Optimising the collection of relative abundance data for the pipi population in New South Wales

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Executive Summary

Overview: This project evaluated fishery-independent and -dependent data sources and sampling strategies to monitor and assess the beach clam (pipi), *Donax deltoides*. This was achieved using novel experiments across several beaches that tested specific hypotheses related to the project objectives. This research was required to determine the most appropriate and rigorous sampling strategies to provide vital demographic information for future managing the sustainable harvesting of the pipi resource in New South Wales (NSW). The project methodologies and results have applicability to other systems and jurisdictions in Australia and elsewhere.

B**ackground:** The pipi has a long history of indigenous, and a more recent history of commercial and recreational, harvesting in NSW. Commercial fishery production increased dramatically from its beginnings in the 1950's to peak at 670 tonnes in 2001, after which it declined to a low of 9 tonne in 2011 despite increasing prices and markets. Whilst the reasons for the decline are not clear and confounded by environmental variability, unregulated fishing probably contributed.

In response to the decade of decline in commercial catches, several management initiatives designed to substantially reduce commercial fishing effort and harvest, and therefore halt further population declines were introduced to the NSW fishery in 2012. These included a six-month total commercial fishing closure, spatially explicit commercial fishing closures of whole beaches and specific zones along particular beaches, a maximum daily catch quota of 40 kg per-commercial fisher, and a minimum legal size limit (45 mm shell length, SL). Concomitant restrictions to recreational and indigenous fishers were also introduced, but these were primarily in response to concerns over human health issues associated with bio-toxins. These latter two groups can still harvest clams year-round across most beaches, but this is limited to 50 clams per day for immediate in-situ bait use only (unless for specific indigenous cultural events).

To assist future management and industry needs for sustainable harvesting of the pipi resource, there was a requirement to develop and test cost-effective and reliable strategies to monitor and assess pipi populations across the breadth of beaches throughout NSW. This included fishery-dependent and -independent data sources and sampling strategies.

A**ims and objectives:** This project aimed to evaluate fishery-independent and -dependent data sources, and provide robust and practical demonstrations of their roles, for the assessment and management of the pipi resource and commercial fishery in NSW. It also provided quantitative information on the across-beach distributions of pipis. The specific project objectives were:

1. Assess fishery-independent and -dependent techniques in developing a practical, cost-efficient and collaborative strategy for surveying the relative abundance and size structure of pipi populations, and
2. Determine the across-beach distribution of pipis.

Methodology: A series of field experiments were done to test specific project objectives. A simple technique to sample pipis in the swash zone was developed, tested and evaluated across a hierarchy of spatial and temporal scales to determine suitable levels of stratification and replication for robust and cost-effective sampling. The developed standardised fishery-independent sampling strategy was used to assess pipis across six beaches in NSW that had different management arrangements. Sampling was strategically stratified to account for small-scale spatial and temporal variability and the flexible across-beach distributions of pipis. Fishery-dependent data sources, including industry logbooks, beach- and port-based sampling, were compared across two regions for their utility and cost-effectiveness in monitoring the commercial fishery.

Results and key findings: The project successfully developed and demonstrated the utility of a standardised fishery-independent sampling strategy in providing robust and reliable data for assessing the densities and size compositions of pipis across beaches throughout NSW. The large-scale sampling detected significant differences in pipi demographics across a range of spatial (among sites on a beach to among beaches) and temporal (days to months) scales across the swash and dry sand habitats. No global differences in the densities and sizes of pipis were detected between commercially fished compared to non-fished beaches, or zones across beaches. This probably resulted from a combination of factors, including current management arrangements and pipis responding to ecological process operating independently across differing spatial and temporal scales across beaches.

Cost-effective monitoring of the commercial pipi fishery could be based on industry logbooks for catch, effort and CPUE information and port-based sampling for size composition. This strategy relies on good management-industry relations, but such data are only applicable to commercially fished beaches. Fishery-independent sampling is the only mechanism to provide reliable and robust data on pipi populations across the breadth of beaches in NSW.

Implications for relevant stakeholders: This study identified the utility and value of standardised fishery-independent and -dependent sampling strategies for assessing the pipi resource and commercial fishery in NSW. This will be valuable to management and stakeholder groups in deliberating future monitoring and assessment strategies and their costs and benefits. The methods and results reported here have applicability to other systems and species in other management jurisdictions in Australia and elsewhere. This project serves as a model for development of sampling strategies in other small-scale fisheries.

Recommendations: Management authorities along with commercial, recreational and indigenous fishing and conservation representatives need to: (1) clearly define the objectives and information requirements for managing the pipi resource and fisheries in NSW, (2) consider the costs and benefits of alternative fishery-independent and -dependent sampling strategies and data sources, (3) use the information provided in this report to design an appropriate long-term monitoring and assessment program that will deliver the necessary data for managing the sustainable and profitable exploitation of the pipi resource in NSW.

Keywords: Beach clam; *Donax deltoides*; Population dynamics; Fishery assessment; Management evaluation; Conservation strategy; Exploitation; Sustainable harvest; Australia

1. Introduction

1.1. Background and Need

Beach clams (burrowing bivalve molluscs of the families Donacidae, Veneridae and Mesodermatidae) inhabit the intertidal and shallow subtidal zones of exposed ocean beaches worldwide, often dominating macrofaunal biomass and contributing greatly to beach ecosystem functioning (McLachlan et al., 1996; Defeo and McLachlan, 2013). In many countries, beach clams are important socially and economically as they are harvested for food and bait (McLachlan et al., 1996). Little is known about the biology and population dynamics of many beach clam species, but populations are often characterised by large temporal and spatial fluctuations in population sizes. Nevertheless, many species have undergone significant and persistent reductions in densities, the reasons for which are unclear but probably exacerbated by unregulated fishing (McLachlan et al., 1996; Defeo, 2003; Ortega et al., 2012). Beach clam fisheries are typically small-scale, low-value and data-poor, which has hampered the development of appropriate management arrangements across fisheries (Defeo et al., 2014).

The pipi, *Donax deltoides*, is endemic to high-energy ocean beaches between Fraser Island (Queensland) and the Eyre Peninsula (South Australia). It has a long history as an important traditional food source to indigenous people and a more recent history of harvesting by commercial and recreational fishers throughout its distribution (Ferguson et al., 2014).

The commercial fishery for pipis in New South Wales (NSW) developed throughout the 1950s and through a period of unrestricted fishing regulations, the numbers of pipi harvesters and beaches accessed increased till total production peaked at 670 tonne (t) in 2001. This was followed by a sharp decline in commercial landings to 9 t in 2011, despite increasing product prices and markets (Rowling et al., 2010). Recreational and indigenous catches throughout this period were also unrestricted and unchecked, and were probably large across many beaches (Murray-Jones and Steffe, 2000; Henry and Lyle, 2003). Although the reasons for the rapid decline in commercial catches remain unclear and potentially related to beach conditions and environmental variability, unrestricted harvesting probably contributed (Ferguson and Ward, 2014).

In response to the decade of decline in commercial catches of pipis, several management initiatives designed to substantially reduce commercial fishing effort and harvest, and therefore halt further population declines were introduced to the NSW fishery in 2012. These included a six-month total commercial fishing closure, spatially explicit commercial fishing closures of whole beaches and specific zones along particular beaches, a maximum daily catch quota of 40 kg per-commercial fisher, and a minimum legal size limit (45 mm shell length, SL). Concomitant restrictions to recreational and indigenous fishers were also introduced, but these were primarily in response to concerns over human health issues associated with bio-toxins. These latter two groups can still harvest pipis year-round across most beaches, but this is limited to 50 pipis per day for immediate in-situ bait use only (unless for specific indigenous cultural events). The current combined harvest from these two sectors is therefore considered to be much smaller than the commercial harvest (Murray-Jones and Steffe, 2000; Rowling et al., 2010). The harvesting of pipis by all sectors is restricted to digging by hand, with no mechanical apparatus permitted.

The NSW commercial pipi fishery is compartmentalised into seven designated regions, with pipi fishers being able to access specified beaches within each region. However, the size, numbers of permitted fishing beaches and commercial harvesters and hence commercial production, differ greatly among regions. Currently, there are 76 licence endorsements to harvest pipis and the current value of the fishery is approximately $AUD 2 million per annum (Rowling et al., 2010).

Pipi fishers that participate in the NSW commercial fishery currently provide catch and effort information via logbooks, but other fishery-dependent data sources such as beach and port-based sampling of catches, which can also provide data on size compositions of catches, warrant investigation as potential data sources for long-term monitoring. Such strategies have been successfully implemented for monitoring other regional small-scale fisheries (Gray, 2008).

Because of the aggregated distribution and harvesting of pipis across beaches, standard catch, effort and catch-per-unit-effort data from fishery-dependent sources can be biased and not indicative of population densities in general. Fishery-independent surveys have therefore been recommended to monitor and assess population demographics of harvested beach clams elsewhere (Defeo and Rueda, 2002). A necessary first step in developing any fishery-independent sampling strategy involves developing and testing appropriate gears and practices (if not already available), and quantifying the relevant habitats and spatial and temporal scales of sampling prior to wide-scale and long-term implementation (Rotherham et al., 2007).

A strategic, rigorous and cost-effective long-term strategy to monitor and assess the pipi resource in NSW is required to assist development of management strategies for sustainable harvesting.

The primary goal of this project was to determine appropriate, cost-efficient and rigorous fishery-dependent and -independent sampling strategies to provide necessary information for managing the sustainable harvesting of the pipi resource in NSW. Therefore, this project aimed to provide robust and practical demonstrations of the roles of fishery-dependent and -independent data for the assessment and management of the pipi resource in NSW. This project serves a model for development of sampling strategies in other small-scale fisheries.

2. Objectives

The objectives of the project were:

1. Assess fishery-independent and -dependent techniques in developing a practical, cost-efficient and collaborative strategy for surveying the relative abundance and size structure of pipi populations, and

2. Determine the across-beach distribution of pipis.

3. Key Methods and Results

3.1. Overview

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

Objective 1:

1. Develop and test fishery-independent sampling gears and protocols
2. Quantify the influences of tide, and scales of time and space on fishery-independent sampling
3. Test the fishery-independent sampling strategy across beaches and time
4. Explore avenues to integrate fisher-knowledge of pipi distributions in a fishery-independent strategy
5. Evaluate alternative fishery-dependent data sources

Objective 2:

1. Determine the across-beach distribution of pipis

Specific details of each experiment are provided in the accompanying appendices. A summary of the key methods and major results of each experiment is provided below.

