1989), and the problems of spatial pseudoreplication (Hurlbert, 1984) are generally well understood. We examined patterns of variation of fish sampled at night using multimesh gillnets at several spatial scales at two depths (shallow: <2 m; deep: 4–8 m) in an estuary. These depths were chosen as "candidates" for standardizing depth in future sampling; they were not chosen for detailed comparisons of depth gradients. The design incorporated spatial scales

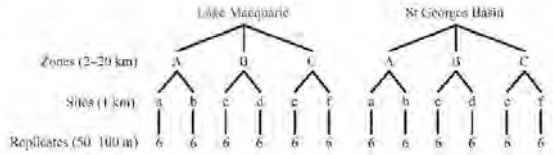

Figure 2. Diagrammatic representation of the hierarchical scales, sampled with multimesh gillnets in an experiment on spatial variation of estuarine fish.

including randomly chosen zones (areas separated by 2–20 km) within estuaries, randomly chosen sites separated by at least 1 km within each zone, and replicate gillnets separated by 50–100 m (Figure 2). Fish abundances were hypothesized to differ at each spatial scale. To provide greater generality, the experiment was done in two large coastal lakes (Lake Macquarie and St Georges Basin), which are relatively well mixed and have no large salinity gradients or tidal ranges.

Components of variation were calculated from separate nested analyses of variance for each estuary and depth. For example, for tailor (*Pomatomus saltatrix*) in deep areas of Lake Macquarie, variability among replicates was greater than among sites or zones that were very similar (Table 1). In contrast, for tarwhine (*Rhabdosargus sarba*) in shallow areas, variability among replicates was similar to that among sites, but variation among zones was less (Table 1).

For most species, some components of variance were negative (Table 1), requiring pooling procedures (Fletcher and Underwood, 2002) or use of residual maximal likelihood (Robinson, 1987), which does not allow negative estimates. The two procedures give identical results when, as here, all levels of replication are balanced (Fletcher and Underwood, 2002). Negative components of variation at scales of sites or zones identifies that numbers of fish are less variable from site to site or zone to zone than among replicate nets. Negative sources of variation are pooled (as done for zones for flat-tail mullet, *Liza argentea*, in Table 1). Pooled components do not need to be sampled, because there is no variation in numbers of fish in this case, from zone to zone.

Table 1. Examples of spatial variation for tailor (*Pomatomus saltatrix*) from Lake Macquarie, deep samples; tarwhine (*Rhabdosargus sarba*) from Lake Macquarie, shallow samples; flat-tail mullet (*Liza argentea*) from Lake Macquarie, shallow samples; and flat-tail mullet from Lake Macquarie, shallow samples, with the mean square for zones pooled (pld), because of the negative estimate of variance.

Source of variation	d.f.	Mean square	VC	Θ
Tailor				
Zones	2	30.1	1.4	0.26
Sites (zones)	3	13.6	1.4	0.26
Residual	30	5.3	5.3	—
Tarwhine				
Zones	2	85.5	2.5	0.33
Sites (zones)	3	56.1	8.1	1.08
Residual	30	7.5	7.5	—
Flat-tail mullet				
Zones	2	28	−23.6	—
Sites (zones)	3	310.8	45.6	1.23
Residual	30	37.2	37.2	—
Flat-tail mullet (zones pld)				
Zones	2	—	—	—
Sites (zones)	3	197.7[(5)]	26.7	0.72
Residual	30	37.2	37.2	—

VC values are components of variance calculated from mean squares; Θ the ratio of estimated variances/residual variance; [(5)] the degrees of freedom after the mean square for zones was pooled.

Negative components of variation were more common on the largest scale, so zones were often pooled with sites and re-analysed (Table 1). Most negative components then disappeared. There was a general pattern of more variation between replicate nets than between sites when components were scaled against the residual (to assess the magnitude of variation of each factor relative to the same reference, rather than relative to each other; see Underwood, 1997). The results indicate that it is unnecessary to include zones in future sampling; more effort should be put into sampling replicate nets and sites in these estuaries.

Experiment 2: temporal variation

Short-term variation (e.g. day-to-day and week-to-week) potentially confounds comparisons across longer scales (e.g. month-to-month and season-to-season). Differences from one time of sampling to the next cannot be interpreted as being associated on any larger scale (e.g. season-to-season), unless it has been demonstrated that differences on smaller scales (i.e. daily or weekly) are not as large (Morrisey *et al.*, 1992b). The only alternative is the costly one of measuring variation at the larger scales (month-to-month or season-to-season) many independent times.

A second experiment was done to examine variation in populations and assemblages of estuarine fish across weeks, months, and seasons, again at two depths (shallow: <2 m; deep: 4–8 m) in a coastal lake (St Georges Basin). Sampling also incorporated two spatial scales within the estuary: two sites 1 km apart nested in each of two zones (2–20 km apart; Figure 3) to measure spatiotemporal interactions and to provide greater generality. We did not attempt to measure variation among the nights sampled, because it takes several nights (in this case four) to sample all four sites across the two zones. Sites were sampled for two consecutive weeks, in 2 months in two consecutive "seasons" (July/August 2005, winter; October/November 2005, spring) to test the hypotheses that abundances of fish vary at each temporal scale.

For frequently abundant species (occurring in >25% of samples), we analysed three factors (seasons, months, and weeks) separately for each site and depth, using nested analyses, then extracted the components of variation. As explained above, "zones" were not included in the model; instead, sites were treated as replicated experiments and analysed separately, giving four independent estimates of each component of variation. For most analyses, components of variation were negative at one or more scales, most frequently at the scales of weeks and months. Therefore, the data were pooled as four times of sampling in each season and re-analysed (Table 2).

Components of variation for species at deep sites were generally larger for seasons than for times within seasons. This pattern, however, was not consistent for the shallow sites. The only consistent pattern was that the residual variation (which is a spatial component) was greater than any temporal scale for every species (Table 2).

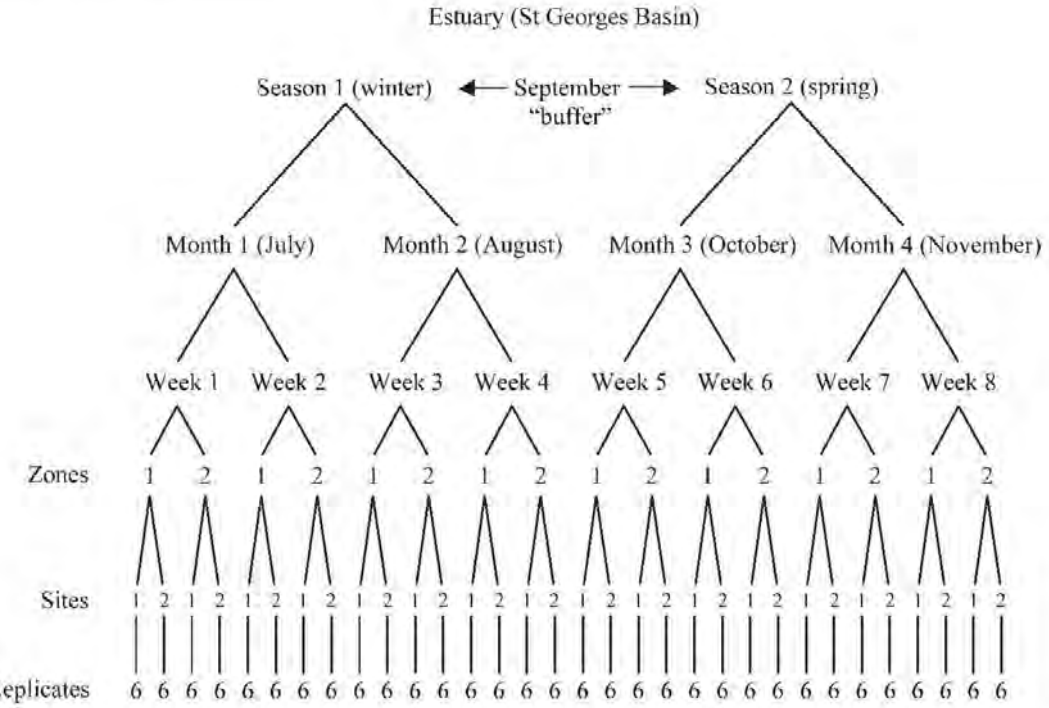

Figure 3. Diagrammatic representation of hierarchical scales sampled, with multimesh gillnets in an experiment on temporal variation of estuarine fish.

Fishery-independent surveys are often time-consuming, labour-intensive, and expensive. Because of these limitations, multiple sites and estuaries often cannot be sampled on the same night or even the same week. Therefore, any spatial comparisons are potentially confounded by time, because most sites (and estuaries) are sampled on different nights or in different weeks. Here, for most species, temporal variation was small compared with spatial variation. Therefore, as long as several sites within an estuary are sampled within a season, it is reasonable to interpret observed differences among sites as representing spatial rather than temporal variation. Further research, however, is needed to test whether similar patterns are observed over temporal scales longer than a few months.

Table 2. Examples of temporal variation and cost–benefit calculations for optimizing the design of sampling.

Source of variation	d.f.	MS	VC	Θ
Three-month periods	2	40.59	1.31	0.23
Times (three-months)	6	9.26	0.58	0.10
Residual	40	5.78	5.78	–
$n = 7$; $t = 4$				

Data are from St Georges Basin, shallow areas. VC values are components of variance; Θ, as in Table 1. n and t the optimal numbers of replicates and times of sampling, respectively, for each 3-month period. Costs were in hours: 1.5 per sample (three people for 0.5 h setting, retrieving net; sorting catch; entering, analysing data); six per time of sampling (three people × 2 h for travel). The example is dusky flathead (*Platycephalus fuscus*). Months within 3-month periods were pooled with weeks (months) because of the former's negative estimate of variance. Mean numbers of fish per sample were 3.4; variances were calculated using 20% (i.e. 0.63).

Step 4: cost–benefit analyses to determine optimal levels of replication

The final step in the strategy uses estimates of spatial and temporal variance (Step 3) to do cost–benefit analyses (e.g. Underwood, 1997). These well-known techniques allow determination of optimal levels of spatial and temporal replication (in terms of minimizing the imprecision of estimated means, the benefit) given restrictions of time, money, or both (the cost). Cost–benefit analyses for the present example are still in progress and will form the basis of a future publication. Applied examples focusing on marine biota, however, can be found in Green and Hobson (1970), Kennelly (1989), and Kennelly *et al.* (1993).

An example is shown for dusky flathead (*Platycephalus fuscus*; Table 2). As described above, weeks and months were pooled. Approximate costs are used solely to illustrate the analyses (costs are in Table 2). It was decided to optimize sampling, based on the precision of sampling, i.e. the standard error around each seasonal estimated mean number of fish. For example, 20% of the mean was chosen as the standard error. The number of samples, n, was then calculated from standard cost–benefit equations. The number of times, t, was determined using the standard error [see Underwood (1997) for all methods].

Of course, a different choice of precision would influence t. The value 20% is a compromise between excessive imprecision and excessive cost. In reality, precision in any sampling programme would be greater because it would be based on sampling from

several seasons, increasing the degrees of freedom in estimated standard error, and reducing confidence intervals around estimates of means. For this example, optimal allocations of sampling in any season are to take $n = 7$ replicates. There should be four times of sampling in each season (Table 2).

Inevitably, different optimal strategies would result for different species, each with different patterns of spatial or temporal variance. Part of the strategy for surveys, therefore, will be to determine appropriate compromises and to understand how to "weight" imprecision for different species in accordance with their importance, commercial value, etc. This will be discussed elsewhere with details of the outcome of analyses for the fisheries in NSW estuaries.

Conclusions

Using appropriately designed experiments to identify the most useful gear and to get estimates of variances at relevant spatial and temporal scales provides the essential information for planning fishery-independent surveys of fisheries resources. Optimal allocation of sampling units as replicates at different scales can then be done using standard methods of experimental design. Using a strategy designed to integrate knowledge of the most useful gear in the most effective sampling designs allows more reliable information from sampling that is independent of the fisheries themselves.

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Appendix 5: Johnson, D.D., Rotherham, D., Gray, C.A. 2008. Sampling estuarine fish and invertebrates using demersal otter trawls: effects of net height, tow duration and diel period. Fish. Res. 93: 315–323.

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Sampling estuarine fish and invertebrates using demersal otter trawls: Effects of net height, tow duration and diel period

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Australia

ABSTRACT

Trawl surveys are done in many parts of the world but few studies have used pilot experiments to examine the effects of gear design and sampling practices on retained samples. We did two separate experiments in two south-eastern Australian estuaries to test the hypotheses that samples of fish and invertebrates taken using a demersal otter trawl would be affected by: (i) vertical net height (standard 0.8 m vs. high-rise 1.2 m) and tow duration (5 vs. 10 vs. 20 min); and (ii) diel period (day vs. night) and tow duration. Mean catch-per-unit-effort (standardised to numbers of individuals caught 5 min^{-1}) was significantly larger in the high-rise net for some variables, but in most cases, CPUE was correlated proportionately between the two heights of net. There were no differences in the structure and composition of assemblages between net heights or among tow durations. The size-frequencies of fish differed between net heights and among tow durations, but there were no clear patterns in the proportions of fish caught across different size classes. Several species were caught mostly, or in larger numbers, at night and contributed to differences in assemblages of fish and invertebrates between diel periods. We stress the importance of using pilot studies to develop more reliable and cost-effective surveys and conclude that future sampling with our trawl should be done at night using the standard net and short (5 min) tows.

1. Introduction

Demersal otter trawls have been used to sample populations and assemblages of fish and invertebrates in many different types of habitats, ranging from shallow estuaries to deep oceanic waters, throughout the world (e.g., Doubleday and Rivard, 1981; Stoner, 1986; Andrew et al., 1997). Although otter trawls are easily utilised and relatively efficient at sampling large numbers and diversities of organisms, the catchability of these gears may be affected by a range of technical and operational factors. Some of these factors include: (i) the design of the net (DeAlteris et al., 1989; Merrett et al., 1991; Stender and Barans, 1994; Wantiez, 1996; West, 2002); (ii) the duration of tows (Godo et al., 1990; Walsh, 1991; Somerton et al., 2002); and (iii) diel period (Engas and Soldal, 1992; Korsbrekke and Nakken, 1999; Petrakis et al., 2001).

Generally, trawl surveys aim to provide representative estimates of the relative abundance and population structure of the species of interest (Gunderson, 1993). So, before implementing large-scale and long-term monitoring surveys, an important first step is to test the effects of different configurations of gear and sampling practices on retained samples (Gunderson, 1993; Rotherham et al., 2007). This is a necessary precursor to designing more accurate, precise and cost-effective sampling strategies (DeAlteris et al., 1989; Kennelly et al., 1993). Unfortunately, there are few examples of this sort of pilot work being done prior to implementing trawl surveys (e.g., Kennelly et al., 1993). In many cases, the effects of technical factors and sampling practices on the catchability of bottom trawls have not been examined until several years after surveys have commenced (e.g., Godo et al., 1990; Alderstein and Enrich, 2003; Weiland and Storr-Paulsen, 2006). This is a problem because changing configurations of trawls and sampling strategies to more efficient designs (e.g., shorter tow durations), may interrupt the time series of data (Somerton et al., 2002).

Much of the previous work examining the effects of trawl design and sampling practices has involved deep-water trawls, particularly in the northern hemisphere (see references above). Thus, many of the predictions of trawl theory (e.g., that long tows are more efficient than short tows at catching larger fish; Wardle, 1986; Godo et al., 1990) have not been tested on species in other sorts of environments, such as estuaries. This lack of information makes designing reliable and efficient surveys difficult.

The aim of the present study was to test the effects of net height (vertical opening of net), tow duration and diel period on catch rates, assemblages and size-frequencies of estuarine fish and invertebrates in an otter trawl in New South Wales (NSW, Australia). Results from these experiments can be used towards developing a

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standardised gear and sampling strategy for future trawl surveys of estuarine fauna in this region. Estuaries in NSW support diverse assemblages of fish fauna (e.g., Gray et al., 1998). So, we used the results of both univariate and multivariate analyses to consider which net height, diel period and tow time would be most effective and reliable in estimating: (i) the abundances of key species across the widest possible range of sizes; and (ii) the species composition and diversity of assemblages.

2. Materials and methods

2.1. Experiment 1—effects of net height and tow duration

This experiment tested the hypothesis that catches rates, assemblages and size-frequencies of fish and invertebrates sampled by an otter trawl would be affected by vertical net height (0.8 and 1.2 m) and tow duration (5, 10 and 20 min). Two "Florida Flyer" (Hughes, 1972) trawl nets were constructed from diamond-shaped polyethylene (PE) netting. Both nets had a headline length of 6 m and nominal mesh sizes of 41 mm (≤2 mm diameter twine) in the body and 40 mm (knotted, ≤3 mm diameter twine) in the codend. These sizes of mesh are similar to those used conventionally in commercial prawn trawls in estuaries of NSW, (which are known to retain a wide diversity of estuarine fauna of various sizes (Liggins and Kennelly, 1996; West, 2002).

The two nets were identical except that in one net, two wedge shaped sections of netting (18 meshes wide and 54 meshes long) were inserted between the upper and lower panels of the wingends so that the net opened to a vertical height of 1.2 m (hereafter called the high-rise net). The other net (hereafter called the standard net) had a vertical opening of 0.8 m.

The size and configuration of our nets were chosen so they could be easily adapted to our research vessels. We could not test net heights larger than 1.2 m because doing so would have required longer upper and lower bridles than used in the standard net, thus confounding any effect of net height. One option was to also use longer bridles in the standard net, but this would have led to operational problems onboard the vessel and may have resulted in a reduction in the vertical opening of the standard net. So, both of our nets were attached to wooden otter boards using upper and lower bridles that were 2 m long. The two nets were towed simultaneously by a 10-m trawler (120 hp) rigged in a twin-gear configuration.

Sampling was done on two days and two nights in the Clarence River (29°42′S 153°37′E; see West (2002) for a description of the study area) during February 2004. Tows during the daytime were done between 0700 and 1300; and during the night between 2000 and 0200 (excluding sunrise and sunset). On each day or night of sampling, four replicate, paired comparisons (high-rise vs. standard net) were done for each tow duration (5, 10 and 20 min), with the order of tows randomly determined. Tow duration was measured from the time the trawls were in contact with the substrate until the beginning of haul back. The two different nets were randomly positioned on either side of the trawler among days and nights to minimize any potential biases. Similarly, all tows were done in a straight line to eliminate any biases caused by changes to the geometry of the trawl during turns. The trawls were towed at speeds of between 0.8 and 1.1 m s^{-1} (depending on current direction) at depths of between 3 and 12 m.

2.2. Experiment 2—effects of diel period and tow duration

This experiment tested the hypothesis that catches rates, assemblages and size-frequencies of fish and invertebrates sampled by the high-rise otter trawl would be affected by diel period and tow time (5, 10, 20 min). To provide greater generality, sampling was done at two sites (separated by 9 km) in the Hawkesbury River (33°57′ S 151°31′ E; see Liggins and Kennelly, 1996). Sampling was done at each site on three days and three nights over a 20-day period in August 2004. Replicate days and nights were selected at random but, to avoid non-independence, were not sampled within the same 24-h period. On each day or night of sampling, four replicates of each tow duration were done (in a random order) using a 6-m (80 hp) single-rig trawler. Otter boards and upper and lower bridles were of similar configurations used during Experiment 1. Day tows were completed between 08:00 and 13:00 and night tows between 19:00 and 01:00 (excluding sunrise and sunset). All sampling was done at depths of between 5 and 15 m; but during individual tows, depth was kept relatively constant.

2.3. Collection of data

In both experiments, organisms caught during each tow were sorted by species. Data collected included: the total weight of the catch; the total weight of prawns (Experiment 1 only); the total numbers of individuals of each species; and the sizes of economically important finfish (fork length—FL to the nearest 0.5 cm), crabs (carapace length – CL – to the nearest mm) and a subsample of 100 prawns (CL to the nearest mm).

2.4. Analyses of data

For both experiments, data were standardised by sampling effort (i.e., numbers 5 min^{-1}) because it was appropriate under the hypotheses tested, i.e., that tow duration affected catch rates (not the total numbers of fish) in the trawl. Analysis of variance (ANOVA) was used to test for differences in CPUE between net heights and among tow durations (both fixed factors) for the total number of individuals and abundant taxa of economic importance (Experiment 1). All ANOVAs were done separately for day and night. Diel period was not included as a factor in the model because days and nights were sampled within the same 24 h period and so, potentially, were not independent (see Underwood, 1997).

For Experiment 2, another ANOVA model was used to test for differences in CPUE (also analysed for total numbers of individuals and abundant species of economic importance) between sites (random factor) and diel periods (day vs. night; fixed factor); and among sampling periods (random factor nested in site × diel period) and tow durations (5, 10 and 20 min; fixed factor).

Prior to all ANOVAs, data were tested for homogeneity of variances using Cochran's test and where necessary, transformed to $\ln(x+1)$. For data that remained heterogeneous, α was set to 0.01 to reduce the risk of Type I errors (Underwood, 1997). Significant differences detected by ANOVA were further investigated using Student–Newman–Keuls (SNK) multiple-comparison tests. Where significant differences were detected between net heights in ANOVA (Experiment 1), Spearman's rank correlations (Sokal and Rohlf, 1995) were used to test the hypothesis that CPUE was correlated proportionately between the standard and high-rise nets.

Non-metric multivariate analyses (Clarke, 1993) from the PRIMER 5 software package (Version 5.2.2, PRIMER-E, Ltd, 2001) were used to investigate the effects of (i) net height and tow duration (Experiment 1) and (ii) diel period and tow duration (Experiment 2), on the structure and composition of assemblages of fish and invertebrates sampled. Data from Experiment 1 were analysed separately for both day and night because potentially, samples were not independent (see above). Further, school prawns (*Metapenaeus macleayi*) were not included in the multivariate anal-

Table 1
Weights (kg) of *M. macleayi* and total numbers of the 14 most abundant species of finfish caught in the standard and high-rise net of an otter trawl towed for 5, 10 and 20 min during the day and night in the Clarence River (Experiment 1)

Tow duration (min)	Day						Night						Total
	Standard net			High-rise net			Standard net			High-rise net			
	5	10	20	5	10	20	5	10	20	5	10	20	
Metapenaeus macleayi (kg)	4.1	7.2	11.5	6.5	8.5	20.6	1.1	4.0	3.8	1.8	5.8	5.2	80
Species of finfish													
Ambassis spp.	16	35	44	39	97	108	157	118	604	260	368	939	2785
Gerres subfasciatus	6	37	216	29	64	162	45	96	296	129	282	665	2027
Herklotsichthys castelnaui	26	43	21	73	142	143	111	62	152	495	169	341	1778
Acanthopagrus australis	43	59	113	49	115	134	70	214	217	124	281	290	1709
Gobiomorphus coxii	10	59	40	29	110	84	20	14	26	20	50	46	508
Sillago ciliata	3	15	43	6	11	50	13	15	111	15	27	165	474
Arius graeffei	1	1	9	4	0	20	6	19	54	20	34	105	273
Brachirus nigra	0	7	11	3	3	14	7	11	24	5	6	13	104
Arenigobius frenatus	1	15	33	4	10	10	1	5	0	8	9	7	103
Platycephalus fuscus	0	3	9	4	4	7	1	3	13	3	5	5	57
Hyperlophus vittatus	0	0	0	1	10	2	5	0	2	4	29	2	55
Synclidopus macleayana	4	5	18	2	3	9	2	0	6	0	0	2	51
Pseudorhombus jenynsii	1	1	8	2	2	6	1	1	5	0	1	1	29
Enoplosus armatus	0	0	1	0	0	1	0	0	7	3	0	13	25
No. of additional species	3	6	4	2	2	5	3	5	4	5	4	5	18
No. of additional individuals	3	8	11	3	2	6	5	8	8	12	11	8	85
Total no. of species	14	19	18	16	15	20	17	17	18	18	17	20	33
Total no. of finfish	114	288	577	248	573	756	444	566	1525	1098	1272	2602	10063

Table 2
Analyses of the effects of net height (Nh, S = standard; H = high-rise) and tow duration (Td, 5, 10 and 20 min) on the mean CPUE (numbers 5 min^{-1}) of the total numbers of individuals and abundant taxa of economic importance (*G. subfasciatus*, *H. castelnaui*, *A. australis* and *M. macleayi*) caught in an otter trawl during the day and night in the Clarence River (Experiment 1)

Source	df	Day			Night		
		MS	*F*	*P*	MS	*F*	*P*
(a) Total individuals							
Nh	1	3.30	9.20	**	4.80	19.61	***
Td	2	0.23	0.65	ns	0.68	2.78	ns
Nh × Td	2	0.22	0.62	ns	0.14	0.56	ns
Residual	42	0.36					
SNK		Nh: H > S			Nh: H > S		
Spearman's rank correlation		H vs. S (df = 22, r^2 = 0.26*)			H vs. S (df = 22, r^2 = 0.67***)		
(b) *G. subfasciatus*							
Nh	1	0.92	1.06	ns	12.27	17.65	***
Td	2	0.87	1.01	ns	0.36	0.52	ns
Nh × Td	2	0.70	0.81	ns	0.46	0.67	ns
Residual	42	0.86			0.69		
SNK					Nh: H > S		
Spearman's rank correlation					H vs. S (df = 22, r^2 = 0.57***)		
(c) *H. castelnaui*							
Nh	1	7.54	7.46	**	7.22	5.64	*
Td	2	1.94	1.92	ns	2.96	2.31	ns
Nh × Td	2	0.03	0.04	ns	0.02	0.02	ns
Residual	42	1.01			1.27		
SNK		Nh: H > S			Nh: H > S		
Spearman's rank correlation		H vs. S (df = 22, r^2 = 0.04)			H vs. S (df = 22, r^2 = 0.69***)		
(d) *A. australis*							
Nh	1	32.09	1.54	ns	232.98	2.09	ns
Td	2	16.43	0.79	ns	228.81	2.05	ns
Nh × Td	2	10.44	0.50	ns	20.11	0.18	ns
Residual	42	20.85			111.54		
(e) *M. macleayi*							
Nh	1	0.60	3.15	ns	0.05	2.34	ns
Td	2	0.15	0.77	ns	0.05	2.49	ns
Nh × Td	2	0.06	0.32	ns	0.00	0.10	ns
Residual	42	0.19			0.02		

Data for *M. macleayi* are weights (kg). Results of SNK and Spearman's rank correlation tests (see text) are shown. ns = not significant; *P < 0.05; **P < 0.01, ***P < 0.001.

yses of data for Experiment 1 because data were collected on the total weight (rather than number) of prawns in each trawl.

For each experiment, centroids were produced by averaging data across replicates from each sampling occasion. Bray-Curtis measures of similarity were then calculated using: (i) raw abundance data that were standardised to test whether assemblages were being sampled in the same relative proportions in different samples; and (ii) presence/absence data to test for differences in the species composition of assemblages. Non-metric multidimensional scaling (nMDS) and two-dimensional ordination plots were used to examine patterns of sample relatedness. Two-way analyses of similarities (ANOSIM) tested the hypotheses that the structure and composition of assemblages differed between (i) net height and tow duration (Experiment 1); and (ii) diel period and tow duration (Experiment 2). Where significant differences were detected by ANOSIM, the species primarily responsible for these differences were identified using the similarity percentages (SIMPER) procedure.

Kolomogorv–Smirnov (K–S) tests were used to test for differences in size-frequency distributions between net heights, tow durations and diel periods for abundant, economically important fish species.

3. Results

3.1. Experiment 1

A total of 10,063 finfish and 80 kg of penaeid prawns (*M. macleayi*) from more than 33 species were caught during the experiment (Table 1). Four taxa comprised more than 80% of the total catch by number and included: Ambassid spp. (28%), *Gerres subfasciatus* (20%), *Herklotsichthys castelnaui* (18%) and *Acanthopagrus australis* (17%). *M. macleayi* was also caught in large numbers and representing 22% of the total catch by weight. Species that were sampled exclusively by either the standard or high-rise net were all caught in relatively low numbers (<5 individuals in total for all tows combined).

ANOVA and SNK tests revealed that mean CPUE was significantly larger ($P < 0.05$) in the high-rise net for the total numbers of individuals, during both the day and night (Table 2a, Fig. 1a). Mean catch rates of *G. subfasciatus* (night) and *H. castelnaui* (day and night) were also significantly larger ($P < 0.05$) in the high-rise net (Table 2b and c; Fig. 1b and c). Despite these differences, Spearman's rank correlations were significant ($P < 0.05$) for all but one test (*H. castelnaui*, day; Table 2); indicating that, overall, CPUE was correlated proportionately between the two types of net. There were no significant differences ($P > 0.05$) between net heights for *A. australis* and *M. macleayi* (Table 2d and e; Figures not shown). Similarly, tow duration had no effect on mean CPUE of any variable analysed in either the day or night. Although diel period was not a factor in the ANOVA model, it is notable that, larger numbers of fish and species were generally caught at night (Table 1; Fig. 1).

MDS ordinations showed no clear groupings of samples from the different net heights and tow durations for either the standardised or presence/absence data (Fig. 2; example shown for standardised data only). These results were consistent between day and night collections and were supported by ANOSIM ($P > 0.25$).

A. australis and *G. subfasciatus* were sufficiently abundant to test the effects of net height and/or tow duration on size-frequency distributions of samples. K–S tests did not detect any differences between net heights for *A. australis* in either the day or the night. Similar comparisons between net heights could not be made for *G. subfasciatus* owing to insufficient data. Size-frequency distributions of *A. australis* were significantly ($P < 0.05$) different between

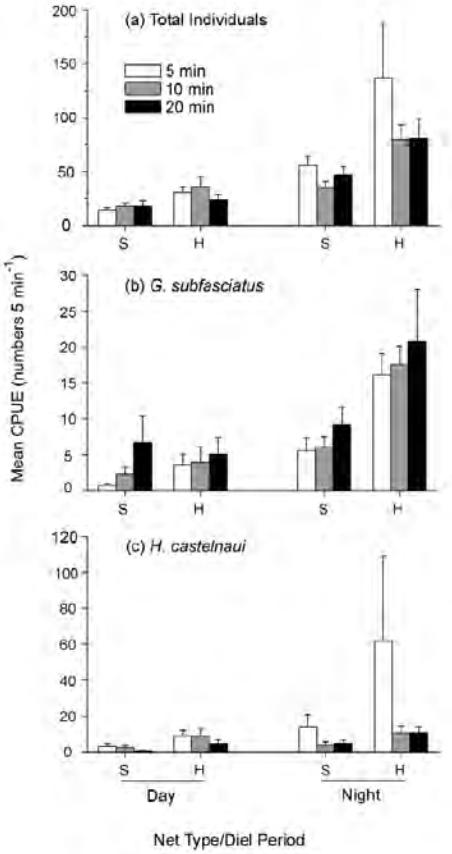

Fig. 1. Mean (+S.E.) CPUE (standardised to numbers 5 min^{-1}) of (a) total numbers of individuals, (b) *G. subfasciatus* and (c) *H. castelnaui* caught in each net type (S: standard; H: high-rise) and tow duration (5, 10 and 20 min) during the day and night in the Clarence River (Experiment 1).

tow durations of 5 and 20 min at night in the high-rise net, but there were no clear patterns in the proportions of size classes caught (figure not shown). K–S tests detected differences in size-frequencies between all tow durations (i.e. 5 vs. 10 min, 5 vs. 20 min, 10 vs. 20 min) for *G. subfasciatus* caught in the high-rise net at night. Again, however, there were no obvious patterns relating to these differences (figure not shown for brevity).

3.2. Experiment 2

A total of 13,567 individuals from more than 39 species were caught in the Hawkesbury River (Table 3). Three species (*Hyperlophus vittatus*, *H. castelnaui* and *A. australis*) accounted for approximately 75% of the total catch by number. In general, species such as *H. vittatus*, *A. australis*, *Pomatomus saltatrix* and *Liza argentea* were caught in greater numbers during the day, while, more individuals of *H. castelnaui*, *Argyrosomus japonicus* and *M. macleayi* were caught at night.

There were no consistent effects of diel period or tow duration on catch rates across the variables examined using ANOVA

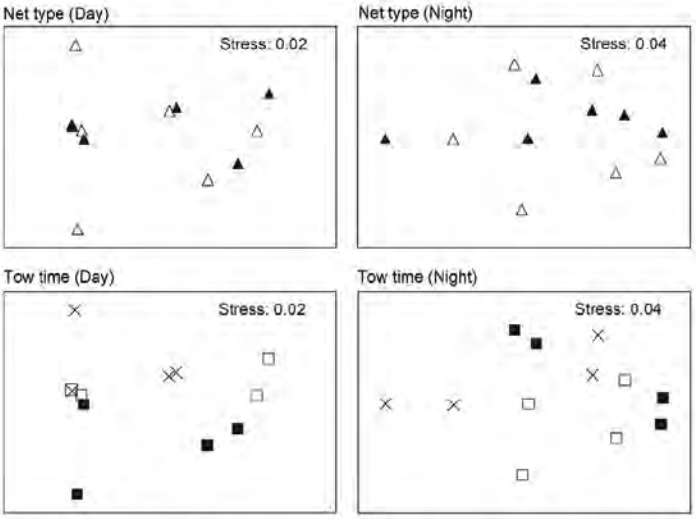

Fig. 2. Two-dimensional nMDS ordinations of standardised, abundance data showing relationships between assemblages of fish and invertebrates caught in each net type: standard (△) and high flyer (▲) and tow duration: 5 min (×), 10 min (□) and 20 min (■), during the day and night in the Clarence River (Experiment 1).

(Table 4); although the following results are worth mentioning. First, differences between diel periods were significant ($P<0.05$) for only two species (Table 4b and c), with catch rates of *A. australis* larger during the day than night (SNK tests, $P<0.01$; Fig. 3a); and *M. macleayi* larger at night than during the day (SNK tests, $P<0.01$; Fig. 3b). Second, there were significant site × diel and/or site × diel × tow interactions for some variables (Table 4; Fig. 3c; *H. vittatus* is shown as an example). Third, for *H. castelnaui* no differences were detected by ANOVA (not shown in Table 4 or Fig. 3).

MDS ordinations and ANOSIM examined differences in the structure and composition of assemblages between diel periods and among tow durations. For both the standardised and presence/absence data, ANOSIM detected significant differences between diel periods ($P<0.01$), but not among tow durations ($P>0.25$). These results were also supported by the ordination plots, with samples from the day and night separating into distinct groupings (Fig. 4; example shown for standardised data only). In contrast, there was no separation of samples from the different tow durations (Fig. 4).

SIMPER analysis revealed that differences in assemblages between diel periods were driven by several species that were caught in larger numbers at night (Table 5). For the standardised data, the more abundant species made a comparatively greater contribution to the average dissimilarity between assemblages. In contrast, taxa that were caught relatively infrequently or in small numbers made greater contributions to differences between diel periods in the case of presence/absence data.

Table 3
Total numbers of each species caught with a high-rise otter trawl using tow durations of 5, 10 and 20 min during the day and night in the Hawkesbury River (Experiment 2).

Tow duration	Day			Night			Total
	5 min	10 min	20 min	5 min	10 min	20 min	
Species							
Hyperlophus vittatus	1499	2144	1718	172	82	55	5670
Herklotsichthys castelnaui	8	38	426	234	416	1302	2424
Acanthopagrus australis	222	490	1017	43	85	193	2050
Metapenaeus macleayi	0	0	0	247	161	297	705
Pomatomus saltatrix	48	264	169	19	35	38	573
Argyrosomus japonicus	12	30	29	84	164	245	564
Sillago maculata	10	19	29	47	46	204	355
Synchidopus macleayana	1	1	6	39	55	249	351
Liza argentea	21	49	76	3	9	13	171
Penaeus plebejus	0	0	0	26	38	67	131
Ambassis spp.	0	3	1	21	27	53	105
Pseudorhombus jenynsii	2	2	10	14	22	53	103
Arenigobius frenatus	0	0	0	11	40	36	87
Gerres subfasciatus	6	17	29	0	3	4	59
No. of additional species	3	8	13	12	12	16	25
No. of additional individuals	9	19	28	37	47	79	219
Total no. of species	13	19	24	25	26	30	39
Total no. of individuals	1838	3076	3538	997	1230	2888	13567

Table 4
Analyses of the effects of site (Si; random), diel period (Di; fixed), sampling period (Pe; random and nested within Si × Di) and tow duration (Td; fixed) on the mean CPUE (numbers 5 min^{-1}) of the total numbers of individuals and abundant taxa of economic importance (*H. vittatus*, *A. australis*, *M. macleayi*, *P. saltatrix* and *A. japonicus*) caught in an otter trawl in the Hawkesbury River (Experiment 2)

Source		MS	F	P	MS	F	P
		(a) Total individuals			(b) *A. australis*		
Si	1	13.14	3.45	ns	3.24[9]	1.44	ns
Di	1	0.53	0.02	ns	44.43[9]	19.72	*
Pe(Si × Di)	8	3.81[126]	5.87	**	2.16[p]		
Td	2	1.24	0.87	ns	0.74[126]	1.13	ns
Si × Di	1	23.15	6.07	*	2.99[p]		
Si × Td	2	1.43[126]	2.20	ns	0.38[p]		
Di × Td	2	0.86[126]	1.32	ns	0.52	0.37	ns
Td × Pe(Si × Di)	16	0.73[p]			0.34[p]		
Si × Di × Td	2	0.04[p]			1.42[126]	2.18	ns
Residual	108	0.65			0.70		
SNK					Di: D > N		
		(c) *M. macleayi*[l]			(d) *H. vittatus*		
Si	1	0.35[9]	0.18	ns	18.20	2.02	ns
Di	1	64.55[9]	33.62	**	66.47	0.90	ns
Pe(Si × Di)	8	2.12[p]			9.02	5.77	***
Td	2	0.32	0.28	ns	6.58	4.21	*
Si × Di	1	0.35[p]			73.63	8.16	*
Si × Td	2	1.17[124]	4.03		1.66[p]		
Di × Td	2	0.32	0.28	ns	0.68	0.14	ns
Td × Pe(Si × Di)	16	0.21[p]			0.74[p]		
Si × Di × Td	2	0.17[124]	4.03		4.91	3.14	*
Residual	108	0.30			1.68		
SNK		Di: N > D					
		(e) *A. japonicus*			(f) *P. saltatrix*		
Si	1	7.73	19.43	**	9.61	5.76	*
Di	1	29.15	12.47	ns	6.67	2.24	ns
Pe(Si × Di)	8	0.40	1.31	ns	1.67[126]	3.72	**
Td	2	0.01[16]	0.03	ns	1.35	3.00	ns
Si × Di	1	2.34	5.88	*	2.98[128]	1.77	ns
Si × Td	2	0.39[p]			0.35[p]		
Di × Td	2	0.08	0.09	ns	1.09[128]	2.42	ns
Td × Pe(Si × Di)	16	0.47			0.39[p]		
Si × Di × Td	2	0.83[16]	1.80	ns	0.13[p]		
Residual	108	0.30			0.47		

Results of SNK tests are shown. ns = not significant; * $P<0.05$; ** $P<0.01$, *** $P<0.001$. [l] = data remained heterogeneous after transformation. Numbers in parentheses are the degrees of freedom following the pooling (denoted by [p]) of non-significant interaction terms at $P>0.25$. Results of analyses are not shown for *H. castelnaui* because there were no significant differences ($P>0.05$).