3.2. Development and testing of fishery-independent sampling gears and protocols (Appendix 2)

This first study component aimed at developing a sampling gear and technique that could be easily used to sample pipis in the swash zone of high-energy ocean beaches. This was based on what commercial fishers use to harvest pipis in this habitat at present, similar in idea to that used in South Australia (pipi rakes) but modified for regional use based on local fishers methodologies.

The experiment was done across Smoky and Killick beaches with industry involvement. The time-based method consisted digging small plots of sand by hand and scooping sediment and pipis into a mesh bag attached to a rigid frame (0.35m long x 0.21m high; area 0.07 m^2). Three digging times (30, 60 and 120 s) and two mesh sizes (12 and 19 mm) were tested and compared to samples obtained using a standard box-quadrat (as developed and tested by other researchers – James and Fairweather, 1995). The sampling design involved using each configuration of method and treatment to sample two separate sites in the swash zone on each of two consecutive days on the two beaches. A total of four replicate samples of each treatment and mesh size were made (total 32 samples) at each site on each sampling day.

The time-based diggings were more effective and efficient in terms of numbers of pipis collected per time taken to do a sample compared to a standard box-quadrat. The timed digging technique was also much simpler and less problematic to use in the swash zone, which is important when industry are involved in sampling. Although a greater total number of pipis were collected in the 120 sec diggings, when the CPUE data were standardized to number per 30 sec, a greater proportion of pipis were collected in the shortest time frame tested. This suggested that most pipis were captured in the first 30 sec of digging, with fewer caught per unit of time thereafter. A major benefit of using the shortest digging time is that a greater number of replicate samples and sites on a beach can be sampled per given unit of time, potentially improving overall precision without large increases in costs. A greater proportion of small pipis (< 20 mm) were retained in the collecting bag with 12 mm compared to 19 mm mesh, and this was recommended for future use.

A 2nd round of sampling was done on each beach that intensively sampled (30 sec dig, 12 mm mesh bag, 24 replicate samples) two sites on each beach to determine optimal levels of sampling. This sampling identified that an optimal design would involve sampling more sites on a beach than replicates within a site. Given a sampling window of 3 hours either side of low tide, future sampling should incorporate 6 replicate 30-sec diggings at each of 8 sites on a beach. This experiment highlighted the importance and value of doing pilot studies to develop appropriate sampling gears and for determining optimal, cost-effective sampling strategies for large-scale surveys.

Greater details of the developed swash sampler and these experiments are given in Appendix 2.

> Gray CA, Johnson DD, Reynolds D, Rotherham D (2014) Development of rapid sampling procedures for an exploited bivalve in the swash zone on exposed ocean beaches. *Fisheries Research* 154, 205-212. (doi:10.1016/j.fishres.2014.02.027)

A standard square box-quadrat (as developed and tested by other researchers – James and Fairweather, 1995) that had 32 cm sides (surface area 0.1 m^2) was adopted for sampling pipis in the dry sand habitat (i.e. above the swash zone).

Other alternative sampling gears (a towed plough, mechanised clam rake) that sample larger areas of substratum than the swash and box-quadrat gears were also tested. However, due to unsolvable (given financial and time constraints) technical issues, further development of this aspect of the project was put aside to focus on completing other components of the project using the two prescribed sampling gears.

3.3. Influences of tide, and scales of time and space on fishery-independent sampling (Appendix 3)

Two separate experiments examined the influences of tide stage and various scales of time (day, week, month, season) and space (site, area, beach) on sampling and estimating the densities and size compositions of pipis in the swash zone of beaches. The experiments were done across Smoky and Killick beaches to test for generality of results. All sampling involved industry and was done using the swash sampler as developed in the 1st study component.

The first experiment specifically tested across both beaches whether the densities and size compositions of pipis sampled in the swash zone significantly differed according to tide stage. Two separate sites (> 500 m apart) on each beach were sampled on each of four randomly selected sampling days. On each day, six temporally and spatially independent replicate samples were made at each site during each of four tide stages (approximately 3 hours apart): low, mid-rising, high, and mid-dropping. Sampling was done during daylight (7.00-19.00 hours) and the first tide stage that occurred after 7.00 hours determined the order in which each tide stage was sampled on each day. All pipis sampled in each replicate were counted and measured.

Densities of pipis in the swash zone were most often least at high tide, which may be a consequence of their restricted shoreward movements and upper beach distributions. In contrast, the size compositions of sampled pipis were the same across all tide stages. This experiment identified that future quantitative population sampling of pipis could be done across a six-hour window around low tide (i.e. 3 hours either side of low tide between mid-tide dropping and mid-tide rising), providing ample time to access and sequentially sample many locations along a beach.

The second experiment specifically tested across both beaches whether the densities and size compositions of pipis significantly differed across a hierarchy of temporal and spatial scales. Pipis were sampled across both beaches on two days in each of two consecutive weeks, in each of two consecutive months in each of two consecutive seasons. On each sampling day on each beach, four sites (located 100-500 m apart) were sampled in each of two separate areas (> 2 km apart) with a total of six replicate samples taken within a 20 m shore-parallel distance at each site. Thus a total of 48 samples were collected each day on each beach. Based on the results from the 1st experiment, all samples were taken within three hours either side of low tide (i.e. between mid-tide dropping and mid-tide rising).

The hierarchical spatio-temporal sampling identified that variation in pipi density in all but a couple of cases was greatest at the lowest temporal and spatial scales sampled; among replicate days and replicate samples at each site. Variation in size compositions was also greatest at the smallest spatial scale examined (among sites), but this was not the case for time. Size compositions were relatively stable across days and the influences of other temporal scales were inconsistent, suggesting that ecological processes influencing size compositions operate at different scales to those influencing densities. The general predominance of small-scale variation was probably a consequence of fine-tuned, local-scale responses of pipis to the variable and dynamic physical and biogenic features of the swash zone habitat. Consequently, future sampling designs for population and management assessments need to account for such small-scale temporal and spatial variation so not to confound larger scale sampling of pipis among beaches. However, since small-scale spatial variability was generally greater than small-scale temporal variability, sampling across a beach over a couple of days to investigate spatial patterns is unlikely to be confounded by small-scale temporal variability.

Future quantitative sampling of pipis in the swash zone across large spatial and temporal scales can be done 3 hours either side of low tide, but must incorporate a strategy that accounts for small-scale spatial and temporal patchiness in pipi populations.

The full results of this project component are provided in Appendix 3.

> Gray CA (2016) Tide, time and space: scales of variation and influences on structuring and sampling beach clams. *Journal of Experimental Marine Biology and Ecology* 474, 1-10. (doi:10.1016/j.jembe.2015.09.013)

3.4. Large-scale testing of the fishery-independent sampling strategy across beaches and time (Appendices 4 and 5)

This component tested the developed fishery-independent sampling strategy across four beaches (South Ballina, Ten Mile, Smoky and Stockton) that were open to commercial pipi fishing and two beaches (Sandon and Illaroo) that commercial operators did not fish. On South Ballina and Stockton beaches, sampling covered areas open and closed to commercial harvesting. Recreational and indigenous anglers could harvest pipis across all beaches and zones.

Sampling was stratified temporally across three distinct periods, before and during the six-month winter-spring (1 June to 30 November 2013) commercial harvesting season. The length of each sampling period and the interval between consecutive sampling periods was six weeks. The 'Before' sampling was in April/May when all beaches were totally closed to commercial pipi harvesting, the 'Early' harvesting in July/August and 'Late' harvesting in October/November, with sampling beginning 6 and 18 weeks, respectively, after the commencement of the harvesting season. To account for short-term variability in pipi densities, in each of the three periods sampling was further stratified across two randomly selected days in each of three randomly selected weeks.

Sampling was also stratified spatially across two habitats, the swash zone and the dry sand belt typically located 10 to 30 m above the swash zone at low tide. To account for small-scale spatial variability, on each sampling day four locations in the swash zone and another four locations in the dry sand clam belt were selected at random within each zone/beach. At each of these locations, six replicate samples were taken so that a total of 96 samples were collected each day of sampling on each beach. Sampling was done during daytime within 3 hours either side of low tide and it took approximately 4 hours to complete sampling each day. The swash sampler and the standard quadrat were used to sample pipis in the swash and dry sand habitats, respectively. All pipis collected in each replicate sample were counted and measured for shell length (SL, mm)

Small-scale spatio-temporal variability in the densities and sizes of pipis was prevalent across the swash and dry sand habitats and the components of variation were generally greatest at the lowest levels examined. Even so, differences in the densities and sizes of pipis among individual beaches and zones on beaches were detected. However, there were no global differences in densities and sizes of pipis across the commercially fished versus non-fished beaches, or between the commercially fished and non-fished zones on beaches, from before to during harvesting. There was no evidence of reduced densities or truncated size compositions of pipis on fished beaches/zones. Greater proportions of small (< 30 mm SL) pipis were observed in the 'early' and 'late' harvest sampling periods, but these were evident across the commercially fished and non-fished beaches and due to the recruitment of small pipis. The results identify the difficulties in detecting fishing-related impacts against natural levels of variability in clam populations and were probably due to a combination of factors, including the current low levels of commercial harvests, the movements and other local scale responses of pipis to ecological processes acting independently across individual beaches and zones on beaches.

There were significant smaller scale spatial and temporal differences in densities of pipis across zones and beaches within each sampling period. For example, significant differences in densities were evident between zones on the commercially fished and control beaches, but they were mostly apparent only across short (day and week) periods before, early and late harvesting. Thus they were most likely pulse responses of pipis to stochastic, non-fishing related events that acted independently across the different zones on each beach.

This large-scale experiment further emphasised the importance and relevance of small-scale spatial and temporal processes in structuring pipi populations. Importantly, it documented the ability of the fishery-independent sampling strategy to detect and identify significant differences in pipi populations across various spatial and temporal scales of interest.

The full results of these large-scale surveys and management implications are provided in Appendices 4 and 5.

Gray CA (2016) Assessment of spatial fishing closures on beach clams. *Global Ecology and Conservation* 5, 108-117. (doi:10.1016/gecco.2015.12.002)

Gray CA (2016) Effects of fishing and fishing closures on beach clams: experimental evaluation across commercially fished and non-fished beaches before and during harvesting. *PLoS One* 11, e0146122. (doi:10.1371/journal.pone.0416122)

3.5. Integration of local fisher knowledge in a fishery-independent sampling strategy (Appendix 6)

This small pilot study specifically examined whether the densities and size compositions of pipis differed between standard survey sites and those chosen by industry representatives. It was done across a 4km section of Lighthouse Beach where pipi fishers actively worked throughout the 2015 season. Pipis were sampled across (1) the swash and dry habitats at each of four standard survey sites, and (2) four separate swash and dry sites chosen by an industry representative. Sampling was in September and October 2015.

In September, the densities of clams significantly differed between the standard and fisher chosen sites in the dry habitat across both sample days, and in the swash on the 1st sampling day. This sampling period corresponded with clams being highly aggregated in small areas, with commercial fishers having a strong knowledge of the location of such aggregations across both habitats. In contrast, there were no significant differences in densities of clams between sampling strategies across either habitat in October. This sampling was done approximately 3 weeks after a large storm event (4 m swell) that modified the morphology of the beach and redistributed clams across and along the section of beach commercial fishers had previously been operating. Consequently, clams were more dispersed, less dense and aggregations less pronounced and commercial fishers had greater difficulty identifying suitable aggregations for sampling.

Across both habitats, the size compositions of pipis significantly differed between the standard and fisher locations in September, but not in October. Differences in the size compositions of pipis in the swash in September may have been due to the fishers choosing areas where larger (commercial sized, > 45 mm SL) pipis predominated. Such differences could impact assessments, especially if the fisher-chosen samples were considered in isolation from the standard samples. This difference in size composition was not evident in October when pipis were more dispersed.

The fisher-chosen sites were generally concentrated across smaller sections of the study zone and were dependent on beach morphology (especially in September), potentially explaining some observed differences in densities and sizes of pipis between sampling strategies. Nevertheless, the data demonstrate how commercial fishers have unique knowledge and ability in identifying pipi aggregations and harvesting 'hotspots' across and along beaches and this was most evident in the September sampling. Clearly, this demonstrated that densities and sizes of pipis were appropriate for commercial harvesting.

Future surveys of pipis could involve a collaborative approach where standard scientific surveys are done across entire beaches along with a set of fisher chosen locations. Along with standard fishery-dependent data (e.g. logbooks), this may provide fishers a greater opportunity to provide input into resource and fishery assessments. It could also help unravel relationships between CPUE and actual stock abundance that typically plague aggregation-based fisheries. Such collaborative sampling could involve commercial, recreational and indigenous fishers depending on the beach and management arrangements. Such an approach may help strengthen cooperation among industry, scientists and managers and assist in the uptake of fishery-independent survey data in assessments and management deliberations.

The full results of this study component are provided in Appendix 6.

3.7. Evaluation of fishery-dependent sampling strategies (Appendix 7)

This experiment specifically compared industry logbooks, and beach- and port-based fishery-dependent data sources for monitoring and assessing catch, effort, CPUE and size compositions of pipis. The evaluation was done across the 2013 fishing season in management regions 1 and 3, serving a model for elsewhere. Specifically, the beach sampling of catches for catch, effort and size composition was done across South Ballina Beach in Region 1 and Smoky, Killick and Goolawah beaches in Region 3. The port-based (cooperative) sampling of size compositions was limited to Region 3.

In general, values of catch, effort and CPUE did not differ significantly between logbooks and beach sampling, and spatial and temporal trends in examined indices were similar across both data sources. Essentially, the beach-based data verified the legitimacy of the logbook catch and effort data, and thus its potential value as a data source in monitoring this fishery. Nevertheless, caution must be exercised in interpreting CPUE in this fishery because it targets aggregations, potentially leading to hyper-stable CPUE rates. Moreover, the CPUE index of catch-per-day would not be an appropriate measure for monitoring the relative densities of pipis due to the current imposed 40 kg daily trip limit.

Beach sampling captured additional data that included the partitioning of fishing effort into search and dig time, and also the number and location of sites fished each day, which could be useful in unravelling complex CPUE-pipi density relationships and potential fishing impacts, and assist in the spatial management of fishing across beaches. For example, commercial fishers do not randomly fish entire beaches, as most observed fishing occurred within a 2km section across each sampled beach. Concentrated fishing as observed here could lead to localised depletions of pipis on beaches. However, the potential mixing and movements of pipis along and across beaches may mitigate or mask such effects. These data could be future sourced from industry and provided on modified logbooks, which would be more cost-effective than employing beach-based observers.

Compared to port sampling, the beach-based size composition data appeared to be biased and influenced by fisher behaviour. In the presence of an observer, pipi fishers may have concentrated gathering their 40 kg catch limit in the shortest possible time (thereby maximising observed CPUE), and in doing so were most likely less selective in the sizes of pipis they retained. In addition to alleviating any possible sampling biases, port- as opposed to beach-based sampling would be cheaper and overall more cost-effective in obtaining size composition data across broader spatial scales for assessment purposes.

Cost-effective future monitoring of the fishery could be done using a combination of logbooks for catch, effort and CPUE that includes strategic periodic validations (e.g. every 5 years) using beach-observers, and port sampling for size compositions. Incorporation of a validation strategy may help mitigate any criticism regarding the reliability of industry-reported data for assessment and management purposes. Nevertheless, the success of any fishery-dependent monitoring strategy is reliant on strong fisher cooperation that requires open and trustful co-management arrangements.

The full results and assessment and management implications of this component are provided in Appendix 7.

3.8. Assessment of the across-beach distribution of pipis (Appendix 8)

This study component examined the across-beach distribution of pipis across three beaches: Lighthouse and North Port Macquarie beaches which were sampled during November and December 2014, and Goolawah Beach sampled in June 2015. Depending on the beach, sampling was stratified according to lunar phase, beach topography and location. Pipis were sampled at intervals along shore-perpendicular transects between the high-tide level and the swash. Opportunistic and qualitative sampling of the surf zone was also done when possible.

The sampling identified that the occurrence of pipis in the dry sand above the swash zone was variable and that the position of the dry clam belt (when present) was not predictable according to lunar phase, beach topography or position along a beach. Indeed, pipis displayed variable patterns of across-beach distributions in space and time across flat and steep beach profiles. Pipis consistently occurred in the swash but displayed extended periods of absence in the dry sand habitat. Pipis can actively alter their across-beach distributions by utilising swash flows and there was some evidence of concordant (landward) movements of pipis across Goolawah Beach.

Qualitative sampling identified that pipis were at times present in shallow gutters and sand banks seaward of the swash zone, but their densities and sizes could not be quantified. It is also not known whether pipis actively inhabited or were passively swept into these seaward habitats. Sampling the surf zone environment will require novel gears and techniques.

The size compositions of pipis differed between the swash and dry habitat but small sample sizes and the pooling of data across sites and sampling dates may have confounded results. Nevertheless, small (< 20 mm SL) pipis were present in swash and dry habitats but a greater proportion was generally sampled in the swash habitat. This concurs with that observed in the other study component sampling.

Sampling and monitoring strategies to assess pipi populations on ocean beaches need to incorporate designs that include both the swash and dry habitats, such as those described in the preceding experiments. Moreover, such sampling needs to be flexible to account for the variable across-beach distributions of pipis. Identification of the position of the dry sand clam belt (when present) using a preliminary sampling methodology is required so that broader-scale sampling among habitats can be stratified accordingly.

The full results and assessment and management implications are provided in Appendix 8.

4. Discussion

4.1. Objective 1. Assess fishery-independent and -dependent techniques in developing a practical, cost-efficient and collaborative strategy for surveying the relative abundance and size structure of pipi populations

This study identified that fishery-dependent monitoring of the pipi fishery in NSW could primarily be based on logbooks for catch, effort and CPUE. It would be beneficial if current logbooks were modified so that locational data could be included and effort partitioned into actual search and dig times. This may help unravel complex CPUE-stock abundance issues that plague aggregation-based fisheries.

Nevertheless, caution must be exercised in interpreting CPUE from such fishery-dependent data sources. For example, the CPUE index of catch-per-day would not be an appropriate measure for monitoring the relative densities of pipis when a daily trip limit is in place. Although the CPUE index of catch-per-hour may provide a better measure of pipi densities in this fishery, it still has potential problems. The fishing of pipi aggregations, particularly across small spatial scales as observed here, can produce CPUE values that may only reflect densities at very local (areas fished) scales and not across an entire beach or region, as reported for similar aggregation-based abalone fisheries across reefs (Prince, 1992).

A long-term strategy could involve a periodic (e.g. every 3 to 5 years) validation program that utilises beach-based sampling to ‘truth’ industry logbook data. This may help mitigate any criticism regarding the reliability of industry-reported data for assessment and management purposes. Such a strategy, however, relies on good industry-management relations.

Sampling at cooperatives could be used for size composition monitoring of fishery harvests. Compared to deploying beach-based observers, this strategy would be most cost-efficient and avoid potential confounding and biases observed in beach-based sampling. Pipi fishers may alter their fishing behaviours in the presence of an observer, an issue common in observer-based studies (Liggins et al., 1997; Faunce and Barbeaux, 2011).

Future monitoring and assessment of the NSW pipi resource cannot solely be based on fishery-dependent data sources as such data are only available for beaches that are commercially fished (and when they are actually fished), which can change both within and among fishing seasons. Consequently, a standardised fishery-independent sampling program is required to (1) assess pipis across the breadth of fished and non-fished beaches and zones across beaches throughout NSW (Gray, 2016b,c), and (2) overcome potential data issues associated with CPUE estimates based on fishery-dependent data.

This study successfully developed and tested a fishery-independent sampling strategy for assessing the densities and size compositions of pipis across beaches. Development of the fishery-independent sampling strategy followed that prescribed by Rotherham et al., (2007). This involved: (1) developing, testing and modifying appropriate sampling gears and practices, (2) quantifying scales of variability across a hierarchy of spatial and temporal scales to identify appropriate scales of sampling, (3) testing the strategy across a range of beaches and management scenarios.

All fishery-independent sampling was done in collaboration with commercial industry representatives and incorporated sampling pipis in the swash zone and in the dry sand habitats, which has generally not been incorporated in other sampling strategies such as that used in South Australia (Ferguson and Ward, 2014). The strategy was tested across several beaches, demonstrating its utility and ability to detect differences in densities and size compositions of pipis within and among beaches and over time.

4.2. Objective 2. Determine the across-beach distributions of pipis

Pipis displayed spatially and temporally variable across-beach distributions between the swash zone and the high tide level, as documented for other species (Mikkelsen, 1981; Leber, 1982; Denadai et al., 2005). Pipis consistently occurred in the swash, but displayed extended periods of absence in the dry sand habitat. The position of the dry clam belt (when present) was not predictable according to lunar phase, beach topography or along-shore location on a beach.

Pipis can actively alter their across-beach distributions by utilising waves and currents, but the processes responsible for driving either shoreward or landward movements could not be identified here. Qualitative sampling identified that pipis were at times present in shallow gutters and sand banks seaward of the swash zone, but their densities and sizes could not be quantified. The movements and linkages of pipis along and across beaches is an important avenue of future research.

Pipis between 5 and 75 mm SL inhabited the swash and dry sand habitats. However, small (< 20 mm SL) pipis generally occurred in greater proportions in the swash, similar to that observed for other species (Mikkelsen, 1981; Leber, 1982).