There was enough data to examine the effects of tow duration on the size-frequency distributions of only *A. australis* (separately for both day and night), *A. japonicus* (night only) and *H. castelnaui* (night only). Significant differences in the size-frequency distributions of *A. australis* were detected between all tow durations (5 vs. 10 min, 5 vs. 20 min, 10 vs. 20 min) during the day; but only between 10 and 20 min tows at night (K–S tests, $P<0.05$). For all of these comparisons, however, there were no clear trends in the proportions of fish caught across different size classes (figures not shown). There were no significant differences in the size-frequency distributions of *A. japonicus* and *H. castelnaui* among tow durations (K–S tests, $P>0.05$). Comparisons of size-frequency distributions between diel periods for each tow duration could not be done for key species owing to insufficient data.

4. Discussion

Mean CPUE was larger in the high-rise net for the total numbers of fish, as well as some individual taxa known to form large schools (e.g., *H. castelnaui* and *G. subfasciatus*; Kuiter, 1996). These differences in catch rates were probably related to the greater volume of water above the substratum sampled by the high-rise net. Similarly, Stender and Barans (1994) reported larger catches of white shrimp, *Penaeus setiferus*, in a tongue net that also sampled a greater proportion of the water column.

We found no differences in the composition of assemblages or the size ranges of fauna caught between the standard and high-rise nets. This contrasts the large differences in the species composition and size structure of samples from different types of trawl nets reported elsewhere (DeAlteris et al., 1989; Merrett et al., 1991; Stender and Barans, 1994; Wantiez, 1996; Wassenberg et al., 1997; West, 2002). Most of these previous studies tested trawls with relatively large differences in their design (e.g., mesh size, ply diameter, length of headlines and footropes) and configuration (e.g., length of sweeps and bridals, single- vs. paired-warp trawls). In our experiment, we controlled for all technical factors other than the vertical net opening. This may explain the similarities in the assemblages and the sizes of fish caught between the two nets tested here.

The decision to use the high-rise net for Experiment 2 was initially based on the larger catch rates of some variables in this net during Experiment 1. Nevertheless, subsequent analyses (Spearman's rank correlation tests) revealed that, for most variables, CPUE was correlated proportionately between the two different nets. Further, there were neither differences in the structure and composition of assemblages, nor in the size-frequencies of abundant taxa between net heights. Thus, we consider the standard net to be more appropriate for future surveys of estuarine fish fauna in NSW because it would reduce: (i) the numbers of organisms that are harmed or killed; and (ii) the need for subsampling, which is time-consuming and can introduce bias and error (Godo et al., 1990).

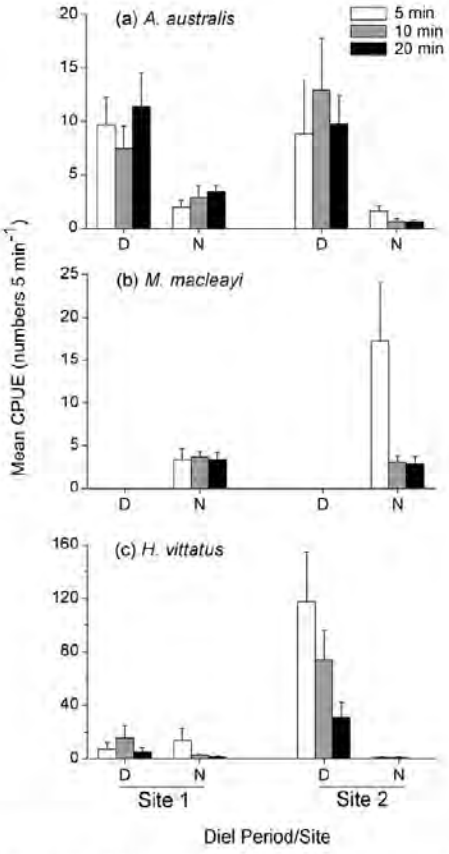

Fig. 3. Mean (±S.E.) CPUE (standardised to numbers 5 min^{-1}) of (a) *A. australis*, (b) *M. macleayi* and (c) *H. vittatus* caught in a high-rise otter trawl for each diel period (D: day; N: night) and tow duration (5, 10 and 20 min) at two sites in the Hawkesbury River (Experiment 2).

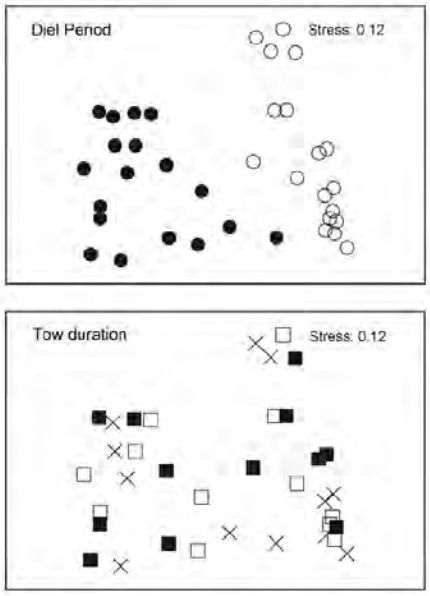

Fig. 4. Two-dimensional nMDS ordinations of standardised, abundance data showing relationships between assemblages of fish and invertebrates caught in an otter trawl for each diel period: day (○) and night (●); and tow duration: 5 min (×), 10 min (□) and 20 min (■) in the Hawkesbury River (Experiment 2).

In theory, shorter tows are expected to underestimate the proportion of larger fish in populations, because some fishes (particularly larger individuals) have a greater capacity to swim in front of a trawl for longer periods of time (Wardle, 1986; Godo et al., 1990). Despite this prediction, previous studies have often found that shorter tows are more efficient than longer tows and that, the duration of tows has no effect on the sizes of fish or crabs retained (Godo et al., 1990; Walsh, 1991; Somerton et al., 2002). These results have been explained by: (i) the catch-by-surprise hypothesis, i.e., fish are more vulnerable to capture in the first few minutes of a tow because they are not alerted by fish that are herding in front of the trawl (Godo et al., 1990); (ii) errors in the measurement of the tow path (Godo et al., 1990); and (iii) time varying escapement of fish and crabs under the footrope owing to changes in trawl geometry (Somerton et al., 2002).

In our experiments, tow duration had no effect on mean CPUE of the total numbers of individuals, or abundant, economically important taxa. There were, however, differences in size-frequencies among tow durations for some species, but no clear patterns in these differences. We also found no effect of tow duration on the size structure and species composition of assemblages, indicating that the sampling efficiency of shorter tows does not appear to be affected by the swimming endurance of the species caught in these experiments. Further, there were no patterns in our data to suggest any catch-by-surprise mechanism (Godo et al., 1990).

Clearly, the effects of tow duration cannot be generalised across all species and systems, which further highlights the need for pilot studies to determine appropriate tow durations prior to commencing research surveys with trawls (although this is rarely done – see Section 1). Based on our results, we consider 5-min tows to be appropriate for future sampling with either the standard, or high-rise net. The main benefit of using 5-min tows is that more replicates can be done than with longer tows, thus improving the overall precision of surveys without large increases in costs (Pennington and Vølstad, 1991). Short tows also reduce large catches and the need for subsampling (Godo et al., 1990).

Diel period affected the structure and composition of assemblages of fish fauna. Dissimilarities between assemblages were related to several species that were caught predominantly, or in larger numbers, at night. These results are similar to previous studies that have used trawls and other sampling gears in shallow, estuarine waters of eastern Australia (Gray et al., 1998; Guest et al., 2003). Although mechanisms explaining observed patterns are beyond the scope of this study, the differences in assemblages between diel periods may be related to the behaviour of fish and invertebrates in response to predators and prey (Helfman, 1993; Gibson et al., 1998), greater avoidance of the trawl during the day owing to enhanced visibility (see Petrakis et al., 2001 and references within), or both.

Given the observed differences in assemblages between diel periods, we conclude that future sampling with the trawl should be done at night. Although sampling at night can be problematic (e.g.,

Table 5
Average abundances of species contributing to dissimilarities in the abundance (standardised data) and composition (presence/absence data) of assemblages of fish and invertebrates caught between the day and night in the Hawkesbury River (Experiment 2)

Species	Day	Night	δ_i
Standardised data			
Hyperlophus vittatus	74.46	4.29	26.99
Acanthopagrus australis	24.00	4.43	18.06
Herklotsichthys castelnaui	6.56	27.11	16.91
Metapenaeus macleayi	0.00	9.79	10.27
Agyrosomus japonicus	0.99	6.85	6.97
Pomatomus saltatrix	6.68	1.28	3.85
Soleichthys microcephalus	0.11	4.76	3.53
Sillago maculata	0.81	4.13	2.81
Penaeus plebejus	0.00	1.79	1.91
Presence/absence data			
Metapenaeus macleayi	0.00	9.79	7.83
Penaeus plebejus	0.00	1.79	7.38
Synchidopus macleayana	0.11	4.76	5.13
Pseudorhombus jenynsii	0.19	1.24	4.86
Ambassis spp.	0.06	1.40	4.56
Liza argentea	2.03	0.35	4.21
Callonymidae spp.	0.10	0.53	4.10
Herklotsichthys castelnaui	6.56	27.11	4.04
Pseudorhombus arsius	0.18	0.51	3.93
Girella tricuspidata	0.17	0.00	3.92
Hyperlophus vittatus	74.46	4.29	3.91
Arenigobius frenatus	0.00	1.11	3.85
Gerres subfasciatus	0.72	0.10	3.78
Platycephalus fuscus	0.00	0.14	3.32
Centropogon australis	0.01	0.17	3.09
Agyrosomus japonicus	0.99	6.85	2.71
Pomatomus saltatrix	6.68	1.28	2.67
Chelidonichthys kumu	0.01	0.11	2.64
Selenotoca multifasciata	0.04	0.06	2.39
Brachirus nigra	0.00	0.15	2.04

Species are listed in order of their contribution, to the average dissimilarity between groups, δ_i, as determined by SIMPER. Only taxa that contributed to at least 2% of dissimilarity are shown.

safety, cost), sampling during the day would not provide representative samples of the diversity of assemblages. Only one abundant species (*A. australis*) was caught in larger numbers during the day (Experiment 2). In Experiment 1, however, larger numbers of *A. australis* were caught during the night (but diel period was not included as a factor in the analyses). Therefore, although not tested in this study, diel-related effects may be inconsistent for some species across different estuaries. This illustrates a problem of sampling multi-species assemblages; strategies may be appropriate for the majority of taxa, but suboptimal for others (Kennelly et al., 1993). So, compromises between costs and effort, logistic constraints and the accuracy and precision of samples, are often required.

The taxa caught in our experiments, which included large numbers and several species of commercial and recreational importance, are typical of the fauna caught in trawls in estuaries of NSW (Gray et al., 1990; Liggins and Kennelly, 1996; West, 2002). Should otter trawls be used as a tool for future sampling of estuarine fauna in NSW, we recommend that sampling be done at night using the standard net and tows of 5 min. Developing robust and optimal designs of sampling for long-term surveys (i.e., the numbers of replicate trawls, sites, sampling occasions, etc.), however, requires additional experiments quantifying spatial and temporal variation in patterns of abundance and the use of cost-benefit analyses (e.g., Kennelly et al., 1993; Rotherham et al., 2007).

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Appendix 6: Gray, C.A., Jones, M.V., Rotherham, D., Broadhurst, M.K., Johnson, D.D., Barnes, L.M., 2005. Utility and efficiency of multi-mesh gill nets and trammel nets for sampling assemblages and populations of estuarine fish. Mar. Freshw. Res. 56: 1077–1088.

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Utility and efficiency of multi-mesh gill nets and trammel nets for sampling assemblages and populations of estuarine fish

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Abstract. Two replicate multi-mesh gill and trammel nets, each comprising five 30 m long panels made from different-sized mesh (38, 54, 70, 90 and 100 mm stretched mesh openings) were fished in a south-east Australian barrier estuary over seven nights to evaluate their potential as sampling gears for fishery-independent surveys of estuarine fish assemblages. There were no differences in composition and structure of assemblages, mean abundance, or diversity of catches between the two types of net. The composition and structure of catches differed between mesh sizes, with the panels made from 38 and 54 mm mesh retaining significantly more fish and species than the larger-sized meshes. The two smallest mesh sizes were important for capturing sub-adults and juveniles of some species. Based on a greater precision of catch per unit effort (CPUE) estimates, less sampling effort and greater ease of use, the multi-mesh gill net was a better sampling unit than the trammel net for assessments of estuarine fish populations.

Extra keywords: CPUE, fishery-independent survey, power, variation.

Introduction

Assessments of fish stocks and assemblages often require information on the relative abundance, size, sex and age compositions of populations, and how they vary spatially and temporally (Hilborn and Walters 1992). For many harvested species, these data traditionally have been derived from fishery-dependent sources, such as fishers' logbooks and/or the scientific sampling of retained landed catches (Doubleday and Rivard 1983). It is well acknowledged, however, that owing to inherent fisher- and fleet-specific variations among gears, practices and the targeted species, such fishery-dependent data are often biased. The preferred method for acquiring data for the assessment of stocks and assemblages involves fishery-independent stratified and randomised research surveys (Pennington and Strømme 1998). Such sampling is already necessary in areas where no fisheries exist and is increasingly being used as a key scientific tool to assess many important fish and invertebrate populations throughout the world (Warnes and Jones 1995; Reid *et al.* 1999; Korsbrekke *et al.* 2001; Kennelly and Scandol 2002).

Estuaries support diverse and abundant ichthyofaunas (Potter and Hyndes 1999; Blaber 2000) that are often heavily exploited by recreational and commercial fishers and subjected to many other anthropogenic perturbations, including habitat degradation, discharge of pollutants and impacts from adjacent land-use. Whereas a plethora of studies have described and compared fish populations in different habitats and estuaries throughout the world (e.g. Claridge *et al.* 1986; Loneragan *et al.* 1987, 1989; Heck *et al.* 1989; Potter *et al.* 1990; Gray *et al.* 1996), few long-term surveys have been used to monitor and assess the status and impacts of changes of different management regimes on exploited estuarine fish stocks.

Estuaries throughout south-eastern Australia support valuable recreational and commercial fisheries, with over 6000 tonnes of fish landed annually in New South Wales (NSW) alone. Although catches comprise a diverse range (>100 species) of fish, relatively few species (<10) account for most (>80%) of the landed catches, but this can vary among gear types and estuaries (Pease 1999; Gray 2002; Gray and Kennelly 2003; Gray *et al.* 2004). Recreational fishers primarily use handlines, whereas commercial fishers mostly use gill and seine nets. At present, fishery-dependent data derived from surveying and sampling catches of recreational (e.g. Steffe and Chapman 2003) and commercial (e.g. Gray *et al.* 2002) fishers using these gears are used to monitor the status of estuarine fish stocks. While such fishery-dependent data

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are invaluable, owing to considerably divergent management strategies among estuaries (i.e. open or closed to different fishing sectors and gears), it is difficult to assess comparatively populations and assemblages of key species. Therefore, a standardised, fishery-independent method of assessing populations and assemblages of fishes across estuaries is desired.

One of the first steps in designing and developing any research survey is to test sampling gears that are efficient and cost-effective in terms of providing representative and standardised samples of the species of interest. Preferably, such sampling should encompass different life history stages of the various key species. A range of towed and static sampling gears, including demersal otter trawls (Smith and Gavaris 1993; Korsbrekke *et al.* 2001), beam trawls (Rogers *et al.* 1997; Hamer and Jenkins 2004), seine and gill nets (Degerman *et al.* 1988; Acosta and Appledoorn 1995; Poulakis and Mitchell 1999), longlines (Simpfendorfer *et al.* 2002) and traps (Kennelly 1992; Smith and Tremblay 2003), have been used to provide independent samples of populations and assemblages of fish and crustaceans in marine and freshwater environments. All of these sampling gears, however, have limitations and the descriptions of populations and assemblages of organisms are invariably confounded by type and selectivity of gear (Miller 1990; Gunderson 1993).

In this paper, we evaluate the utility and efficiency of two types of passive sampling gears – multi-mesh trammel and gill nets – as potential research sampling gears for fishery-independent research surveys of key estuarine fish species in south-eastern Australia. These sorts of gears have been used extensively to sample fish populations in many coastal and freshwater regions throughout the world (e.g. Loneragan *et al.* 1987; Degerman *et al.* 1988; Mattson and Mutale 1992; Acosta 1997) primarily because they are inexpensive, easily deployed and retrieved, and can be successfully fished with minimal environmental impact across a range of habitats. Whilst previous studies have assessed differences in the selectivity of various gill and trammel nets used in specific fisheries throughout the world (Fabi *et al.* 2002; Moth-Poulsen 2003), few studies have compared their utility as tools for independently sampling a wide range of fish species of differing sizes and morphologies (but see Acosta 1997). Our specific aims in this study were to: (*i*) compare the compositions and relative abundances of species and lengths of key species captured in the multi-mesh trammel and gill nets, and (*ii*) determine the precision of estimated mean catch per unit effort (CPUE) and (*iii*) estimate levels of optimal replication for each sampling gear. The latter estimates provide an indication of the level of effort required to use each net type for future large-scale surveys of estuarine fish assemblages.

Materials and methods

Sampling

Sampling was done over seven nights between 8 and 19 September 2003, in Tuggerah Lake (151°30′E 33°21′S), a relatively large (70 km^2 surface area) and shallow (average depth of 1.9 m) barrier estuary in central NSW. The lake supports valuable commercial and recreational fisheries for fish and crustaceans, with the main commercial fish species caught being *Mugil cephalus*, *Girella tricuspidata* and *Platycephalus fuscus*.

Two replicate multi-mesh gill and trammel nets (i.e. a total of four nets) were constructed. Each net measured 162 m in total length and comprised five panels (1.5 m deep × 30 m long), all made from polyamide (PA), monofilament netting with the same hanging ratio (0.5), but different stretched mesh sizes (38, 54, 70, 90 and 100 mm, respectively). Each panel of the trammel net had two outer panels (1.5 deep × 30 m long) and an inner panel (2.4 deep × 30 m long) of 300 mm (stretched mesh) PA monofilament netting all secured to the same head- and foot-rope. Twine diameter was 0.15 mm for the 38 and 54 mm mesh, and 0.2 mm for all other meshes. Each 30 m panel had a head-rope (patent floatline No. 22, Oy Lindeman Ab, Finland) that incorporated 100 evenly spaced 16 × 72 mm floats and a 34 m, 0.75 kg foot-rope (patent leadline No. 28, Oy Lindeman Ab). To ensure independence among replicate panels and to restrict the potential effect of the smaller-mesh panels 'leading' larger fish to adjacent larger-meshed panels, the head- and foot-rope of each adjacent panel were separated by 3 m lengths of 4 mm diameter twisted polyethylene rope. The order of the panels of different-sized mesh in each net was randomised daily.

The two replicate trammel and gill nets were simultaneously fished in the same area of the lake for each of the seven nights. All nets were bottom set (depth 1.5–4 m) between 1700 and 1800 hours (to avoid predation by birds) and retrieved 3 h later. Catches in each panel were separated and sorted by species. The number, weight (g), fork length (FL, to nearest 0.5 cm) and the method of retention (i.e. entangled or gilled) were recorded. Crabs were identified, counted and weighed. The time spent sorting the catch from each type of net was also recorded.

Statistical analyses

Analysis of variance (ANOVA) was used to test the hypothesis of no differences in catches of fish (expressed as mean CPUE and analysed separately for the total number of individuals, the total number of species and the most abundant economically important species—*M. cephalus*, *Liza argentea* and *G. tricuspidata*), between net types and mesh sizes (both fixed factors). Data were tested for normality (Shapiro–Wilk test) and homogeneity of variances (Cochran's test) and where necessary, transformed using log (x + 1). For data that remained heterogeneous, α was set to 0.01 to reduce the risk of Type I errors (Underwood 1981). Student–Newman–Keuls (SNK) tests were used to determine significantly different mean values.

Where significant results were detected in ANOVA and SNK tests for the total numbers of fish and species in the present study, Spearman rank correlations were used to test whether or not catches from replicate samples were correlated (caught in the same relative proportions) between relevant treatments. Correlations were not done for abundant, economically important species, because differences in the size distributions of individual fish species were evident between mesh sizes, and future use of the different mesh sizes may be required to sample individuals across a wide range of sizes that incorporate different stages of life-history. Differences between size-frequency distributions of key species caught in the trammel and gill net were tested using Kolmogorov–Smirnov (KS) tests (Sokal and Rohlf 1995).

Non-parametric multivariate analyses from the PRIMER 5 package (Version 5.2.2, PRIMER-E Ltd, 2001) were used to test for differences in the composition and structure of catches of fish between net type and mesh size. The general procedures used followed those outlined in Clarke (1993) and Clarke and Warwick (1994). Replicate samples for each night were pooled and grouped into net type/mesh size combinations. Data were transformed to presence/absence to test whether

Table 1. List of species and number of individuals caught with multi-mesh gill and trammel nets in Tuggerah Lake (NSW)
$n = 14$ for each net type

Family	Scientific name	Common name	Gill	Trammel
Belonidae	*Strongylura leiura*	Longtom	3	0
Bothidae	*Pseudorhombus jenynsii*	Small-tooth flounder	3	1
Clupeidae	*Herklotsichthys castelnaui*	Southern herring	108	186
Dasyatidae	*Dasyatis brevicaudata*	Estuary stingray	1	4
Diodontidae	*Dicotylichthys punctulatus*	Three-bar porcupinefish	0	2
Gerreidae	*Gerres subfasciatus*	Silver biddy	36	11
Girellidae	*Girella tricuspidata*	Luderick	379	265
Hemiramphidae	*Hyporhamphus regularis*	River garfish	17	7
Monacanthidae	*Meuschenia freycineti*	Six-spine leatherjacket	0	1
Monacanthidae	*Meuschenia trachylepis*	Yellowfin leatherjacket	2	1
Mugilidae	*Mugil cephalus*	Sea mullet	365	586
Mugilidae	*Mugil georgii*	Fantail mullet	67	69
Mugilidae	*Liza argentea*	Flat-tail mullet	444	524
Mugilidae	*Myxus elongatus*	Sand mullet	27	119
Percichthyidae	*Macquaria colonorum*	Estuary perch	1	0
Platycephalidae	*Platycephalus fuscus*	Dusky flathead	3	8
Plotosidae	*Cnidoglanis macrocephalus*	Estuary catfish	5	2
Pomatomidae	*Pomatomus saltatrix*	Tailor	9	20
Portunidae	*Portunus pelagicus*	Blue-swimmer crab	3	5
Sillaginidae	*Sillago ciliata*	Sand whiting	94	96
Sillaginidae	*Sillago maculata*	Trumpeter whiting	1	2
Soleidae	*Synaptura nigra*	Black sole	1	2
Sparidae	*Acanthopagrus australis*	Yellowfin bream	110	63
Sparidae	*Rhabdosargus sarba*	Tarwhine	3	2
Sphyraenidae	*Sphyraena obtusata*	Striped sea-pike	0	1
Terapontidae	*Pelates quadrilineatus*	Trumpeter	1	0
Tetraodontidae	*Tetractenos hamiltoni*	Common toadfish	0	7
		Total number	1683	1984

the species composition of catches was the same in both net types and mesh sizes. Analyses were also done on raw data and standardised catch data (proportion of total catch) to test whether the structure of assemblages differed according to the relative abundance of catches and to test whether both nets sampled assemblages in similar proportions respectively. Non-metric multidimensional scaling (nMDS) was used as a graphical representation of sample relatedness from two-dimensional ordination plots. Two-way crossed analysis of similarity (ANOSIM) was used to test the *a priori* hypothesis that the composition and structure of catches differed between net type and mesh size. Where differences were significant at a global level, pairwise comparisons were subsequently done. Similarity percentages (SIMPER) were then used to determine those species primarily responsible for the observed dissimilarities between groups.

The coefficient of variation (CV) was used to compare estimates of relative variation in mean CPUE estimates for each net type. Dividing the total catch by the total number of worker hours taken to process each net type compared relative operational cost efficiency. Power analyses (S-Plus statistical software package) were used to determine the number of replicates required to detect differences in mean abundance (CPUE), which may be influenced by spatial variation between sites within an estuary, or temporal differences between sampling times. For each net type, the mean numbers of fish (and standard deviations) were determined from the 14 replicate samples collected over the study period. These variables were used to estimate sample sizes needed to detect differences (effect size) of between 10 and 50% in mean CPUE, for each type of net. Statistical powers of 0.7, 0.8 and 0.9 were modelled for each effect size.

Results

Composition of samples

A total of 3667 individuals representing 27 species (26 finfish and 1 portunid crab) were caught during the experiment (Table 1). Catches were significantly correlated (Spearman rank correlation test, $r^2 = 0.8$, $P < 0.01$) between the gill net (1683 individuals of 23 species weighing a total of 616.9 kg) and trammel net (1984 individuals of 24 species weighing 799.2 kg). Twenty species of fish were caught in both gear types. The remaining seven species, which were caught in only either net type, occurred in very small numbers (<7 individuals of each species) and comprised species of little commercial or recreational significance, such as *Strongylura leiura*, *Sphyraena obtusata* and *Meuschenia freycineti* (Table 1). Eight species accounted for 90% of all individuals caught, with the three most abundant species, *Liza argentea*, *Mugil cephalus* and *Girella tricuspidata* comprising 70% of the total catch.

Effect of net type and mesh size on the numbers and species of fish caught

Although the trammel net retained a greater overall number of fish than the gill net (Table 1), ANOVA failed to detect

Table 2. Analyses testing for significant differences in mean numbers of individuals, species and abundant economically important fish species (*Girella tricuspidata*, *Mugil cephalus* and *Liza argentea*), caught between net type (multi-mesh gill and trammel nets) and mesh size in Tuggerah Lake (NSW)

Treatment	d.f.	Total individuals *F*-value	Total species *F*-value	*M. cephalus* *F*-value	*G. tricuspidata* *F*-value	*L. argentea*† *F*-value
Net type (N)	1	3.45	0.89	3.23	0.31	0.58
Mesh size (M)	4	18.14***	26.67***	0.55	6.79***	49.77***
N × M	4	1.34	1.31	0.35	0.64	0.02

†Data remained heterogeneous after log (x + 1) transformation and α was set at 0.01. ***$P < 0.001$.

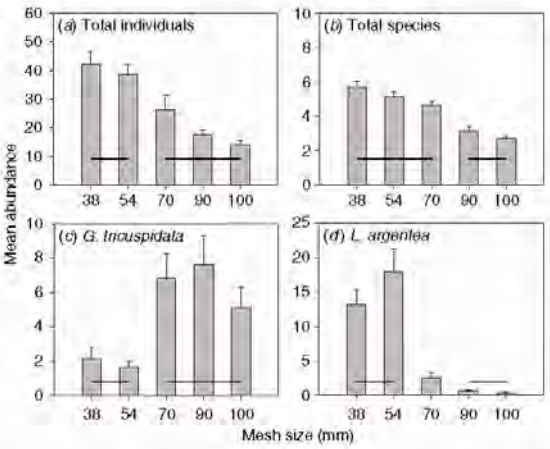

Fig. 1. Mean number of (*a*) individuals, (*b*) species, (*c*) *Girella tricuspidata* and (*d*) *Liza argentea* caught in 38, 54, 70, 90 and 100 mm mesh panels of multi-mesh gill and trammel nets (pooled) in Tuggerah Lake (NSW). Standard error shown. Horizontal lines join means with no significant difference (Student–Newman–Keuls tests).

significant differences between net types for the mean number of total individuals caught (Table 2). Catches did differ according to mesh size (Table 2), with the 38 and 54 mm mesh panels retaining significantly more fish than the 70, 90 and 100 mm mesh panels (SNK tests, Fig. 1*a*). A similar result was observed for the mean number of species caught, with significant *F*-ratios detected for mesh size, but not net type (Table 2). The 38, 54 and 70 mm mesh panels captured significantly more species than the panels made from either 90 or 100 mm meshes (SNK tests, Fig. 1*b*). Spearman rank correlations between pairs of statistically different means (SNK tests, Fig 1*a*, *b*) for the total number of individuals and species were not significant ($P > 0.05$), indicating that the different mesh sizes sampled fish in different proportions.

There was considerable variation of the effects of mesh size and net type on the catches of individual economically important species (Table 2). For example, ANOVA failed to detect a significant difference in the numbers of *M. cephalus* caught between the trammel and gill nets, nor between the five different mesh sizes of each net type. This result for *M. cephalus* probably was confounded by a large number of individuals entangled in the smaller mesh (38 and 54 mm) panels of each type of net (Fig. 3). Results of ANOVA for *G. tricuspidata* and *L. argentea* revealed a significant effect due to mesh size (Table 2), but contrasting patterns of catch distributions. For example, the mean numbers of *G. tricuspidata* were significantly greater in the larger-meshed (70, 90 and 100 mm) than the small-meshed (38 and 54 mm) panels, while the opposite was true for *L. argentea* (SNK tests, Figs 1*c*, *d*).

Effect of net type and mesh size on the composition and structure of catches

Non-metric multidimensional scaling ordination plots and ANOSIM examined differences in the composition and structure of catches between net type and mesh size. ANOSIM detected significant differences between mesh size ($P < 0.01$), but not between net type ($P > 0.95$) for the raw, standardised and presence/absence data. This was evident in the ordinations as samples from gill and trammel nets did not separate into distinct assemblages (Fig. 2*a*). In contrast, samples from the smaller-mesh panels (38 and 54 mm) tended to separate from the larger-meshed panels (70, 90 and 100 mm) for both net types (Fig. 2*b*). This was supported by the pairwise comparisons, which showed that assemblages caught in the 38 and 54 mm mesh panels differed significantly from those caught in the 70, 90 and 100 mm mesh panels for the standardised data. Pairwise comparisons for the raw and presence/absence data were the same except in three cases: (*i*) the 54 mm mesh did not differ from 70 mm mesh, (*ii*) the 70 mm mesh differed significantly from the 90 mm mesh, and (*iii*) the 70 mm mesh differed significantly from the 100 mm mesh (raw data only).

Similarity percentages analyses identified those species responsible for dissimilarities between pairs of different-sized mesh, which related to variability in catch rates and the presence or absence of each particular fish species. For the raw and standardised data, much of the dissimilarity between fish assemblages was attributed to common species, including *L. argentea*, *Herklotsichthys castelnaui*, *Sillago ciliata* and *Myxus elongatus*, which characterised the smaller mesh sizes (38 and 54 mm). In contrast, *M. cephalus*,

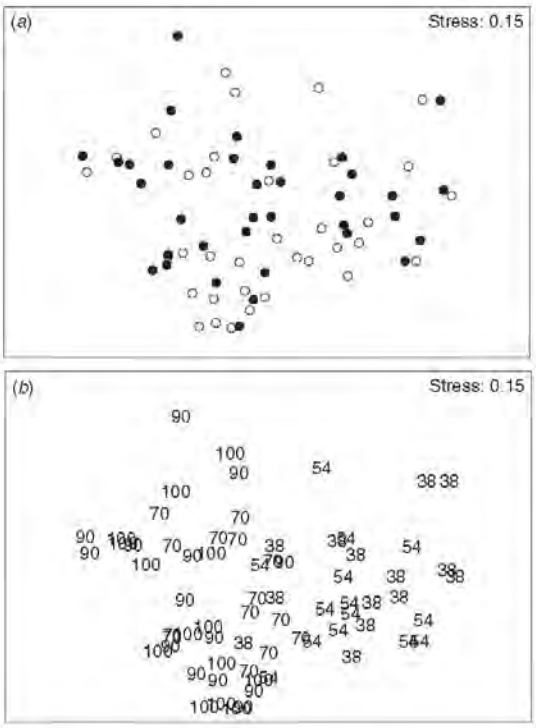

Fig. 2. Two-dimensional nMDS ordination plots of standardised catch data showing relationships between fish communities for (*a*) gear type: multi-mesh gill net (•) and multi-mesh trammel net (○) and (*b*) mesh size: 38, 54, 70, 90 and 100 mm, in Tuggerah Lake (NSW). Each individual point represents two replicate samples pooled for each night (see text for further details).

G. tricuspidata and *Acanthopagrus australis* characterised catches from the larger-meshed panels (70, 90 and 100 mm mesh). Differences in fish assemblages between the 70 and 100 mm mesh panels were primarily due to species such as *M. cephalus*, *L. argentea*, *A. australis* and *G. tricuspidata*, which were more abundant in the 70 mm mesh.

Species that occurred in relatively small numbers and were only sporadically caught made a greater contribution towards dissimilarities between mesh sizes for presence/absence data. For example, *Gerres subfasciatus*, *Hyporhamphus regularis*, *Pomatomus saltatrix*, *Portunus pelagicus* and *Platycephalus fuscus* were primarily captured in 38 and 54 mm meshes and were important discriminating species between smaller and larger mesh sizes (i.e. 38 *v.* 70, 90 and 100 mm and 54 *v.* 90 and 100 mm). Dissimilarities between medium and large mesh sizes (i.e. 70 *v.* 90 and 100 mm) were also influenced by relatively rare species including *S. ciliata*, *M. elongatus*, *P. fuscus*, *P. pelagicus*, *P. saltatrix* and *A. australis*, which, with the exception of one species (*P. pelagicus*), were generally more abundant in the 70 mm mesh panels.

Effect of net type and mesh size on the sizes of fish caught

Differences in length–frequency distributions between treatments were tested for the three most abundant species: *M. cephalus*, *G. tricuspidata* and *L. argentea*. Length–frequency distributions differed significantly between net types for catches of *M. cephalus* (KS test, $P < 0.01$). The main differences were largely related to a greater proportion of individuals <25 cm FL being caught in the 38 and 54 mm mesh panels of the gill net (Fig. 3). The mean sizes of *M. cephalus* increased with mesh size for individuals that were captured by gilling in both the gill and trammel nets. For example, irrespective of type of net, the mean size of gilled *M. cephalus* was ~26 cm FL in the 38 mm mesh and 36 cm FL in the 90 mm mesh. In contrast, the mean sizes of entangled fish were similar for all mesh sizes and for both types of net, and were primarily comprised of individuals >30 cm FL (Fig. 3).

Liza argentea were primarily caught in the 38 and 54 mm mesh panels, with fewer than ten individuals caught in each of the 90 and 100 mm mesh panels of either net type. The length–frequency distributions of *L. argentea* differed significantly between the gill and trammel nets (KS test, $P < 0.01$), with greater numbers of smaller individuals caught in the trammel net, particularly for the 54 mm mesh size (Fig. 4). The mean length of gilled *L. argentea* retained in the two smallest meshes ranged from 24 to 26 cm FL for both gears, with entangled individuals slightly larger and ranging from 26 to 30 cm FL (Fig. 4).

Girella tricuspidata were primarily caught in the 70, 90 and 100 mm mesh panels and the length–frequency distributions of *G. tricuspidata* were not significantly different between net types (KS test, $P > 0.70$). The mean size of gilled individuals increased from 25 cm FL in the 70 mm mesh panel to 32 cm FL in the 100 mm mesh panel of the gill net, and from 27 to 30 cm FL in the 70 and 100 mm meshes of the trammel net. Most entangled *G. tricuspidata* were >30 cm FL in both the gill and trammel nets (Fig. 5).