Sampling and monitoring strategies to assess pipi populations on ocean beaches need to be flexible to incorporate the variable across-beach distributions of pipis.

5. Conclusions

This project (1) successfully developed and tested a standardised fishery-independent sampling strategy to assess the pipi resource in NSW, and (2) evaluated alternative fishery-dependent data sources typically used by agencies to monitor fisheries and assess harvested populations (Objective 1). It further examined the across-beach distributions of pipis (Objective 2).

The study identified that logbooks and port monitoring (i.e. fishery-dependent data sources) would be the most cost-effective means to monitor commercial harvests of pipis. However, a standardised fishery-independent sampling strategy will provide the only consistent framework to deliver robust and reliable data essential for assessing and managing the pipi resource across the breadth of NSW. The fishery-independent sampling needs to be appropriately stratified and replicated in space and time to account for small-scale spatial and temporal variability as well as the variable across-beach distributions of pipis.

Assessments of the pipi resource and commercial fishery would best include a combination of fishery-dependent and -independent data sources. Management and relevant stakeholders need to determine available finances and clearly articulate the objectives and information needs for managing the commercial pipi resource and associated fisheries so that an appropriate cost-effective and robust long-term monitoring and assessment program be designed and implemented. The success of such a program will rely on good management-stakeholder relations.

6. Implications

This study has identified the utility and value of fishery-dependent and standardised fishery-independent data sources for assessing the NSW pipi resource and associated commercial fishery. This is of immense value to management authorities, the commercial fishing industry and other stakeholder groups in determining a cost-effective and collaborative long-term monitoring and assessment program for the sustainable and profitable harvesting of the pipi resource in NSW and elsewhere.

The methodologies and results reported here have applicability to other systems and species in other management jurisdictions in Australia and elsewhere.

7. Recommendations

Future monitoring and assessments of the pipi resource and fishery in NSW should include a combination of fishery-dependent and -independent data sources. Industry logbooks and port-based sampling of catches would be the most cost-effective means to monitor commercial harvests of pipis. This type of strategy could potentially be extended to the indigenous sector, but this requires further investigation. Similarly, a profile of the recreational pipi fishery is required to develop a monitoring strategy of harvests from this sector. A standardised fishery-independent sampling strategy will provide the only consistent and robust data to assess the pipi resource across the breadth of beaches in NSW.

Management and relevant pipi resource stakeholders need to consider the results from this study to develop an appropriate monitoring and assessment program that will provide the necessary demographic information required for managing the sustainable harvesting of the pipi resource in NSW. The overall strategy needs to be clearly aligned with defined management objectives.

Further development

Further fisheries-related ecological and biological research is required for greater understanding of pipi population dynamics, and to evaluate the effectiveness of management regulations and harvesting strategies as well as other anthropogenic impacts on the resource. Specifically, these include:

1. Quantify the recreational and indigenous harvest of pipis across beaches in NSW,
2. Quantification of population linkages including the along-shore movements of pipis across and among beaches and relationships with dominant ecological processes,
3. Stock-recruitment relationships and ecological drivers of juvenile recruitment variability,
4. Development of an index of recruitment,
5. Levels of flexibility in growth and longevity and size at maturity,
6. Depth of burrowing of pipis in the swash and surf zone,
7. Determination of ecological drivers of local-scale aggregations,
8. Effects of harvesting on local-scale aggregations.

8. Extension and Adoption

Most fieldwork involved commercial pipi harvesters and indigenous and recreational fishing representatives were present on some occasions.

The Project Advisory Committee that included government and commercial, recreational and indigenous fishing representatives was kept abreast of developments and provided summary documentation throughout the study. Presentations and summary information were provided to stakeholder representatives, individual fishers and management. Several scientific publications have been published and the project report will be available to all interest groups including the broader community.

A positive outcome from this project is that NSW DPI has adopted some project developments. Specifically, NSW DPI (1) has used the fishery-independent gears and strategy developed here to sample and assess pipis across some beaches, and (2) is in discussion with commercial industry concerning future sampling and monitoring requirements. The outputs from this project will further assist in these developments.

Project coverage

There were no known media reports concerning this project.

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10. Project materials developed

Scientific Publications:

1. Gray CA, Johnson DD, Reynolds D, Rotherham D (2014) Development of rapid sampling procedures for an exploited bivalve in the swash zone on exposed ocean beaches. *Fisheries Research* 154, 205-212. (doi:10.1016/j.fishres.2014.02.027)

2. Gray CA (2016) Tide, time and space: scales of variation and influences on structuring and sampling beach clams. *Journal of Experimental Marine Biology and Ecology* 474, 1-10. (doi:10.1016/j.jembe.2015.09.013)

3. Gray CA (2016) Assessment of spatial fishing closures on beach clams. *Global Ecology and Conservation* 5, 108-117. (doi:10.1016/gecco.2015.12.002)

4. Gray CA (2016) Effects of fishing and fishing closures on beach clams: experimental evaluation across commercially fished and non-fished beaches before and during harvesting. *PLoS One* 11, e0146122. (doi:10.1371/journal.pone.0416122)

11. Appendices

List of Appendices:

1. Intellectual property

2. Gray CA, Johnson DD, Reynolds D, Rotherham D (2014). Development of rapid sampling procedures for an exploited bivalve in the swash zone on exposed ocean beaches. *Fisheries Research* 154, 205-212. (doi:10.1016/j.fishres.2014.02.027)

3. Gray CA (2016). Tide, time and space: scales of variation and influences on structuring and sampling beach clams. *Journal of Experimental Marine Biology and Ecology* 474, 1-10. (doi:10.1016/j.jembe.2015.09.013)

4. Gray CA (2016). Assessment of spatial fishing closures on beach clams. *Global Ecology and Conservation* 5, 108-117. (doi:10.1016/gecco.2015.12.002)

5. Gray CA (2016). Effects of fishing and fishing closures on beach clams: experimental evaluation across commercially fished and non-fished beaches before and during harvesting. *PLoS One* 11, e0146122. (doi:10.1371/journal.pone.0416122)

6. Gray CA. Integration of local fisher knowledge in a fishery-independent sampling strategy: a pilot assessment

7. Gray CA. Evaluation of fishery-dependent sampling strategies for monitoring a small-scale beach clam fishery

8. Gray CA. Assessment of the across-beach distributions of the beach clam, *Donax deltoides*.

Appendix 1.

Intellectual property

No intellectual property that results in commercialisation was developed from this project.

Appendix 2.

Gray CA, Johnson DD, Reynolds D, Rotherham D (2014) Development of rapid sampling procedures for an exploited bivalve in the swash zone on exposed ocean beaches. *Fisheries Research* 154, 205-212. (doi:10.1016/j.fishres.2014.02.027)

Development of rapid sampling procedures for an exploited bivalve in the swash zone on exposed ocean beaches

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Abstract

We assessed a time-based technique for estimating the relative abundance and size composition of populations of the beach clam, *Donax deltoides*, in the swash zone of exposed ocean beaches by comparing it with a standard quadrat-based method. The time-based method consisted digging small plots of sand by hand and scooping sediment and clams into a mesh bag attached to a rigid frame. We tested three digging times (30, 60 and 120 s) and two mesh sizes (12 and 19 mm). Compared to a standard box-quadrat, the time-based diggings were more effective and efficient in terms of numbers of clams collected per time taken to do a sample. The timed digging technique was also much simpler and less problematic to use in the swash zone, which is important when industry are involved in sampling. Although a greater total number of clams were collected in the 120 s diggings, when the CPUE data were standardized to number per 30 s, a greater proportion of clams were collected in the shortest time frame tested. This suggests most clams were captured in the first 30 s of digging, with fewer caught per unit of time thereafter. A major benefit of using the shortest digging time is that a greater number of replicate samples and patches of clams on a beach can be sampled per given unit of time, potentially improving overall precision without large increases in costs. An optimal sampling design would involve sampling more patches on a beach than replicates within a patch. Given a sampling window of 3 hours either side of low tide, we suggest that future sampling should incorporate 6 replicate 30-s diggings at each of 8 patches on a beach. We further recommend that a 12 mm mesh bag be used as it retained a greater proportion of small clams (< 20 mm). This study highlights the importance of doing pilot studies to develop appropriate sampling gears and for determining optimal, cost-effective sampling strategies for large-scale surveys.

Keywords: Beach clam; Donacidae; Mollusc; Sample design; Benthic fauna; Exploitation; Population Assessment; Variability; Pilot study

1. Introduction

Burrowing bivalve molluscs of the families Donacidae, Mesodermatidae, Solenidae and Veneridae, commonly known as beach or surf clams, are found on exposed sandy beaches worldwide (Ansell, 1983: McLachlan et al., 1995, 1996: Castilla and Defeo, 2001). Beach clams can occur in high densities, contribute greatly to overall benthic faunal biomass, and thus play important trophic and ecological roles in ocean beach ecosystems (McLachlan et al., 1996: Defeo and McLachlan, 2005). Many species are of anthropogenic importance both socially and economically, as they are harvested for human consumption and bait (McLachlan et al., 1996; Murray-Jones and Steffe, 2000; Defoe, 2003).

The beach clam *Donax deltoides* (Lamarck, 1818) is the largest and most common burrowing bivalve inhabiting exposed, coastal, sandy beaches in eastern and southern Australia (James and Fairweather, 1995; Murray-Jones and Ayre, 1997; Ferguson and Mayfield, 2006). They are primarily distributed in the swash zones (shallow subtidal to high intertidal areas) of high-energy dissipative beaches (King, 1985), with dense aggregations often accounting for up to 85% of the total benthic faunal biomass on some beaches (Murray-Jones, 1999). Like other burrowing bivalves, *D. deltoides* can exhibit large temporal and spatial fluctuations in abundance and are potentially vulnerable to anthropogenic disturbances including over-harvesting (Defeo and Alava, 1995; Ferguson and Mayfield, 2006; Ortega et al., 2012).

Population levels and commercial catches of *D. deltoides* have declined across numerous east Australian beaches over the past decade, with the New South Wales (NSW) total landed commercial catch decreasing from over 500 tonnes to less than 100 tonnes between 2004 and 2010 (Rowling et al., 2010; O'Connor and O'Connor, 2011). This occurred despite increasing product prices and markets (Rowling et al., 2010). The paucity of current demographic information on east coast populations has hampered assessments and development of harvest plans for the species (Murray-Jones, 1999; Murray-Jones and Steffe, 2000). This contrasts the situation in southern Australia where the major commercial fishery is regularly assessed and managed by annual quota controls (Ward et al., 2010; Ferguson 2013).