Effect of net type on precision of CPUE and required sampling effort

While a slightly greater number of individuals were caught by the trammel net, estimates of the mean CPUE for the abundance of all species were less precise for the trammel net compared with the gill net (as indicated by CV, Table 3). Estimates and the level of precision of mean CPUE for the seven most abundant species were variable between net types, with CV values ranging from 18 to 56 (Table 3). For species such as *L. argentea*, *H. castelnaui* and *M. cephalus*, the gill net provided more precise estimates of CV, whereas for *M. elongatus* and *S. ciliata*, the trammel net proved to be more precise. For some species (*G. tricuspidata* and *A. australis*), the precision of mean CPUE was very similar between the two types of gears (Table 3). In terms of the time taken to process

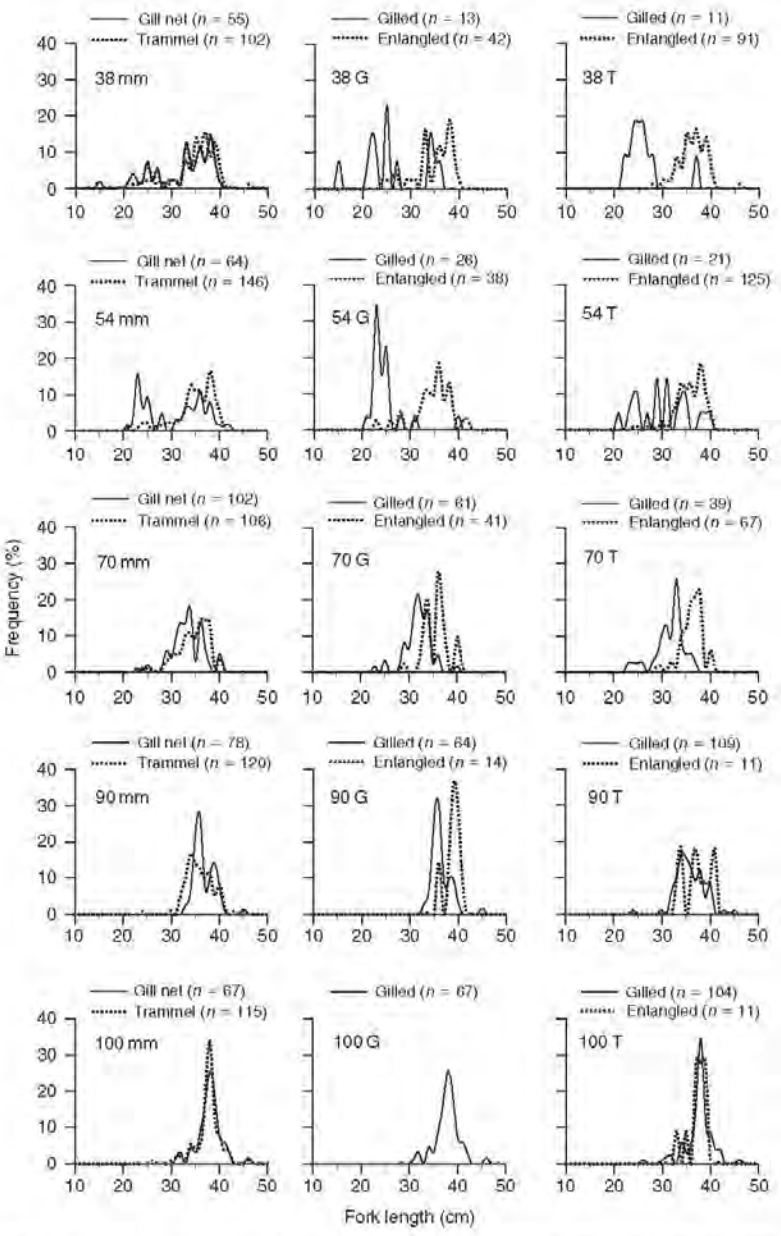

Fig. 3. Length–frequency distributions of *Mugil cephalus* caught in multi-mesh gill (G) and trammel (T) nets in Tuggerah Lake (NSW) and the frequency of gilled and entangled individuals caught in each mesh size.

the nets (i.e. removing gilled and entangled fish, and measuring and weighing individuals), the gill net proved more efficient (sorting time for the trammel nets was 35% greater, Table 3).

Table 4 shows the minimal number of replicates required to detect differences of between 10 and 50% in mean CPUE for each net type and three different levels of power (0.7, 0.8, 0.9). The sample sizes required to detect changes in mean

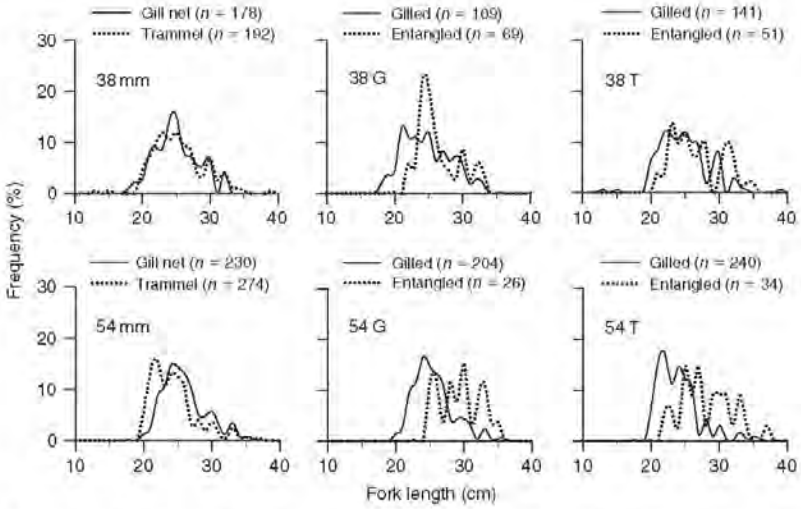

Fig. 4. Length–frequency distributions of *Liza argentea* caught in multi-mesh gill (G) and trammel (T) nets in Tuggerah Lake (NSW) and the frequency of gilled and entangled individuals caught in each mesh size.

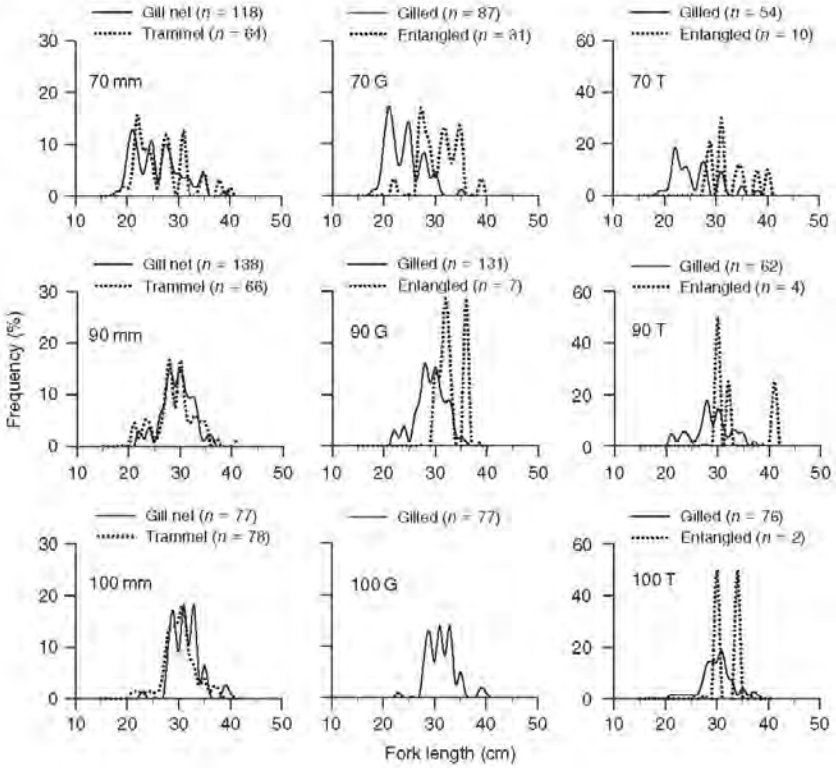

Fig. 5. Length–frequency distributions of *Girella tricuspidata* caught in multi-mesh gill (G) and trammel (T) nets in Tuggerah Lakes (NSW) and the frequency of gilled and entangled individuals caught in each mesh size.

Table 3. Summary of catch statistics for multi-mesh gill and trammel net sampling in Tuggerah Lake (NSW)

Catch per unit effort (CPUE) is number of fish caught per net per night ($n = 14$) and s.d. is the standard deviation. The coefficient of variation (CV = 100 × s.d./mean) represents the relative variation for each net type. Total catch (TC) is the total number of individuals caught during the study. Total sorting time (TST) is the time taken to remove fish from nets, and to measure and weigh individuals

Net type	Mean CPUE	s.d.	CV	Total catch (TC)	Total sorting time (TST)	TC/TST
Gill net						
All species	120.2	31.5	26.2	1683	339 min	5.0
Liza argentea	31.7	6.9	21.9	444		
Herklotsichthys castelnaui	7.7	3.0	38.6	108		
Girella tricuspidata	27.4	7.4	26.9	379		
Myxus elongatus	1.9	1.0	52.2	27		
Mugil cephalus	26.1	5.3	20.4	365		
Sillago ciliata	9.7	3.7	38.0	94		
Acanthopagrus australis	8.9	1.6	17.6	110		
Trammel net						
All species	141.8	43.0	30.3	1984	457 min	4.3
Liza argentea	48.4	17.2	35.5	524		
Herklotsichthys castelnaui	13.3	7.4	55.8	186		
Girella tricuspidata	19.0	4.9	26.0	265		
Myxus elongatus	8.5	3.7	43.9	119		
Mugil cephalus	42.7	9.4	21.9	586		
Sillago ciliata	10.1	3.2	31.5	96		
Acanthopagrus australis	4.5	0.8	17.7	63		

Table 4. Sample sizes (n) required using multi-mesh gill and trammel nets to detect differences (minimal detectable change) of between 10 and 50% in mean CPUE at three levels of power

Net type	Power	Sample size (n)				
		Minimum detectable change (%)				
		10%	20%	30%	40%	50%
Gill net	0.7	34	9	4	3	2
	0.8	63	16	7	4	3
	0.9	118	30	14	8	5
Trammel net	0.7	45	12	5	3	2
	0.8	83	21	10	6	4
	0.9	158	40	18	10	7

CPUE were different for each net type, and large for gill and trammel nets in the 10 to 20% range (Table 4). For effect sizes greater than 10%, there was a large decrease in the number of replicates needed, with an approximate 50% decrease in the sample size needed with each 10% increase in effect size (Table 4). In all scenarios, trammel nets require more replicate sets than gill nets to detect a similar effect size.

Discussion

This study has provided valuable insights into the potential use of multi-mesh gill and trammel nets as sampling gears for undertaking larger-scale and longer-term fishery-independent research surveys of estuarine ichthyofauna in south-eastern Australia. Most of the predominant species captured (*Mugil cephalus*, *Girella tricuspidata*, *Acanthopagrus australis*, *Liza argentea*) in our research nets are the same species as those captured and retained in the NSW estuarine commercial gill-net fishery, which is restricted to using nets with a mesh size >80 mm (Gray 2002; Gray *et al.* 2004). Several species, including *A. australis* and *G. tricuspidata*, are also important to recreational fishers in this region. Both multi-mesh nets captured a range of sizes of all these species; thus, sampling with these nets may be suitable for use to assess the status of populations of these important species. Gill and trammel nets can be species and size selective (Hamley 1975). We therefore acknowledge that before implementing any large-scale and long-term sampling programme, further pilot work is required to assess the potential limitations of these types of sampling gears on providing robust estimates of the size and age structure, and relative abundance, of populations of different species across different types of habitats and estuaries, and among seasons.

The univariate and multivariate analyses showed that the composition of catches, the relative proportion that each species was captured and estimates of the relative abundances of the common species were similar in both net types. Acosta (1997) also reported that, except for some pelagic species, basic patterns in the structure of fish catches from coral reefs and mangroves in Puerto Rico were similar between samples collected in gill and trammel nets. In contrast, Thomas *et al.* (2003) found that trammel nets were more effective in catching penaeid prawns than were gill nets. The great similarity

between catches of fishes from both these net types reflects how both gears are constructed and fish in a very similar manner. It is unknown, however, what the multi-mesh gill and trammel nets did not sample in our study. Acosta (1997) found that both these net types did not sample specific species that were present in an area as evidenced from visual surveys.

There were significant differences between samples caught in the different mesh sizes of the two gear types. For the multivariate analyses, the differences detected depended on the data transformation and the hypothesis being tested. For example, the species most responsible for determining differences between mesh sizes varied for the raw, standardised and presence/absence data. Nevertheless, the smaller (38 and 54 mm) mesh sizes we used caught several important species, notably *Herklotsichthys castelnaui*, *Sillago ciliata* and *Gerres subfasciatus*, which are generally caught in small numbers in the commercial gill-net fishery. *Sillago ciliata* and *G. subfasciatus* are, however, important in the estuarine beach-seine fishery in NSW (Gray and Kennelly 2003), and *H. castelnaui* is an important baitfish species. Further, the 38 and 54 mm mesh panels captured juveniles (<current minimum legal length, MLL) of some important species, notably *M. cephalus* (MLL = 27 cm TL), *G. tricuspidata* (MLL = 25 cm TL), and *L. argentea* (no MLL). Use of these mesh-sizes may provide indices of relative abundance of fish before recruitment into recreational and commercial fisheries.

Fishing with experimental multi-mesh gill or trammel nets at different times of the year should yield important information on the abundances of juvenile, sub-adult and adult fishes. Such sampling may be influenced by seasonal and ontogenetic changes in the behavioural patterns of individual species. Due to differences in the sizes of fish caught among mesh sizes, future use of all mesh sizes tested here is warranted because other species and sizes of fish may be captured at different times of the year. Multi-mesh gill nets have previously been used to a limited extent to sample estuarine fish populations in south-eastern Australia (Middleton *et al.* 1984; Pollard 1994), but have been useful sampling gears for examining the demographic patterns of several estuarine fish populations in other parts of Australia, including temperate and tropical waters of Western Australia and Queensland (Potter *et al.* 1983; Blaber *et al.* 1985, 1989; Loneragan *et al.* 1987, 1989; Potter and Hyndes 1994; Ley *et al.* 2002) and in freshwater and marine environments in other parts of the world (Marias 1981; Degerman *et al.* 1988; Mattson and Mutale 1992; Acosta 1997; Appelberg 2000). The most appropriate times (e.g. months/seasons) to sample the different life-history stages of fish with gill or trammel nets in south-eastern Australia still need to be determined.

Based on previous studies, we hypothesised that the trammel net would capture more fish through entanglement than the gill net (von Brandt 1984; Losanes *et al.* 1992; Fabi *et al.* 2002), resulting in a greater diversity and abundance of fish, and a wider size range of individuals. This, however, did not prove true because the total number of individuals caught was comparable between the two nets, with a relatively similar percentage of total fish entangled in the gill net and trammel net (22% and 29% respectively). The amount of entanglement of fish in each net type appeared to be mesh-size and species specific. For example, a large proportion of *M. cephalus* were entangled in the smaller mesh-sizes (38 to 70 mm, particularly in the trammel net), but not in the larger mesh sizes (90 and 100 mm). In contrast to some other studies (e.g. Hamley 1975; Fabi *et al.* 2002), entangled fish made a difference to the length distribution of both gill and trammel net catches by contributing significantly to the number of large fish caught. Hence, the mechanism by which fish are caught (gilled or entangled) is an important factor affecting selectivity in these types of nets (see also Hamley 1975; Madsen *et al.* 1999; Fabi *et al.* 2002).

The gill net provided more precise estimates of mean CPUE than did the trammel net for all species combined and for most individual species. In general, this means that compared with trammel nets, fewer replicate gill nets would be required to obtain similarly precise estimates of mean CPUE. Previous studies in other regions have also reported smaller coefficients of variation (CV) for gill nets compared with trammel nets (Bagenal 1979; Acosta 1997). Moreover, the CV values for multi-mesh gill nets reported in the present study are well below the value of 77% calculated by Craig *et al.* (1986), which was averaged across several studies in freshwater lakes in Canada. Acosta (1997) reported small CV's for gill nets used to sample fish assemblages near coral reefs and mangroves in south-west Puerto Rico. The present research was also done in coastal habitats, which maintain essentially saline conditions. Hence, the above differences may represent an increased level of variability in the distribution and abundance of fishes in freshwater environments.

The precision of estimates of CPUE for individual species varied; each type of net gave less precise estimates for some of the species caught. This is likely to be the case for most sampling gears aimed at representatively sampling multi-species assemblages. Decisions concerning the type and allocation of sampling effort required to sample multi-species assemblages require identifying needs and prioritisation of assessing different species (Leaman 1981). This may result in the use of a sampling gear and a corresponding level of replication that is sub-optimal for certain species (Kennelly *et al.* 1993). In such cases, compromises between the overall performance of gear, cost and effort, and logistic constraints, are necessary, together with a focus on the key species of interest.

The *a priori* analyses of power and sample size have provided an indication of the level of replication that is required for larger-scale surveys of estuarine fishes in south-eastern Australia. From the present study, it was determined that nine (replicate) gill net samples would be required to detect a 20% difference in abundance between two populations (e.g. between sites or sampling occasions) and four gill net samples

to detect a 30% effect size at a power of 0.7. Determination of acceptable power requires consideration of Type II error rates (i.e. the probability of not detecting a difference in mean CPUE when one does exist) (Peterman 1990; Lester *et al.* 1996). The large-scale fishery-independent sampling programme of freshwater fishes in Sweden uses precision sufficient to make it highly probable that a 50% difference would be detected between sampling occasions (Degerman *et al.* 1988). These authors also reported that the number of gill nets required to achieve a certain precision of CPUE for a species (perch, *Perca fluviatilis*) was significantly correlated with the area of the lake sampled. Further, Kurkilahti and Rask (1996) found that the catching efficiency of two net types was different in two lakes. Our results may, therefore, only be applicable to estuaries that are similar in size and type to Tuggerah Lake. Further experiments using experimental multi-mesh gill nets to determine optimal levels of replication and precision for estimates of CPUE may need to be done in several different types and sizes of estuaries throughout south-eastern Australia.

Based on (*i*) precision of CPUE estimates, (*ii*) required sampling effort, and (*iii*) ease of sampling, the multi-mesh gill net was a better sampling tool than the multi-mesh trammel net for estimating sizes of populations of estuarine fish. While the two types of net caught similar species and their sizes in relatively similar occurrences and abundances, the gill net was more efficient at retaining smaller individuals of some fish species (e.g. *M. cephalus* and *L. argentea*). The degree of entangling of some fishes was greater in the trammel net, making them harder and more time-consuming to remove. This caused more stress and damage to several species and therefore, the probability of greater post-release mortalities. The gill net was also easier to set and retrieve, handle, and repair than the trammel net. Given the three general advantages above, further studies to elucidate the most appropriate multi-mesh gill net configurations (e.g. range and number of mesh sizes, length of panels) and fishing practices (e.g. setting and soak-times) (e.g. Hamley 1975; Acosta 1994; Olin *et al.* 2004) for sampling estuarine ichthyofauna and on estimating the relative abundance, size and age distributions of different species are warranted. Further work also needs to compare the relative efficiency of multi-mesh gill nets at sampling assemblages of estuarine fish compared with other sampling gears such as trawls, seines and fyke nets. Ideally, these types of studies should to be stratified across different times of the year, habitats and estuaries.

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Sampling estuarine fish using multi-mesh gill nets: Effects of panel length and soak and setting times

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Abstract

Experiments on the effects of soak (1, 3 and 6 h) and setting times (18:00, 22:00 and 3:00 h), and panel length (20, 50, 120 m) on catches of fish in multi-mesh (38, 54 and 89 mm) gill nets were done in a south-east Australian estuary to develop an optimal, representative and standardized sampling methodology for future fishery-independent surveys of estuarine ichthyofauna. Univariate analysis revealed that while the longest soak times and panels often caught significantly more numbers and species of fish, there were no differences between short and intermediate soak times. Further, where differences between soak time and panel length treatments were detected, fish were often being caught in the same relative proportions. Standardized catch-per-unit-of-effort (CPUE) (numbers of fish caught 20 m^{-1} of net h^{-1}) decreased significantly with increasing soak time, but there were no differences in CPUE between different panel lengths for the total numbers of fish and key species caught. Multivariate analyses failed to detect any differences in fish assemblages between soak times and panel lengths for 38 and 89 mm mesh. However, inconsistent and species-specific differences in the abundances of common species accounted for differences in assemblage structure for the 54 mm mesh. Similarly, while the size ranges of most species of economic importance were comparable between different panel lengths and soak times, there were inconsistent differences in the proportions of fish captured across size classes for some species. Setting time had no effect on the mean numbers of fish or species caught, structure of assemblages or size-ranges of most species investigated. Based on these results, the use of 20-m panels soaked for 1 h at any time during the night was considered optimal for future surveys. The benefits of this uniform sampling methodology are discussed in terms of increased replication, reduced costs and the potential for lower fish mortality.

Keywords: Estuarine ichthyofauna; Fishery-independent survey; Multi-mesh gill net; Net length; Soak time; Standardized sampling

1. Introduction

Ecological- and fisheries-related assessments of the relative abundances and population compositions of fish and invertebrates often rely on catch-per-unit-of-effort (CPUE) data (Gunderson, 1993; Helser and

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Hayes, 1995). To provide accurate, reliable and cost-effective estimates of CPUE, research surveys should employ unbiased, standardized and optimal sampling methodologies that are appropriately stratified and replicated in space and time (Andrew and Mapstone, 1987; Morrisey et al., 1992a,b). This typically requires the use of initial pilot studies incorporating manipulative experimental approaches to test specific hypotheses concerning the design and deployment of sampling gears (e.g., Miller, 1983; Kennelly, 1989; Montgomery, 2000).

Gill nets have been extensively used to sample marine and freshwater fish worldwide (Craig et al., 1986; Loneragan et al., 1987; Degerman et al., 1988; Mattson and Mutale, 1992; Acosta, 1997). It is well documented that gill nets can be size- and species-selective (Hamley, 1975), and many studies have focussed on quantifying the extent to which this is influenced by mesh size (Jensen, 1990). Fewer studies have assessed the effects of other technical factors, such as soak (duration a gear is fished) and setting time (time of day/night gear is deployed) and net length on estimates of gill net CPUE. Such factors are important because they have the potential to directly influence the catchability or efficiency of sampling gears. Specifically, as fish accumulate in a gill net, catching efficiency declines until eventually the net becomes 'saturated' and no additional individuals are retained (Hamley, 1975; Hubert, 1983). Gill net saturation depends on several technical and biological factors, including the amount of available, unoccupied meshes and the abundance, morphology and behaviour of fish (Kennedy, 1951; Beverton and Holt, 1954; Minns and Hurley, 1988). Spatial, temporal and species-specific variations in the influence of these factors mean that the accumulation of catches in gillnets may not be uniformly correlated to net length or soak time (Hamley 1975, Hubert, 1983).

Without knowledge about the effects of net length and soak and setting time on CPUE, research surveys employing gill nets to sample fish may be sub-optimal and at worst, non-representative; leading to unreliable estimates of relative abundances. Standardized methods of sampling fishes with multi-mesh gill nets (i.e., nets containing continuous or discrete panels of different-sized meshes) have been utilized in surveys of Nordic lakes for over 30 years (Appleberg et al., 1995) and adapted elsewhere (Mattson and Mutale, 1992). Despite these types of gill nets being widely used to sample fish communities in different regions and habitats throughout Australia (e.g., Potter et al., 1983; Pollard, 1994; Blaber et al., 1995), there have been no investigations of appropriate gear configurations and sampling practices. Indeed, studies of gill net saturation effects have generally been limited to enclosed freshwater lakes in North America (Kennedy, 1951; Minns and Hurley, 1988; Hansen et al., 1998) and experimental tanks elsewhere (Losanes et al., 1992), which are typically characterized by a lower diversity and abundance of fish and a less heterogenous substratum than open marine systems, such as coastal estuaries (sensu Acosta, 1994).

The present study is part of a research program to develop a complementary suite of fishery-independent sampling tools for surveying the widest possible size range and diversity of key species of fish, throughout estuaries in New South Wales (NSW), Australia. Gray et al. (2005) evaluated the utility and efficiency of multi-mesh gill and trammel nets for sampling estuarine fishes in this region and found that although both gears caught similar species of fish in comparable abundances, multi-mesh gill nets were considered more appropriate due to (i) greater precision of CPUE estimates, (ii) less required sampling effort and (iii) ease of sampling. In this paper, we test the hypotheses that catches and catch rates of estuarine fish in multi-mesh gill nets will be affected by different soak and setting times and panel lengths. We then determine the most optimal (in terms of the quantity and structure of the catch obtained and efficiency of sampling) and representative gear configuration and fishing practices for use in future surveys.

2. Materials and methods

2.1. Study sites

Experiments were done in Tuggerah Lake (151°30′E, 33°21′S) in central NSW, Australia. Tuggerah Lake is a large (70 km^2 surface area) and shallow barrier estuary (average depth of 1.9 m) supporting valuable commercial and recreational fisheries that mostly target *Mugil cephalus*, *Girella tricuspidata*, *Platycephalus fuscus*, *Acanthopagrus australis* and *Sillago ciliata* (Henry and Lyle, 2003; NSW Fisheries, 2003).

2.2. General gill net construction

All multi-mesh gill nets were made from polyamide monofilament netting. Each net comprised three separate panels made from mesh with nominal stretched openings of 38, 54 and 89 mm. These mesh sizes were chosen because Gray et al. (2005) demonstrated that

they were effective at retaining a range of species and sizes important to estuarine fisheries. Twine diameter was 0.15 mm for the 38 and 54 mm mesh and 0.2 mm for the 89 mm mesh. All panels had a stretched depth of 1.5 m and were attached to head-ropes (patent floatline No. 22–Oy Lindeman Ab Finland) and foot-ropes (patent leadline No. 28–Oy Lindeman Ab Finland) at hanging ratios of 0.5 (two lengths of stretched mesh to one length of head-rope). To restrict the potential effect of the smaller-mesh panels 'leading' larger fish to adjacent larger-meshed panels, the head- and foot-rope of each adjacent panel were separated by 5 m lengths of 4 mm diameter twisted polyethylene rope.

2.3. Experiment 1 — effects of soak time and panel length on catches

This experiment tested the null hypothesis that catches and catch rates of estuarine fish (teleost and crab species) did not differ between 3 panel lengths (20, 50 and 120 m) and 3 soak times (1, 3 and 6 h). Three replicate gill nets were constructed. Each net comprised the above 3 panel lengths and different mesh sizes, except for 120-m panels made from 38 mm mesh, which were not used owing to the potential for large numbers of small non-commercial fish species to be caught and killed. Therefore, each replicate multi-mesh gill net comprised 8 panels with a total length of 485 m. The order of the different panels in each net was randomized daily. On each of 9 sampling occasions between 19th November and 5th December 2003, the 3 replicate gill nets were set on the bottom of Tuggerah Lake between 17:00 and 18:00 h. A randomly selected replicate gill net was retrieved at the end of each of the three treatment soak times (i.e., at 1, 3 and 6 h). A total of 9 replicates for each soak time and panel length were completed over the study period (i.e. 9 nights × 1 replicate night^{-1}). Catches of fish and crabs were sorted according to mesh size, panel length and soak time and counted. All fish were measured to the nearest 0.5 cm fork-length (FL).

Table 1
List of fish species and number of individuals caught in each mesh size (38, 50 and 89 mm) and panel length (20, 50 and 120 m) of multi-mesh gill nets in Tuggerah Lake (NSW)

Species	38 mm		54 mm			89 mm			Total
	20 m	50 m	20 m	50 m	120 m	20 m	50 m	120 m	
Mugil cephalus	50	72	99	261	521	173	368	869	2413
Liza argentea	94	435	221	385	1108	8	6	7	2264
Herklotsichthys castelnaui	346	929	10	16	45	3	1	1	1351
Girella tricuspidata	4	22	40	84	165	74	137	479	1005
Acanthopagrus australis	7	26	60	108	248	32	67	176	724
Sillago ciliata	21	86	46	113	277	3	15	30	591
Gerres subfasciatus	71	207	14	29	43	–	–	12	376
Pomatomus saltatrix	32	53	6	17	36	2	9	13	168
Portunus pelagicus	3	11	7	22	28	7	19	54	151
Platycephalus fuscus	18	20	9	17	56	1	6	16	143
Myxus elongatus	9	39	2	10	52	–	4	1	117
Mugil georgii	12	25	1	1	24	–	1	–	64
Cnidoglanis macrocephalus	3	1	4	5	6	–	14	14	47
Ambassis spp.	13	28	–	1	1	–	–	–	43
Rhabdosargus sarba	1	1	4	8	17	–	–	–	31
Strongylura leiura	2	11	–	2	5	–	–	–	20
Hyporhamphus regularis	3	10	2	1	–	–	–	2	18
Monodactylus argenteus	1	1	–	1	4	3	–	–	10
Pranesus ogilbyi	1	1	–	3	5	–	–	–	10
Meuschenia trachylepis	–	1	1	1	4	–	–	–	7
Pseudorhombus jenynsii	–	–	–	1	–	–	1	2	6
Meuschenia freycineti	–	3	–	–	–	–	–	–	4
Pseudorhombus arsius	1	–	–	–	–	1	1	1	4
Sphyraena obtusata	3	1	–	–	–	–	–	–	4
Dicotylichthys punctulatus	–	–	–	–	1	–	1	–	2
Pseudocaranx dentex	1	–	–	–	1	–	–	–	2
Monacanthus chinensis	–	1	–	–	–	–	–	–	1
Tetractenos hamiltoni	–	1	–	–	–	–	–	–	1
Total individuals	696	1985	526	1086	2650	307	650	1677	9577

2.4. Experiment 2 — effects of setting time on catches

This experiment tested the null hypothesis that catches of estuarine fish (teleost and crab species) did not differ between 3 nocturnal setting times: 18:00, 22:00 and 3:00 h. Four replicate multi-mesh gill nets each comprising three 30-m panels of the three different mesh sizes were used. On each sampling occasion, 2 replicate gill nets were deployed at two zones (separated by at least 500 m) in Tuggerah Lake at one of the three randomly selected setting times (i.e., 18:00, 22:00 or 3:00 h), and left to fish for 3 h. A total of 9 nights sampling were completed between the 24th May and 8th June 2004, providing 6 replicate gill nets for each setting time in each zone (i.e. 3 nights × 2 replicates night^{-1}).

2.5. Analyses of data

Two-factor orthogonal analyses of variance (ANOVA) was used to test for differences in the mean numbers of individuals and catch rates (standardized CPUE, defined as the number of fish captured 20 m^{-1} of net h^{-1}) between soak times and panel lengths (both fixed factors) for each mesh size (analysed separately for the total number of individuals, total number of species and the five most abundant species in each case). Prior to calculating catch rates for the total number of species, rarefaction analysis (Heck et al., 1975) was performed using EcoSim (Gotelli and Entsminger, 2004), to control for longer nets and soak times catching more species when more individuals were caught.

For experiment 2, another ANOVA model was used to test for differences in relative abundances of fish (also analysed separately for the total number of individuals, total number of species, and for the 5 most abundant species) between setting times (fixed factor) and zones (random factor). Since the interest was in the effects of setting time and whether patterns changed between zones within an estuary, data were summed across panels made from the different mesh sizes in each gill net. Where interaction terms were non-significant at $P<0.25$, they were pooled with the residuals to increase power for the main effect of setting time.

Prior to all ANOVAs, assumptions of normality and homogeneity of variances were tested using Shapiro–Wilk and Cochran's tests, respectively and where necessary, transformed to $\log(x+1)$. For data that remained heterogeneous, α was set to 0.01 to reduce the risk of Type 1 errors (Underwood, 1981). Multiple comparisons among means were done using Student–Newman–Keuls (SNK) tests.

Where significant results were detected in ANOVA and subsequent SNK tests for the mean numbers of individuals and species, Spearman's rank correlations were used to determine if replicate catches were correlated between relevant treatments. Significant correlations (at the 5% level) indicated that, despite differences in relative abundances, fish were caught in the same relative proportions by each treatment. This is appropriate for research sampling, since the aim is to obtain optimal and representative samples of fish populations and assemblages, rather than to employ gears or fishing strategies that catch the most organisms.

Differences in the structures of fish assemblages between soak times, panel lengths and setting times were investigated using non-parametric multivariate analyses

Table 2
Results of ANOVA (F ratios shown) testing for significant differences in the mean numbers of individuals, total species and 5 most abundant species of fish (*M. cepahalus*, *L. argentea*, *H. castelnaui*, *G. tricuspidata* and *A. australis*) caught between different soak times (1, 3, 6 h) and panel lengths (20, 50, 100 m) in 38, 54 and 89 mm mesh panels of multi-mesh gill nets in Tuggerah Lake (NSW)

Treatment	38 mm		54 mm		89 mm	
	df	*F* ratio	*df*	*F* ratio	*df*	*F* ratio
Total individuals						
Soak time (*S*)	2	4.64*	2	14.77***	2	3.13
Panel length (*P*)	1	26.68***	2	60.87***	2	54.00***
S×*P*	2	0.01	4	0.58	4	0.34
Total species						
Soak time (*S*)	2	10.09***	2	11.46***	2	2.55
Panel length (*P*)	1	37.33***	2	48.94***	2	28.09***
S×*P*	2	1.49	4	1.84	4	1.25
M. cephalus						
Soak time (*S*)	2	0.04	2	0.86	2	0.67
Panel length (*P*)	1	1.34	2	16.77***	2	13.07***
S×*P*	2	0.61	4	0.15	4	0.02
L. argentea						
Soak time (*S*)	2	5.60**	2	8.85***	2	0.62
Panel length (*P*)	1	7.56**	2	7.90***	2	0.01
S×*P*	2	0.12	4	0.19	4	1.07
H. castelnaui						
Soak time (*S*)	2	1.09	2	15.22***	na	na
Panel length (*P*)	1	6.61*	2	6.15**	na	na
S×*P*	2	0.05	4	2.18	na	na
G. tricuspidata						
Soak time (*S*)	2	1.29	2	5.42**	2	0.43
Panel length (*P*)	1	7.38*	2	9.82***	2	4.93**
S×*P*	2	0.86	4	0.42	4	0.49
A. australis						
Soak time (*S*)	2	2.68	2	1.93	2	0.73
Panel length (*P*)	1	8.20**	2	16.18***	2	36.94***
S×*P*	2	0.13	4	0.97	4	1.48

na=not analysed due to small numbers of individuals (<10).
* $P<0.05$.
** $P<0.01$.
*** $P<0.001$.

(Clarke 1993; Clarke and Warwick, 1994) from the PRIMER 5 package (Version 5.2.2, PRIMER-E Ltd, 2001). Data were standardized (proportions of total catch in each group) to test whether the different soak and setting times and panel lengths sampled fish assemblages in the same relative proportions. Non-metric multidimensional scaling (nMDS) was used to generate two-dimensional ordination plots. Analysis of similarities (ANOSIM) were used to test a priori hypotheses that the structure of catches differed between soak times and panel lengths (two-way, experiment 1), and between the three setting times (one-way, experiment 2). Where differences were significant at a global level ($P<0.05$), pairwise comparisons between sample groups were done. The species primarily responsible for the observed dissimilarities between sample groups were identified using similarity percentage analyses (SIMPER).

Differences in length–frequency distributions between soak and setting times and panel lengths for economically important species were examined using Kolmogorov–Smirnov (K–S, $P=0.05$) (Sokal and Rohlf, 1995). Fork lengths for each fish species were pooled across mesh sizes (to allow for small numbers or an absence of individuals in some mesh sizes) and two-sample K–S tests done between pairs of samples from each soak time (e.g., 1 vs. 3 h, 3 vs. 6 h and 6 vs. 1 h) and panel length (e.g., 20 vs. 50 m, 50 vs. 120 m and 120 vs. 20 m) in experiment 1 and setting times (e.g., 18:00 vs. 22:00, 22:00 vs. 3:00 and 3:00 vs. 18:00) in experiment 2.

3. Results

3.1. Experiment 1 — effects of soak time and panel length on catches

3.1.1. Numbers of fish and species

A total of 9577 individuals representing more than 28 species were caught during the experiment (Table 1). Five species (*M. cephalus*, *Liza argentea*, *Herklotsichthys castelnaui*, *G. tricuspidata* and *A. australis*) accounted for over 80% of the total catch. Results of ANOVA on the numbers of total individuals and species were similar, with significant differences detected between soak times and panel lengths for the 38 and 54 mm meshes (Table 2) and panel length for the 89 mm mesh. SNK tests of these means revealed that as soak time increased, more individuals and species were generally caught (Fig. 1a,b), with significant differences between

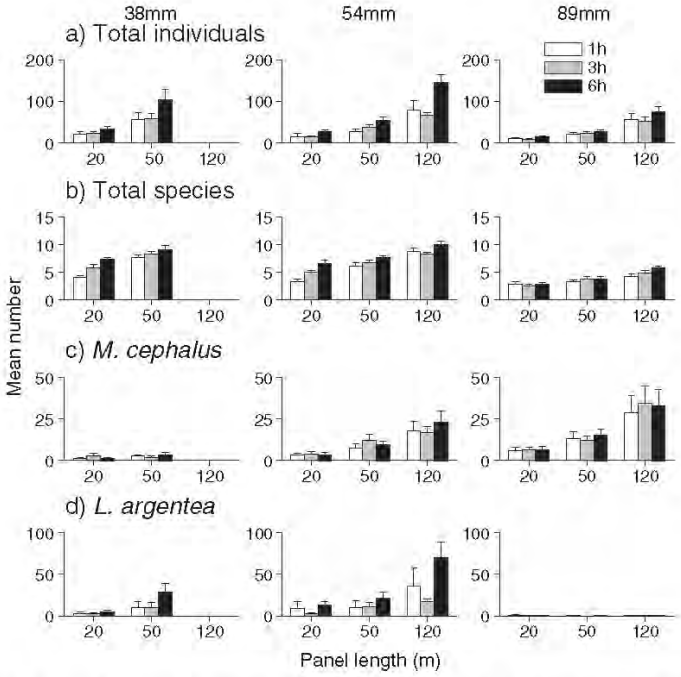

Fig. 1. Mean number of (a) total individuals, (b) total species and two of the five most abundant species, (c) *M. cephalus* and (d) *L. argentea* caught in different panel lengths (20, 50, 120 m) of multi-mesh gill nets (38, 54 and 89 mm) after soak times of 1 (white fill), 3 (grey fill) and 6 (black fill) h in Tuggerah Lake (NSW). Standard errors are shown.

soak times of 1 and 6 h or 3 and 6 h, but never between 1 and 3 h. Spearman's rank correlations between replicate samples of these statistically different means were significant (except for the 38 mm mesh set at 1 vs. 6 h), indicating that for the most part, individuals and species were caught in the same relative proportions.

The effects of soak time on the mean numbers of the 5 most abundant species were species-specific and inconsistent between mesh sizes (Table 2). For some species (e.g., *M. cephalus*, Fig. 1c), there was no discernable soak time effect for any mesh size, while for other species, there was a significant effect for either one (e.g., *G. tricuspidata*) or two (*L. argentea*, Fig. 1d) mesh sizes (Table 2). In all significant cases, however, SNK tests indicated that catches were greater after a 6-h soak time, with no differences in mean numbers between 1 and 3 h soak times. Spearman's rank correlations between replicate samples of these statistically different means were not significant ($P>0.05$).

The numbers of total individuals and species significantly increased with each panel length across all mesh sizes (SNK tests, Fig. 1). For the mean number of total individuals, Spearman's rank correlations between replicate samples of statistically different means were all significant ($P<0.05$), which means that although longer panels caught more fish, catches were proportional across all panel lengths. For the mean number of total species, significant correlations (Spearman's rank correlation test, $P<0.05$) between replicate samples occurred only for the 54 mm mesh, and only between some treatments (e.g., 20 vs. 120 m and 50 vs. 120 m), suggesting that longer panels generally caught more species, and in different relative proportions. However, this is most likely the result of longer panels catching few relatively uncommon species, as total catches of species combined over the entire study period (Table 1) were significantly correlated between different panel lengths (Spearman's rank correlation tests, $P<0.001$).

The effects of panel length on the mean numbers of abundant species were more consistent than soak time (Table 2), with significant differences detected across all species and mesh sizes except for *M. cephalus* and *L. argentea* caught by the 38 and 89 mm mesh, respectively (Table 2). For some species, catches significantly increased with longer panels and larger mesh sizes (SNK tests, 120>50>20 m) (e.g., *M. cephalus* caught in 54 and 89 mm mesh, Fig. 1c). For other species, there were no differences between 20- and 50-m panels or 50- and 120-m panels (e.g., *L. argentea* caught in 54 mm mesh, Fig. 1d) or both. SNK tests revealed that for all abundant species, 120-m panels caught significantly more individuals than 20-m panels. Spearman's rank correlations between replicate samples of those statistically different means in SNK tests were significant ($P<0.05$) for all species and mesh sizes except for *S. ciliata* (38 mm mesh) and *A. australis* (54 and 89 mm mesh). This result is similar to that observed for the numbers of total individuals and species (see above) and suggests that despite longer panels often catching significantly more fish, catches of abundant species were proportionally similar between panels of different lengths.

3.1.2. Standardized CPUE of fish and species

ANOVA on the mean CPUE for the number of total individuals revealed significant differences due to soak

Table 3
Results of ANOVA (*F*-ratios shown) testing for significant differences in the mean CPUE (standardized catch 20 m^{-1} of net h^{-1}) of individuals, total species and 5 most abundant species of fish (*M. cepahalus*, *L. argentea*, *H. castelnaui*, *G. tricuspidata* and *A. australis*) caught between different soak times (1, 3, 6 h) and panel lengths (20, 50, 100 m) in 38, 54 and 89 mm mesh panels of multi-mesh gill nets in Tuggerah Lake (NSW)

Treatment	38 mm		54 mm		89 mm	
	df	*F* ratio	*df*	*F* ratio	*df*	*F* ratio
Total individuals						
Soak time (*S*)	2	12.00***	2	20.33***	2	41.96***
Panel length (*P*)	1	0.16	2	0.04	2	0.27
S×*P*	2	0.02	4	0.61	4	0.23
Total species						
Soak time (*S*)	2	147.08***	2	101.13***	2	190.58***
Panel length (*P*)	1	41.22***	2	36.60***	2	76.59***
S×*P*	2	1.92	4	0.45	4	11.00***
M. cephalus						
Soak time (*S*)	2	5.45**	2	12.97***	2	13.56***
Panel length (*P*)	1	0.84	2	0.09	2	0.26
S×*P*	2	0.41	4	0.15	4	0.05
L. argentea						
Soak time (*S*)	2	0.60	2	0.96	2	1.70
Panel length (*P*)	1	0.72	2	0.11	2	1.61
S×*P*	2	0.04	4	0.24	4	1.29
H. castelnaui						
Soak time (*S*)	2	3.10	2	1.81	na	na
Panel length (*P*)	1	0.25	2	1.41	na	na
S×*P*	2	0.00	4	1.20	na	na
G. tricuspidata						
Soak time (*S*)	2	0.42	2	0.87	2	5.34**
Panel length (*P*)	1	1.11	2	0.07	2	0.03
S×*P*	2	0.59	4	1.37	4	0.34
A. australis						
Soak time (*S*)	2	1.70	2	5.69**	2	16.71***
Panel length (*P*)	1	2.24	2	0.08	2	0.68
S×*P*	2	0.76	4	2.21	4	1.78

na=not analysed due to small numbers of individuals (<10).
*$P<0.05$.
** $P<0.01$.
*** $P<0.001$.

time across all mesh sizes (Table 3). Mean CPUE was inversely correlated to soak time (Fig. 2a), with significant differences between 1 and 3 h and 1 and 6 h, but not between 3 and 6 h (SNK tests, $P<0.05$). For the mean number of total species, both soak time and panel length had a similar significant effect on CPUE across all mesh sizes (Table 3 and Fig. 2b); however, for the 89 mm mesh, there was also a significant interaction. There were no other main effects or interactions for panel length.

ANOVA on the mean CPUE of the 5 most abundant species (*M. cephalus*, *L. argentea*, *H. castelnaui*, *G. tricuspidata* and *A. australis*) revealed three general patterns (examples of each pattern are shown in Fig. 2 for selected species only). First, *M. cephalus* displayed a similar pattern to the numbers of total individuals, with significant differences in mean CPUE between soak times (1>3=6 h or 1>3>6 h, SNK tests) across all mesh sizes (Table 3 and Fig. 2a). Second, *L. argentea* and *H. castelnaui* showed no differences in mean CPUE between soak time and panel length across all meshes, although a trend of higher mean CPUE for a 1-h soak time was evident (Fig. 2c). Third, there were differences in mean CPUE between soak times (1>3=6 h or 1>6, 1=3, 3=6 h, SNK tests) for *G. tricuspidata* and *A. australis*, but not across all mesh sizes (Table 3 and Fig. 2d).

3.1.3. Structure of assemblages

Catch structures for each of the three mesh sizes used with different soak times and panel lengths did not separate into distinct groups (Fig. 3). ANOSIM generally supported the above lack of patterns in nMDS ordinations, with no significant differences in the structures of fish assemblages between panel lengths for any mesh size. Similarly, there were no significant differences between soak times for 38 and 89 mm meshes. The only significant difference detected by ANOSIM was between soak times for 54 mm mesh (global $R=0.08$, $P<0.05$; Fig. 3b). Pairwise comparisons showed that assemblages of fish were significantly different between 1 and 3 h and 1 and 6 h, but not between 3 and 6 h. SIMPER analysis indicated that these differences were largely driven by common species; some of which were caught in greater proportions after short soak times and others after longer soak times (SIMPER tables not shown for brevity).

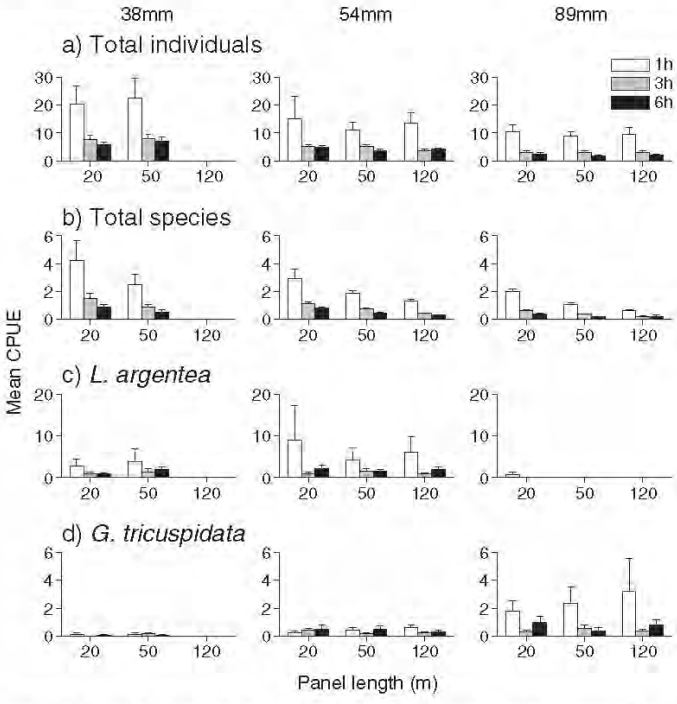

Fig. 2. Mean CPUE (standardized catch 20 m^{-1} of net h^{-1}) of (a) total individuals, (b) total species and two of the five most abundant species, (c) *L. argentea* and (d) *G. tricuspidata* caught in different panel lengths (20, 50, 120 m) of multi-mesh gill nets (38, 54 and 89 mm) after soak times of 1 (white fill), 3 (grey fill) and 6 (black fill) h in Tuggerah Lake (NSW). Standard errors are shown.

3.1.4. Lengths of economically important fish

Kolmogorov–Smirnov tests comparing the size–frequency distributions of key economically important fish (*M. cephalus*, *L. argentea*, *G. tricuspidata*, *S. ciliata* and *A. australis*) revealed significant differences between soak times and panel lengths for all species except *A. australis* (Table 4). Differences primarily occurred between soak times of 1 and 3 h and 1 and 6 h. Size ranges were similar between soak times for species such as *M. cephalus* and *G. tricuspidata*, with significant differences in K–S tests related to greater proportions of fish retained across certain size classes (Fig. 4). In contrast, significant differences between soak times for species such as *L. argentea* and *S. ciliata* were due to differences in actual size ranges or differences in the proportions of fish caught or both (Fig. 4).

For panel lengths, K–S tests detected significant differences between the 20- and 120-m panels and 50- and 120-m panels (Table 4). In general, 120-m panels caught greater proportions of larger individuals of most of the species investigated (Fig. 5). Despite this trend, fish in the largest size classes for some species (e.g., *L. argentea* and *G. tricuspidata*) were actually caught with the shortest panel length (20 m) (Fig. 5). The shorter panels (20 and 50 m) also generally retained greater proportions of fish in smaller size classes (e.g., *M. cephalus*, *G. tricuspidata*) and for

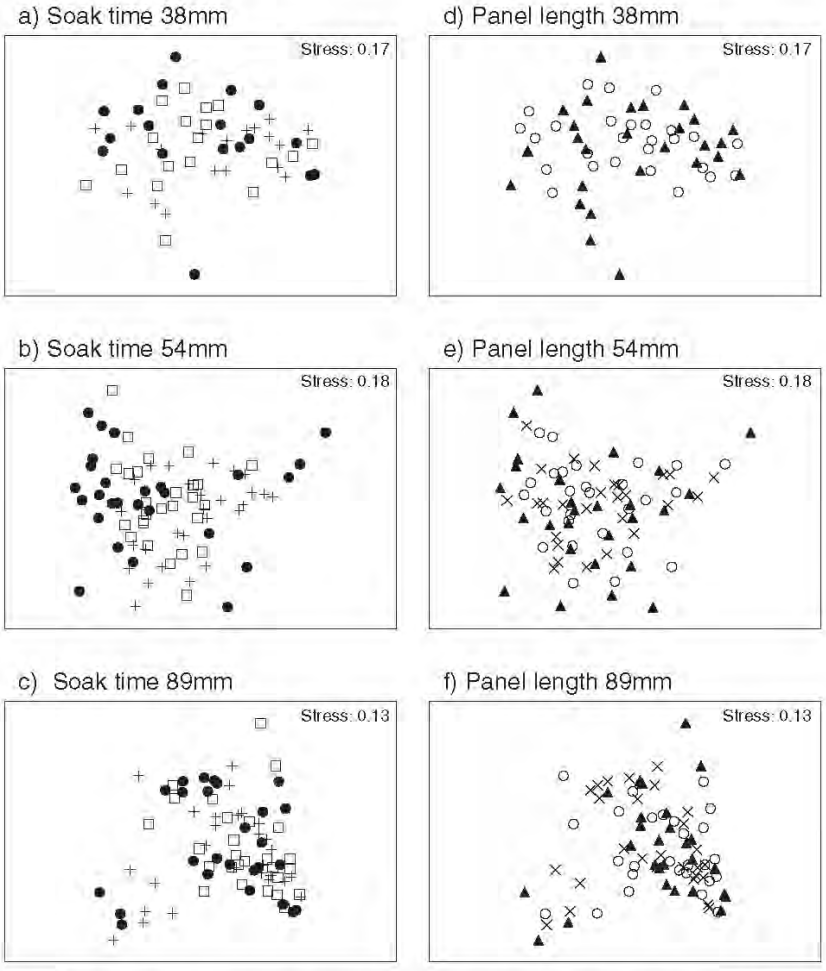

Fig. 3. Two dimensional nMDS ordinations of standardized proportional catch data showing relationships between fish communities caught in 38, 54 and 89 mm mesh panels of multi-mesh gill nets in Tuggerah Lake (NSW) for different soak times (graphs a–c: ●=1 h, □=3 h, +=6 h) and different panel lengths (graphs d–f: ▲=20 m, ○=50 m, ×=120 m). Individual data points represent replicate samples.

Table 4
Results of Kolmogorov–Smirnov (K–S) tests (*D* values shown) comparing length frequency distributions of abundant economically important species between different soak times (1, 3, 6 h) and panel lengths (20, 50, 120 m)

Species	Comparisons					
	Soak time (h)			Panel length (m)		
	1 vs. 3	1 vs. 6	3 vs. 6	20 vs. 50	20 vs. 120	50 vs. 120
M. cephalus	0.042	0.056	0.076*	0.056	0.055	0.076**
L. argentea	0.151***	0.081**	0.073	0.078	0.104**	0.12***
G. tricuspidata	0.165**	0.092	0.116	0.126	0.156*	0.075
S. ciliata	0.307***	0.254***	0.101	0.129	0.123	0.161**
A. australis	0.089	0.071	0.045	0.069	0.106	0.09

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

some species (e.g., *L. argentea* and *S. ciliata*), caught the smallest individuals (Fig. 5).

3.2. Experiment 2 — effects of setting time on catches

3.2.1. Numbers of fish and species and structure of assemblages

A total of 3350 individuals of 20 species (18 teleost and 2 crab) were caught during experiment 2 (Table 5). Fourteen species were caught across all setting times, with 5 species unique to a particular time (e.g., *Cnidoglanis macrocephalus*, *Monodactylus argenteus*, *Gerres subfasciatus* and *Pseudocaranx dentex*). However, with the exception of *C. macrocephalus*, these species were all caught in very low numbers (≤5 individuals). Seventeen species were caught during the 22:00 and 3:00 h setting times compared to 16 species during the 18:00 time. The total catches were similar among setting times (Table 5). ANOVA on mean numbers of individuals, species and the 5 most abundant fish species failed to detect any significant differences between setting times or zones (Table 6). The only significant

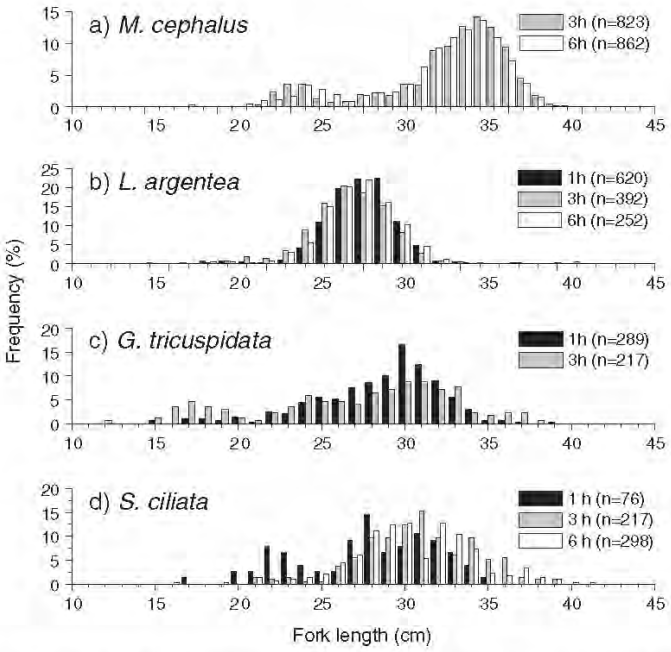

Fig. 4. Fork length–frequency distributions of (a) *M. cephalus*, (b) *L. argentea*, (c) *G. tricuspidata* and (d) *S. ciliata* caught from different soak times (1, 3, 6 h) in multi-mesh gill nets at Tuggerah Lake, NSW. Data are only presented for soak times where significant differences were detected in Kolmogorov–Smirnov tests (Table 4).

result was an interaction between these factors for *A. australis*, which indicated that the effect of soak time was dependent on each particular zone. There were also no significant differences in the structures of fish assemblages between different setting times evident in nMDS ordinations (Fig. 6) or ANOSIM (global $R=0.003$, $P>0.05$).

3.2.2. Lengths of economically important fish

Differences in the size–frequency distributions between setting times were tested for the 5 most abundant economically important species (*G. tricuspidata*, *S. ciliata*, *L. argentea*, *A. australis* and *Myxus elongatus*). Significant differences occurred between 18:00 and 22:00 (K–S=0.238, $P<0.001$) for *G. tricuspidata*, and although these setting times caught a similar size range of fish, differences appeared to be related to greater proportions of large fish (>30 cm FL) caught earlier (18:00) and greater proportions of smaller fish (20–27 cm FL) caught later (22:00) (Fig. 7). Differences in size–frequency distributions of *S. ciliata* occurred between 22:00 and 3:00 (K–S=0.164, $P<0.05$) and were opposite to those above; apparently driven by greater proportions of smaller (<33 cm FL) and larger (>33 cm FL) fish caught earlier (22:00 h) and later (3:00 h) in the night, respectively (Fig. 7).

4. Discussion

The development of sampling tools that provide representative estimates of relative abundance and demographic parameters is a logical starting point in the design of cost-effective and reliable ecological and fisheries surveys (Andrew and Mapstone, 1987). By quantifying the influences of soak and setting time and panel length on multi-mesh gill nets used to sample estuarine fishes in south eastern Australia, this study has (i) identified optimal strategies for local surveys and equally importantly (ii) illustrated the importance of this type of work as a precursor to fishery-independent studies in general.

Gill nets soaked for 1 h were more efficient at sampling fish and species than those soaked for longer periods. This soak time also provided a sufficient, representative sample of fish assemblages and was considered optimal for providing estimates of the relative abundance of key species. The choice of an appropriate soak time, however, was not as definitive when

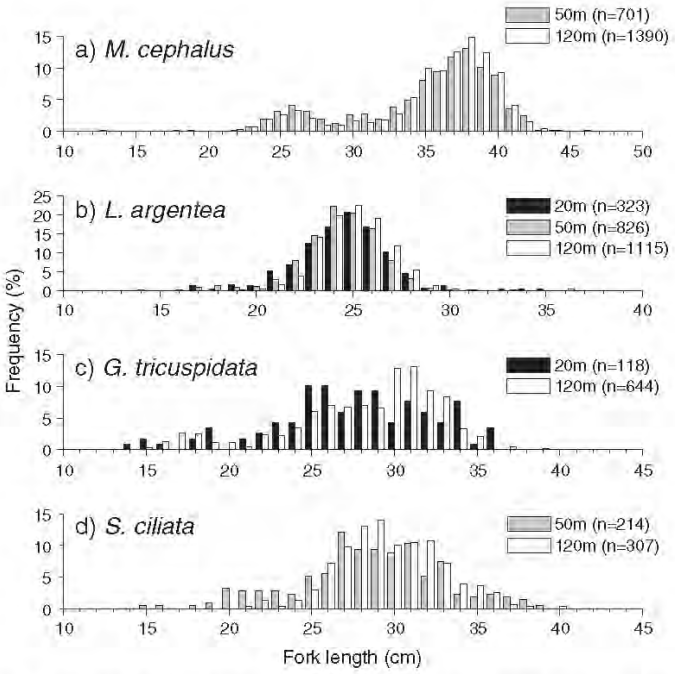

Fig. 5. Fork length–frequency distributions of (a) *M. cephalus*, (b) *L. argentea*, (c) *G. tricuspidata* and (d) *S. ciliata* caught from different panel lengths (20, 50, 120 m) in multi-mesh gill nets at Tuggerah Lake, NSW. Data only presented for panel lengths where significant differences were encountered in Kolmogorov–Smirnov tests (Table 4).

Table 5
List of fish species and number of individuals caught in multi-mesh gillnets (38, 54, and 89 mm mesh pooled) set for 3 h but at 3 different times (18:00, 22:00 and 3:00 h) in Tuggerah Lake (NSW)

Species	Setting time			
	18:00	22:00	3:00	Total
Girella tricuspidata	136	313	173	622
Sillago ciliata	149	199	204	552
Liza argentea	247	57	210	514
Acanthopagrus australis	125	144	171	440
Herklotsichthys castelnaui	122	183	71	376
Myxus elongatus	172	113	74	359
Mugil cephalus	113	122	100	335
Platycephalus fuscus	4	12	14	30
Meuschenia trachylepis	3	14	7	24
Pomatomus saltatrix	10	4	9	23
Cnidoglanis macrocephalus	19	–	–	19
Mugil georgii	2	3	6	11
Macquaria colonorum	1	2	8	11
Portunus pelagicus	6	4	1	11
Hyporhamphus regularis	7	1	1	9
Monodactylus argenteus	–	5	–	5
Rhabdosargus sarba	–	2	2	4
Gerres subfasciatus	–	–	3	3
Pseudocaranx dentex	–	–	1	1
Scylla serrata	–	1	–	1
Total	1116	1179	1055	3350

considering the lengths of economically important species. While the shortest soak time (1 h) caught similar size ranges for most species (Fig. 4), there were inconsistent and species-specific differences in the proportions of sizes captured between some soak times. These results highlight the difficulty in representatively sampling multi-species fish assemblages across a wide range of life-history stages, as allocation of sampling effort may be sub-optimal for certain species at particular times in their development (Kennelly et al., 1993). In future surveys, decisions concerning optimal sampling effort may involve prioritization of the key species of interest, together with compromises between power, cost and effort, and logistic constraints.

Table 6
Results of ANOVA (F ratios shown) testing for significant differences in the mean numbers of total individuals, species and the 5 most abundant species of fish (*M. cephalus*, *L. argentea*, *H. castelnaui*, *G. tricuspidata*, *S. ciliata* and *A. australis*) caught between setting times (18:00, 22:00 and 3:00 h) and zones (1 and 2) for multi-mesh gillnets (catch summed for 38, 54 and 89 mm mesh in each replicate gill net) used in Tuggerah Lake (NSW)

	Source of variation		
	Setting time (S)	Zone (Z)	$S \times Z$
	df (2)	df (1)	df (2)
Total individuals	0.09	0.45	1.67
Total species	1.11	0.93	Pld
G. tricuspidata	1.19	0.93	1.51
S. ciliata	0.34	0.42	Pld
L. argentea	1.06	0.13	Pld
A. australis	0.07	3.23	4.18*
H. castelnaui	0.05	0.33	Pld
M. elongatus	0.56	2.83	Pld.

"Pld" indicates the interaction term was non-significant at $P<0.25$ and pooled with the residual.
* $P<0.05$.

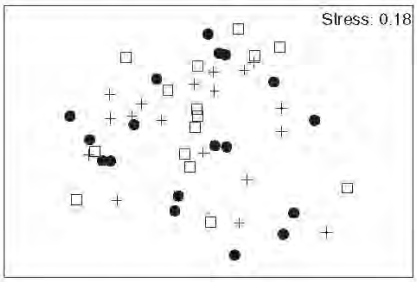

Fig. 6. Two dimensional nMDS ordination of standardized proportional catch data showing relationships between fish communities caught using multi-mesh gill nets set at 18:00 (●) 22:00 (□) and 3:00 (+) h in Tuggerah Lake, NSW.

While longer nets caught significantly more fish and species than shorter nets, this is a direct consequence of the longer nets covering greater areas and therefore encountering more fish (Acosta, 1994). Acosta (1994) suggested that longer panels may be more efficient than shorter panels at leading fish into nets and retaining them. The results presented here do not support this hypothesis (Fig. 2 and Table 3), since there were no differences in mean CPUE of the number of total individuals between different lengths of panels. In fact, mean CPUE of the numbers of total species was significantly greater in smaller panels. Similar results have been reported in previous studies using nets comparable in size to those tested here (e.g., 23–183 m, Minns and Hurley, 1988), although Acosta (1994) demonstrated no differences in CPUE between different net lengths. However, in this latter study, sampling was done during both day and night, and with much larger nets (e.g., 183–549 m) than those examined here. Direct observations by Acosta (1994) also revealed that after a fish detected a net, it swam alongside the meshes before eventually tiring and attempting to pass through. Sampling during daylight hours using relatively short nets is warranted to test if the results obtained in the present study are consistent over diel cycles. This may provide further insight into the relationships between net length and fish behaviour.

Few studies have directly quantified the specific mechanisms that contribute towards a reduction in the catchability of gill nets. Space limitation, in which

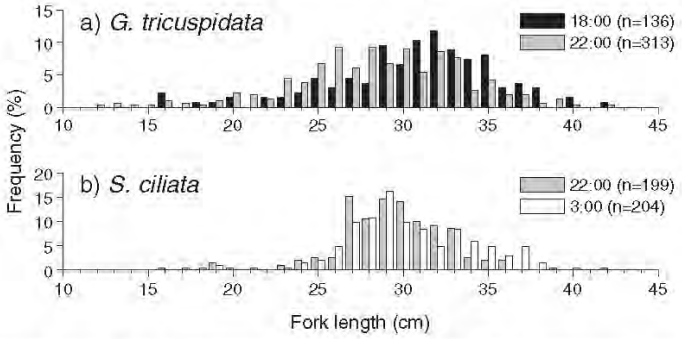

Fig. 7. Fork length–frequency distributions of (a) *G. tricuspidata* and (b) *S. ciliata* caught from different setting times (18:00, 22:00 and 3:00 h) in multi-mesh gill nets at Tuggerah Lake, NSW. Data are only presented for setting times where significant differences were detected in Kolmogorov–Smirnov tests (see text).

occupied and surrounding meshes are unable to retain fish, is often cited as a main factor in the saturation of nets (Minns and Hurley, 1988; Hansen et al., 1998). However, it is unlikely that a lack of unoccupied meshes was responsible for the reduction in CPUE or catch efficiency associated with longer soak times observed in the present study, since during retrieval there were often sections of free meshes observed across most mesh sizes, particularly for the largest mesh size (89 mm). In fact, the total numbers of fish were significantly greater after a soak time of 6 h. A more likely explanation for the observed reductions in efficiency is the presence of retained, struggling fish, which may have induced an avoidance reaction in fish at liberty (Kennedy, 1951; Hamley, 1975). Olin et al. (2004) presented a similar conclusion in a recent study of multi-mesh gill nets in Finland, but also hypothesized that a reduction in catching efficiency may have occurred due to accumulations of fish causing gill nets to sag, which altered the hanging ratio and allowed some fish to escape. Another possible factor that contributed towards the observed reduction in gill net efficiency with increasing soak time in the present study may have been some spatial reduction in fish availability owing to depletion around the gear (Minns and Hurley, 1988). Further research is required to address the potential for this and the other effects discussed above.

Given the results presented here, multi-mesh gillnets comprising 20-m panels and deployed for 1 h are considered optimal for use in future surveys of estuarine fish populations in NSW. Clearly, the benefit of short panels (20 m) compared to larger ones is that the number of replicate gill nets can be doubled on each sampling occasion, which increases the power of detecting within- and between-site variation, without concomitant increases in cost. Greater replication may also alleviate concerns relating to short panels (20 m) not catching large and representative samples of some species. For example, the total amount of net deployed in 6 replicate gill nets comprised of 20-m panels is essentially the same as that in 3 replicate gill nets comprised of 50-m panels (i.e., 120 vs. 150 m). The benefit of a 1-h soak time is also obvious in terms of expanding the scope for increased replication in future studies. Furthermore, a short soak time is more likely to reduce fish mortality (Buchanan et al., 2002), particularly if fish are processed and released as a gill net is retrieved.

The temporal deployment of nets within nights had no significant effect on the numbers of fish or species caught, or on the structure of assemblages. However, for some species, there were small and inconsistent differences in the proportions of individuals caught in certain size cohorts. Previous studies done in north American waters have suggested that gill net saturation within a single night might be confounded by variations in fish behaviour (Kennedy, 1951; Minns and Hurley, 1988). For example, a gill net set earlier in the night (i.e., 18:00 h) might fill with day-active and crepuscular species, while a gill net set later might fill with nocturnally active species. The lack of differences between setting times in the present study is probably related to inherent dissimilarities in the behaviours of North American and south-eastern Australia fish. Alternatively, the use of multi-mesh gill nets here may preclude the saturation of certain panels, particularly if there are prominent differences in size and morphology between species that are active at different times. In any case, the present study has shown that setting multi-mesh gill nets at different times during the night is unlikely to significantly bias estimates of relative abundances in future surveys.

The two experiments described in this paper have shown that multi-mesh gill nets constructed of 20-m panels and soaked for 1 h at any time during the night, are sufficient in sampling populations, assemblages and sizes of most species able to be caught by this type of gear. These findings, however, may not be applicable to all other types of estuaries in NSW, such as coastal rivers, or to a wide range of different habitats, without further research. An understanding of spatial and temporal variation in catches of fish in multi-mesh gill nets, across hierarchical scales, is also required to determine optimal levels of replication for future surveys.

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Appendix 8: Gray, C.A., Rotherham, D., Chapman, M.G., Underwood, A.J., and Johnson, D.D. Spatial scales of variation of assemblages of fish in coastal lakes sampled with multi-mesh gillnets: Implications for designing research surveys. Fish. Res. In press.

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Spatial scales of variation of assemblages of fish in coastal lakes sampled with multi-mesh gillnets: Implications for designing research surveys

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ABSTRACT

A key element in designing surveys to assess populations and assemblages of fish is to understand their spatial and temporal scales of variation. As part of a strategy to develop fishery-independent surveys, a pilot experiment was done in two coastal lakes in south-eastern Australia to examine scales of spatial variation among assemblages of fish caught in multi-mesh gillnets. A hierarchical, nested sampling design measured spatial variation among zones (2–20 km apart), sites within zones (1 km apart) and replicate gillnets at each site (50–100 m apart), in shallow and deep habitats. Assemblages differed between lakes. In both lakes, most species were caught in greater numbers in shallow samples, which had greater proportions of small fish of some species. Spatial variation in uni- and multivariate analyses was greatest at the smallest spatial scale (i.e. among replicates), which was consistent across species, depths and lakes. Variation among samples was also generally greater among sites than zones. It would be unnecessary to include the scale of zones in future sampling and more effort should be placed on sampling replicate nets and sites within lakes. The importance of doing pilot experiments and their general applicability to designing surveys of fisheries resources is illustrated.

1. Introduction

Much research has focused on understanding how fauna and flora vary over a range of spatial and temporal scales (Morrisey et al., 1992; Fraschetti et al., 2005; Stewart-Koster et al., 2007). Measuring spatial and temporal variation is important for understanding the mechanisms and processes that structure assemblages of species, assessing environmental impacts and for identifying appropriate scales of sampling (Levin, 1992; Underwood et al., 2000). Many studies have measured the spatial and temporal variation of organisms using hierarchical analyses of variance (Morrisey et al., 1992; Murdoch and Aronson, 1999), but there are few such studies of fish (Anderson and Millar, 2004). Moreover, few of these studies have tested for consistency of patterns across habitats.

Coastal lakes contain abundant and diverse fish faunas, which support commercial and recreational fisheries (e.g. Albaret and Lae, 2003; Gray and Kennelly, 2003; Rueda and Defeo, 2003). These environments are subject to strong environmental fluctuations (Kjerfve, 1994) and the fish in coastal lakes are also affected by many biotic factors (e.g. recruitment, predation, and competition; Livingston, 1987). Interactions between abiotic and biotic factors in time and space can strongly affect patterns of abundance, causing patchiness at a range of scales (Morrisey et al., 1992).

The problems of sampling at single spatial scales have been well documented (Andrew and Mapstone, 1987). Despite this, fish in lakes and estuarine systems are not generally sampled over multiple spatial scales; sampling is usually stratified across particular sites or habitats (Jones and West, 2005), along environmental gradients, such as salinity or flow regimes (Loneragan et al., 1986), or according to pre-determined spatial grids (Rueda and Defeo, 2003). Thus, potential confounding effects of small-scale variation on larger scale comparisons and, therefore, the appropriateness of many survey designs for sampling fish in lakes and estuaries, remain unknown.

Here, we use a hierarchical sampling design to examine spatial variation in populations and assemblages of fish in two coastal lakes in south-eastern Australia as a key step in developing sampling tools for surveys in coastal lakes and estuaries (Gray et al., 2005; Rotherham et al., 2007). Sampling, including the spatial scales of replicates (50–100 m), sites (1 km) and zones (2–20 km), was done in two depths and in two lakes (separated by hundreds of km) to assess the generality of results. Specifically, we tested the hypotheses that: (i) populations and assemblages vary at each spatial scale examined and (ii) patterns of variation are consistent across depths

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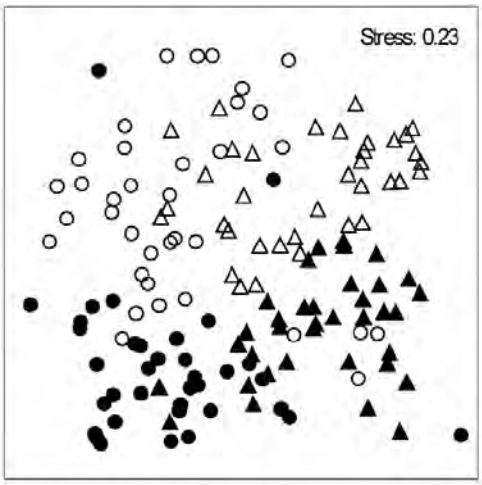

Fig. 1. Two-dimensional nMDS ordination plot of standardised data showing relationships among assemblages of fish sampled with multi-mesh gillnets in Lake Macquarie shallow (○), Lake Macquarie deep (●), St Georges Basin shallow (△) and St Georges Basin deep (▲). Stress value = 0.23.

and lakes. The purpose of the study was to determine what scales should be included in future sampling designs.

2. Methods

2.1. Study locations

Sampling was done in Lake Macquarie (LM) (151°36′E, 33°06′S) and St Georges Basin (SGB) (150°36′E, 35°08′S), in New South Wales (NSW, south-eastern Australia). These lakes are relatively shallow (mean depth <2 m), micro-tidal, well mixed and have no large salinity or tidal gradients. LM has a surface water area of 125 km^2; SGB has a surface water area of 44 km^2 (Roy et al., 2001).

2.2. Sampling design and methods

In each lake, three zones (2–20 km apart) and two sites within each zone (1 km apart), were selected randomly in shallow (<2 m) and deep (4–8 m) habitats. Depths were chosen as potential strata for future surveys. On each night of sampling, 12 replicate multi-mesh gillnets (6 shallow and 6 deep), separated by 50–100 m, were fished at a single site chosen at random. A different site was sampled on each night and the two lakes were sampled over a 13-day period in May/June 2005. Nets were bottom-set at dusk, left to soak for 1 h and then retrieved (Rotherham et al., 2006). Fish caught in each net were identified and counted; economically important species were measured for fork length (FL, to the nearest 0.5 cm).

Each multi-mesh gillnet consisted of 7 m × 20 m panels of different sizes of stretched mesh (36, 44, 54, 63, 76, 89 and 102 mm). The 36 and 44 mm panels were made from monofilament netting with twine diameter 0.15 mm; the remaining panels had multifilament netting with twine diameter 0.4 mm (Gray et al., 2005). All panels were 2 m deep and had a hanging ratio of $E = 0.5$. Adjacent panels were separated by 5 m of rope to maintain independence and restrict the smaller mesh panels 'leading' larger fish to adjacent larger meshed panels. The order of the panels of different-sized mesh in each net was randomly allocated.

Table 1
Total numbers of individuals of the 10 most abundant species sampled by multi-mesh gillnets in shallow and deep strata in Lake Macquarie and St Georges Basin

	No.	%
Lake Macquarie		
Shallow		
Herklotsichthys castelnaui	763	32.1
Girella tricuspidata	324	13.6
Gerres subfasciatus	319	13.4
Liza argentea	217	9.1
Mugil cephalus	132	5.6
Rhabdosargus sarba	91	3.8
Pomatomus saltatrix	75	3.2
Pelates sexlineatus	68	2.9
Acanthopagrus australis	65	2.7
Platycephalus fuscus	60	2.5
All other 22 spp.	260	11.0
Total	2174	100
Deep		
G. subfasciatus	289	35.0
P. sexlineatus	110	13.3
H. castelnaui	96	11.6
Sillago maculata	86	10.4
P. saltatrix	68	8.2
A. australis	58	7.0
L. argentea	24	2.9
Dinolestes lewini	21	2.5
P. fuscus	14	1.7
Pagrus auratus	14	1.7
All other 14 spp.	46	5.6
Total	826	100
St Georges Basin		
Shallow		
Sillago ciliata	604	26.3
L. argentea	361	15.7
M. cephalus	306	13.3
Myxus elongatus	295	12.8
G. subfasciatus	224	9.8
G. tricuspidata	137	6.0
P. saltatrix	88	3.8
Pseudocaranx dentex	78	3.4
A. australis	69	3.0
P. fuscus	51	2.2
All other 14 spp.	84	3.7
Total	2297	100
Deep		
A. australis	218	25.8
S. ciliata	142	16.8
P. saltatrix	132	15.6
G. subfasciatus	110	13.0
R. sarba	64	7.6
P. auratus	40	4.7
G. tricuspidata	38	4.5
P. dentex	26	3.1
S. maculata	18	2.1
L. argentea	14	1.7
All other 7 spp.	42	5.0
Total	844	100

% = percent contribution to total sample.