In response to the declines in commercial catches of *D. deltoides* in NSW, restrictions on minimum legal shell length (MLSL - 40 mm shell length), a daily possession (trip limit) of 40 kg per fisher per day and a 6-month (December – May inclusive) closure to commercial harvesting were implemented in 2011. To further promote stock recovery, the MLSL was increased to 45 mm in June 2012. These harvest restrictions elevated the need for an assessment of populations of *D. deltoides* across beaches throughout NSW to assess the effectiveness of these management measures.

A challenge in low value, data-poor fisheries is obtaining cost-effective data required for assessment and management (Smith et al., 2009). Catch sampling by industry representatives has proven to be an efficient way of collecting data for incorporation in stock assessments in such fisheries (Starr and Vignaux, 1997). Whilst standard scientific techniques to sample beach clams such as corers and box-quadrats are available and widely used (Defeo and Alava, 1995; James and Fairweather, 1995; Laudien et al., 2003), they are not always easily transferable to industry representatives and using these methods directly in the swash zone can be problematic.

Industry-based monitoring programs require sampling techniques that are simple, rapid and efficient (Starr, 2010; Lordan et al., 2011; Kraan et al., 2013). Before implementation of any such program, there is a need to develop and test appropriate sampling techniques with industry representatives. Ideally, any sampling technique adopted should allow quantitative, comparative assessments of populations of beach clams to be made across multiple spatial and temporal scales by industry, scientists and managers. Rapid sampling techniques have been successfully developed and implemented in monitoring and assessment programs in both aquatic (Besley and Chessman, 2008; Parravicini et al., 2010) and terrestrial (Tomlinson, 1981; Abril and Gomex, 2013) systems.

In this study, we tested the effectiveness of digging small plots of sand by hand for 3 time intervals as a way of quickly and quantitatively sampling populations of *D. deltoides* in the swash zone of exposed beaches. We compared this technique to a standard box-quadrat sampling tool. A key focus was determining the efficiency of the 3 sampling times as opposed to ascertaining which sample time caught most individuals. The time-based digging method is similar to that used by commercial operators to harvest beach clams in the swash zone, whereby sand is scooped into a mesh bag attached to a rigid frame.

Estimates of relative abundance based on catch per-unit effort (CPUE) data are directly influenced by spatial heterogeneity in the distribution and abundance of the target species (Kennelly, 1989). To account for the inherent spatial patchiness of beach clam populations, analyses (Underwood, 1981; Kennelly, 1989) were performed to estimate the optimal levels of spatial replication at different scales to ensure that future sampling strategies provide the most representative results given the available time for sampling. The study was done across two beaches to provide greater generality of results. The developmental approach and general outcomes of our study are applicable to sampling other species of bivalves, as well as other organisms in other environments (Rotherham et al., 2007).

2. Methods

2.1. Beaches

This study was done on two adjacent, high-energy, exposed ocean sandy beaches in eastern Australia: (1) Killick (-31°07' S, 153° 00' E) and (2) Smoky (-30° 58' S, 153° 02' E) (Short, 2007). Historically, commercial, recreational and indigenous fishers have harvested *D. deltoides* on both of these beaches. Sampling was done in February and March 2013 during the 6-month (December to June) temporary closure to commercial fishing. Throughout sampling, the swell ranged between 0.5 and 1.8 m in height and was generally from an easterly direction. Three licensed commercial beach-clam harvesters were involved in sampling activities.

2.2. Comparisons of sampling techniques

We specifically tested whether hand digging for different lengths of time (30, 60 and 120 s) in the swash zone caught different numbers and sizes of *D. deltoides* and compared these samples with those obtained concurrently using a standard, square box-quadrat (0.32m long x 0.32m wide x 0.20m high; area 0.1 m^2; James and Fairweather, 1995). The time-based, hand-digging technique involved scooping sediment and clams from a specified area of approximately 0.1 m^2 into a mesh bag (1.2 m long, 1.1 m circumference), which was constructed from either 12 or 19 mm (2 mm–twine diameter) polyethylene netting hung on the bar (i.e square-shaped) and attached to an aluminum frame (0.35m long x 0.21m high; area 0.07 m^2). Each frame was held on the substratum and the bags kept open by wave–mediated water flow. Square-shaped mesh was used to ensure that meshes stayed open and each net remained a fixed sampling unit as the geometry and selectivity of diamond-shaped mesh can be affected by many factors including the weight of accumulated catch (Broadhurst et al., 2004). The contents of each box-quadrat were also scooped by hand into both mesh bag configurations. Industry members under strict supervision of scientific staff did the time-based digging, with the same member doing all sampling on a particular beach.

The sampling design involved using each configuration of method and treatment to sample two identified patches of clams in the swash zone on each of two consecutive days (18 and 19 February 2013) on the two beaches. On each day, the two patches of clams sampled on each beach were picked at random from all swash zone patches on the beach visually identified by commercial fishers and scientific staff who drove along each beach prior to sampling. The actual position sampled within a patch was haphazardly selected, with each individual sample being interspersed at least 1 m along a narrow strip parallel to the shore. A total of 4 replicate samples of each treatment and mesh size were made (total 32 samples) within each patch. The order in which replicate samples of each treatment were collected was randomly selected. It took approximately 2 hours to sample each patch, with all sampling occurring within 3-hours either side of predicted low tide (total 6 h period). All retained clams in each replicate sample were counted and measured for maximum shell length (to the nearest mm below) using digital Vernier calipers. The time taken to complete each replicate sample was recorded.

2.3. Determination of optimal levels of sampling

Two-randomly selected patches of clams on each of Killick and Smoky beaches were sampled on two randomly selected days (19 and 14 March 2013, respectively). On each day, 24 replicate diggings (each lasting 30 s) were completed using the 12-mm mesh bag in each patch of beach clams as described above. We used two teams of people who each sampled two patches within 1.5 hours either side of the predicted low tide. All retained clams in each replicate sample were counted and measured for length as described above. To determine optimal numbers of patches and replicate samples within a patch, standard cost-benefit analyses were performed (Underwood, 1981; Kennelly, 1989).

2.4. Data analyses

A five-factor analysis of variance (ANOVA) model was used to test for differences in raw abundance and standardized CPUE (catch per 30-s) for the mean number of *D. deltoides* caught between beaches (Killick v Smoky: fixed factor), among days (random factor nested in beach), among patches (random factor nested in day and beach), among mesh sizes (12 v 19 mm: orthogonal fixed) and among sample times (30, 60 and 120 s: orthogonal fixed). Data were standardized (catch per 30-s) because we were interested in testing hypotheses about the efficiency of sample times, rather than about which sample time caught the most individuals of *D. deltoides*.

The CPUE data for the box-quadrat were analysed using a four-factor ANOVA model (same design as five-factor model without the factor 'sample time'). Prior to all ANOVAs, data were tested for homogeneity of variance using Cochran's test and where necessary, transformed ln (x+1). Student-Newman-Keuls (SNK) multiple comparison test were used to investigate significant differences detected by ANOVA. CPUE data from the two methods were not compared directly due to the different volumes of sediment sampled by each technique. The size compositions of samples obtained on each beach from the timed diggings (pooled across all time periods) and box-quadrat using the 12 and 19 mm mesh were compared using Kolmogorov-Smirnov (K-S) tests.

Standard cost-benefit analyses were done on data pooled across beaches and sampling days. The total cost (time) of each sampling period and the variance of the estimated mean of each sampling period were minimized to determine the optimal number of patches and replicate samples within patches. The restricting costs in this study were the total time available to sample a beach (6 hr; i.e. 3 hr either side of predicted low tide), the time taken to sample a patch (30 min) and an individual replicate (1 min), and travel time between patches (10 min). Variance for estimated means were calculated from a two-factor ANOVA: Beach (random) and Patch (Random; nested in beach) following methods discussed in Kennelly et al. (1993).

3. Results

A total of 1,314 and 2,170 *D. deltoides* were sampled across the two days of sampling on Killick and Smoky beaches, respectively (Table 1). The total number of sampled clams segregated by beach, patch, sampling technique, time and mesh size is shown in Table 1. Greater total numbers of clams were sampled in the timed diggings (regardless of time) compared to the box-quadrat across most patches on both beaches. Further, for each technique, a greater total number of clams were collected on Smokey Beach than on Killick Beach.

The average (±SE) number of clams collected per patch using the box-quadrat ranged between 0.25 (0.25) to 4.75 (1.60) on Killick Beach and between 4.25 (2.62) to 16.25 (2.21) on Smoky Beach. In comparison, the average number of clams collected per patch in the 30 s timed diggings ranged between 4.25 (1.03) to 17.5 (2.22) on Killick Beach and 9.00 (2.04) to 17.25 (4.62) on Smoky Beach. Similarly, for the 120 s timed diggings the average number of clams collected per patch ranged from

8.75 (3.09) to 27.75 (11.84) on Killick Beach, and between 18.50 (8.21) to 38.50 (6.03) on Smokey Beach.

The average time (± 1SE) it took to complete a replicate sample (pooled across both beaches and mesh sizes) was 67.81 (1.49) s, 97.81 (1.89) s and 158.87 (1.76) s to complete a replicate 30, 60 and 120 s timed sample and 106.91 (3.44) s for the box-quadrat. The average diameter and depth (± SE) of 20 measured plots sampled in the 30 s timed diggings was 57.4 (0.94) and 18.3 (0.47) cm, respectively.

3.1. Time-based digging

3.1.1. Mean number of D. deltoides

For the mean number of *D. deltoides*, ANOVA detected significant ($P<0.05$) interactions between mesh size and patch ((Me x Pa(Be x Da)); and between mesh size and sample time (Me x Ti) (Table 2). For six out of the eight patches, SNK tests revealed no differences in mean numbers of clams between the 12 and 19 mm mesh (i.e. 6=12). But results were inconsistent for the remaining two patches (i.e. 12>19 and 12<19). Similarly, for the Me x Ti interaction, there were no differences between mesh sizes for the 30- and 120-s sample times. However, for the 60-s period, significantly fewer *D. deltoides* were caught using the 12 mm mesh (i.e. 12<19). By comparison, differences between sampling times were consistent between mesh sizes, with the mean number of *D. deltoides* increasing significantly with time (30<60<120).

3.1.2. Mean standardized CPUE of D. deltoides

When data were standardized to catch-per-30 s, ANOVA detected significant differences between sampling times (Table 2). SNK tests revealed that CPUE of *D. deltoides* was significantly greater in the 30- than 120-s sample. There were, however, no significant differences in CPUE between 60 and 120 s samples. Thus, the majority of clams were collected in the first 30 s of digging, with diminishing numbers collected thereafter. A significant mesh x patch interaction (Me x Pa(Da)) was also detected with subsequent SNK tests revealing similar patterns for the mean number of *D. deltoides,* in which there were no differences between mesh size for 6 of the 8 patches (i.e. 12=19) and inconsistent differences for the remaining two patches (i.e. 12<19 and 12>19).