2.3. Analyses of data

Data were analysed using standard multivariate and univariate techniques. Multivariate analyses were done on Bray-Curtis dissimilarity matrices and non-metric multidimensional scaling (nMDS) was used to display multivariate patterns of assemblages. Analyses of similarity (ANOSIM; Clarke, 1993) and permutational multivariate analysis of variance (PERMANOVA; Anderson, 2005) tested hypotheses about differences in assemblages between lakes, depths and at the different spatial scales (i.e. zones and sites). Analyses were done using: (i) untransformed abundance data (to test hypotheses about the composition and abundance of species in the

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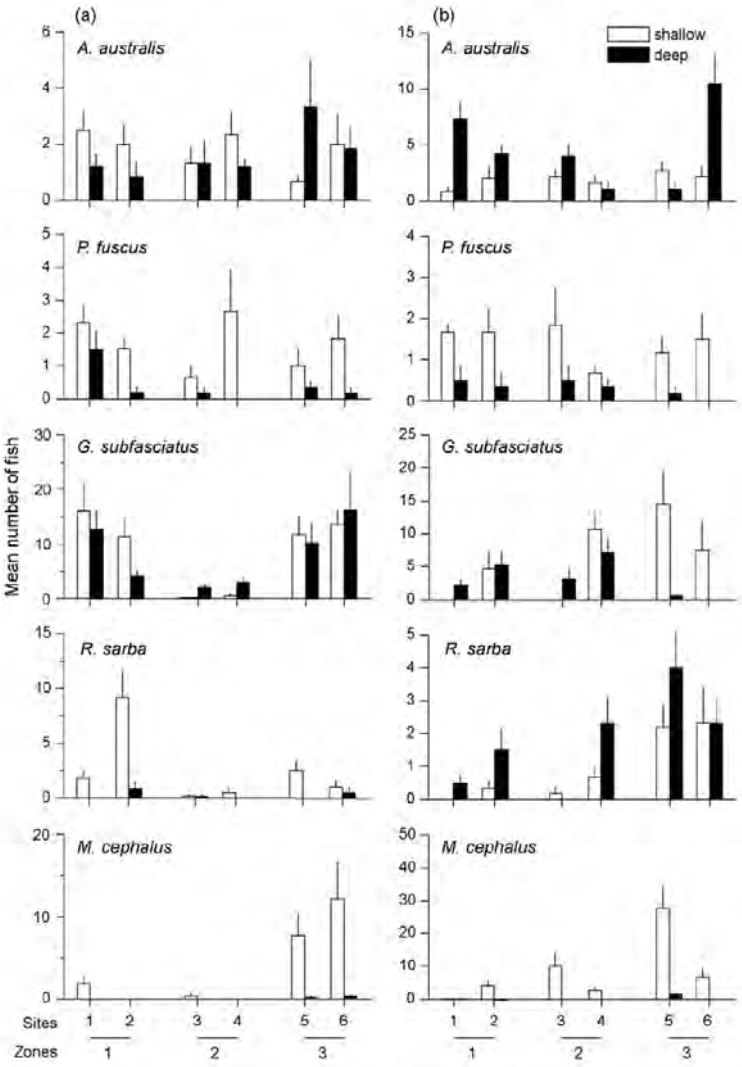

Fig. 2. Mean (+S.E.; $n = 6$) numbers of common species of fish sampled with multi-mesh gillnets in shallow and deep strata at six sites (nested in three zones) in (a) Lake Macquarie and (b) St Georges Basin.

assemblages); (ii) standardised data (to determine whether species were sampled in similar proportions in different samples); (iii) presence/absence data (to test hypotheses about the composition of assemblages and frequencies of occurrence of species).

Abundances of individual species were analysed separately for each lake and depth using nested ANOVA to obtain independent measures of components of variation for each spatial scale (Underwood, 1997). Analyses were restricted to species that were caught in more than 25% of samples at each depth in each lake. All spatial factors were considered random; analyses were done on untransformed data. Components of variation were calculated for each spatial scale (zones, sites and replicates) as in Underwood (1997). Negative components of variation were removed using pooling procedures (Fletcher and Underwood, 2002). Multivariate components of variation were similarly calculated for each of the different spatial scales from the PERMANOVA analyses.

Size-frequency distributions of abundant species (>50 individuals in both shallow and deep samples pooled across sites) were compared between depths in each lake using Kolmogorov–Smirnov (K–S) tests.

3. Results

3.1. General

Similar numbers of fish were sampled across depths and lakes, but more species were sampled at both depths in LM than in SGB

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Table 2
Results of components of variance derived from ANOVA for common species sampled at each depth in Lake Macquarie and St Georges Basin

	Lake Macquarie				St Georges Basin			
	Deep		Shallow		Deep		Shallow	
	VC	ø	VC	ø	VC	ø	VC	ø
A. australis								
Zones	0.3	0.06	0	–	12.1	1.02	0	–
Sites	0.0	–	0	–	1.4	0.12	0	–
Res	4.5	–	3	–	11.8	–	3.2	–
G. subfasciatus								
Zones	19.5	0.26	50	0.87	4.2	0.35	4	0.07
Sites	5.7	0.08	0	–	2.4	0.2	20.8	0.35
Res	75	–	57.5	–	12	–	60	–
G. triscuspidata								
Zones	na	na	0	–	2.5	0.64	0	0
Sites	na	na	0	–	0	–	4	0.67
Res	na	na	118.2	–	3.9	–	6	–
L. argentea								
Zones	0	–	0	–	0.2	0.5	129.9	0.88
Sites	0.8	0.36	26.7	0.72	0	–	20.8	0.14
Res	2.2	–	37.2	–	0.4	–	146.9	–
M. cephalus								
Zones	na	na	27	0.94	na	na	17.1	0.23
Sites	na	na	0	–	na	na	70.6	0.96
Res	na	na	28.7	–	na	na	73.4	–
P. fuscus								
Zones	0	–	0	–	na	na	0	–
Sites	0.2	0.5	0.1	0.04	na	na	0	–
Res	0.4	–	2.8	–	na	na	1.7	–
P. saltatrix								
Zones	1.4	0.26	0	–	1.4	0.12	0	–
Sites	1.4	0.26	3.3	1	1	0.09	2.4	0.36
Res	5.3		3.3	–	11.6	–	6.6	–
R. sarba								
Zones	0	–	2.5	0.33	0.7	0.25	1.2	0.71
Sites	0	–	8.1	1.08	1.1	0.39	0	–
Res	0.7	–	7.5	–	2.8	–	1.7	–
S. ciliata								
Zones	na	na	0	–	0	–	25	0.2
Sites	na	na	0.8	0.42	1.3	0.18	121.7	1.4
Res	na	na	1.9	–	7.2	–	86.6	–
S. maculata								
Zones	0	–	0	–	0.2	0.4	na	na
Sites	0.9	0.16	0	–	0	–	na	na
Res	5.7	–	2.1	–	0.5	–	na	na
Multivariate								
Zones	232.6	0.12	419	0.27	483.1	0.43	124.8	
Sites	477.2	0.26	1045.5	0.68	347.4	0.31	901.5	
Res	1863.3	–	1541.6	–	1117.6	–	1078.7	

VC = variance component; Ø = the ratio estimated magnitude of variance: estimated residual variance; na = no analysis.

(Table 1). More individuals and species were sampled in shallow than in deep water for both lakes. Samples were dominated by relatively few species; the 10 most abundant species accounted for 89–96% of all individuals sampled (Table 1).

3.2. Differences in assemblages between lakes and depths

The structure and composition of assemblages differed significantly between lakes and depths within each lake (ANOSIM, $P < 0.05$; see Fig. 1). Some species were caught in large numbers and proportions in only one lake (e.g. *Herklotsichthys castelnaui*, *Sillago maculata* and *Pelates sexlineatus* in LM; *Sillago ciliatia* and *Myxus elongatus* in SGB (Table 1), but most species found in only one lake were caught in small numbers (e.g. *Macquaria colonorum*, *Hyporhamphus regularis ardelio*).

Assemblages differed between depths primarily because of species being more abundant in shallow samples. In each lake, 10 species accounted for a very large percentage of the dissimilarity between depths (82% in LM; 91% in SGB from SIMPER analyses, Clarke, 1993). Eight of the species in the set of 10 from each lake were more abundant in shallow than deep samples.

3.3. Scales of spatial variation

Abundances of common species varied between depths and among zones and sites within each estuary (Fig. 2). Separate analyses of spatial components of variation were therefore done for individual species and the assemblages for each depth in each lake, separately. Of the 38 analyses, 16 had negative components of variation at the scale of zones, 13 at the scale of sites and 6 at more than 1

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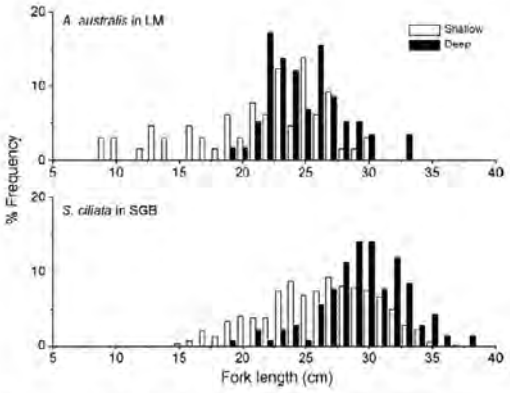

Fig. 3. Examples of fish sampled with multi-mesh gillnets that differed significantly in length between depths.

spatial scale (Table 2). The negative sources of variation were pooled (shown as zero variance in Table 2) and the other components were recalculated.

Most variation was at the smallest spatial scale of replicate nets (residual variance; Table 2) across species, depths and lakes for univariate and for multivariate data. Further, when components were scaled against the residual (to assess the magnitude of variation of each factor relative to the same reference; see Underwood, 1997), variation among zones was generally smaller than among sites. Not all species, however, displayed similar patterns of variation (e.g. *Gerres subfasciatus*; Table 2); nor were patterns among zones and sites consistent for all species across depths (e.g. *Acanthopagrus australis*) or lakes (e.g. *Liza argentea*).

There were significant differences (Permanova, $P < 0.05$) in the composition, abundance and proportions of species in assemblages (raw and standardised data) and the frequency of occurrence of species (presence/absence data) among sites for each depth and lake, but not between zones.

3.4. Size frequencies of fish

K–S tests detected significant differences in the size frequencies of *G. subfasciatus*, *A. australis* and *Pomatomus saltatrix* between depths in LM and of *S. ciliatia* in SGB (examples in Fig. 3). Greater proportions of smaller individuals were generally caught in shallow water (e.g. *G. subfasciatus* <12 cm, *A. australis* <20 cm and *S. ciliate* <25 cm FL). Greater proportions of larger individuals (*S. ciliatia* >27 cm and *P. saltatrix* >42 cm FL) were caught in the deep strata.

4. Discussion

The fish sampled here included several species (e.g. *G. subfasciatus*, *A. australis*, *Mugil cephalus* and *Girella tricuspidata*) that are important in the commercial estuarine fisheries of NSW (Gray, 2002; Gray and Kennelly, 2003). Our sampling with multi-mesh gillnets has provided additional data on (i) smaller sizes of these species; (ii) species not retained by commercial fishers (e.g. important baitfish species such as *H. castelnaui*) and (iii) species important in the recreational fishery (e.g. *P. saltatrix*).

The greater diversity of fish fauna in LM than in SGB accords with commercial beach-seine catches in these lakes (Gray and Kennelly, 2003). Similarly, the predominance of certain species in one lake, such as *S. maculata* in LM and *S. ciliatia* in SGB, is consistent with the beach-seine fisheries (Gray and Kennelly, 2003). Our sampling was not designed to ascertain the mechanisms and processes that caused the observed differences, but our results reinforce previous hypotheses that there are intrinsic dissimilarities in the fish fauna of these lakes. Similar patterns have been observed for benthic invertebrate fauna in coastal lakes (Dye and Barros, 2005). Thus, no particular lake could be classified or used as a 'reference' or 'control' for comparative purposes (see also Sheaves, 2006).

An important result was that most variability occurred at the smallest spatial scale investigated (among replicate nets), a pattern that was consistent across different species, depths and lakes. Another important result was that variation was generally larger among sites than among zones, although this pattern was not the same for all species, depths and lakes. Small-scale patchiness has also been observed for reef fishes in different habitats (Anderson and Millar, 2004) and is a general property of benthic assemblages in coastal habitats (review Fraschetti et al., 2005). Our data support the hypothesis that small-scale variability is common for a wide range of taxa in estuaries (Livingston, 1987). A similar understanding of scales of temporal variation (e.g. days, weeks, months and seasons) is required in designing more reliable long-term surveys (Morrisey et al., 1992).

Overall, our data show that including the scale of zones in future sampling is unnecessary; more effort should be put into sampling replicate nets and sites in future surveys of coastal lakes. Unless replication is sufficient at the relevant scales, future surveys are unlikely to be reliable for assessing patterns of abundance and detecting environmental impacts on assemblages of commercially important fish (Underwood and Chapman, 2003). In addition to identifying the scales at which the populations show most variation, the components of variation can also be used to do cost-benefit analyses, which will determine the most cost-efficient and optimal levels of spatial replication for future surveys (e.g. Kennelly et al., 1993; Underwood, 1997). This is the next step in our strategy to develop fishery-independent sampling tools (Rotherham et al., 2007). Pilot experiments are essential for designing cost-efficient and robust sampling strategies; here we have provided an example of the type of pilot studies that need to be done to assist in designing future research surveys.

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Appendix 9: Summary of experiment 5: Temporal variation of fish fauna sampled with multi-mesh gill nets

Aim

To provide information on the need to sample at different temporal scales and provide data for cost-benefit analyses to determine optimal levels of replication for future surveys.

Methods

Design of sampling and gear

We examined variation in populations and assemblages of estuarine fish across weeks, months and seasons, at two depths (shallow: < 2 m and deep: 4 – 8 m) in a coastal lake (St Georges Basin). Sampling also incorporated two spatial scales within the estuary: two sites 1 km apart nested in each of two zones (2 – 20 km apart; Fig 1) to measure temporal-spatial interactions and to provide greater generality. We did not attempt to measure variation among the nights sampled because it takes several nights (in this case 4) to sample all 4 sites across the two zones. Sites were sampled for two consecutive weeks, in two months in two consecutive 'seasons' (July – August, winter and October – November 2005, spring) to test the hypotheses that abundances of fish vary at each temporal scale.

Each multi-mesh gillnet consisted of 7 individual, 20-m panels of different sizes of stretched mesh (36, 44, 54, 63, 76, 89 and 102 mm), connected together in a random order. Adjacent panels were separated by 5 m of rope to minimise the potential effect of the smaller-mesh panels 'leading' larger fish to adjacent larger-meshed 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 E = 0.5.

Nets were bottom-set at dusk, left to soak for 1 hour and then retrieved (Rotherham et al., 2006). Catches in each net were then identified and counted; economically-important species of fish were measured for fork length (FL, to the nearest 0.5 cm).

Analyses of data

For frequently abundant species (occurring in >25 % of samples), we analysed 3 factors (seasons, months, weeks) separately for each site and depth using nested ANOVA and then extracted the components of variation (Underwood, 1997). Zones were not included in the model; instead, sites were treated as replicated experiments and analysed separately, giving 4 independent estimates of each component of variation. Multivariate components of variation were calculated for each of the different spatial scales using permutational multivariate analysis of variance (PERMANOVA; Anderson, 2001, 2005; Anderson and Millar, 2004). Components of variation were then averaged across the four sites. Negative components of variation were removed using pooling procedures (Fletcher and Underwood 2002).

Differences in assemblages between deep and shallow habitats were examined using multivariate analyses done on Bray-Curtis dissimilarity matrices. Centroids were calculated from the 6 replicate gill-nets in each depth for each sampling time. Analyses were done using: (i) untransformed abundance data (to test if the abundance of assemblages differed); (ii) standardised, untransformed abundance data (to test if assemblages were sampled in similar proportions); and (iii) presence/absence data (to test if the species composition of assemblages and frequencies of

occurrence of the different taxa differed). Analyses of similarity (ANOSIM; Clarke, 1993) tested the hypothesis that assemblages differed between depths. The SIMPER procedure was used to identify species that contributed to differences between assemblages (Clarke, 1993). Non-metric multidimensional scaling (nMDS) was used to display multivariate patterns of data.

Size-frequency distributions of abundant, economically-important species (> 50 individuals in both shallow and deep samples pooled across sampling times) were compared between depths in each lake using Kolmogorov-Smirnov (K-S) tests. Analyses were done for *A. australis*, *G. subfasciatus*, *G. triscuspidata*, *H. castelnaui*, *L. argentea*, *M. cephalus*, *P. dentex*, *P. fuscus*, *P. saltatrix*, *R. sarba*, *S. ciliata*, *S. maculata.*

Main results

A total of 13 484 fish and invertebrates from more than 38 species were sampled during the experiment. More individuals and species were caught in the shallow (8807 individuals from 36 species) than in the deep (4677 individuals from 28 species). Twenty six species were caught in both depths; ten species were caught only in shallow and 2 species only in the deep.

For most analyses, components of variation were negative at one or more scales; most frequently at the scales of weeks and months (Table 2). Therefore, the data were pooled as 4 times of sampling in each season and re-analysed (Table 3). Components of variation for species at deep sites were generally larger for seasons than for times within seasons. This pattern was, however, not consistent for the shallow sites. The only consistent pattern across both depths was that the residual variation (which is a spatial component) was greater than any temporal scale for every species (Table 3).

There were significant differences (ANOSIM, $p<0.05$) in the abundance, structure and composition of assemblages between shallow and deep water, which were obvious in nMDS ordinations (an example is shown in Fig 2 for the presence/absence data only). Differences were primarily driven by species that were caught in greater numbers (abundance data), proportions (standardised data) or more frequently (presence/absence data), in shallow water (e.g., *Sillago ciliata*, *Gerres subfasciatus*, *Girella tricuspidata*, *Herklotsichthys castelnaui*, *Mugil cephalus*, *Myxus elongatus*, Table 4). Some taxa were, however, caught predominantly, or in larger numbers and proportions in deep water (Table 4, e.g., *Acanthopagrus australis*, *Pomotomus saltatrix*, *Sillago maculata*, *Chelidonicthys kumu*, *Pagrus auratus*, *Pseudocaranx dentex*).

K-S tests detected significant differences in the size-frequencies of most species between depths (Table 5). In general, greater proportions of smaller individuals of many species were caught in shallow depths (Fig. 3). By comparison, greater proportions of larger individuals were caught in the deep strata (Fig. 3).

Main conclusions

Fishery-independent surveys are often time-consuming, labour-intensive and expensive. Because of these limitations, multiple sites and estuaries often cannot be sampled on the same night, or even the same week. Therefore, any spatial comparisons are potentially confounded by time because most sites (and estuaries) are sampled on different nights (or in different weeks). Here, for most species, temporal variation was small compared to spatial variation. So, as long as several sites within an estuary are sampled within a season, it is reasonable to interpret observed differences among sites as representing spatial rather than temporal variation. Further research is, however, needed to test whether similar patterns are observed over temporal scales longer than a few months.

There were differences in the structure and composition of assemblages and sizes of key species between deep and shallow water. Thus, in future studies, sampling across these depth strata would

be necessary to ensure that sampling is representative of the underlying populations and assemblages.

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Table 1. Total numbers of individuals of the 20 most abundant species sampled in multi-mesh gillnets in shallow and deep strata in St Georges Basin. % = percent contribution to the total numbers.

Shallow	**No.**	**%**	**Deep**	**No.**	**%**
Sillago ciliatia	2048	23.25	*Sillago ciliatia*	954	20.40
Gerres subfasciatus	1448	16.44	*Gerres subfasciatus*	624	13.34
Girella tricuspidata	757	8.60	*Girella tricuspidata*	504	10.78
Herklotsichthys castelnaui	669	7.60	*Herklotsichthys castelnaui*	474	10.13
Myxus elongatus	655	7.44	*Acanthopagrus australis*	443	9.47
Platycephalus fuscus	648	7.36	*Pomatomus saltatrix*	288	6.16
Liza argentea	603	6.85	*Rhabdosargus sarba*	229	4.90
Mugil cephalus	530	6.02	*Sillago maculata*	211	4.51
Pomatomus saltatrix	260	2.95	*Platycephalus fuscus*	200	4.28
Acanthopagrus australis	231	2.62	*Mugil cephalus*	184	3.93
Pelates sexlineatus	229	2.60	*Pseudocaranx dentex*	181	3.87
Rhabdosargus sarba	224	2.54	*Liza argentea*	155	3.31
Pseudocaranx dentex	167	1.90	*Pagrus auratus*	58	1.24
Sillago maculata	117	1.33	*Chelidonicthys kumu*	36	0.77
Meuschenia trachylepis	60	0.68	*Dasyatis thetis*	25	0.53
Monodactylus argenteus	36	0.41	*Myxus elongatus*	23	0.49
Monacanthus. chinensis	36	0.41	*Pelates sexlineatus*	19	0.41
Dasyatis thetis	25	0.28	*Trachurus novaezelandiae*	15	0.32
Portunus pelagicus	17	0.19	*Synaptura nigra*	14	0.30
Sphyraena obtusata	7	0.08	*Portunus pelagicus*	12	0.26
All other 15 spp.	40	0.45	All other 8 spp.	28	0.60
Total	8807	100	Total	4677	100

Table 2. Components of variation calculated from mean squares in ANOVA for abundant species caught in deep (D) and shallow (S) habitats. Temporal scales are seasons (S), months (M) and weeks (W).

		D	S			D	S			D	S
A. australis	S	1.36	0.09	*G. subfasciatus*	S	-0.39	35.56	*G. tricuspidata*	S	0.13	10.77
	M (S)	-0.39	0.18		M (S)	-1.45	0.42		M (S)	0.14	3.47
	W (M, S)	0.50	0.00		W (M, S)	8.16	14.02		W (M, S)	3.78	1.37
	Res	4.39	2.62		Res	18.60	43.22		Res	9.40	27.67
H. castelnaui	S	27.04	59.75	*C. kumu*	S	0.17	na	*L. argentea*	S	2.47	-5.70
	M (S)	-7.86	14.10		M (S)	-0.11	na		M (S)	1.23	0.54
	W (M, S)	12.62	35.09		W (M, S)	0.05	na		W (M, S)	-1.06	20.85
	Res	36.20	345.98		Res	1.10	na		Res	15.43	32.90
M. cephalus	S	4.80	7.69	*M. elongatus*	S	na	3.58	*M. trachylepis*	S	na	0.00
	M (S)	-0.19	-1.58		M (S)	na	0.38		M (S)	na	-0.01
	W (M, S)	-0.13	3.33		W (M, S)	na	1.63		W (M, S)	na	-0.01
	Res	5.02	7.38		Res	na	13.32		Res	na	0.31
P. auratus	S	1.43	na	*P. dentex*	S	1.03	-0.36	*P. fuscus*	S	-0.19	1.46
	M (S)	-0.08	na		M (S)	-0.01	1.03		M (S)	1.72	-0.47
	W (M, S)	-0.08	na		W (M, S)	0.53	-0.11		W (M, S)	0.27	0.89
	Res	2.18	na		Res	1.74	3.59		Res	1.37	5.78
P. saltatrix	S	0.13	0.46	*P. sexlineatus*	S	na	0.17	*R. sarba*	S	0.42	0.48
	M (S)	-0.16	-1.29		M (S)	na	0.57		M (S)	-0.26	0.55
	W (M, S)	0.75	3.24		W (M, S)	na	-2.21		W (M, S)	0.29	0.26
	Res	3.83	3.48		Res	na	33.39		Res	6.25	2.82
S. ciliata	S	7.01	-4.58	*S. maculata*	S	-0.26	0.60	Multivariate	S	571.62	590.87
	M (S)	-5.84	-1.34		M (S)	0.54	0.12		M (S)	16.11	37.69
	W (M, S)	16.59	6.65		W (M, S)	0.56	0.17		W (M, S)	468.30	223.01
	Res	17.40	90.13		Res	2.43	2.07		Res	1560.90	1408.30

Table 3. Components of variation calculated from mean squares in ANOVA for abundant species caught in deep (D) and shallow (S) habitats. Months and weeks were pooled as 4 times (T) of sampling within each season (S) because of negative estimates of variance.

		D	S			D	S			D	S
A. australis	S	1.23	0.15	*G. subfasciatus*	S	0.70	35.70	*G. tricuspidata*	S	0.17	11.93
	T (S)	0.24	0.13		T (S)	6.30	14.30		T (S)	3.87	3.68
	Res	4.39	2.62		Res	18.60	43.22		Res	9.40	27.67
H. castelnaui	S	24.42	64.45	*C. kumu*	S	0.13	na	*L. argentea*	S	2.85	0.39
	T (S)	7.39	44.49		T (S)	0.00	na		T (S)	0.23	17.83
	Res	36.20	345.98		Res	1.08	na		Res	15.03	32.90
M. cephalus	S	4.77	7.17	*M. elongatus*	S	na	3.70	*M. trachylepis*	S	na	16.07
	T (S)	0.60	2.28		T (S)	na	1.89		T (S)	na	0.00
	Res	5.53	7.38		Res	na	13.32		Res	na	45.71
P. auratus	S	1.37	na	*P. dentex*	S	1.03	0.01	*P. fuscus*	S	0.39	1.31
	T (S)	0.00	na		T (S)	0.52	0.57		T (S)	1.42	0.58
	Res	2.08	na		Res	1.74	3.58		Res	1.37	5.78
P. saltatrix	S	0.08	0.04	*P. sexlineatus*	S	na	0.71	*R. sarba*	S	0.33	0.66
	T (S)	0.64	2.38		T (S)	na	0.00		T (S)	0.11	0.62
	Res	3.83	3.48		Res	na	31.96		Res	6.25	2.82
S. ciliata	S	5.07	0.00	*S. maculata*	S	0.16	0.64	Multivariate	S	576.99	603.43
	T (S)	12.70	6.52		T (S)	0.79	0.24		T (S)	479.03	248.14
	Res	17.40	87.37		Res	2.42	2.07		Res	1560.90	1408.30

Table 4. Results of SIMPER analyses. Average abundance of species contributing to differences in assemblages of fish fauna between shallow and deep strata for a) raw abundance data; b) standardised abundance data (proportions); and c) presence/absence data. Species are listed in order of their contribution to the average dissimilarity (Av. Diss) between the two groups at a cumulative cut-off of 80%.

Species	Shallow Av. Abund	Deep Av. Abund	Av. Diss	Contrib%	Cum. %
a) raw abundance data					
Sillago ciliatia	10.67	4.97	13.05	20.26	20.26
Gerres subfasciatus	7.54	3.25	9.00	13.97	34.23
Herklotsichthys castelnaui	3.48	2.47	6.31	9.80	44.03
Girella tricuspidata	3.94	2.62	4.93	7.66	51.69
Myxus elongatus	3.41	0.12	4.66	7.23	58.92
Liza argentea	3.14	0.81	4.42	6.86	65.78
Platycephalus fuscus	3.37	1.04	4.38	6.80	72.57
Mugil cephalus	2.76	0.96	3.94	6.12	78.69
Acanthopagrus australis	1.2	2.31	2.52	3.91	82.60
b) standardised abundance data (proportions)					
Sillago ciliatia	10.67	4.97	9.38	15.89	15.89
Gerres subfasciatus	7.54	3.25	6.49	11.01	26.90
Girella tricuspidata	3.94	2.62	5.80	9.84	36.74
Herklotsichthys castelnaui	3.48	2.47	5.57	9.44	46.18
Acanthopagrus australis	1.2	2.31	5.24	8.88	55.05
Mugil cephalus	2.76	0.96	3.61	6.13	61.18
Myxus elongatus	3.41	0.12	3.43	5.81	66.99
Liza argentea	3.14	0.81	3.42	5.80	72.79
Platycephalus fuscus	3.37	1.04	3.37	5.71	78.50
Pomatomus saltatrix	1.35	1.5	2.70	5.58	83.08
c) Presence/absence data					
Meuschenia trachylepis	0.31	0.03	2.60	8.30	8.30
Myxus elongatus	3.41	0.12	2.28	15.58	15.58
Pelates sexlineatus	1.19	0.10	2.13	22.37	22.37
Mugil cephalus	2.76	0.96	1.78	28.06	28.06
Monacanthus chinensis	0.19	0.02	1.78	33.74	33.74
Herklotsichthys castelnaui	3.48	2.47	1.76	39.36	39.36
Liza argentea	3.14	0.81	1.75	44.95	44.95
Chelidonicthys kumu	0.01	0.19	1.60	50.05	50.05
Pagrus auratus	0.01	0.30	1.57	55.06	55.06
Pseudocaranx dentex	0.87	0.94	1.41	59.55	59.55
Sillago maculata	0.61	1.10	1.34	63.81	63.81
Dasyatis thetis	0.13	0.13	1.28	67.91	67.91
Synaptura nigra	0.03	0.07	1.28	72.00	72.00
Portunus pelagicus	0.09	0.06	1.11	75.33	75.53
Rhabdosargus sarba	1.17	1.19	0.88	78.35	78.35
Cnidoglanis macrocephalus	0.03	0.04	0.88	81.15	81.15

Table 5. Results of Kolmogorov-Smirnov tests (D values shown) for differences in the size frequency distributions of abundant species between deep and shallow habitats. **P*<0.05; ***P*<0.01; ****P*<0.001.

Species	Deep vs. Shallow	
	D	P
Acanthopagrus australis	0.233	***
Gerres subfasciatus	0.232	***
Girella. triscuspidata	0.217	***
Herklotsichthys castelnaui	0.301	***
Liza argentea	0.152	**
Mugil cephalus	0.138	*
Pseudocaranx dentex	0.2	**
Platycephalus fuscus	0.126	*
Pomatomus saltatrix	0.218	***
Rhabdosargus sarba	0.307	***
Sillago ciliatia	0.267	***
Sillago maculata	0.128	ns

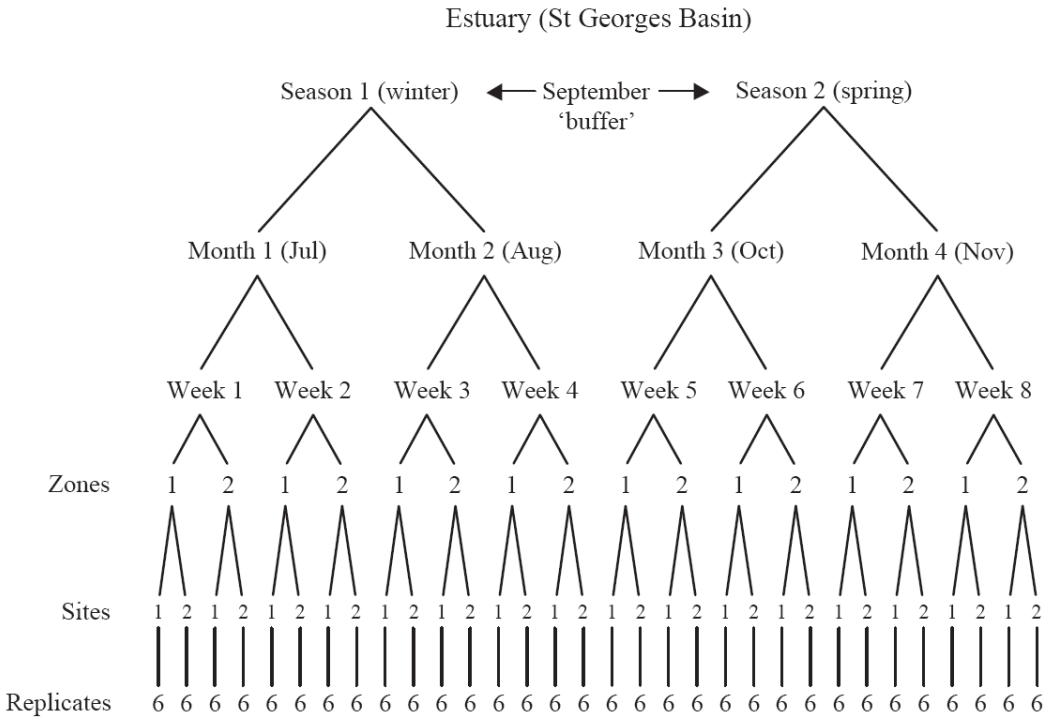

Figure 1. Diagrammatic representation of the design of the experiment

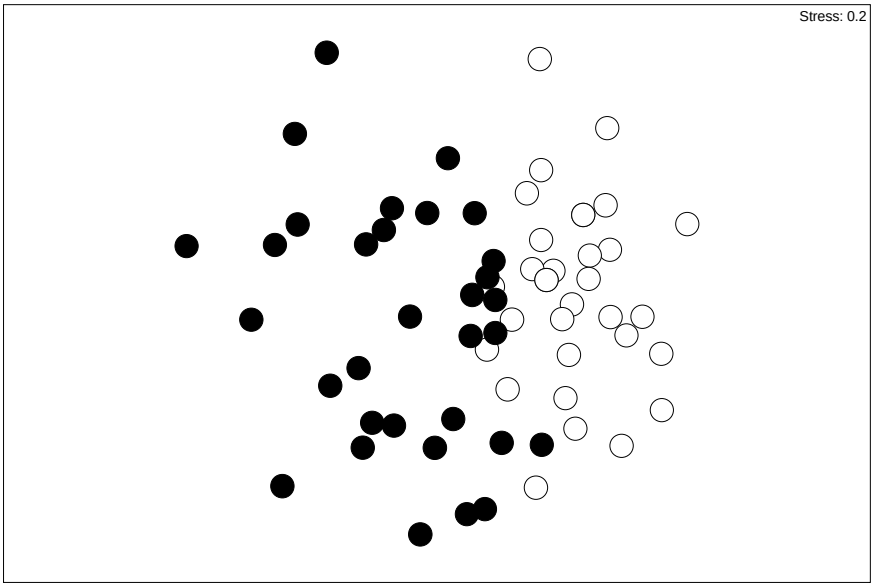

Figure 2. Two-dimensional nMDS plot of presence/absence data showing relationships between assemblages of fish sampled in deep (dark circles) and shallow (light circles) habitats with multi-mesh gill nets.

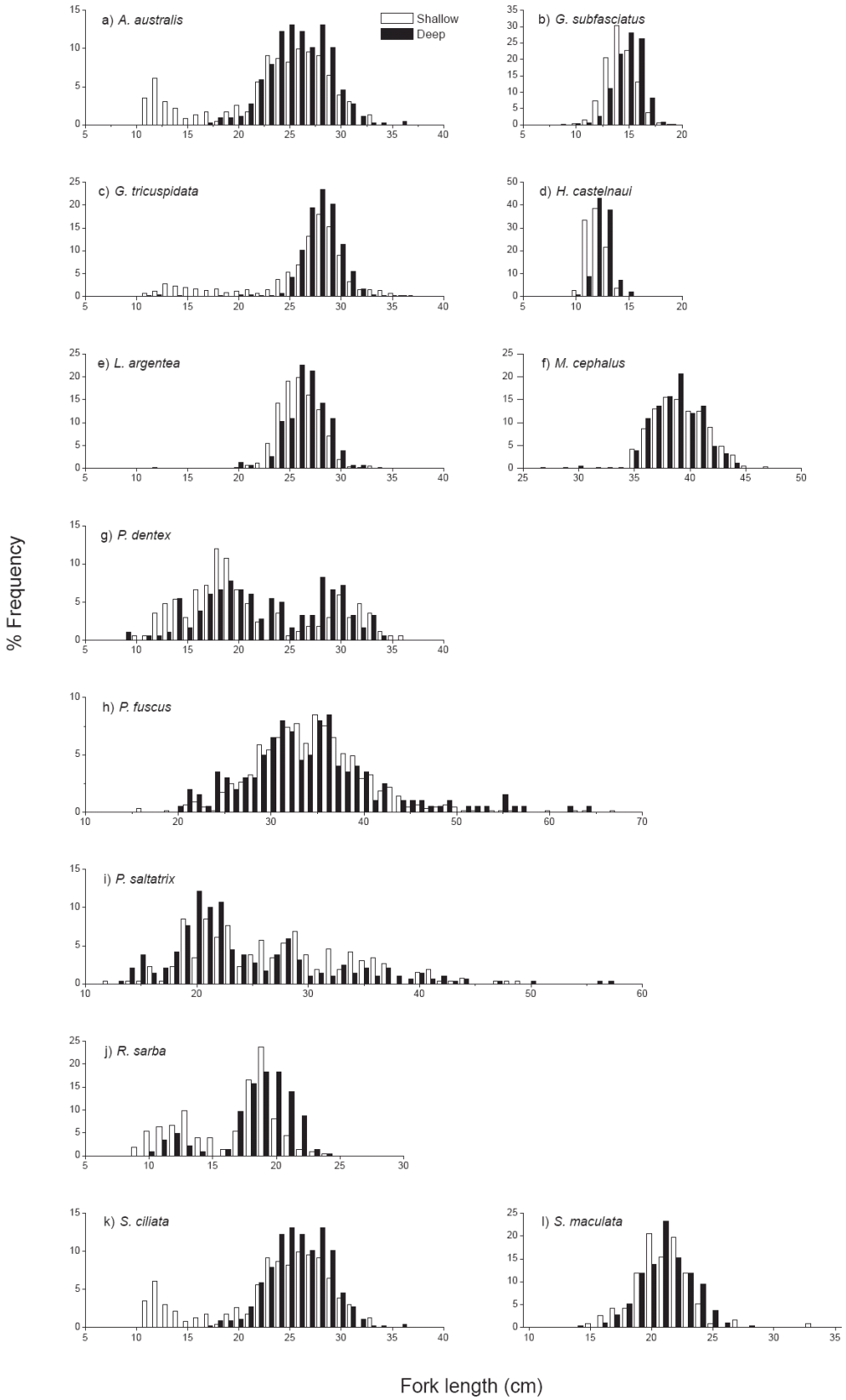

Figure 3. Size-frequency distributions of abundant economically-important species caught in deep and shallow habitats with multi-mesh gill nets. Data are pooled across sites and sampling times.

Appendix 10: Rotherham, D., Broadhurst, M.K., Gray, C.A. and Johnson, D.D. 2008. Developing a beam trawl for sampling estuarine fish and crustaceans: Assessment of a codend cover and effects of different sizes of mesh in the body and codend. ICES J. Mar. Sci. 65: 687–696.

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Developing a beam trawl for sampling estuarine fish and crustaceans: assessment of a codend cover and effects of different sizes of mesh in the body and codend

Douglas Rotherham, Matt K. Broadhurst, Charles A. Gray, and Daniel D. Johnson

Rotherham, D., Broadhurst, M. K., Gray, C. A., and Johnson, D. D. 2008. Developing a beam trawl for sampling estuarine fish and crustaceans: assessment of a codend cover and effects of different sizes of mesh in the body and codend. – ICES Journal of Marine Science, 65: 687–696.

An experiment was carried out in the Clarence River (New South Wales, Australia) to test the hypotheses that fish and crustacean catches in an experimental beam trawl were affected by a codend cover and the sizes of mesh in the body and codend. The cover had no obvious effects on the catches retained in the codend. Similarly, in comparisons between trawl bodies made from 26- and 41-mm diamond-shaped mesh, there were no differences in the assemblages of fish caught, or in the mean numbers entering the codends. For one species of fish (*Acanthopagrus australis*), however, there were differences in the proportions caught between the trawl bodies across different size classes. There was also some evidence to suggest that mesh size in the body of the trawl influenced the size selection of school prawns (*Metapenaeus macleayi*). For most finfish, there were no differences in catches between codends made from 20-mm and from 29-mm mesh hung on the bar (i.e. square-shaped mesh). In contrast, mesh size in the codend was important for the size selectivity of school prawns, with smaller carapace lengths at 50% retention in the 20-mm codend. We conclude that use of a 41-mm mesh in the body and a 20-mm square mesh in the codend of the beam trawl would be appropriate for future sampling with this gear in estuaries of New South Wales. A similar experimental approach to ours is needed in adapting the beam trawl to estuaries in other parts of the world, or in developing other types of research trawl.

Keywords: beam trawl, body effect, cover effect, fishery-independent surveys, sampling gear.

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Introduction

Conventional otter trawls have been used widely to sample populations and assemblages of demersal fish and crustaceans (Gunderson, 1993; Kennelly *et al.*, 1993; Korsbrekke *et al.*, 2001; Petrakis *et al.*, 2001). Nevertheless, the use of these gears for scientific sampling is problematic because (i) catches of fish are often enhanced by the herding effects of the otter boards and rigging, and (ii) the horizontal opening of trawls can be highly variable, particularly in scaled-down adaptations of commercial gears (Gunderson and Ellis, 1986; Wardle, 1986). Although otter trawls are used in estuaries throughout the world (Stokesbury *et al.*, 1999; Araújo *et al.*, 2002; Richardson *et al.*, 2006), full-scale commercial gears typically require large vessels, which often precludes sampling in relatively shallow water (West, 2002).

As sampling tools, beam trawls overcome many of the inherent inadequacies of otter trawls. Their rigid opening ensures a constant swept area, and the absence of otter boards limits any herding of fish (Gunderson and Ellis, 1986; Gunderson, 1993). Scaled-down versions of these gears also remain effective when deployed from small boats, and are ideal for sampling in shallow estuarine habitats. Consequently, small beam trawls (~1-m horizontal opening and 1–2-mm mesh opening) have been used extensively to sample juvenile fish and epifaunal crustaceans in seagrass beds (Gray and Bell, 1986; Loneragan *et al.*, 1995; Guest *et al.*, 2003). Although not commonly used in estuaries, larger beam trawls (~3-m horizontal opening), specifically developed to target a wider size- and species-range of icthyofauna, have potential as sampling tools for both ecological and fishery-independent studies across a range of habitats and depths.

Understanding the effects of different configurations of gear is essential for the development of standardized sampling tools that are optimal, reliable, and cost-efficient (Kennelly and Craig, 1989; Rotherham *et al.*, 2007). The size and configuration of mesh, especially in the codend, strongly affects the efficiency and selectivity of towed gears, and has been the focus of extensive research aimed at reducing unwanted catches in commercial otter trawl fisheries (Wileman *et al.*, 1996; Millar and Fryer, 1999; Broadhurst, 2000). Comparatively fewer studies have quantified similar changes to other towed gears, such as beam trawls, used for research sampling (e.g. Mous *et al.*, 2002). There is also limited information on the effects of changing the size of mesh used in other sections of towed gears on their retained catches

(DeAlteris *et al*., 1990; Broadhurst *et al*., 2000, 2005), which has implications for both improving gear selectivity and developing useful and cost-effective sampling gears.

The general experimental procedures for assessing changes to the sizes or configurations of mesh used in towed gears involve either towing the treatment and control configurations to be examined in alternate or simultaneous hauls, or attaching fine-meshed covers to the treatment gears to retain the escaping organisms (Pope *et al*., 1975; Wileman *et al*., 1996). The covered-codend technique is generally preferred, providing that the cover does not affect either the performance of the gear or the behaviour of the targeted organisms (Wileman *et al*., 1996). Despite the widespread and routine use of codend covers on both towed and static gears, most studies have ignored the potential for any confounding effects on the efficiency or selectivity of the gear (but see Madsen and Holst, 2002; Macbeth *et al*., 2005). Nevertheless, such effects are real (Wileman *et al*., 1996), so require examination in any study that seeks to estimate selection using this methodology.

Here, we investigate the utility of a beam trawl (3-m horizontal opening) for sampling key species of estuarine fish and crustaceans. Specifically, we first tested the effects of a codend cover on the catching efficiencies of a beam trawl rigged with two different sizes of mesh in both the body and the codend. We then examined the effects of different sizes of mesh in these sections of the gear on the size and species selection of fish and crustaceans in the codend. The results from this experimental work were then used towards developing an appropriately configured beam trawl for future research surveys.

Material and methods

Study site and construction of gears

The study was done in the Clarence River, New South Wales (NSW; 29°27′S 153°12′E; see West, 2002, for a description of the study area) in November and December 2005 using a chartered commercial prawn trawler (10 m long), rigged to tow two trawls in a standard twin-gear configuration. Two identical, stainless-steel beam trawl frames, (3 × 0.8 m; Figure 1) were constructed, along with two trawl bodies of identical metric dimensions (headropes, 3.7 m; groundropes, 4.1 m), twine characteristics (green polyethylene [PE], 3-strand, twisted twine of ~1.1 mm diameter) and mesh orientation (diamond-shaped), but different nominal mesh sizes (26 and 41 mm; Figure 2a).

The 41-mm mesh was chosen because it is similar to the minimum size of mesh used in commercial prawn trawls in estuaries of NSW (~40 mm) and is known to retain key species of fish and crustaceans of different size (Liggins and Kennelly, 1996; West, 2002). Although sizes of mesh other than 26 and 41 mm were available, differences in twine diameters precluded their comparison between trawl bodies. We also considered sizes of mesh smaller than 26 mm in the body of the trawl to be unsuitable because (i) sorting times would increase through retention of large numbers of very small, unwanted taxa (e.g. *Acetes* shrimp), and (ii) the trawl would be more difficult to tow through the water, which would increase the use of fuel and potentially affect the proper operation of the codend and cover. Buraschi S146R zippers 1-m long were sewn into the posterior ends of the trawl bodies to facilitate changing the codends and covers.

Four codends (all 1-m long and in circumference) were constructed from knotless, polyamide (PA) netting (braided twine, 2.5 mm diameter) hung on the bar (i.e. square-shaped) (Figures 1 and 2b). As the geometry of conventional diamond-shaped mesh can be affected by several parameters, including the volume or weight of the catch (Robertson, 1986), square mesh was used so that each codend remained a fixed sampling unit. Two of the codends were made from 20-mm mesh (100 × 100 bars) and the other two from 29-mm mesh (70 × 70 bars) (Figure 2b). A zipper 1-m long was sewn into the anterior edge of each codend to allow attachment and removal from the trawl bodies.

Two identical, hooped covers were designed to fit over the codends (Figure 1). Both covers were made from 16-mm knotless PA, diamond-shaped mesh (0.9 mm diameter, 3-strand, twisted twine) and measured 200 meshes in both the transverse (T) and normal (N) directions. Three aluminium hoops (600 mm diameter) were distributed along the length of the cover immediately adjacent to the codend (Figure 1). Zippers were used to attach the anterior edge of the covers to the trawl bodies.

Experimental procedure

At the start of the study, the 26- and 41-mm trawl bodies were randomly assigned to either side of the trawler (to eliminate any potential biases). Four codend configurations (20- and 29-mm

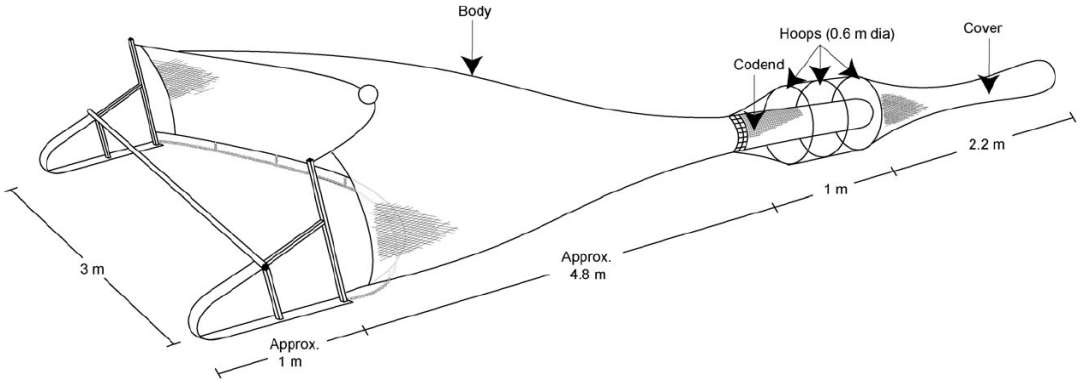

Figure 1. Diagrammatic representation of the experimental beam trawl used in the study.

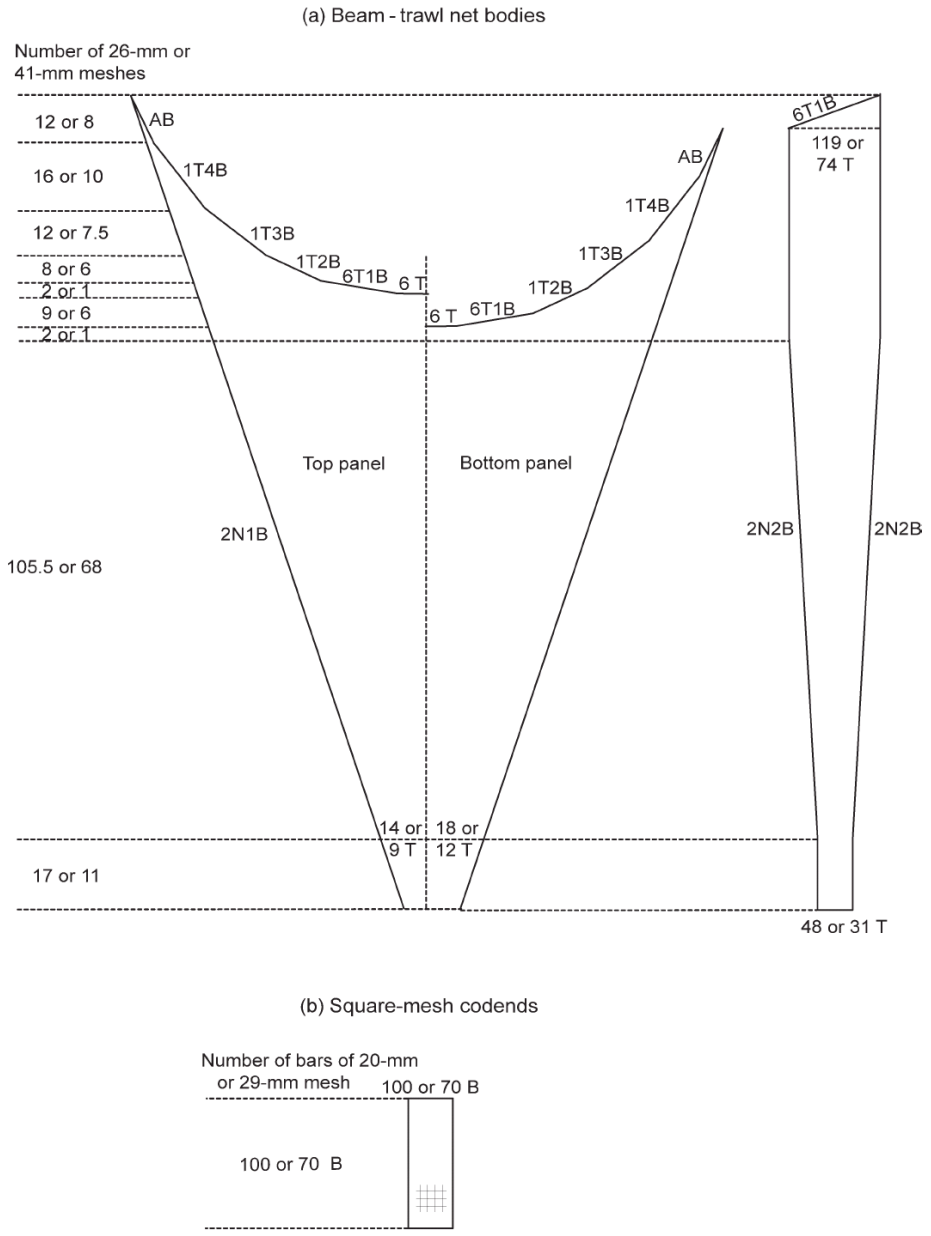

Figure 2. Plans of the (a) 26- and 41-mm net bodies, and (b) 20- and 29-mm square-mesh codends used in the experimental beam trawl. T, transversals; N, normals; B, bars.

codends, with and without covers) were tested for each trawl body (Table 1). For each deployment, two identical examples of one, randomly selected codend configuration were zippered to the 26- and 41-mm trawl bodies, then towed for 10 min at a speed of $\sim$1.2 m s^{-1}. Identical codends were attached to the different trawl bodies during each deployment so as to preclude the need for additional experiments (which are costly and may harm or kill more organisms than necessary), if it was later found that the cover had no significant effects on retained catches. Data from the covered codends could then be used to test the effects of mesh size in the body (26 vs. 41 mm, twin trawl) and codend (20 vs. 29 mm, alternate haul) on catches retained in the beam trawl.

The testing order of the codend configurations was randomized in blocks (each block consisting of one replicate of the four configurations). On each day, we attempted four replicate

Table 1. Summary of the different beam trawl treatments and their abbreviations used in the study.

Treatment	Abbreviation
26-mm diamond-mesh body with 20-mm square-mesh codend	26B–20C
26-mm diamond-mesh body with 20-mm square-mesh codend and 16-mm diamond-mesh cover	26B–20CC
26-mm diamond-mesh body with 29-mm square-mesh codend	26B–29C
26-mm diamond-mesh body with 29-mm square-mesh codend and 16-mm diamond-mesh cover	26B–29CC
41-mm diamond-mesh body with 20-mm square-mesh codend	41B–20C
41-mm diamond-mesh body with 20-mm square-mesh codend and 16-mm diamond-mesh cover	41B–20CC
41-mm diamond-mesh body with 29-mm square-mesh codend	41B–29C
41-mm diamond-mesh body with 29-mm square-mesh codend and 16-mm diamond-mesh cover	41B–29CC

deployments of each treatment configuration (i.e. 4 blocks × 4 codends). On the first day, however, only three replicate blocks were completed, resulting in a total of 15 replicates of each treatment configuration for each trawl body.

After each tow, the contents of the codends and covers (where appropriate) were emptied onto a partitioned tray and sorted by species. Data collected included the total numbers of individuals of each species, and the sizes of economically important fish (fork length, FL, to the nearest 0.5 cm), crabs (carapace width, to the nearest mm), and prawns (carapace length, CL, to the nearest millimetre). When the catch of prawns in a tow was large, the total weight and a subsample of 150 prawns were used to estimate total numbers, then CL measurements taken.

Analyses of data

Two-factor, orthogonal analyses of variance (ANOVA) were used to test the null hypotheses of no differences in retained catches (analysed for variables that included the total numbers of individuals and species, and selected abundant species) between (i) codend configurations (covered vs. uncovered; fixed factor) and days (random factor), separately for each codend (20 and 29 mm) attached to each trawl body, and (ii) covered codends (20 vs. 29 mm; fixed factor) and days (random factor) separately for each trawl body (26 and 41 mm). To ensure balanced analyses, data from day 1 were excluded.

Before ANOVAs, data were $\ln(x+1)$ transformed, to model treatment effects as approximately multiplicative. Data for prawns (*Metapenaeus macleayi*) from the comparison between 20- and 29-mm codends were, however, expressed as the proportion retained in each codend (i.e. not passing through the meshes of the codend and into the cover) and transformed to arcsin of the square root of the proportion. All transformed data were then tested for heteroscedasticity using Cochran's test. For data that remained heterogeneous, α was set to 0.01 to reduce the risk of type 1 error (Underwood, 1997). Multiple comparisons among means were carried out with Student–Newman–Keuls (SNK) tests. Differences between days were noted, but not investigated further. Where interaction terms were not significant at $p > 0.25$, they were pooled with the residual to increase the power of the test for the main effect of codend configuration.

Paired *t*-tests (two-tailed) were used to test the null hypothesis of no differences in the numbers and species of fish (total numbers of fish and species and selected abundant species) caught by the two trawl bodies (26 vs. 41 mm) attached to each of the covered codends (20 and 29 mm). Catches were expressed as totals in the codends and covers.

Differences in the structures of fish assemblages between (i) codend configurations (covered vs. uncovered), separately for each codend (20 and 29 mm) attached to each trawl body (26 and 41 mm), (ii) trawl bodies (26 vs. 41 mm), separately for each covered codend (20 and 29 mm), and (iii) codends (20 vs. 29 mm), separately for each trawl body (26 and 41 mm) were investigated using non-parametric multivariate analyses from the PRIMER 5 package (Version 5.2.2, PRIMER-E Ltd, 2001). Data were $\log(x+1)$ transformed to reduce the influence of abundant species and ordination plots generated from non-metric multidimensional scaling (nMDS) of Bray–Curtis similarity matrices. Analyses of similarities (ANOSIM) were used to test the *a priori* hypothesis that the structure of fish communities differed between covered and uncovered codends, and trawl bodies and codends with different sizes of mesh.

Differences in the size frequency distributions for abundant, economically important species were tested across all appropriate combinations of treatments using Kolmogorov–Smirnov (K–S) tests. For each replicate haul of a trawl configuration with a covered codend (see Table 1), the size frequencies of prawns (*M. macleayi*) caught in each replicate of the codend and cover were scaled where necessary, then vertically stacked (see Millar *et al*., 2004). Parametric selection curves (logistic and Richards) were then fitted using maximum likelihood. To account for between-haul variation, standard errors of the selectivity parameter estimates (i.e. CL at 50% retention, L_{50}, and selection range, SR) were corrected using the replicate estimate of dispersion (Millar and Fryer, 1999; Millar *et al*., 2004). The most appropriate model was determined by likelihood-ratio tests and visual examination of residual plots. The bivariate form of Walds *F*-test was used to test for differences between selection curves for comparisons of mesh size in the (i) trawl body for each codend (26B–20CC vs. 41B–20CC; 26B–29CC vs. 41B–29CC), and (ii) codend for each trawl body (26B–20CC vs. 26B–29CC; 41B–20CC vs. 41B–29CC). To preserve an overall type I error rate of 5%, a Bonferroni corrected *p*-value of 0.0125 (0.05/4) was used for each comparison.

Results

Numbers of fish and species

In all, 8241 fish and crustaceans (5634 and 2607 individuals in the codends and covers, respectively), consisting of >26 species, were caught during the experiment (Table 2). Four species accounted for >90% of the total catch: *M. macleayi* (65%), *Gerres subfasciatus* (11%), *Herklotsichthys castelnaui* (7%), and *Acanthopagrus australis* (7%). Catches in the codend covers mainly consisted of *M. macleayi* (~95%) and *Ambassid* spp. (~2%). Only small numbers (<20 individuals) and proportions (<0.1%) of most economically important finfish species passed through the meshes in the codends and into the covers. These were mainly small fish, including juveniles of species with low dorsal profiles (e.g. *Hyperlophus vittatus*, *Pomatomus saltatrix*, and *H. castelnaui*).

Table 2. List of the species and numbers of individuals caught with the experimental beam trawls in the Clarence River (NSW).

Family	Scientific name	Common name	Number
Ambassidae	*Ambassis* spp.	Glassy perchlet	395
Ariidae	*Arius graeffei*	Fork-tailed catfish*	70
Bothidae	*Pseudorhombus arsius*	Large-toothed flounder*	47
Chaetodontidae	*Selenotoca multifasciata*	Old maid	12
Carcharhinidae	*Carcharhinus* spp.	Shark*	1
Clupidae	*Herklotsichthys castelnaui*	Southern herring*	604
	Hyperlophus vittatus	Whitebait*	20
Dasyatididae	*Dasyatis thetidis*	Estuary stingray	1
Gerreidae	*Gerres subfasciatus*	Silver biddy*	937
Hemiramphidae	*Arrhamphus sclerolepis*	Snub-nose garfish*	1
	Hyporhamphus regularis	River garfish*	8
Monodactylidae	*Monodactylus argenteus*	Diamond fish	11
Mugilidae	*Liza argentea*	Flat-tail mullet*	1
Penaeidae	*Metapenaeus bennettae*	Greasyback prawn*	1
	Metapenaeus macleayi	School prawn*	5 321
	Penaeus monodon	Black tiger prawn*	3
	Penaeus plebejus	Eastern king prawn*	20
Platycephalidae	*Platycephalus fuscus*	Dusky flathead*	6
Plotosidae	*Euristhmus lepturus*	Long-tailed catfish	3
	Plotosis lineatus	Striped catfish	31
Pomatomidae	*Pomatomus saltatrix*	Tailor*	108
Portunidae	*Scylla serrata*	Mud crab*	1
Soleidae	*Aseraggodes macleayanus*	Narrow-banded sole	25
Sparidae	*Acanthopagrus australis*	Yellowfin bream*	577
	Rhabdosargus sarba	Tarwhine*	30
Teraponidae	*Pelates sexlineatus*	Striped trumpeter*	7
		Total numbers	8 241

Asterisks denote an economically important species.

For other finfish species (e.g. *A. australis* and *Rhabdosargus sarba*), no individuals passed into the covers.

Table 3. Results of ANOVA (*F*-ratios shown) testing for significant differences in the mean numbers of fish and crustaceans retained in the uncovered and covered codends (gear configuration) for different sizes of mesh in the body (26- and 41-mm mesh; 26B and 41B) and codend (20- and 29-mm mesh; 20C and 29C) of the beam trawls.

Source of variation	d.f	Mesh size in trawl (body-codend)			
		26B – 20C	41B – 20C	26B – 29C	41B – 29C
Total individuals					
Gear configuration (G)	1	0.19	0.05	2.60	0.08
Days (D)	2	0.69	1.20	0.14	0.42
G×D	2	Pld	Pld	Pld	1.60
Residual	18	–	–	–	–
Total species					
Gear configuration (G)	1	0.06	0.04	0.00	1.83
Days (D)	2	0.14	0.19	1.11	0.02
G×D	2	Pld	1.54	2.46	Pld
Residual	18	–	–	–	–
M. macleayi					
Gear configuration (G)	1	0.01	0.00	0.98	0.29
Days (D)	2	1.26	0.87	2.05	1.58
G×D	2	Pld	Pld	Pld	Pld
Residual	18	–	–	–	–
G. subfasciatus					
Gear configuration (G)	1	0.25	1.11	0.38	0.09
Days (D)	2	2.03	2.45	1.79	1.78
G×D	2	Pld	1.55	Pld	Pld
Residual	18	–	–	–	–
A. australis					
Gear configuration (G)	1	0.67	0.25	1.09	1.31
Days (D)	2	0.87	0.99	1.37	0.82
G×D	2	Pld	Pld	Pld	Pld
Residual	18	–	–	–	–
H. castelnaui					
Gear configuration (G)	1	0.01	0.19	0.37	0.00
Days (D)	2	0.16	1.14	0.07	0.17
G×D	2	Pld	Pld	Pld	Pld
Residual	18	–	–	–	–

Variables analysed included the total numbers of individuals and species, and abundant economically important species (*M. macleayi*, *G. subfasciatus*, *A. australis*, and *H. castelnaui*). "Pld" indicates that the *F*-ratio for the interaction term was not significant at $p > 0.25$ and was pooled with the residual. No significant differences ($p < 0.05$) were detected.

Assessments of the codend cover

ANOVA did not detect any significant differences in retained catches between the covered and uncovered codends attached to either of the trawl bodies, for any of the variables examined (Table 3). Similarly, there were no differences in the structures of fish assemblages between gear configurations (covered and uncovered codends) and days for any combinations of the trawl bodies and codends (ANOSIM, $p > 0.25$). Ordinations generated from nMDS scaling supported these results, because samples did not separate into distinct groupings for any codend or net body. For brevity, ordination plots are not shown for these or for any other multivariate analyses below. Always, stress values of the ordinations ranged between 0.13 and 0.18, which are generally considered adequate for reliable representations of data (Clarke, 1993).

Table 4. Results of K–S tests (*D* values shown) comparing the length frequency distributions of abundant economically important species between the covered and uncovered 20- and 29-mm codends (20C and 29C) attached to the 26- and 41-mm bodies (26B and 41B) of the beam trawls.

Species	Uncovered codend vs. covered codend			
	26B–20C	41B–20C	26B–29C	41B–29C
M. macleayi	0.233**	0.101	0.168*	0.540**
G. subfasciatus	0.163	0.061	0.209*	0.130
A. australis	0.263	0.217	0.451**	0.124
H. castelnaui	0.618**	0.176	0.249**	0.09

*$p < 0.05$; **$p < 0.001$.

K–S tests comparing the size frequencies of abundant economically important fish (*G. subfasciatus*, *A. australis*, and *H. castelnaui*) and crustaceans (*M. macleayi*) between covered and uncovered codends detected significant differences (Table 4). Nevertheless, results were inconsistent among species and across the different codends and trawl bodies (Table 4). For example, (i) the size frequency distributions and proportions of *M. macleayi* were variable between treatments (Figure 3a–c), (ii) the uncovered and covered codends caught greater proportions of small and large individuals of *A. australis*, respectively, but only for the 29-mm codend attached to the 26-mm trawl body (Figure 3d), and (iii) the uncovered codends caught greater proportions of small *G. subfasciatus* and *H. castelnaui* than the covered codends, but only for the 26-mm trawl body (Figure 3e; for brevity, data are only shown for *G. subfasciatus*).

Effects of mesh size in the body of the beam trawl

There were no significant differences in catches retained in the codends and covers between the 26- and 41-mm trawl bodies, for any of the variables examined (paired *t*-tests, $p > 0.05$; Table 5). These results were also consistent for both the small- and large-meshed codends. Multivariate analyses failed to detect significant differences between the trawl bodies for either codend (ANOSIM, $p > 0.25$). These results were supported by a lack of species groupings in the two nMDS ordinations.

K–S tests comparing the size frequency distributions of *A. australis* from the 26- and 41-mm trawl bodies detected significant differences for the 29-mm codend (the 26- and 41-mm bodies retained greater proportions of small and larger fish, respectively; Figure 4). No other significant differences were detected ($p > 0.05$).

Effects of mesh size in the codend

ANOVA detected a significant difference between codends (20 vs. 29 mm) for the mean number of *H. castelnaui* caught with the 26-mm trawl body (Table 6). Subsequent SNK tests showed that catches of *H. castelnaui* were significantly greater in the 20-mm codend (Figure 5). No other differences between codends were detected by ANOVA for the remaining variables analysed (Table 6; Figure 5). There was, however, a significant difference between days for *G. subfasciatus* (41-mm trawl body).

There were no significant differences in the assemblages of fish and crustaceans between codends for either trawl body (ANOSIM, $p > 0.05$) and there were no clear groupings of samples in nMDS ordinations. K–S tests comparing size frequency distributions of fish between codends detected significant differences in the

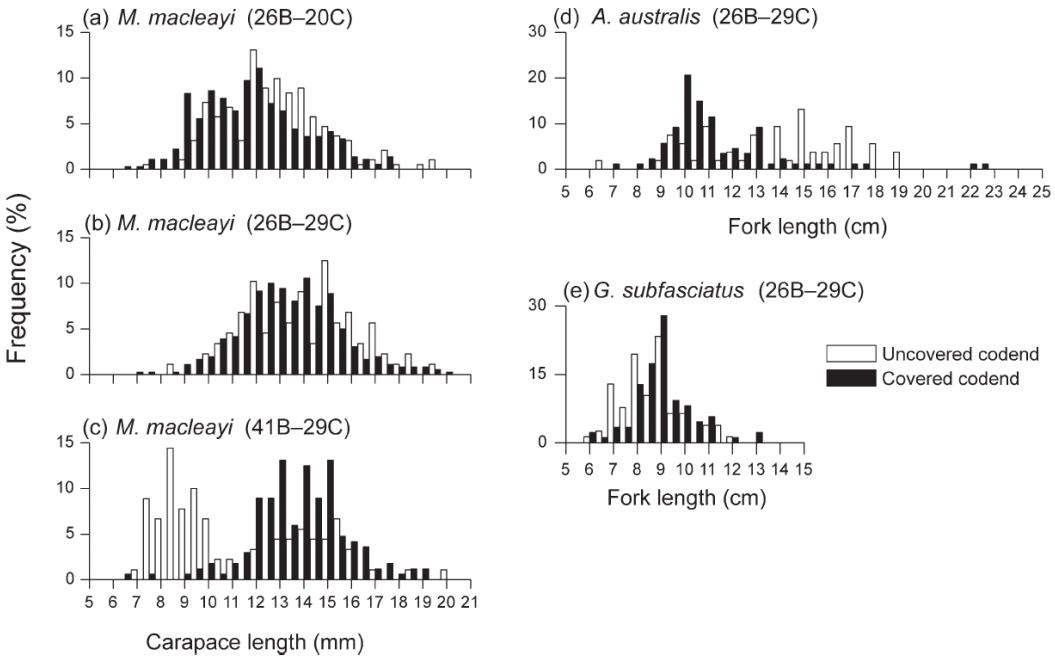

Figure 3. Size frequency distributions of (a–c) *M. macleayi*, (d) *A. australis*, and (e) *G. subfasciatus* retained in the uncovered and covered 20- and 29-mm codends (20C and 29C) attached to the 26- and 41-mm bodies of the beam trawls (26B and 41B). Data are only presented for trawl configurations where significant differences were detected by K–S tests (Table 3).

Table 5. Summaries of two-tailed paired *t*-tests comparing the numbers of total fish and species, and abundant economically important species caught in the covered 20- and 29-mm codends (i.e. codend and covers) attached to the 26- and 41-mm bodies of the beam trawls.

Species	26- vs. 41-mm trawl body					
	20-mm covered codend			29-mm covered codend		
	t	*p*	*n*	*t*	*p*	*n*
Total fish	0.965	0.351	15	1.323	0.207	15
Total species	−0.610	0.553	15	−1.468	0.164	15
M. macleayi	0.555	0.587	15	1.459	0.167	15
G. subfasciatus	−1.963	0.07	15	−1.943	0.072	15
A. australis	−1.896	0.079	15	−0.069	0.945	15
H. castelnaui	1.997	0.066	15	1.07	0.303	15

n, numbers of replicates. No significant differences ($p < 0.05$) were detected.

size compositions of *A. australis* for the 41-mm trawl body. There were, however, no clear patterns in the range of sizes or proportions of fish caught for this species (not shown for brevity). No other significant differences were detected by K–S tests ($p > 0.05$).

Size selection of *M. macleayi*

Size-selectivity curves were successfully converged for each codend attached to each trawl body. The logistic model was used always because there were no significant reductions in deviances with the Richards model (likelihood-ratio tests; $p > 0.05$). The selection curves representing the comparison between codends for each net body (i.e. 26B–20CC vs. 26B–29CC, and 41B–20CC vs. 41B–29CC) were significantly different (Wald's tests, $p < 0.001$; Figure 6, Table 7). There were, however, no significant differences between selection curves representing the comparison between trawl bodies for each codend (i.e. 26B–20CC vs. 41B–20CC, and 26B–29CC vs. 41B–29CC; Wald's tests, $p > 0.01$, Bonferroni corrected). Nevertheless, although the estimated L_{50}s for school prawns caught with different trawl bodies were similar for both the 20- and 29-mm codends, the SRs of the 26-mm trawl bodies were slightly wider than for the 41-mm trawl bodies (Figure 6, Table 7).

Table 6. Results of ANOVA (*F*-ratios shown) testing for significant differences in the mean numbers of total individuals and species, and abundant economically important species of fish and crustaceans (*M. macleayi*, *G. subfasciatus*, *H. castelnaui*, and *A. australis*) caught between codends (20 and 29 mm) attached to the 26- and 41-mm bodies of the beam trawls.

Source of variation	d.f	26-mm body	41-mm body
Total individuals			
Codend (C)	1	0.09	1.45
Days (D)	2	0.85	3.98
C × D	2	Pld	Pld
Residual	18	–	–
Total species			
Codend (C)	1	0.22	1.05
Days (D)	2	1.39	0.86
C × D	2	Pld	Pld
Residual	18	–	–
M. macleayi			
Codend (C)	1	1.68	1.10
Days (D)	2	0.61	2.98
C × D	2	Pld	Pld
Residual	18	–	–
G. subfasciatus			
Codend (C)	1	2.34	0.33
Days (D)	2	1.09	4.75*
C × D	2	Pld	Pld
Residual	18	–	–
H. castelnaui			
Codend (C)	1	97.15*	0.35
Days (D)	2	0.05	0.92
C × D	2	0.00	Pld
Residual	18	–	–
A. australis			
Codend (C)	1	2.65	0.05
Days (D)	2	0.71	0.64
C × D	2	Pld	Pld
Residual	18	–	–

"Pld" indicates that the *F*-ratio for the interaction term was not significant at $p > 0.25$ and was pooled with the residual. *$p < 0.05$.

Discussion

Our study has provided evidence to support the utility of the covered codend method for assessing the selectivity of an experimental beam trawl. Equally important, by testing hypotheses concerning the sizes of mesh used in the beam trawl we have contributed towards (i) understanding the influence of different sections of this gear on its selectivity, (ii) demonstrating the practical importance of pilot studies for developing cost-effective and reliable sampling tools, and (iii) developing an appropriate configuration for sampling estuarine fish and crustaceans in southeastern Australia.

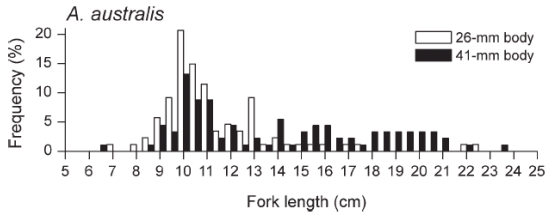

Figure 4. Size frequency distributions of *A. australis* retained in the covered 29-mm codend attached to the 26- and 41-mm bodies of the beam trawls.

The use of rigid hoops in covernets can alleviate the physical masking of codend meshes (Wileman *et al.*, 1996). Nevertheless, the presence of hooped covers can still potentially affect organisms entering the trawl and escaping through codend meshes owing to visual stimuli, changes in water flow through the gear, or both (Dahm *et al.*, 2002; Macbeth *et al.*, 2005). For example, any cover-induced reduction in the flow of water through a trawl may: (i) result in fewer numbers of smaller organisms being washed

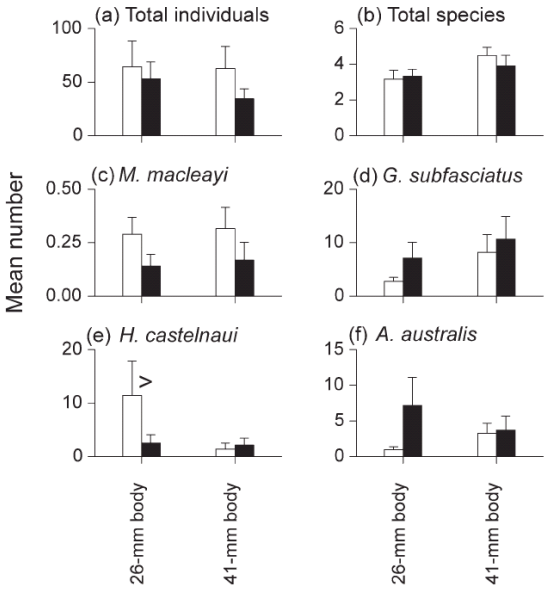

Figure 5. Mean numbers (±s.e.) of (a) total individuals, (b) total species and abundant economically important species, (c) *M. macleayi* (expressed as proportions retained in codend), (d) *G. subfasciatus*, (e) *H. castelnaui*, and (f) *A. australis* retained in covered 20-mm (white histograms) and 29-mm (black histograms) codends attached to the 26- and 41-mm bodies of the beam trawls. The directions (>) of significant differences detected in SNK tests are shown, where applicable.

through the codend meshes and into the cover, and (ii) because of concomitant increases in anteriorly displaced water, allow larger actively swimming fish to avoid capture. Such effects may manifest as differences in the numbers, proportions, and sizes of individuals caught in comparisons between covered and uncovered codends.