3.2 Box-quadrat

For the mean number of *D. deltoides* caught in the box-quadrat, ANOVA detected a significant ($P<0.05$) mesh x patch interaction (Me x Pa(Da)). SNK tests revealed a significant difference between mesh sizes for only one patch (12<19 mm), with no differences detected for each of the remaining seven patches.

3.3. Size compositions of samples

There were no significant differences on either beach in the size compositions of retained clams sampled with the 12 and 19 mm mesh in the timed diggings (KS Tests, Fig. 2). Despite this, a greater proportion of clams between 10 and 20 mm were retained in the 12 mm compared to the 19 mm mesh on both beaches (Fig. 2). Because of low sample sizes, no meaningful analyses could be done for samples collected using the box-quadrat. Nevertheless, there appeared no obvious mesh size related difference in size compositions of clams collected with the box-quadrat. A similar size range of individuals was collected in the timed diggings and the box-quadrat on each beach (Fig. 2).

3.4 Optimal levels of sampling

The estimated variability among patches ($\theta^2 = 47.93$) was similar to that observed between replicate samples within a patch ($\theta^2 = 59.62$). The estimated percentage standard error of the mean for several different sampling configurations (numbers of patches and replicate samples within a patch) is shown in Table 3. This analysis identified that the precision and ability (power) of detecting differences in numbers of clams between patches (locations) was more dependent on the numbers of patches sampled compared to replicate samples within a patch. For example, when 6 patches were sampled the estimated SE reduced from 26.65 to 25.72 % when 6 or 10 replicates were done at each patch. In contrast, when 6 replicate samples were done across either 6 or 10 patches, the estimated percentage SE reduced from 26.65 to 20.65 %. This means it would be more effective to place greater effort into sampling more patches than replicates at each patch given the inherent variability observed in this study. With a total cost of 360 minutes (i.e. 6 hour window) to sample clams on any given beach, we determined the optimal sampling configuration to be 6 replicate timed-diggings at each of 8 patches. This design will allow adequate time to enter, locate, travel between and sample different patches along a beach; and then exit a beach 3-hours either side of the predicted low tide.

4. Discussion

Simple, robust and cost-effective resource surveys are needed (Andrew and Mapstone, 1987) and this study provides an example of the development of a simple method for sampling an exploited bivalve in a problematic sampling environment. Compared to a standard box-quadrat (James and Fairweather, 1995), the timed diggings in this study were more efficient in terms of numbers of *D. deltoides* collected per time taken to do each replicate sample. Although both methods sampled a similar size range of clams, the timed digging technique was simpler and less problematic to use in the swash zone as waves (albeit small) made it difficult to use the box-quadrat at times. This is important given potential uptake of this technique as an industry-based sampling tool. Further, the net and frame are light, less bulky and more readily transportable than the box-quadrat. Industry members were accustomed to using the timed technique, as it was similar to how they harvest clams in the swash zone at present. Such a sampling system may not be transferable to all field situations and further investigation is required prior to implementing any collaborative or industry-based sampling program in this fishery. This includes outreach and training, and testing and implementing data audit and quality control systems, so that industry involvement is optimized and information collected is reliable, useful and accepted by all stakeholders (Lordan et al., 2011; Mackinson et al., 2011; Kraan et al., 2013).

An industry-assisted sampling program that utilizes a rake-based sampling tool (local commercial fishers harvesting method) has been developed for a beach clam fishery in southern Australia (Ward et al., 2010). Whilst direct comparisons cannot be made between techniques in the two regions at present, it is important to develop and test sampling gears and practices to suit the species and environment under study (Andrew and Mapstone, 1987). This is particularly important when involving industry. Notably, commercial fishers operate in different ways between these regions and the scale and dynamics of the fisheries also differ, with densities and concomitant production of clams being a magnitude greater in the southern compared to the eastern fishery (Rowling et al., 2010; Ferguson 2013).

Beach clams are typically patchily distributed and abundance (CPUE) data can be highly variable even when a precise volume of sand is sampled using fixed-sized quadrats (Defeo and Rueda, 2002). Our work suggests that variation in the area sampled in the 30 s diggings was relatively negligible (SE approximately 2%). We suggest the timed technique is analogous to other scientific sampling strategies, including trawl surveys based on tow duration (Johnson et al., 2008) and trap, gillnet and baited underwater video techniques reliant on soak-time (Kennelly, 1989; Acosta, 1994; Gray et al., 2009; Cappo et al., 2007). In duration-based trawl surveys, the area swept by each trawl is not always constant, depending on a multitude of factors such as tow speed, water currents and depth (Misund et al., 1999; Petrakis et al., 2001). Similarly, the area of attraction and subsequent capture of organisms in baited trap and video surveys can vary considerably even when a standard soak-time is employed due to a plethora of abiotic and biotic factors (Robertson, 1989; Groeneveld et al., 2003; Lowry et al., 2012). Nevertheless, the data collected in these surveys provide valuable indices of relative abundance (CPUE), which are commonly used worldwide in scientific assessments of aquatic fauna (Gunderson, 1993; Pennington and Stromme, 1998; Kennelly and Scandol, 2002).

A greater number of clams were collected in the longer timed diggings, which was probably the result of slightly bigger areas being sampled as well as clams from adjacent areas being washed into the sampling frame. In contrast, when the CPUE data were standardized to number sampled per 30 s, greatest numbers of clams were collected in the shortest (30 s) time frame tested. This suggests that most clams were collected in the first 30 s with diminishing returns per unit of effort thereafter; a common result in time-based samplings of aquatic biota (Kennelly, 1989; Johnson et al., 2008). Further, clams once disturbed can rapidly burrow (Ellers, 1995) and this could also explain diminishing catches with digging time and lower catches in the box-quadrat. We consider 30-s diggings to be appropriate for future sampling of beach clams in the swash zone. A major benefit of using the shorter timed diggings is that more replicates and patches can be done compared to longer diggings per given time period, thus improving the overall precision of surveys without large increases in costs (Pennington and Vølstad, 1991; Rotherham et al., 2006). Shorter timed samplings can potentially also reduce large catches and the need for subsampling (Godo et al., 1990), which would also simplify any industry involvement in sampling (Lordan et al., 2011).

The precision and ability to detect differences among populations of beach clams was more dependent on the number of patches or locations sampled rather than the absolute number of replicates taken within a patch. Future sampling would benefit from designs that place greater emphasis on sampling more patches (locations) within and among different beaches rather than replicates within a patch. This is common with other sampling approaches in variable environments; again allowing greater precision of sampling and ability to detect significant differences among populations at the most relevant scales (Rotherham et al., 2006; Gray et al., 2009). Industry acknowledgement and understanding of sampling designs is necessary for successful participatory assessment programs and having industry assist in patch/location selection as well as digging could help this development process (Lordan et al., 2011).

Whilst the size compositions of samples were similar between mesh sizes, a greater proportion of smaller-sized (> 10 to < 20 mm) clams were retained in the 12-mm mesh bags. Although collecting bags made of smaller mesh may retain smaller clams, prior to this study we found that 5 mm mesh bags became clogged very fast, particularly when courser shell grit was encountered, negatively impacting sampling and sorting of catches. Retention of large quantities of sediment in smaller meshed bags could also bias retention of small clams. We suggest that mesh sizes < 12 mm may neither be practical or reliable for use in this sampling system and other sampling techniques are required to sample beach clams < 10 mm long.

In conclusion, the timed digging technique (30 s, 12 mm mesh nets) has the potential to provide estimates of relative abundances and size compositions of beach clams. This approach will enable effective sampling of multiple locations and patches of clams along a beach. Given the constraints of tides, finding and moving between locations and measuring samples, we suggest it would be optimal to sample 8 patches or locations each with 6 replicate diggings across a beach. Prior to implementation of any large-scale study however, further experimentation is required to determine the optimal temporal frequencies of sampling for longer-term population assessments of *D. deltoides* across beaches throughout eastern Australia.

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Fig. 1. Mean number and standardized number of *Donax deltoides* sampled in the time-based diggings and in the box quadrat for each mesh size on Killick and Smoky beaches.

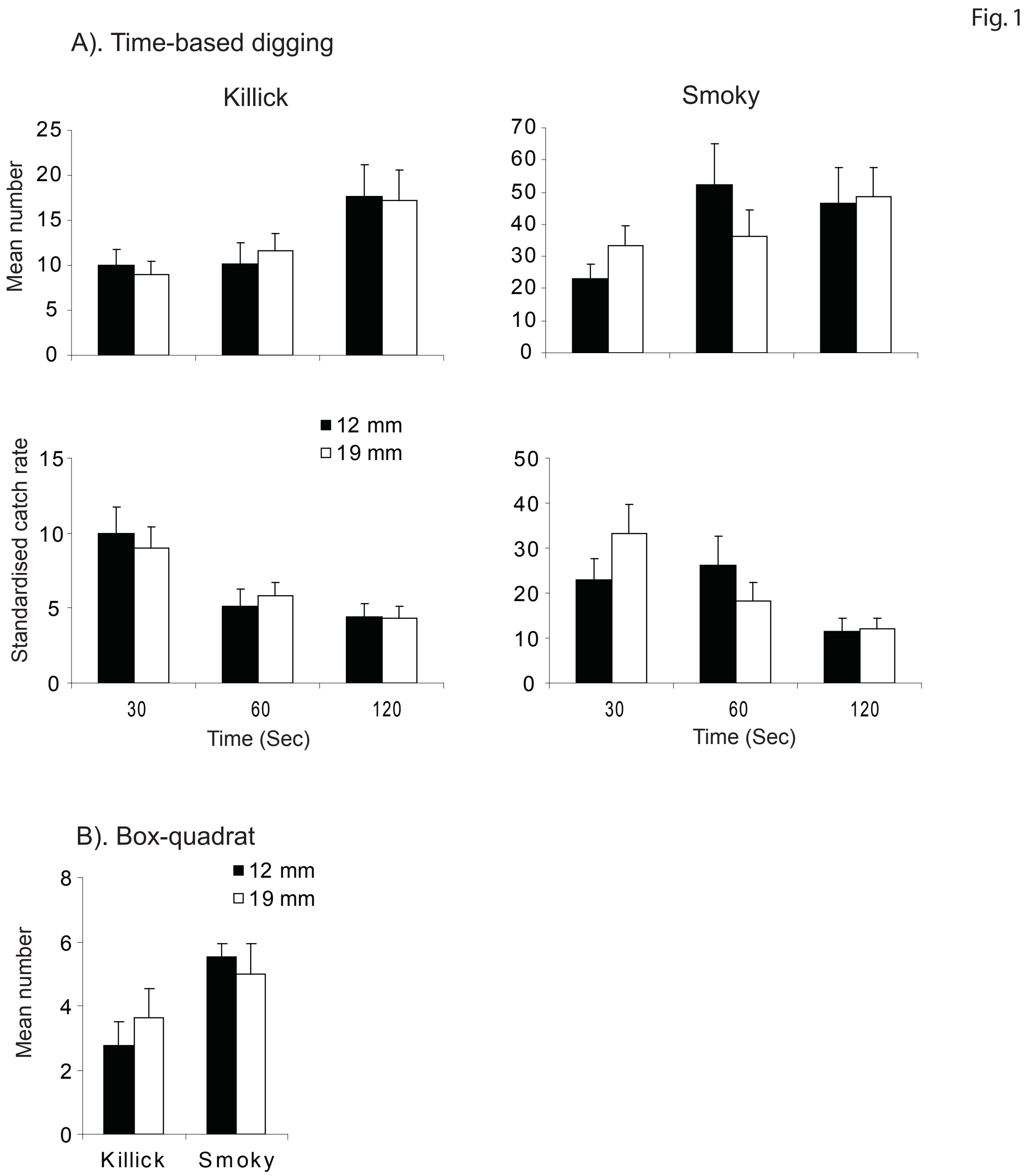

Fig. 2. Size composition of *Donax deltoides* sampled in the time-based diggings and in the box quadrat for each mesh size on Killick and Smoky beaches.