We found no differences between covered and uncovered codends for the mean numbers of fish and species retained, or the structures of their assemblages. Further, differences in the size frequency distributions of abundant economically important species were inconsistent among treatments and species, which

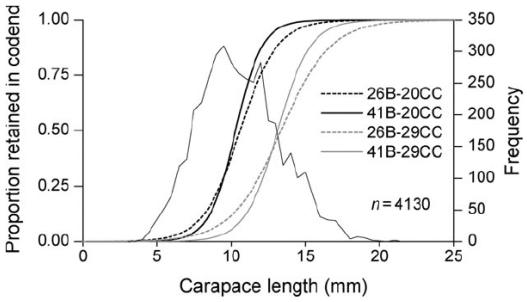

Figure 6. Logistic selection curves for school prawns (*M. macleayi*) caught in the covered 20- and 29-mm codends (20CC and 29CC) attached to the 26- and 41-mm bodies (26B and 41B) of the beam trawls. The size frequency distribution is for all school prawns retained in the codends and covers combined.

Table 7. CL (mm) at 50% probability of retention (L_{50}) and SRs of school prawns (*M. macleayi*) caught in the covered 20- and 29-mm codends (20CC and 29CC) attached to the 26- and 41-mm bodies of the 26B and 41B beam trawls.

Parameter	Mesh size in trawl (body-codend)			
	26B–20CC	41B–20CC	26B–29CC	41B–29CC
L_{50} (mm)	10.60 (0.15)	10.30 (0.15)	13.36 (0.38)	13.14 (0.26)
SR	2.73 (0.27)	1.97 (0.20)	3.67 (0.55)	2.42 (0.37)

Standard errors are shown in parentheses.

probably reflected spatial and temporal variability among replicate tows, rather than the mechanisms described above. These results indicate that the cover had minimal, if any, effects on the sampling performance of the beam trawl, irrespective of the sizes of mesh in the body or codend. Similar results have been reported in the few studies that have included assessments of cover effects on other towed (Madsen and Holst, 2002) and static (Macbeth *et al.*, 2005) gears.

Changing the size of mesh in the body of the beam trawl had no effect on catches entering the codend for most of the variables examined. Similarly, there were no differences in the structure of assemblages between trawl bodies. For one species (*A. australis*), however, greater proportions of larger individuals were retained in the 29-mm codend when it was attached to the 41-mm trawl body. This result may indicate that the 26-mm trawl body enhanced avoidance reactions in larger individuals of this species, possibly owing to more visual impact, some reduction in relative water flow through the meshes, or both. Alternatively, the difference may reflect natural between-haul variation, given that (i) the smaller-meshed trawl body also caught greater proportions of smaller individuals of *A. australis* (which were between 8 and 13 cm FL and too large to escape through the meshes of the 41-mm trawl body), and (ii) differences were inconsistent between small- and large-meshed codends.

Several of the few previous studies comparing mesh sizes in the bodies of trawls have demonstrated no differences in the retained catches of mobile finfish (e.g. DeAlteris *et al.*, 1990; Broadhurst *et al.*, 2005). Nevertheless, some results are inconsistent between different studies and have been related to species-specific behavioural responses of fish or fishery-specific operational characteristics of the gears (Broadhurst *et al.*, 2000, 2005). In contrast, several studies have demonstrated significant influences of the bodies of both otter and beam trawls on the size and species selection of crustaceans (e.g. Thorsteinsson, 1981; Hillis and Earley, 1982; Dremière *et al.*, 1999; Polet, 2000).

We found no differences in the selectivity parameters of *M. macleayi* between codends attached to the 26- and 41-mm trawl bodies examined here, although SRs were slightly wider for both codends when they were attached to the 26-mm trawl body. As for other penaeids (e.g. *Penaeus latisulcatus*; Broadhurst *et al.*, 2000), this result probably reflected some influence of the trawl body on the size selection of *M. macleayi*. Further research, perhaps using covers or pockets placed strategically over the body of the trawl (e.g. Polet, 2000), is required to validate these observations. The estimates of L_{50} that we obtained for *M. macleayi* were, however, similar to those estimated for this species using the same codends (i.e. 20- and 29-mm square mesh) attached to other types of gear (e.g. single and twin otter trawls; Broadhurst *et al.*, 2004; Macbeth *et al.*, 2004).

It is generally considered that most of the selection in towed gears occurs in the codend (DeAlteris *et al*., 1990; Wileman *et al*., 1996), and many studies have investigated the differences in catches between codends comprising different sizes and configurations of mesh (Broadhurst, 2000). In our study, mesh size in the codend was important for the size selectivity of school prawns, evidenced by the significant differences in parameter vectors, and in particular the lower estimate of L_{50} for the 20-mm codend. There were, however, no differences in the structure of assemblages between codends. Moreover, we found that, for most finfish, both codends were virtually non-selective for the sizes caught at the sampled sites, because only few individuals escaped through codend meshes and were retained in the cover. Therefore, the difference between codends for numbers of *H. castelnaui* (a species known to form large schools; Kuiter, 1996) was most likely the result of spatial and temporal variability among replicate tows.

We conclude that mesh sizes of 41 mm in the body and 20 mm in the codend of a beam trawl are appropriate for sampling the relative abundance, diversity, and sizes of fish and crustaceans caught in this study. The species caught in this short-term experiment are a typical subset of the fauna caught in towed gears in estuaries of NSW (Liggins and Kennelly, 1996; West, 2002). The beam trawl developed here may not, however, be appropriate for sampling species that (i) were not encountered during the experiment, and (ii) occur in estuaries elsewhere in the world. In fact, representative sampling of multispecies assemblages often requires several methods (Olin and Malinen, 2003), which should be developed for each specific region using properly designed pilot experiments (Andrew and Mapstone, 1987; Underwood, 1997; Rotherham *et al*., 2007). Our pilot work in developing fishery-independent sampling tools in estuaries of NSW has also included the use of multi-mesh gillnets (Gray *et al*., 2005; Rotherham *et al*., 2006).

Testing the effects of different configurations of gear is only one part of a broader strategy for developing scientific sampling tools for fishery-independent surveys (see Rotherham *et al*., 2007). Before starting large-scale, long-term surveys with a beam trawl, additional experiments are needed to test the effects of different sampling practices (e.g. tow duration, diel period) on retained catches. Moreover, determining optimal levels of replication for future surveys requires an understanding of spatial and temporal variation in fish fauna, across appropriate scales and strata.

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Appendix 11: Rotherham, D., Gray, C.A., Johnson, D.D. and Lokys, P. 2008. Effects of diel period and tow duration on estuarine fauna sampled with a beam trawl over bare sediment. Consequences for designing more reliable and efficient surveys. Est. Coast. Shelf Sci. 78: 179–189.

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Effects of diel period and tow duration on estuarine fauna sampled with a beam trawl over bare sediment: Consequences for designing more reliable and efficient surveys

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Abstract

The effects of diel period and tow duration (5, 10 and 20 min) on samples of estuarine fauna in a beam trawl, were tested over bare sediment in Tuggerah Lake (New South Wales, Australia). Mean catch rates (numbers of fish caught 5 min^{-1}) were significantly larger at night for the total numbers of individuals and abundant, economically important species of fish and invertebrates (e.g. *Gerres subfasciatus, Metapenaeus macleayi, Penaeus plebejus*). Greater proportions of larger fish were also caught at night for some species (e.g. *G. subfasciatus, Acanthopagrus australis, Rhabdosargus sarba*), but not across all tow durations. Multivariate analyses detected dissimilarities in the composition and structure of assemblages between diel periods, which were driven by species caught predominately, or in larger proportions, at night. Short tows (5 min) were more efficient than longer tows (10 or 20 min) for sampling the diversity of species (i.e. most species were caught in the first 5 min of a tow). There were, however, no clear or consistent patterns relating to the effect of tow duration on the catch rates of other variables, the size ranges of abundant species, or the structure and composition of assemblages. Our data confirm that at night, bare sediment is an important habitat for a wide size- and species-range of estuarine fish and invertebrates. In future, more cost-effective and reliable information concerning these taxa would be achieved by sampling with the beam trawl at night, using tow durations of 5 min. We also highlight a problem inherent in the design of many studies of diel variation of fauna (i.e. the potential non-independence of data among day and night periods) and discuss its solution.

Keywords: diurnal variations; estuaries; brackishwater fish; tow time; unvegetated habitat; south-eastern Australia

1. Introduction

Diel variation of fauna in estuarine, inshore and oceanic waters has been well documented (e.g. Ross et al., 1987; Walsh, 1988; Gray et al., 1998). Differences in patterns of distribution and abundance between day and night periods are, however, often inconsistent among taxa and habitats owing to a number of factors such as: (1) size- and species-specific variation in the behaviour of fish and invertebrates in relation to predators, competitors and prey (Burrows et al., 1994; Pillar and Barange 1997; Gibson et al., 1998; Nagelkerken et al., 2000); and (2) the type of sampling gear that is used (Olin and Malinen, 2003; Guest et al., 2003). The effects of these (and other) factors have important consequences for the accuracy and precision of samples obtained from ecological and fishery-independent surveys. Ideally, the decision to sample during the day, night, or both, should be determined a priori using properly designed experiments (Andrew and Mapstone, 1987; Underwood, 1997). Nevertheless, many studies have relied (and continue to rely) on daytime sampling, simply for pragmatic reasons (e.g. cost, safety and past practice).

In estuarine systems, many studies have used small, fine-meshed seine nets and beam trawls to examine diel variation of fish and invertebrates in seagrass beds (reviewed by Guest et al., 2003). In general, these types of gears are selective

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for small fish and crustaceans. Less is known of the effects of diel period across wider size ranges of organisms (including species of economic importance) and in other habitats within estuaries, such as bare sediment (e.g. Gray et al., 1998; Miller and Skilleter, 2006).

For towed sampling gears, such as trawls, the duration of tows is another factor that may affect the diversity, abundance and size ranges of retained fauna (Godo et al., 1990; Somerton et al., 2002); which also has implications for designing reliable and cost-effective sampling strategies. For example, reducing tow duration and increasing the number of replicate trawls may: (1) increase the precision of surveys (Pennington and Vølstad, 1991); (2) decrease the need for subsampling when catches are very large (Somerton et al., 2002); and (3) potentially reduce the mortality of sampled organisms.

Pilot experiments should be the preferred method of determining appropriate tow durations for trawling gears that are used as sampling tools (e.g. Kennelly et al., 1993). Unfortunately, for many trawl surveys, the effect of reducing the duration of tows has not been investigated until a number of years after their commencement (e.g. Godo et al., 1990; Wieland and Storr-Paulsen, 2006). In some cases, despite the benefits of using shorter, more-efficient tows, the original longer tows have been retained in order to preserve the continuity of the time series of data (Somerton et al., 2002).

Previous studies examining the effects of tow duration have mostly focussed on deep-water trawling grounds in the northern hemisphere (e.g. Godo et al., 1990). Although tow duration often has no effect on the mean sizes of fish and crustaceans, catch-per-unit-of-effort (CPUE) is generally higher for shorter tows (Godo et al., 1990; Somerton et al., 2002; Wieland and Storr-Paulsen, 2006). These results may not, however, be applicable to the fauna of other aquatic environments (e.g. estuaries); or to other types of towed gears (e.g. beam trawls).

In this experiment, we tested the hypotheses that diel period and tow duration affected catch rates, assemblages and size ranges of estuarine fauna retained in an experimental beam trawl. We then used the results of this pilot work to decide on an appropriate diel period and tow duration for future sampling with the beam trawl.

2. Materials and methods

2.1. Study area

The experiment was done in Tuggerah Lake (151°30′E; 33°21′S) in central NSW. Tuggerah Lake is a relatively large (70 km^2 surface area), shallow (average depth of about 2 m), microtidal, barrier estuary that consists of a central mud basin (Roy et al., 2001). Marine- and fluvial-delta sands are also located along the seaward and landward margins, respectively (Roy et al., 2001). Although substrates within Tuggerah Lake are predominately unvegetated and planar (Roy et al., 2001), the seagrass *Zostera capricorni* grows around the fringe of most of the shoreline and in some protected bays. The estuary also supports valuable commercial and recreational fisheries.

2.2. Design of experiment

Three sites separated by 1–5 km were selected over predominately flat, unvegetated sediment. Replicate sites were included to provide greater generality of results, as many previous studies investigating the effects of diel period have only sampled at a single site or location within an estuary (e.g. Guest et al., 2003). Sampling was done using a 3-m, stainless-steel beam trawl that was 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.

A total of 9 days and 9 nights (3 days and 3 nights at each of 3 sites) were sampled over a 6-week period during May and June, 2006. To avoid non-independence of data (see Underwood, 1997), replicate days and nights were sampled at random, but not within the same 24-h period. Tows during the day were done between 0700 and 1300 h and at night between 1800 and 0100 h. On each sampling occasion (a day or night period), four non-overlapping replicates of each tow duration (5, 10 and 20 min), were done at a randomly selected site. The order of tows was also assigned at random.

The beam trawl was towed at speeds of about 1.2 m s^{-1} in depths of water ranging from 1.5 to 2 m. 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), crabs (carapace width to the nearest mm) and prawns (carapace length (CL) to the nearest mm).

2.3. Analyses of data

2.3.1. Univariate

A four-factor analysis of variance (ANOVA) model was used to test for differences in standardised CPUE (defined as the number of individuals captured 5 min^{-1} and analysed for numbers of individuals, numbers of species and the six most abundant species of economic importance) among sites (random factor), between diel periods (day vs. night; fixed factor), among sampling periods (random factor nested in site and diel period) and among tow durations (5, 10 and 20 min; fixed factor). We standardised data by sampling effort (i.e. numbers 5 min^{-1}) because we were interested in testing hypotheses about the efficiency (i.e. catch rates) of different tow durations, rather than about which tow duration caught the most individuals or species.

Prior to standardising CPUE for the number of species it was necessary to correct for differences in the number of individuals caught among the different tow durations. Longer tows often catch more individuals. This is a problem because the likelihood of collecting more species increases when more individuals are collected (Simberloff, 1972; Gotelli and Colwell, 2001). So, we performed rarefaction analysis (Simberloff, 1972) using PAST (Palaeontologigal Statistics; Hammer et al., 2007). Rarefaction uses the number of species collected in the sample with the largest number of individuals, to generate the expected number of species in samples with smaller numbers of individuals (Simberloff, 1972; Gotelli

and Colwell, 2001). Following rarefaction, data were standardised to numbers of species 5 min^{-1}.

Prior to ANOVAs, data were tested for homogeneity of variances using Cochran's test and where necessary, transformed to $\ln(x+1)$. Heterogeneous data were still analysed, but with α set to 0.01 to reduce the risk of Type I errors (Underwood, 1997). Non-significant ($P > 0.25$) interaction terms were pooled to increase the power of tests for the main effects (Underwood, 1997). Student–Newman–Keuls multiple comparison tests were used to examine significant differences detected by ANOVA.

The variances of estimated means were used as measures of precision. Pooling of data across sampling periods provided replicate estimates of variance for each combination of site, diel period and tow duration (sensu Andrew and Mapstone, 1987). For abundant species that occurred in at least 75% of samples, a three-factor ANOVA was used to test the effect of site (three levels, random factor), diel period (two levels, fixed factor) and tow duration (three levels, fixed factor, $n = 3$ replicates) on estimates of precision. Some species were mainly caught during the day or night and so the precision of means were compared among tow times for the particular diel period, using two-factor ANOVA (sites: random, 3 levels; tow duration: fixed, 3 levels). All data were fourth-root transformed to reduce the influence of zeroes on estimates of precision and to facilitate comparisons among species. Post-hoc pooling and multiple comparison tests were done using the methods described above. Size–frequency distributions of abundant, economically important species were compared between diel periods and tow times using Kolmogorov–Smirnov (K-S) tests.

2.3.2. Multivariate

We used permutational multivariate analysis of variance (PERMANOVA; Anderson, 2001, 2005; previously called NP-MANOVA) to test the null hypotheses of no differences in assemblages between diel periods and among tow durations. Analyses were done using: (1) standardised, untransformed, abundance data (testing whether assemblages were sampled in similar proportions between diel periods and tow durations); and (2) presence/absence data (testing whether the species composition of assemblages and frequencies of occurrence of the different taxa differed between treatments).

Owing to difficulties in the interpretation of interactions involving more than two factors in multivariate space, it was considered necessary to reduce the number of factors compared to the full univariate design. Therefore, the effect of tow duration was first tested separately for each sampling occasion (i.e. 18 lots of independent, one-way analyses). There was no effect of tow duration for any of the 18 analyses done on the standardised data ($P > 0.05$). For the presence/absence data, there was no effect ($P > 0.05$) of tow duration for 14 of the 18 analyses. Based on the results of these analyses, we concluded that, overall, tow duration was not important and that replicates could be pooled and the factor eliminated from the analyses. Separate analyses were then done on the standardised and presence/absence data for each site using diel period (fixed) and sampling period (nested within diel period) as factors.

All multivariate analysis were done on Bray–Curtis dissimilarity matrices. Non-metric multidimensional scaling (nMDS) was done on the similarity matrices to visualise multivariate patterns of assemblages. The SIMPER procedure was used to identify species that contributed to differences between assemblages (Clarke, 1993).

3. Results

3.1. General

A total of 29 582 individuals from more than 41 species of fish and invertebrates were caught during the experiment (Table 1). Three species of finfish (*Gerres subfasciatus*, *Acanthopagrus australis* and *Rhabdosargus sarba*), and three species of penaeid prawn (*Metapenaeus macleayi*, *Penaeus plebejus* and *Metapenaeus bennettae*) comprised more than 80% of the total catch. Larger numbers of species and individuals were caught at night than during the day. There were also more species caught only at night (9 species), than only during the day (2 species). In most cases, however, species caught exclusively in either diel period were captured in relatively low numbers (<5 individuals; e.g. *Pseudorhombus jenynsii* and *Argyrosomus japonicus*). Longer tows caught more species, but these were few and in small numbers (Table 1).

3.2. Mean catch rates of total individuals, total species and economically important fish and invertebrates

The mean CPUE of the total number of individuals was significantly different between diel periods and among sites and sampling occasions (Table 2). Subsequent SNK tests revealed that catch rates were larger at night than during the day (Table 2; Fig. 1a). Mean catch rates of some abundant economically important species including *Gerres subfasciatus*, *Metapenaeus macleayi* and *Penaeus plebejus* were also significantly larger at night (SNK tests; Table 2). By comparison, mean CPUE of the total number of species was not affected by diel period; but there were significant differences among sites, sampling occasions and tow durations (Table 2). SNK tests revealed that as tow duration increased, the mean CPUE of the total numbers of species declined significantly (SNK tests, Table 2; Fig. 1b).

For other abundant economically important taxa (e.g. *Acanthopagrus australis* and *Rhabdosargus sarba*), there were no significant effects of diel period, tow duration, or any interactions between these factors (Fig. 1c, example shown for *A. australis* only). Although only very low numbers of *Metapenaeus bennattae* were caught during the day (<10 individuals) than at night, ANOVA failed to detect differences between diel periods for this species (Table 2; Fig. 1d). This was probably related to the low power of the test (df = 1, 2), as none of the interaction terms could be pooled.

3.3. Precision of sampling

The abundance of four species (*Gerres subfasciatus*, *Acanthopagrus australis*, *Rhabdosargus sarba* and *Metapenaeus*

Table 1
Total numbers of individuals of each species caught in a beam trawl towed for 5, 10 and 20 min during the day and night in an experiment in Tuggerah Lake (NSW). Data are pooled across sites and sampling periods

Species name	Day			Night			Total
	5 min	10 min	20 min	5 min	10 min	20 min	
Gerres subfasciatus[a]	134	260	317	1908	3924	8743	15286
Acanthopagrus australis[a]	336	680	1368	421	907	1593	5305
Metapenaeus macleayi[a]	39	88	160	281	541	1032	2141
Rhabdosargus sarba[a]	144	206	418	101	208	586	1663
Penaeus plebejus[a]	0	1	1	153	352	643	1150
Metapenaeus bennettae[a]	1	2	4	173	255	552	987
Pelates quadrilineatus[a]	3	17	30	79	152	257	538
Loligo spp.	58	85	132	42	55	92	464
Callionymidae spp.	1	0	2	37	127	269	436
Ambassis spp.	11	47	81	35	105	117	396
Sillago maculata[a]	0	3	7	37	92	215	354
Hyperlophus vittatus[a]	3	9	15	43	77	72	219
Herklotsichthys castelnaui[a]	8	20	15	22	56	70	191
Platycephalus fuscus[a]	1	9	10	11	22	51	104
Meuschenia trachylepis[a]	9	17	24	3	0	1	54
Pseudorhombus arsius[a]	0	3	6	4	10	29	52
Synaptura nigra[a]	0	0	0	4	15	20	39
Portunus pelagicus[a]	0	1	5	4	4	16	30
Portunus sanguinolentus[a]	0	1	2	5	5	14	27
Sillago ciliata[a]	0	0	0	1	10	12	23
Pomatomus saltatrix[a]	3	1	2	3	2	12	23
Arenigobius frenatus	1	0	1	3	5	10	20
Centropogon australis	0	0	0	1	4	8	13
Pagrus auratus[a]	0	0	2	2	3	5	12
Hyporhamphus regularis[a]	1	2	2	2	2	1	10
Liza argentea[a]	2	0	1	2	1	3	9
Monacanthus chinensis[a]	1	2	1	0	1	1	6
Cnidoglanis macrocephalus[a]	2	1	0	0	0	1	4
Dasyatis thetidis	0	2	0	0	1	1	4
Pseudorhombus jenynsii[a]	2	0	2	0	0	0	4
Penaeus esculentus[a]	0	0	0	1	2	1	4
Girella tricuspidata[a]	0	0	0	1	0	2	3
Plotosus lineatus	0	0	3	0	0	0	3
Monodactylus argenteus[a]	0	0	0	1	0	0	1
Siganus nebulosus[a]	0	0	0	0	0	1	1
Argyrosomus japonicus[a]	0	0	0	0	0	1	1
Dinolestes lewini[a]	0	0	0	0	0	1	1
Meuschenia freycineti[a]	0	0	0	0	0	1	1
Aseraggodes macleayanus	0	0	0	1	0	0	1
Tetractenos hamiltoni	0	0	0	0	0	1	1
Enoplosus armatus	0	0	0	0	1	0	1
Total	760	1457	2611	3381	6939	14434	29582

[a] Species of economic importance.

macleayi) was frequent enough (occurring in >75% of samples) to compare estimates of precision using the full 3-factor ANOVA model (sites, diel periods and tow durations). Variances were significantly smaller (i.e. means were more precise) during the day than at night for *G. subfasciatus* (Table 3a). For other species, there were either: (1) significant interactions between diel periods and sites (e.g. *A. australis*; Table 3b); or (2) no significant differences (e.g. *R. sarba*; Table 3c). There were no differences in the precision of means among tow durations for any of the analyses (Table 3a–f).

3.4. Size–frequency distributions

The size–frequency distributions of *Gerres subfasciatus*, *Acanthopagrus australis* and *Rhabdosargus sarba* were significantly different between diel periods, but only for short (5 min) and intermediate (10 min) tow durations (K-S tests, Table 4). These differences were generally driven by greater proportions of larger fish that were caught during the night (Fig. 2a; *G. subfasciatus* is shown as an example). There were also significant differences in size–frequency distributions between tow durations at night for a number of species. For example, larger proportions of small individuals of *A. australis* (<10 cm FL) were caught in 5-min tows, than in 10- or 20-min tows (Fig. 2b; example shown for 5 vs. 10 min only). The proportions of larger individuals of *A. australis* were similar between 5- and 10-min tows, and 5- and 20-min tows (data shown for 5 vs. 10 min only). Similarly, there were no other patterns in the proportions of *G. subfasciatus* or *R. sarba* caught between

Table 2
Analyses of the effects of site (Si; random), diel period (Di; fixed), sampling period (Pe; random and nested within Si and Di) and tow duration (To; fixed) on the mean CPUE (numbers 5 min^{-1}) of the total numbers of individuals, species and abundant taxa of economic importance (*Gerres subfasciatus*, *Acanthopagrus australis*, *Metapenaeus macleayi*, *Rhabdosargus sarba*, *Penaeus plebejus* and *Metapenaeus bennettae*) caught in a beam trawl in Tuggerah Lake (NSW). Results of SNK tests are shown. ns, not significant; $^{*}P < 0.05$; $^{**}P < 0.01$; $^{***}P < 0.001$. *F* ratios are not shown for interaction terms that were pooled (or eliminated) at $P > 0.25$. Degrees of freedom are shown in parentheses following the pooling of non-significant interaction terms

Source		MS	*F*	*P*	MS	*F*	*P*
		(a) Total individuals			(b) Total species[a]		
Si	2	29.32	29.52	***	2.82	29.98	***
Di	1	138.51	55.87	*	13.01	38.48	ns
Pe (Si × Di)	12	0.99(194)	6.63	***	0.09(186)	6.08	***
To	2	0.01(194)	0.06	ns	7.31	145.33	***
Si × Di	2	2.48	2.50	ns	0.34	3.59	ns
Si × To	4	0.02			0.05(186)	3.25	ns
Di × To	2	0.02(194)	0.12	ns	0.16	6.63	ns
To × Pe(Si × Di)	24	0.11			0.01		
Si × Di × To	4	0.17			0.02(186)	1.57	
Residual	162	0.16			0.02		
SNK		Di: N > D			To: 5 > 10 > 20		
		(c) *Gerres subfasciatus*			(d) *Acanthopagrus australis*		
Si	2	38.67(14)	10.96	**	2773.82(14)	14.46	***
Di	1	344.07(14)	97.53	***	300.45(14)	1.57	ns
Pe(Si × Di)	12	3.86			198.70		
To	2	0.37(32)	0.90	ns	10.30(194)	0.50	ns
Si × Di	2	1.53			150.89		
Si × To	4	0.26			2.01		
Di × To	2	0.43(32)	1.05	ns	11.38(194)	0.55	ns
To × Pe(Si × Di)	24	0.47			13.89		
Si × Di × To	4	0.21			3.19		
Residual	162	0.28			22.54		
SNK		Di: N > D					
		(e) *Metapenaeus macleayi*[a]			(f) *Rhabdosargus sarba*		
Si	2	10.69(14)	3.75	ns	74.01(14)	2.71	ns
Di	1	99.05(14)	34.71	***	0.00(14)	0.00	ns
Pe(Si × Di)	12	3.06			25.35		
To	2	0.29	0.92	ns	7.91(32)	0.78	ns
Si × Di	2	1.60			39.07		
Si × To	4	0.31(190)	1.65	ns	4.18		
Di × To	2	0.99(190)	0.53	ns	25.10(32)	2.48	ns
To × Pe(Si × Di)	24	0.17			10.32		
Si × Di × To	4	0.21			14.82		
Residual	162	0.19			6.66		
SNK		Di: N > D					
		(g) *Penaeus plebejus*[a]			(h) *Metapenaeus bennettae*[a]		
Si	2	0.55(14)	0.78	ns	2.96	2.27	ns
Di	1	134.42(14)	188.75	***	92.71	40.13	ns
Pe (Si × Di)	12	0.76			1.31	6.25	***
To	2	0.14(194)	1.28	ns	0.15	0.29	ns
Si × Di	2	0.45			2.31	1.77	ns
Si × To	4	0.07			0.52	2.05	ns
Di × To	2	0.10(194)	0.97	ns	0.15	0.27	ns
To × Pe (Si × Di)	24	0.06			0.26	1.22	ns
Si × Di × To	4	0.05			0.55	2.15	ns
Residual	162	0.12			0.21		
SNK		Di: N > D					

[a] Data remained heterogeneous after transformation.

different tow durations during the night (data not shown). Only one significant difference was detected between tow durations in the daytime (K-S tests); greater proportions of smaller and larger individuals of *G. subfasciatus* were caught in 5- and 10-min tows, respectively (Fig. 2c).

3.5. Multivariate analyses

There were significant differences in the structure (standardised data) and composition (presence/absence data) of assemblages between diel periods and among sampling

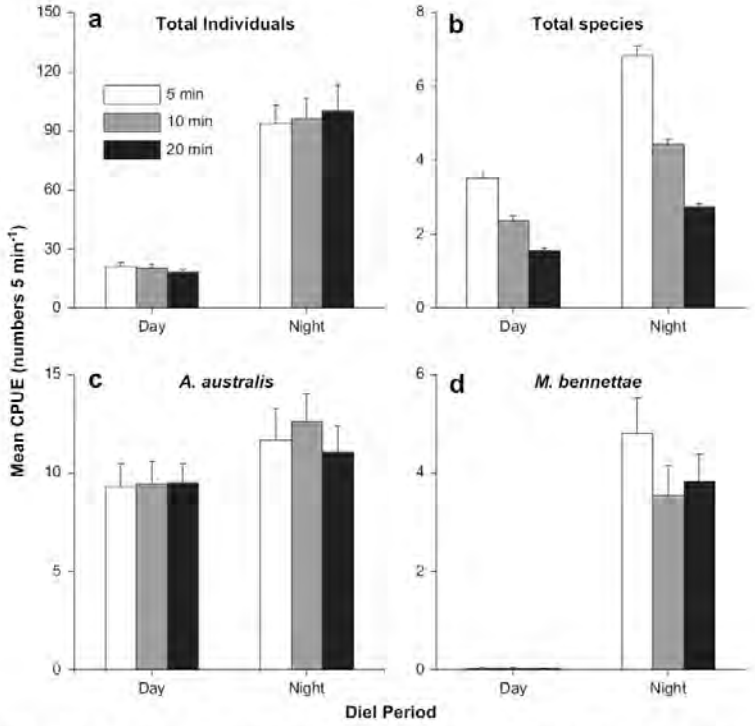

Fig. 1. Mean CPUE (numbers 5 min $^{-1}$) of the (a) total numbers of individuals, (b) total numbers of species, and abundant economically important species (c) *Acanthopagrus australis* and (d) *Metapenaeus bennettae* caught in 5-, 10- and 20-min tows of a beam trawl during the day and night in Tuggerah Lake (NSW). Data are pooled across sites and sampling periods. Standard errors are shown.

periods across all sites (Table 5). These results were supported by the nMDS ordinations, which showed clear differences between assemblages sampled during the day and night (Fig. 3). Most of the species that contributed to differences in the abundance of assemblages were caught in greater proportions at night and included *Gerres subfasciatus*, *Metapenaeus macleayi*, *Penaeus plebejus* and *Metapenaeus bennettae* (SIMPER). Patterns of abundance of some species were inconsistent among sites (e.g. *Acanthopagrus australis* and *Rhabdosargus sarba*; SIMPER).

Differences in the composition of assemblages (presence/absence) between diel periods were driven by a larger number of species, which were also predominately caught at night (SIMPER). This included common taxa (e.g. *Penaeus plebejus* and *Metapenaeus bennettae*), as well as species that were caught infrequently or in small numbers (e.g. *Platycephalus fuscus*, *Callionymidae* spp., *Pelates quadrilineatus*, *Pomatomus saltatrix*, *Sillago maculata* and *Portunus pelagicus*).

4. Discussion

Pilot experiments are essential for designing more reliable and cost-efficient sampling strategies (Andrew and Mapstone, 1987). In this study, the testing of hypotheses about the effects of diel period and tow duration on samples of estuarine fauna in a beam trawl has: (1) provided new data on the effects of these factors on assemblages and size ranges of fauna sampled over bare sediment; and (2) led to the development of an appropriate methodology for future sampling with this gear.

In many parts of the world, more organisms and species are often caught in estuarine seagrass beds at night than during the day (Greening and Livingston, 1982; Roblee and Zieman, 1984; Guest et al., 2003; Unsworth et al., 2007). Although fewer comparable studies have been done over bare sediments in estuaries, similar diel patterns have also been reported (e.g. Gray et al., 1998; Miller and Skilleter, 2006). For example, in northern NSW (Australia), Gray et al. (1998) found that several species and sizes of fish were caught mainly at night over bare sand. Similarly, in a subtropical estuary in Queensland (Australia), Miller and Skilleter (2006) observed a larger diversity and abundance of nekton over bare sediment at night, but gave no information on the sizes of fish caught. Our results are consistent with previous studies, as differences between diel periods were driven by taxa that were caught in larger numbers, proportions and sizes at night.

Factors explaining the differences between diel periods in our experiment are equivocal, because we did not test any hypotheses related to them. The increased activity of many

Table 3
Analyses of the effects of site (Si; random), diel period (Di; fixed) and tow duration (To; fixed) on estimates of mean precision (variances) for abundant economically important taxa (*Gerres subfasciatus*, *Acanthopagrus australis*, *Rhabdosargus sarba*, *Metapenaeus macleayi*, *Penaeus plebejus* and *Metapenaeus bennettae*) caught in a beam trawl in Tuggerah Lake (NSW). Data from each sampling period provided replicate estimates of variance. Diel period was not included as a factor in the analyses for *P. plebejus* and *M. bennettae*, because these species were only caught in very small numbers during the day (<10), and so only data from the night were analysed. Results of SNK tests are shown. ns, not significant; $^{*}P < 0.05$; $^{**}P < 0.01$; $^{***}P < 0.001$

Source		MS	*F*	*P*	MS	*F*	*P*
		(a) *Gerres subfasciatus*			(b) *Acanthopagrus australis*		
Si	2	$14.70^{(46)}$	7.38	**	$1.12^{(44)}$	3.09	ns
Di	1	$115.09^{(46)}$	57.80	***	0.005	0.00	ns
To	2	$0.54^{(46)}$	0.27	ns	$0.77^{(44)}$	2.13	ns
Si × Di	2	0.77			$2.12^{(44)}$	5.85	**
Si × To	4	1.25			0.06		
Di × To	2	$0.18^{(46)}$	0.09	ns	$0.11^{(44)}$	0.29	ns
Si × Di × To	4	1.24			0.43		
Residual	36	2.22			0.39		
SNK		Di: D<N					
		(c) *Rhabdosargus sarba*			(d) *Metapenaeus macleayi*		
Si	2	$60.64^{(46)}$	0.74	ns	$3.73^{(44)}$	25.81	***
Di	1	$219.81^{(46)}$	2.68	ns	11.04	14.09	ns
To	2	$50.83^{(46)}$	0.62	ns	$0.24^{(44)}$	1.65	ns
Si × Di	2	56.05			$0.78^{(44)}$	5.42	**
Si × To	4	55.78	0.26	ns	0.07		
Di × To	2	$21.46^{(46)}$			$0.01^{(44)}$	0.09	ns
Si × Di × To	4	75.48			0.04		
Residual	36	87.20			0.16		
SNK							
		(e) *Penaeus plebejus*			(f) *Metapenaeus bennettae*		
Si	2	$0.00^{(22)}$	0.17	ns	$0.30^{(22)}$	3.46	ns
To	2	$0.01^{(22)}$	2.43	ns	$0.02^{(22)}$	0.19	ns
Si × To	4	0.004			0.04		
Residual	18	0.005			0.10		
SNK							

species of fish and invertebrates at night is, however, well known and often related to foraging activity and the avoidance of predators (e.g. Helfman, 1981; Gibson et al., 1998). Gray et al. (1998) hypothesised that differences between diel periods over bare sand were the result of larger individuals of some species of fish (e.g. *Gerres subfasciatus*) moving from deeper habitats in the day, to shallow sand at night to feed. We also found greater numbers and proportions of larger individuals of *G. subfasciatus* and other species at night. This included sizes (>100 mm) of economically important finfish (e.g. *Acanthopagrus australis*, *Rhabdosargus sarba*, *Platycephalus fuscus*, *Sillago maculata*), which were not caught in the seine net used by Gray et al. (1998).