Fig. 2

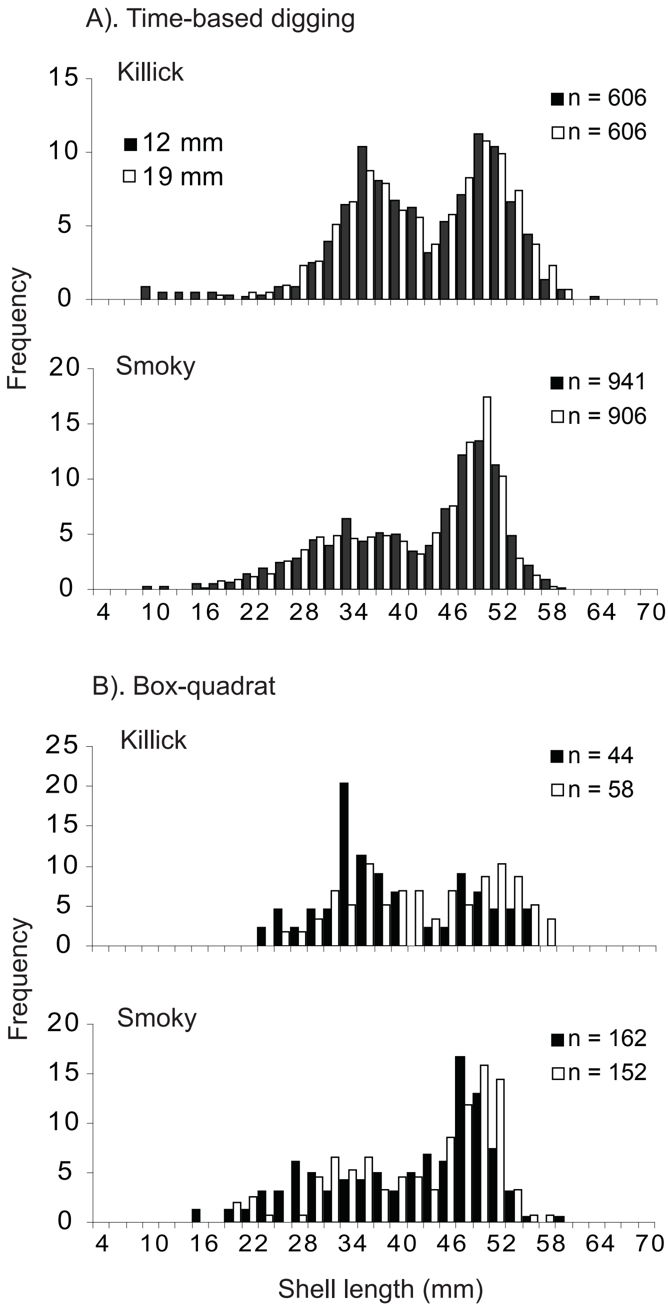

Table 1. Summary of the total numbers of *Donax deltoides* sampled in the box quadrat and in each level of time-based digging for each mesh size at each sampled patch on Killick and Smokey beaches. n = 4 replicates.

Killick Beach			Timed-based digging		
Patch	Mesh Size (mm)	Box quadrat	30 sec	60 sec	120 sec
1	12	15	70	37	75
1	19	16	37	59	74
2	12	1	17	22	52
2	19	15	27	27	56
3	12	19	19	16	45
3	19	12	19	24	35
4	12	9	54	88	111
4	19	15	61	76	111
Total		102	304	349	559
Smoky Beach					
Patch	Mesh Size (mm)	Box quadrat	30 sec	60 sec	120 sec
5	12	17	36	51	88
5	19	49	45	46	74
6	12	42	63	75	124
6	19	19	48	95	90
7	12	38	69	103	102
7	19	43	43	49	76
8	12	65	44	75	111
8	19	50	66	120	154
Total		323	414	614	819

Table 2. Results of analyses of variance (ANOVA) testing for differences in catches of *Donax deltoides* among geographic (Beach, Patch), temporal (Day) and sample technique (Mesh, Time) configurations for the time-based digging and the box quadrat. Summary results of relevant SNK tests are given next to each ANOVA. Factors in ANOVA in bold represent significant test results (P<0.05).

A. Time-based digging (number)	ANOVA				SNK		
Source	DF	MS	F	P	Factor	Level	Result
Beach	1	10.311	28.61	**0.033**	**Me x Ti**		
Day(Beach)	2	0.360	0.06	0.942	Me(Ti)	30 sec	12=19
Patch(Beach*Day)	4	5.898	19.10	**0.000**		60 sec	12<19
Mesh	1	0.000	0.00	0.976		120 sec	12=19
Time	2	6.274	19.75	**0.048**			
Beach*Mesh	1	0.077	1.84	0.308	Ti(Me)	12 mm	30<60<120
Beach*Time	2	0.318	1.60	0.309		19 mm	30<60<120
Mesh*Day(Beach)	2	0.042	0.05	0.953			
Time*Day(Beach)	4	0.199	0.80	0.558	**MeXPa(BeXDa)**	Patch 1	12=19
Mesh*Patch(Beach*Day)	4	0.863	2.79	**0.028**		Patch 2	12=19
Time*Patch(Beach*Day)	8	0.249	0.81	0.598		Patch 3	19<12
Mesh*Time	2	0.186	19.39	**0.049**		Patch 4	12<19
Beach*Mesh*Time	2	0.010	0.05	0.949		Patch 5	12=19
Time*Mesh*Day(Beach)	4	0.179	1.27	0.358		Patch 6	12=19
Time*Mesh*Patch(Beach*Day)	8	0.142	0.46	0.884		Patch 7	12=19
Residual	144	0.309				Patch 8	12=19
Total	191						

B. Time-based digging (standardised CPUE)							
Source	DF	MS	F	P			
Beach	1	492.481	10.11	0.086	**Time**		120=60<30
Day(Beach)	2	48.714	0.20	0.830			
Patch(Beach*Day)	4	249.128	13.41	**0.000**	**MeXPa(BeXDa)**	Patch 1	12=19
Mesh	1	2.297	6.10	0.245		Patch 2	12=19
Time	2	557.822	30.22	**0.032**		Patch 3	19<12
Beach*Mesh	1	0.376	0.44	0.577		Patch 4	12<19
Beach*Time	2	18.457	3.56	0.129		Patch 5	12=19
Mesh*Day(Beach)	2	0.863	0.01	0.989		Patch 6	12=19
Time*Day(Beach)	4	5.185	0.22	0.922		Patch 7	12=19
Mesh*Patch(Beach*Day)	4	76.657	4.13	**0.003**		Patch 8	12=19
Time*Patch(Beach*Day)	8	23.874	1.29	0.256			
Mesh*Time	2	6.481	6.90	0.127			
Beach*Mesh*Time	2	0.939	0.08	0.925			
Time*Mesh*Day(Beach)	4	11.779	0.51	0.734			
Time*Mesh*Patch(Beach*Day)	8	23.290	1.25	0.273			
Residual	144	18.578					
Total	191						

C. Box quadrat (number)							
Source	DF	MS	F	P			
Beach	1	763.141	10.12	0.086	**MeXPa(BeXDa)**	Patch 1	12<19
Day(Beach)	2	75.391	3.32	0.142		Patch 2	12=19
Patch(Beach*Day)	4	22.734	1.25	0.304		Patch 3	12=19
Mesh	1	2.641	0.75	0.545		Patch 4	12=19
Beach*Mesh	1	3.516	0.36	0.607		Patch 5	12=19
Mesh*Day(Beach)	2	9.641	0.16	0.854		Patch 6	12=19
Mesh*Patch(Beach*Day)	4	58.797	3.22	**0.020**		Patch 7	12=19
Residual	48	18.234				Patch 8	12=19
Total	63						

Table 3. Summary of cost-benefit-analyses showing the estimated percentage standard error (SE) of the mean and the total number of samples (N) for different sampling configurations.

Sample Configuration	SE (%)	Total N
6 patches, 6 replicates	26.65	36
6 patches, 8 replicates	26.08	48
6 patches, 10 replicates	25.72	60
8 patches, 6 replicates	23.08	48
8 patches, 8 replicates	22.58	64
8 patches, 10 replicates	22.28	80
10 patches, 6 replicates	20.65	60
10 patches, 8 replicates	20.20	80
10 patches, 10 replicates	19.92	100

Appendix 3.