An improvement in the detection and subsequent avoidance of the beam trawl by fishes during the day (Stoner, 1991; Petrakis et al., 2001), may also explain the diel patterns observed here. This hypothesis would, however, be unlikely for most penaeid prawns (e.g. *Penaeus plebejus*, *Metapenaeus bennettae*, *Metapenaeus macleayi*), because these and other species of this taxa have a limited ability to avoid approaching

Table 4
Results of Kolmogorov–Smirnov tests comparing size–frequency distributions of abundant, economically important species caught with a beam trawl in Tuggerah Lake (NSW) between diel periods (day vs. night; separately for each tow time) and tow durations (5, 10 and 20 min; separately for each diel period). na, not analysed ($n = <50$); ns, not significant; $^{*}P < 0.05$; $^{**}P < 0.01$, $^{***}P < 0.001$

Species	Diel period			Tow duration					
	5 min	10 min	20 min	Day			Night		
	D vs. N	D vs. N	D vs. N	5 vs. 10	5 vs. 20	10 vs. 20	5 vs. 10	5 vs. 20	10 vs. 20
Gerres subfasciatus	**	ns	ns	*	ns	ns	ns	ns	*
Acanthopagrus australis	*	ns	ns	ns	ns	ns	***	**	*
Rhabdosargus sarba	**	*	ns	ns	ns	ns	ns	**	ns
Sillago maculata	na	na	na	na	na	na	na	ns	ns
Metapenaeus macleayi	na	ns	ns	ns	ns	ns	ns	ns	ns
Penaeus plebejus	na	na	na	na	na	na	ns	ns	ns
Metapenaeus bennettae	na	na	na	na	na	na	ns	ns	*

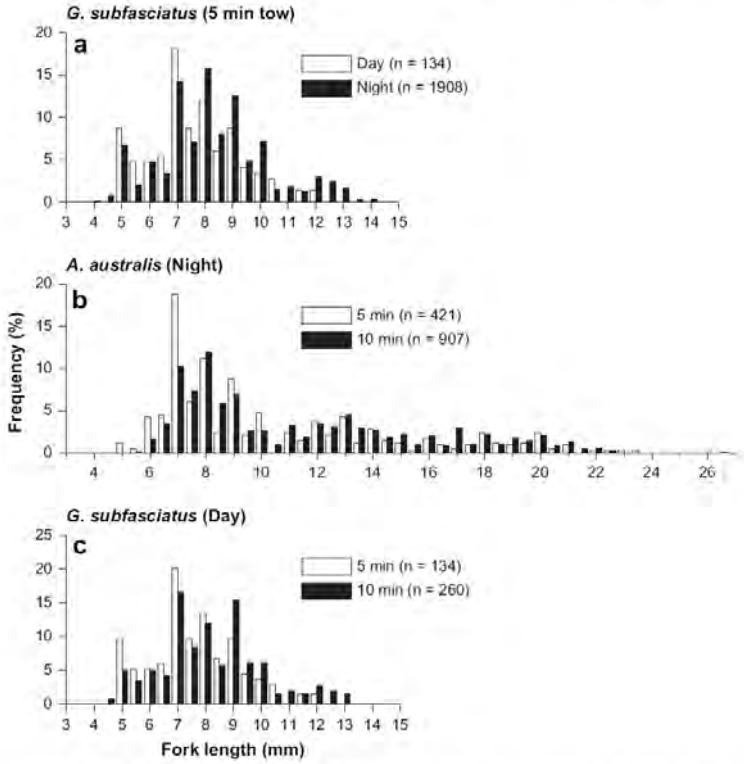

Fig. 2. Size–frequency distributions of abundant, economically important species caught between different diel periods and tow durations in a beam trawl in Tuggerah Lake (NSW): (a) *Gerres subfasciatus* caught between the day and night in 5-min tows; (b) *Acanthopagrus australis* caught between 5- and 10-min tows during the night; and (c) *G. subfasciatus* caught between 5- and 10-min tows during the day. Data are pooled across sites and sampling periods and presented as examples of significant differences detected in Kolmogorov–Smirnov tests (Table 4).

Table 5
Comparisons of assemblages of fish and invertebrates between diel periods (Diel; fixed) and among sampling periods (Period; nested within Diel) at three sites in Tuggerah Lake using permutational multivariate analysis of variance (PERMANOVA) done on: (a) standardised, abundance data; and (b) presence/absence data. $^{**}P < 0.01$; $^{***}P < 0.001$. For each test, 4999 permutations were used. When the number of unique permutations was low (e.g. 10 out of 4999), Monte Carlo (mc) *P*-values were used instead of permutation (perm) *P*-values (see Anderson, 2005)

Source	df	Standardised			Presence/absence		
		MS	*F*	*P*	MS	*F*	*P*
Site 1							
Diel	1	40819	12.8	** (mc)	21677	15.0	*** (mc)
Period (Di)	4	3178	6.6	*** (perm)	1441	3.4	*** (perm)
Residual	66	484			417		
	71						
Site 2							
Diel	1	57852	30.2	*** (mc)	24361	23.5	*** (mc)
Period (Di)	4	1915	6.4	*** (perm)	1038	2.5	** (perm)
Residual	66	297			407		
	71						
Site 3							
Diel	1	75288	13.8	*** (mc)	45175	16.5	*** (mc)
Period (Di)	4	5437	5.4	*** (perm)	2744	2.9	*** (perm)
Residual	66	1001			942		
	71						

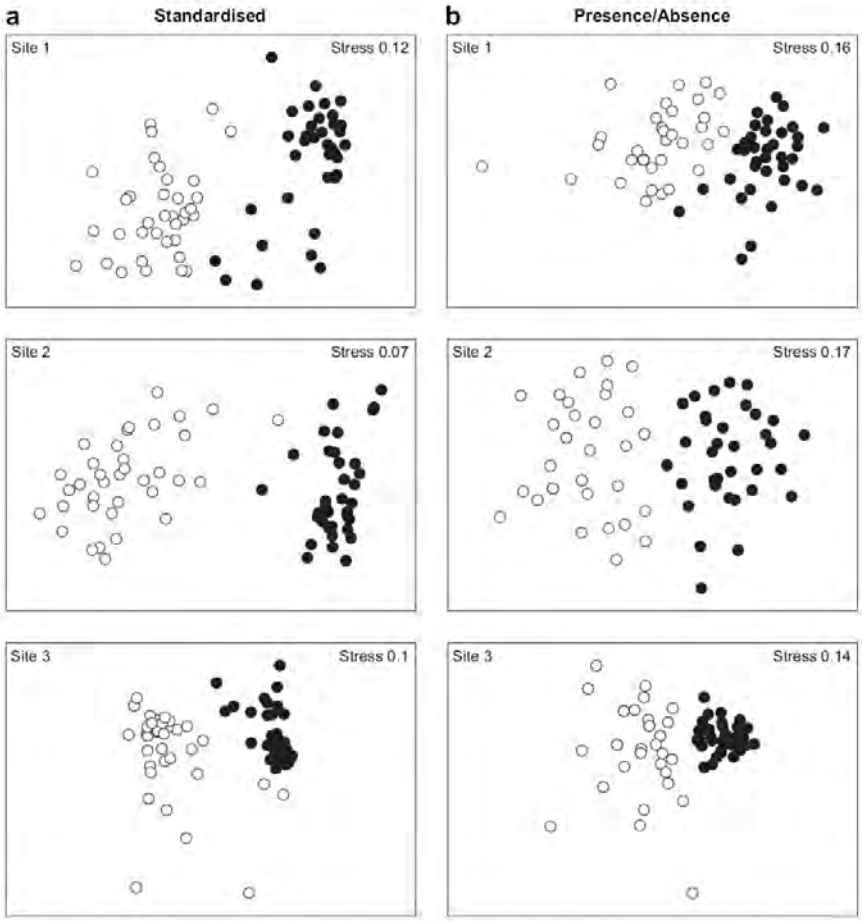

Fig. 3. Two-dimensional nMDS ordination plots of (a) standardised data; and (b) presence/absence data showing relationships between assemblages of estuarine fauna caught using a beam trawl during the day (open circles) and night (closed circles) at three sites in Tuggerah Lake (NSW). Data are pooled across sites and sampling periods.

trawls (Broadhurst, 2000). Moreover, these species generally bury in substrates during the day and emerge at night (Allen, 1966; Greening and Livingston, 1982; Vance, 1992).

Regardless of the underling processes that caused the observed patterns, we conclude that future sampling with the beam trawl should be done at night. This decision also takes into account the precision of CPUE estimates, which were similar between diel periods for most variables. Bare sediments are often perceived as being less valuable than other habitats, such as seagrass, and this may be partly because sampling is often done only during the day (Gray et al., 1998). Nevertheless, the diversity of fauna caught in this experiment, which was typical of the taxa caught in trawls in estuaries of NSW (Gray et al., 1990; Liggins and Kennelly, 1996), confirms the importance of bare sediment for a wide size and species range of fish and invertebrates (Gray et al., 1998).

Sampling a day and night within the same 24-h period is common in previous studies examining the effects of diel period on patterns of abundance of aquatic organisms (e.g. Greening and Livingston, 1982; Gray et al., 1998; Griffiths, 2001; Morrison et al., 2002; Ribeiro et al., 2006). Nevertheless, designing experiments in this way has serious implications for the independence of data (Underwood, 1997). For example, weather conditions (or other unknown factors) during the day may affect the distribution and abundance of fauna during subsequent sampling at night (and vice versa), potentially causing samples among treatments to be either positively or negatively correlated (Underwood, 1997). Therefore, any tests of the effects of diel period on organisms would be done on non-independent data, which increases the risk of either Type I (positive correlation) or Type II (negative correlation) errors (see Underwood, 1997 for a detailed discussion).

To avoid non-independence of data, future studies examining the effects of diel period on fauna should ensure that replicate days and nights are sampled at random within a given study period (sensu Underwood, 1997), as done here.

Despite the hypothesis that longer tows result in larger catch rates owing to catch-by-exhaustion (i.e. fish herding in front of a trawl eventually tire and fall back into the codend; Wardle, 1986), previous studies have generally found that shorter tows are more efficient at sampling fish and invertebrates than longer tows (Godo et al., 1990; Somerton et al., 2002). These results have been explained by: (1) the catch-by-surprise hypothesis (i.e. fish are more vulnerable to capture in the first few minutes of a tow because they are not alerted by other fish herding in front of the trawl; Godo et al., 1990); (2) errors in the measurement of the tow path (Godo et al., 1990); and (3) the escapement of organisms under the footrope (Somerton et al., 2002). In our study, tow duration did not affect catch rates of any individual species.

Longer tows often catch more species because a greater proportion of the substrate is sampled (e.g. Carothers and Chittendon, 1985; Kennelly et al., 1993). In our study, longer tows did catch more species, but only few and in very small numbers (<5 individuals). Consequently, the mean CPUE of the number of species (5 min^{-1}) decreased significantly as the duration of tows increased; indicating that most species were caught in 5-min tows. These results were supported by the multivariate analyses, which did not detect any differences in the structure and composition of assemblages among tow durations. Thus, in future studies, sampling with additional 5-min tows would be more efficient than sampling with 10- or 20-min tows. The benefit of increasing the number of replicate tows is that means can be estimated more precisely (e.g. Pennington and Vølstad, 1991).

Larger fish generally have a greater capacity to swim in front of a trawl for longer periods of time, than smaller fish (Wardle, 1986). Consequently, shorter tows are expected to underestimate the proportion of larger fish in a population (Godo et al., 1990). Despite this hypothesis, most of the previous studies investigating the effects of tow duration have found no differences in the sizes of organisms retained (Godo et al., 1990; Somerton et al., 2002; Wieland and Storr-Paulsen, 2006). Similarly, we found no clear or consistent patterns relating to differences in the size ranges of organisms caught among tow durations. These results further support the use of 5-min tows.

In conclusion, future sampling with our beam trawl should be done at night using 5-min tows. This strategy may not be appropriate across all types of estuaries in NSW. Nevertheless, the consistency of our results with previous studies in other estuaries, systems and countries, provides some confidence that our results can be generalised.

Shorter tows provide greater scope for increasing levels of replication and precision in surveys, without large increases in costs (Pennington and Vølstad, 1991). Determining levels of replication that are both optimal and reliable, however, requires additional experiments measuring spatial and temporal variation, across a range of different scales (e.g. Morrisey et al., 1992a,b; Kennelly et al., 1993; Rotherham et al., 2007).

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Appendix 12: Summary of experiment 8: Spatial and temporal variation of fish fauna sampled with a beam trawl.

Summary of experiment 8: Spatial and temporal variation of fish fauna sampled with a beam trawl

Aim

To investigate spatial and temporal patterns of variation in the abundance of estuarine fish and crustaceans sampled with an experimental beam trawl. The hypotheses tested were that: (1) most spatial variation in abundance is at the smallest scale (replicate tows); (2) abundances of fish and crustaceans vary from night to night, week to week, month to month and season to season; and (3) patterns of spatial and temporal variation are consistent between estuaries.

Methods

Design of experiment and gear

The experiment was done in two estuaries (Tuggerah Lake and Lake Macquarie). These estuaries we selected because they are relatively close together; allowing sampling to be done during the same time periods. Sampling was done at three sites (separated by kilometres) within each estuary. Two nights were sampled within each of two weeks, within each of two months, within each of two seasons (see Fig. 1). Nights and weeks were sampled at random within each month and season to remove any potential effects of moon phase and to minimise any possible non-independence among nights and weeks. Given that there are generally no sites deeper than 4 m in Tuggerah Lake, sampling was not stratified into deep and shallow. Sites deeper than 4m exist in Lake Macquarie, but we only sampled areas < 4 m in depth to facilitate comparisons with Tuggerah Lake.

Sampling was done at night using a beam trawl (horizontal opening of 3m) configured with 41-mm diamond-shaped mesh in the body and 20-mm mesh hung on the bar (i.e., squared shaped; bar length of 10mm) in the codend (see Appendix 10). On each night of sampling, 6 replicate tows of the beam trawl (5 min duration; see appendix x) were done at each site for a total of 18 trawls. The beam trawl was towed at speeds of about 1.2 ms^{-1}.

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), crabs (carapace width to the nearest mm) and prawns (carapace length – CL to the nearest mm).

Analyses of data

For each estuary separately, we analysed variation among sites and replicates for each night of sampling using nested ANOVA and then calculated the components of variation (Underwood, 1997). For species that were found in more than 25% of samples on each night, this gave 16 independent estimates of variation among these spatial scales (i.e., 2 nights x 2 weeks x 2 months x 2 seasons = 16 times of sampling). Components of variation for spatial scale were then averaged across the times of sampling.

Variation among nights, weeks, months and seasons was also analysed for frequently abundant species (occurring in more than 25% of samples) using nested ANOVA. Components of variation calculated from mean square estimates. This was done separately for each site and estuary, providing 3 independent estimates of the components of variation, which were then averaged across sites. Negative components of variation were removed using pooling procedures (Fletcher and Underwood 2002).

Multivariate components of variation were also calculated for each of the different spatial and temporal scales using permutational multivariate analysis of variance (PERMANOVA; Anderson, 2001, 2005; Anderson and Millar, 2004).

Main results

A total of 46 113 fish and invertebrates from more than 45 species were caught in the experiment. More individuals and species were caught in Tuggerah Lake (31 032 individuals from more than 40 species) than in Lake Macquarie (15 081 individuals from 32 species). More than 27 species were caught in both estuaries; 13 species were caught only in Tuggerah Lake and 5 species only in Lake Macquarie.

Spatial variation

In Tuggerah Lake, for most species there was a general pattern of more variation among replicates than among sites (Table 1). There were, however, no general patterns of variation in Lake Macquarie; variation was greater among sites for some species and among replicates for others (Table 2). These results suggest that the scales of sites and replicates need to be incorporated into future sampling because patterns of variation are not consistent even between estuaries that are close together.

Temporal variation

Patterns of temporal variation were not consistent between estuaries. In Tuggerah Lake, there was generally more variation among nights and weeks for most species Table 3). In Lake Macquarie, however, variation was often greater among seasons than among months, weeks or nights (Table 4). For most species across both estuaries, temporal variance was generally small compared to spatial variation among replicates (residual variance).

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Table 1. Total numbers of individuals of the 20 most abundant species sampled in a beam trawl in Tuggerah Lake and Lake Macquarie. % = percent contribution to the total numbers.

Tuggerah Lake	**No.**	**%**	**Lake Macquarie**	**No.**	**%**
Gerres subfasciatus	12 538	40.40	*Metapenaeus bennettae*	5319	35.27
Metapenaeus bennettae	5292	17.05	*Gerres subfasciatus*	3923	26.01
Penaeus plebejus	2740	8.83	*Ambassid* spp.	2754	18.26
Pseudorhombus jenynsii	2432	7.84	*Metapenaeus macleayi*	1125	7.46
Acanthopagrus australis	1911	6.16	*Sillago maculata*	603	4.00
Pelates sexlineatus	1808	5.83	*Apogon* spp.	554	3.67
Metapenaeus macleayi	1568	5.05	*Pelates sexlineatus*	240	1.59
Rhabdosargus sarba	794	2.56	*Acanthopagrus australis*	135	0.90
Sillago maculata	693	2.23	*Loligo* spp.	87	0.58
Ambassid spp.	303	0.98	*Pomatomus saltatrix*	56	0.37
Loligo spp.	239	0.77	*Pseudorhombus arsius*	47	0.31
Herklotsichthys castelnaui	106	0.34	*Hyperlophus vittatus*	36	0.24
Synaptura nigra	100	0.32	*Siganus fuscescens*	34	0.23
Platycephalus fuscus	73	0.24	*Herklotsichthys castelnaui*	33	0.22
Hyporhamphus regularis	65	0.21	*Portunus pelagicus*	27	0.18
Portunus pelagicus	60	0.19	*Penaeus plebejus*	25	0.17
Centropogon australis	43	0.14	*Pagrus auratus*	18	0.12
Pomatomus saltatrix	42	0.14	*Monacanthus. chinensis*	13	0.09
glassy	40	0.13	*Pseudorhombus jenynsii*	9	0.06
Cnidoglanis macrocephalus	34	0.11	*Atherinosoma microstoma*	7	0.05
All other 20 spp.	151	0.48	All other 12 spp.	36	0.22
Total	31032	100	Total	15081	100

Table 2. Components of spatial variation calculated from mean squares in ANOVA for abundant species in Tuggerah Lake. Spatial scales are Sites and replicates (Res).

A. australis	Sites	9.02	*A. jacksoniensis*	Sites	1.06	*C. macrocephalus*	Sites	0.14
	Res	10.93		Res	3.56		Res	0.67
G. subfasciatus	Sites	310.11	*H. castelnaui*	Sites	0.45	*H. regularis*	Sites	0.02
	Res	364.47		Res	1.00		Res	0.38
Loligo spp.	Sites	0.34	*M. bennettae*	Sites	154.37	*M. macleayi*	Sites	8.97
	Res	2.22		Res	152.64		Res	14.44
P. fuscus	Sites	0.00	*P. jenynsii*	Sites	50.95	*P. pelagicus*	Sites	0.03
	Res	0.40		Res	34.98		Res	0.52
P. plebejus	Sites	22.44	*P. saltatrix*	Sites	0.08	*P. sexlineatus*	Sites	17.54
	Res	36.81		Res	0.43		Res	12.77
R. sarba	Sites	2.13	*S. maculata*	Sites	1.35	*S. nigra*	Sites	0.02
	Res	3.60		Res	2.88		Res	0.47
Multivariate	Sites	388.12						
	Res	478.26						

Table 3. Components of spatial variation calculated from mean squares in ANOVA for abundant species in Lake Macquarie. Spatial scales are sites and replicates (Res).

A. australis	Sites	0.16	*A. jacksoniensis*	Sites	102.67	*Apogon* spp.	Sites	13.03
	Res	0.51		Res	56.61		Res	11.29
G. subfasciatus	Sites	411.33	*H. vittatus*	Sites	0.00	*Loligo* spp.	Sites	0.07
	Res	241.61		Res	3.52		Res	1.22
M. bennettae	Sites	513.99	*M. macleayi*	Sites	479.00	*P. arsius*	Sites	0.03
	Res	280.39		Res	332.30		Res	0.29
P. auratus	Sites	0.53	*P. pelagicus*	Sites	0.13	*P. sexlineatus*	Sites	1.11
	Res	0.36		Res	0.12		Res	9.45
S. fuscescens	Sites	3.38	*S. maculata*	Sites	1.92	Multivariate	Sites	1032.04
	Res	5.12		Res	3.02		Res	1231.49

Table 4. Components of temporal variation calculated from mean squares in ANOVA for abundant species caught in Tuggerah Lake. Temporal scales are seasons (S), months (M) nested in seasons, weeks (W) nested in months and seasons, and nights (N) nested in weeks, months and seasons.

A. australis	S	1.06	*A. jacksoniensis*	S	0.96	*G. subfasciatus*	S	55.73
	M (S)	0.00		M (S)	0.14		M (S)	55.72
	W (M, S)	2.02		W (M, S)	0.57		W (M, S)	10.54
	N (W, M, S)	1.94		N (W, M, S)	0.11		N (W, M, S)	285.16
	Res	10.93		Res	2.28		Res	359.22
H. castelnaui	S	0.01	*H. regularis*	S	0.02	*Loligo* spp.	S	0.28
	M (S)	0.00		M (S)	0.02		M (S)	0.01
	W (M, S)	0.04		W (M, S)	0.00		W (M, S)	0.01
	N (W, M, S)	0.05		N (W, M, S)	0.26		N (W, M, S)	0.16
	Res	0.56		Res	0.19		Res	1.76
M. bennettae	S	11.08	*M. macleayi*	S	1.40	*P. fuscus*	S	0.00
	M (S)	49.80		M (S)	0.40		M (S)	0.00
	W (M, S)	159.06		W (M, S)	5.28		W (M, S)	0.05
	N (W, M, S)	47.57		N (W, M, S)	5.91		N (W, M, S)	0.00
	Res	152.64		Res	14.44		Res	0.31
P. jenynsii	S	12.11	*P. plebejus*	S	4.10	*R. sarba*	S	0.00
	M (S)	25.96		M (S)	7.38		M (S)	0.02
	W (M, S)	6.35		W (M, S)	10.51		W (M, S)	1.04
	N (W, M, S)	24.94		N (W, M, S)	14.21		N (W, M, S)	0.26
	Res	34.98		Res	35.43		Res	3.57
S. nigra	S	0.01	Multivariate	S	70.95			
	M (S)	0.00		M (S)	171.95			
	W (M, S)	0.02		W (M, S)	163.01			
	N (W, M, S)	0.00		N (W, M, S)	220.83			
	Res	0.44		Res	478.26			

Table 5. Components of temporal variation calculated from mean squares in ANOVA for abundant species caught in Lake Macquarie. Temporal scales are seasons (S), months (M) nested in seasons, weeks (W) nested in months and seasons, and nights (N) nested in weeks, months and seasons.

A. australis	S	0.00	*A. jacksoniensis*	S	103.11	*G. subfasciatus*	S	119.25
	M (S)	0.04		M (S)	15.31		M (S)	181.61
	W (M, S)	0.00		W (M, S)	6.70		W (M, S)	37.86
	N (W, M, S)	0.05		N (W, M, S)	35.18		N (W, M, S)	78.40
	Res	0.62		Res	56.22		Res	241.61
M. bennettae	S	298.54	*Apogon* spp.	S	13.12	*P. saltatrix*	S	0.00
	M (S)	0.00		M (S)	0.00		M (S)	0.00
	W (M, S)	85.77		W (M, S)	5.18		W (M, S)	0.00
	N (W, M, S)	91.33		N (W, M, S)	0.51		N (W, M, S)	0.09
	Res	280.39		Res	8.48		Res	0.52
S. maculata	S	0.56	*P. sexlineatus*	S	0.46	Multivariate	S	793.66
	M (S)	0.08		M (S)	0.00		M (S)	137.10
	W (M, S)	0.01		W (M, S)	0.29		W (M, S)	192.54
	N (W, M, S)	0.80		N (W, M, S)	0.27		N (W, M, S)	319.10
	Res	3.02		Res	8.18		Res	1234.97

Appendix 13: Summaries of the spatial cost-benefit analyses for multi-mesh gill nets and beam trawl.

Table 1. Summary of cost-benefit analyses for abundant species caught in multi-mesh gill nets in Lake Macquarie. θ^2_e = variance due to replicate nets; θ^2_S = variance due to sites; cost of a deep replicate = 40 min; cost of a shallow replicate = 50 min; cost of a site = 480 min; limiting cost = 2400 min (5 nights, 1 site per night). N = optimal number of replicate gill nets; B = optimal number of replicate sites; $B_{10\%}$ = number of sites required to detect changes of 10% of the mean; $B_{30\%}$ = number of sites required to detect changes of 30% of the mean. See Underwood (1997) for cost-benefit formulae. Actual replication = 6 replicate gill nets at each of 6 replicate sites.

Lake Macquarie

Species	θ^2_e	θ^2_S	N	B	Mean	$B_{10\%}$	$B_{30\%}$	Estimated SE using optimal replication (% of mean)	Estimated SE using actual replication (% of mean)
Deep									
Gerres subfasciatus (Silver biddy)	74.99	21.27	6.50	3.24	8.03	50.90	5.66	39.62	29.55
Acanthopagrus australis (Yellowfin bream)	4.73	0.03	43.50	1.08	1.61	5.35	0.59	22.24	22.92
Platycephalus fuscus (Dusky flathead)	0.44	0.23	4.80	3.57	0.39	214.29	23.81	77.46	58.03
Liza argentea (Flat-tail mullet)	2.19	0.85	5.57	3.41	0.67	278.82	30.98	90.36	67.39
Pomatomus saltarix (Tailor)	5.29	2.48	5.06	3.52	1.89	98.81	10.98	53.00	39.63
Rhabdosargus sarba (Tarwhine)	0.71	0.00	67.58	0.75	0.25	19.67	2.19	51.08	56.44
Sillago maculata (Trumpeter whiting)	5.72	0.88	8.83	2.88	2.39	26.74	2.97	30.47	23.13
Multivariate	1863.00	477.00	6.85	3.18					
Shallow									
Gerres subfasciatus (Silver biddy)	60.68	36.28	4.01	3.53	8.86	65.49	7.28	43.09	31.38
Mugil cephalus (Sea mullet)	29.71	21.05	3.68	3.61	3.67	216.60	24.07	77.42	56.77
Platycephalus fuscus (Dusky flathead)	2.81	0.12	14.99	1.95	1.67	11.07	1.23	23.81	18.79
Liza argentea (Flat-tail mullet)	37.22	26.75	3.65	3.62	6.03	101.65	11.29	52.98	38.88
Sillago ciliata (Sand mullet)	1.86	0.84	4.61	3.38	1.14	95.86	10.65	53.27	38.44
Pomatomus saltarix (Tailor)	3.33	3.34	3.09	3.78	2.08	102.08	11.34	51.96	38.74
Rhabdosargus sarba (Tarwhine)	7.52	10.06	2.68	3.91	2.53	201.02	22.34	71.71	54.27
Multivariate	1542.00	1046.00	3.76	3.59					

Table 2. Summary of cost-benefit analyses for abundant species caught in multi-mesh gill nets in St Georges Basin. θ^2_e = variance due to replicate nets; θ^2_S = variance due to sites; cost of a deep replicate = 40 min; cost of a shallow replicate = 50 min; cost of a site = 480 min; limiting cost = 2400 min (5 nights, 1 site per night). N = optimal number of replicate gill nets; B = optimal number of replicate sites; $B_{10\%}$ = number of replicate sites required to detect changes of 10% of the mean; $B_{30\%}$ = number of sites required to detect changes of 30% of the mean. See Underwood (1997) for cost-benefit formulae. Actual replication = 6 replicate gill nets at each of 6 replicate sites.

St Georges Basin

Species	θ^2_e	θ^2_S	N	B	Mean	$B_{10\%}$	$B_{30\%}$	Estimated SE using optimal replication (% of mean)	Estimated SE using actual replication (% of mean)
Deep									
Gerres subfasciatus (Silver biddy)	12.04	5.74	5.02	3.53	3.06	87.19	9.69	49.73	37.19
Acanthopagrus australis (Yellowfin bream)	11.83	11.12	3.57	3.85	6.06	39.36	4.37	31.96	24.39
Liza argentea (Flat-tail mullet)	0.37	0.12	6.08	3.32	0.39	119.57	13.29	60.03	44.74
Girella tricuspidata (Luderick)	4.14	1.69	5.42	3.44	1.06	220.21	24.47	79.96	59.67
Sillago ciliata (Sand mullet)	7.24	1.34	8.04	2.99	3.94	14.43	1.60	21.96	16.53
Pseudocaranx dentex (Silver trevally)	1.21	0.82	4.21	3.70	0.72	212.36	23.60	75.74	57.14
Pomatomus saltatrix (Tailor)	11.56	2.12	8.09	2.99	3.67	26.40	2.93	29.73	22.40
Rhabdosargus sarba (Tarwhine)	2.79	1.62	4.55	3.63	1.78	70.67	7.85	44.15	33.16
Sillago maculata (Trumpeter whiting)	0.46	0.10	7.43	3.09	0.50	64.77	7.20	45.80	34.32
Multivariate	1118.00	347.00	6.22	3.29					
Shallow									
Gerres subfasciatus (Silver biddy)	59.99	23.95	4.90	3.31	6.22	93.46	10.38	53.14	38.23
Mugil cephalus (Sea mullet)	73.39	84.29	2.89	3.84	8.50	151.80	16.87	62.85	47.19
Liza argentea (Flat-tail mullet)	146.88	124.67	3.36	3.70	10.03	167.41	18.60	67.24	49.72
Girella tricuspidata (Luderick)	6.07	3.97	3.83	3.57	3.81	38.36	4.26	32.76	23.95
Sillago ciliata (Sand mullet)	86.59	141.72	2.42	3.99	16.78	63.05	7.01	39.74	30.41
Pseudocaranx dentex (Silver trevally)	6.32	2.59	4.84	3.32	2.17	83.01	9.22	49.97	35.97
Pomatomus saltatrix (Tailor)	6.61	2.38	5.16	3.25	2.44	61.33	6.81	43.43	31.18
Sillago maculata (Trumpeter whiting)	1.79	0.78	4.69	3.36	0.94	130.20	14.47	62.27	44.89
Multivariate	1079.00	901.00	3.39	3.69					

Table 3. Summary of cost-benefit analyses for abundant species caught in the beam trawl in Tuggerah Lake. θ^2_e = variance due to replicate trawls; θ^2_S = variance due to sites; cost of a replicate trawl = 10 min; cost of a site = 45 min; limiting cost = 480 min; N = optimal number of replicate trawls; B = optimal number of replicate sites; $B_{10\%}$ = number of replicate sites required to detect changes of 10% of the mean; $B_{30\%}$ = number of sites required to detect changes of 30% of the mean. See Underwood (1997) for cost-benefit formulae. Actual replication = 6 replicate trawls at each of 3 replicate sites.

Tuggerah Lake

Species	θ^2_e	θ^2_S	N	B	Mean	$B_{10\%}$	$B_{30\%}$	Estimated SE using optimal replication (% of mean)	Estimated SE using actual replication (% of mean)
Ambassid spp.	3.56	1.06	3.89	5.72	37.56	0.14	0.02	1.56	1.98
Gerres subfasciatus (Silver biddy)	364.47	310.11	2.30	7.06	43.53	24.72	2.75	18.71	25.54
Acanthopagrus australis (Yellowfin bream)	10.93	9.02	2.33	7.02	6.64	31.13	3.46	21.05	28.66
Synaptura nigra (Black sole)	0.47	0.02	10.13	3.28	0.35	56.05	6.23	41.33	52.53
Platycephalus fuscus (Dusky flathead)	0.40	0.00	22.04	1.81	0.25	34.02	3.78	43.37	60.42
Cnidoglanis macrocephalus (Estuary catfish)	0.67	0.14	4.61	5.27	0.12	2070.37	230.04	198.18	246.78
Metapenaeus bennettae (Greasyback prawn)	152.64	154.37	2.11	7.26	18.38	67.15	7.46	30.41	42.13
Herklotsichthys castelnaui (Southern herring)	1.00	0.45	3.16	6.26	0.37	565.37	62.82	95.01	123.16
Melicertus plebejus (Eastern king prawn)	36.81	22.44	2.72	6.65	9.51	39.76	4.42	24.45	32.44
Hyporhamphus regularis (river garfish)	0.38	0.02	9.58	3.41	0.23	113.77	12.64	57.77	73.02
Metapenaeus macleayi (school prawn)	14.44	8.97	2.69	6.67	5.44	48.34	5.37	26.91	35.76
Loligo spp. (Squid)	2.22	0.34	5.45	4.82	0.83	108.02	12.00	47.32	58.48
Pseudorhombus jenynsii (small-toothed flounder)	34.98	50.95	1.76	7.67	8.44	99.35	11.04	35.99	51.52
Portunus pelagicus (Blue swimmer crab)	0.52	0.03	9.00	3.56	0.21	199.62	22.18	74.92	94.17
Pomatomus saltatrix (Tailor)	0.43	0.08	5.07	5.02	0.15	759.10	84.34	123.01	152.38
Rhabdosargus sarba (Tarwhine)	3.60	2.13	2.76	6.62	2.76	45.25	5.03	26.15	34.62
Sillago maculata (Trumpeter whiting)	2.88	1.35	3.09	6.32	2.41	39.44	4.38	24.98	32.48
Pelates sexlineatus (Striped trumpeter)	12.77	17.54	1.81	7.61	6.28	62.40	6.93	28.64	40.78
Multivariate	1863.00	477.00	4.19	5.52					

Table 4. Summary of cost-benefit analyses for abundant species caught in the beam trawl in Lake Macquarie. θ^2_e = variance due to replicate trawls; θ^2_S = variance due to sites; cost of a replicate trawl = 45 min; cost of a site = 45 min; limiting cost = 480 min; N = optimal number of replicate trawls; B = optimal number of replicate sites; $B_{10\%}$ = number of replicate sites required to detect changes of 10% of the mean; $B_{30\%}$ = number of sites required to detect changes of 30% of the mean. See Underwood (1997) for cost-benefit formulae. Actual replication = 6 replicate trawls at each of 3 replicate sites.

Lake Macquarie

Species	θ^2_e	θ^2_S	**N**	**B**	**Mean**	$B_{10\%}$	$B_{30\%}$	**Estimated SE using optimal replication (% of mean)**	**Estimated SE using actual replication (% of mean)**
Ambassid spp.	56.61	102.67	1.58	7.90	9.56	0.91	8.23	43.80	63.93
Gerres subfasciatus (Silver biddy)	241.61	411.33	1.63	7.84	13.62	1.86	16.70	62.06	90.07
Acanthopagrus australis (Yellowfin bream)	0.51	0.16	3.73	5.83	0.47	0.00	0.02	48.39	61.41
Synaptura nigra (black sole)	280.39	513.99	1.57	7.91	18.47	3.41	30.70	50.67	74.02
Siganus fuscescens (Black trevally)	5.12	3.38	2.61	6.75	0.12	0.00	0.00	753.37	1006.00
Pseudorhombus arsius (Large-toothed flounder)	0.29	0.03	7.08	4.14	0.16	0.00	0.00	77.75	96.29
Apogon spp. (Cardinal fish)	11.29	13.03	1.97	7.41	1.92	0.04	0.33	82.68	115.92
Pagrus auratus (snapper)	0.36	0.53	1.73	7.70	0.06	0.00	0.00	495.49	711.12
Metapenaeus macleayi (school prawn)	332.30	479.00	1.77	7.66	3.91	0.15	1.37	238.91	341.67
Loligo spp. (Squid)	1.22	0.07	9.00	3.56	0.30	0.00	0.01	79.24	99.61
Portunus pelagicus (Blue swimmer crab)	0.12	0.13	2.07	7.30	0.09	0.00	0.00	170.58	237.04
Sillago maculata (Trumpeter whiting)	3.02	1.92	2.66	6.71	2.09	0.04	0.39	32.24	42.93
Pelates sexlineatus (Striped trumpeter)	9.45	1.11	6.20	4.49	0.83	0.01	0.06	91.90	113.46

Appendix 14: Gray, C.A., Rotherham, D., Underwood, A.J., Chapman, M.G. and Johnson, D.D. 2007. A strategy for developing fishery-independent sampling tools. Poster presentation at the 2007 "*Fish Stock Assessment Methods for Lakes and Reservoirs: Towards the true picture of fish stock*", conference Ceske Budejovice, Czech Republic.

A strategy for developing fishery-independent sampling tools

NSW DEPARTMENT OF PRIMARY INDUSTRIES
Charles Gray
Doug Rotherham
Daniel Johnson
Tony Underwood
Gee Chapman

An example for estuarine fish in New South Wales, Australia

Fishery-independent surveys can play an important role in the assessment and management of populations of fish and invertebrates. Developing more reliable, robust and cost-effective surveys can be done using pilot experiments to test specific hypotheses about the design and deployment of sampling gears and scales of variation of target species. We present a strategy for doing this pilot work.

1 Identifying suitable sampling gear for target species

Example: Testing the utility of multi-mesh gillnets and trammelnets
Gillnets were considered to be the superior method because they provided greater precision of CPUE, were easier to use and required less sampling effort.
See Gray et al. (2005). *Marine & Freshwater Research* 56, 1077-1088.

2 Testing different configurations of gear and sampling practices

Example: Testing the effects of soak (1, 3 and 6 hours) and setting times (18:00, 22:00, 3:00) and lengths of panels of multi-mesh gillnets on samples of fish. Analyses identified that nets constructed of 20-m panels, soaked for 1 hour at any time of the night provided the most optimal and representative samples. The benefits of short soak times and panels are greater replication, smaller costs and potential for lower fish mortality.
See Rotherham et al. (2006). *Journal of Experimental Marine Biology and Ecology* 331, 226-339.

3 Understanding spatial and temporal scales of variability

Example: Hierarchical experiments were used to measure: (1) spatial variation of fish among zones (2-20 km apart), sites within zones (1 km apart) and replicate multi-mesh gillnets at each site (50-100 m apart) across shallow and deep strata; and (2) temporal variation across weeks, months and seasons. Spatial variation was generally greatest at the smallest spatial scale (among replicates); suggesting that future sampling need not include the scale of zones and that more effort should be put into sampling replicate nets and sites. Temporal variation was generally greater across seasons than across shorter time intervals; indicating that samples obtained at different sites within a particular season were more representative of spatial variation than short-term temporal variation.

4 Cost-benefit analyses to determine optimal levels of replication

Spatial and temporal variances identified in Step 3 can be used to do standard cost-benefit analyses to determine optimal levels of spatial and temporal replication (with regard to maximising precision and minimising costs). This can be done on a species-by-species basis or on a whole assemblage basis by weighting the importance of different species (e.g. according to their commercial/recreational value). This is currently being done for the present example.

Conclusion: Pilot experiments can be used to: (1) identify the most suitable sampling gears; (2) develop the most optimal and representative configurations of gear and sampling practices; (3) measure spatial and temporal variation; and (4) determine optimal levels of replication so that sampling is more cost-effective and reliable. Our strategy could be applied to other systems.

See Rotherham et al. (2007). *ICES Journal of Marine Science*, doi:10.1093/fsm096

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