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Tide, time and space: scales of variation and influences on structuring and sampling beach clams

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ABSTRACT

Scales of spatio-temporal variability in the densities and sizes of organisms need to be quantified so that the relative importance of different ecological processes that structure assemblages can be determined and future sampling strategies optimized. This study examined variability in the densities and size compositions of the beach clam, *Donax deltoides*, across tide stages and various hierarchical scales of time (day, week, month, season) and space (patch, area, beach). Densities of clams in the swash zone were most often least at high tide, which may be a consequence of their restricted shoreward movements and upper beach distributions. Regardless, the size compositions of sampled clams were the same across all tide stages. Future quantitative population sampling could be done across a six-hour window around low tide, providing ample time to access and sequentially sample many locations along a beach. The components of variation in densities of clams were in all but a couple of cases, greatest at the smallest temporal and spatial scales examined; among days and among replicate samples within a patch. Variation in size compositions was also greatest at the smallest spatial scale examined (among patches), but this was not the case for time. Size compositions were relatively stable across days and the influences of other temporal scales were inconsistent, suggesting that ecological processes influencing size compositions operate at different scales to those influencing densities. Nevertheless, the general predominance of small-scale variation was probably a consequence of fine-tuned, local-scale responses of clams to the variable and dynamic physical and biogenic features of the swash zone habitat. These results concur with studies in other habitats that show small-scale temporal and spatial variability is a dominant feature of aquatic benthic organisms and challenges the paradigm that ocean beach ecology is primarily driven by large-scale hydrodynamic forces. Small-scale patchiness of clams in time and space need to be taken into consideration in future sampling strategies for population and impact assessments. Moreover, the ecological processes that drive small-scale temporal and spatial structuring of organisms on ocean beaches require greater investigation.

1. Introduction

The densities and sizes of aquatic benthic organisms are often inherently variable in time and space due to the interacting effects of numerous biotic and abiotic factors (Thrush, 1991; Morrisey et al., 1992a,b; Fraschetti et al., 2005; Chapman et al., 2010). Knowledge of the levels of such demographic variation across different temporal and spatial scales is required for understanding the mechanisms and processes that structure populations and assemblages, and for determining appropriate spatio-temporal scales of sampling for ecological, environmental and population assessment studies (Andrew and Mapstone, 1987; Osenberg et al., 1994; Underwood et al., 2000; Fraschetti et al., 2001). Consequently, a number of studies have used hierarchical sampling designs to investigate levels of variation in the densities and structure of benthic faunal assemblages inhabiting the intertidal and subtidal zones of hard (Underwood and Chapman, 1996; Benedetti-Cecchi, 2001; Fraschetti et al., 2001, 2005; Olabarria and Chapman, 2002) and soft substrata (Morrisey et al., 1992a,b; Chapman et al., 2010). Whilst many studies on ocean beach fauna have also investigated patterns in time and space (Defeo and McLachlan, 2005), few have incorporated hierarchical sampling schemes (James and Fairweather, 1996).

Sandy beaches dominate the worlds coastlines (McLachlan and Brown, 2006) and are highly valued for their ecological and anthropogenic importance, yet many are extremely modified and under increasing anthropogenic threat (Schlacher et al., 2007, 2008; Defeo et al., 2009). Although ocean sandy beaches can be extremely dynamic and physically harsh habitats for organisms to occupy, they support diverse and abundant faunas (Defeo and McLachlan, 2005; Defeo et al., 2009). The strong hydrodynamic (e.g. wave, current, tide and wind) conditions that directly impact beach topographies, water circulation and sediment distributions are often hypothesized to drive ocean beach ecology (McLachlan et al., 1993; Defeo and McLachlan 2005; Nel et al., 2014). Because of this, it is commonly regarded that the along-shore structuring of organisms on ocean beaches primarily occurs at large spatial scales (100’s m – km’s), and that the dynamic physical conditions and constant fluxing of beaches prohibit structuring across small spatial scales. ~~S~~ome studies have challenged this paradigm, demonstrating that alongshore structuring of some ocean beach organisms occurs across small spatial scales (m < 10’s m) (Gimenez and Yannicelli, 2000; Cooke et al., 2015). Indeed, abiotic and biotic processes that affect benthic organisms inhabiting hard and soft substrata elsewhere often occur at local spatial and temporal scales, leading to considerable small-scale patchiness in faunal distributions and assemblage structures (Fraschetti et al., 2005; Chapman et al., 2010). Moreover, such small-scale spatio-temporal variability can confound and potentially mask larger-scale patterns if not properly accounted for in sampling designs (Morrisey et al., 1992a,b; Gray et al., 2009; Chapman et al., 2010).

Beach clams (burrowing bivalves – Families Donacidae, Veneridae and Mesodermatidae) are among the most dominant macrobenthic fauna common to tropical and temperate ocean sandy beaches throughout the world (McLachlan et al., 1996; Defeo and McLachlan, 2005). Beach clams not only play important ecological roles as consumers and prey on beaches (McLachlan et al., 1981; Defeo and McLachlan, 2005), but also are harvested by humans with many species having experienced severe population declines (McLachlan et al., 1996; Defeo, 2003; Ortega et al., 2012). Rigorous assessments of beach clam populations are required for provision of appropriate conservation and fisheries management strategies for resource sustainability (Defeo, 2003). But, because beach clams are often clumped and patchily distributed along and across beaches (McLachlan et al., 1996; Dugan and McLachlan, 1999, Denadai et al., 2005), and can respond rapidly to changes in beach abiotic conditions (Leber, 1982; McLachlan et al., 1995), sampling for population and impact assessment purposes can be problematic. It is therefore imperative that spatio-temporal scales of variation of the densities and size compositions of clam populations are understood so that sampling for assessment purposes is done across appropriate scales that account for natural population variability.

This study investigated the influences of tidal stage, and various scales of time and space on structuring and sampling the densities and size compositions of beach clams inhabiting the swash zone on ocean sandy beaches. The specific aims were: 1) test for the effects of tide stage on sampling the densities and size compositions of beach clams, and 2) quantify levels of variation in the densities and size compositions of beach clams across a hierarchy of temporal (day, week, month, season) and spatial (patch, area, beach) scales. Sampling was done across two beaches to test for generality of patterns and the data are discussed it terms of ocean beach ecology and sampling strategies.

2. Methods

2.1. Species and sampling

The study beach clam examined was *Donax deltoides*, the largest and most common burrowing bivalve inhabiting the swash zones of high-energy ocean sandy beaches in eastern and southern Australia (Gray et al., 2014). Like other clams, *D. deltoides* can occur in dense aggregations but they exhibit large temporal and spatial fluctuations in densities (Ferguson and Ward, 2014; Gray et al., 2014). The species has been heavily fished and experienced population declines throughout its distribution over the past decade, the reasons for which are not fully clear (Ferguson and Ward, 2014; Gray et al., 2014).

Two independent experiments were done across Smokey (−30°58'S, 153°02'E; 14 km long) and Killick (−31°07'S, 153°00'E; 16 km long) beaches in eastern Australia. Both beaches lie between rocky headlands, are fronted by rip-dominated bar systems and typically arc shaped but primarily face the east and southeast such that they are exposed to seas and winds from the north, east and south directions (Short, 2007). Sampling in each experiment was orientated towards the middle of each beach (> 2 km away from each headland), in seemingly homogeneous areas devoid of topographic features such as large berms and cusps. In both experiments, *D. deltoides* inhabiting the swash zone were sampled by finger digging for 30 sec a small area of sand (average diameter 57 cm, depth 18 cm) and scooping it into a net that had 12 mm mesh hung on a frame measuring 35 x 21cm (see Gray et al., 2014 for greater details and discussion of the technique). This technique sampled clams > 10 mm shell length (SL) (Gray et al., 2014; and see results). All clams collected in each replicate sample were counted and measured for SL (mm) using digital calipers. Operational information including time of sampling and general beach and ocean conditions were recorded.

2.2. Experiment 1. Tide

This experiment was done in March 2013 and specifically tested across both beaches whether the densities and size compositions of clams sampled in the swash zone significantly differed according to tide stage. On each of four randomly selected sampling days on each beach, two separate patches of clams (each patch visually extending 30-50 m along the beach and separated more than 500 m apart) occurring in the 'swash zone clam belt' were selected from the many observable patches on each beach. A total of 6 temporally and spatially independent replicate samples randomly allocated within a shore-parallel 20 m distance were made within each patch during each of four tide stages (approximately 3 hours apart): low, mid-rising, high, mid-dropping. Sampling was done during daylight (7.00-19.00 hours) and the first tide stage that occurred after 7.00 hours determined the order in which each tide stage was sampled on each day. This differed among sampling days across both beaches as sample days were spread across a three-week period. It took approximately 10 minutes to sample each patch and 30 min to sample both patches on a beach during each tide stage on each day.

A four factor permutational analysis of variance (PERMANOVA, Anderson 2001; Anderson et al., 2008) with the factors Beach (2 levels, random and fully orthogonal), Day (4 levels, random and nested in beach), Patch (2 levels, random and nested in beach and day) and Tide (4 levels, fixed and fully orthogonal) was used to determine if the densities of clams significantly differed according to tide stage and interactions with day and patch. The analysis was based on the euclidean distance measure with Type III (partial) sums-of-squares calculated using 9999 unrestricted permutations of the raw data. When significant interactions involving the factor tide were detected by the initial PERMANOVA, pair-wise tests between levels of this factor were done separately for each level of the other factor using a separate run of the PERMANOVA routine. PERMANOVA was also used to determine whether the size compositions of sampled clams for each tide stage (combined across sampling days and patches) on each beach significantly differed. The percent contribution of clams in each one mm length class was used to classify distributions using the Bray Curtis dissimilarity matrix with Type III (partial) sums-of-squares calculated using 9999 unrestricted permutations of the raw data.

2.3. Experiment 2. Time and space

This experiment was done in April/May and July/August 2013 and specifically tested across both beaches whether the densities and size compositions of clams significantly differed across a hierarchy of temporal and spatial scales. Clams were sampled across both beaches on two days in each of two consecutive weeks, in each of two consecutive months in each of two consecutive seasons. On each sampling day on each beach, four patches of clams (each 30-50 m long and located 100-500 m apart) were sampled in each of two separate areas (> 2 km apart) with a total of 6 replicate samples taken within a 20 m shore-parallel distance within each patch (Gray et al., 2014). Thus a total of 48 samples were collected each day on each beach. Based on the results from experiment 1, all samples were taken within three hours either side of low tide (i.e. between mid-tide dropping and mid-tide rising).

PERMANOVA was used to analyse the data collected across the hierarchy of spatial and temporal scales examined. Although autocorrelation is a problem in ecological studies (McArdle and Blackwell, 1989; Legendre, 1993), it is negated in PERMANOVA as the analyses and construction of Pseudo-F ratios are based on non-parametric permutation-based (exchangeability) tests, meaning that potential autocorrelations among data are eliminated (Anderson, 2001; Anderson et al., 2008). A six factor PERMANOVA was used to test for differences in the densities of clams sampled on each beach. The factors were: Season (2 levels, Random), Month (4 levels, Random and nested in Season), Week (2 levels, Random and nested in Month), Day (2 levels, Random and nested in Week), Area (2 levels, Random) and Patch (4 levels, Random and nested in Area and Day). Each analysis was based on the euclidean distance measure with Type III (partial) sums-of-squares calculated using 9999 unrestricted permutations of the raw data.

The components of variation for each temporal and spatial scale were determined separately by running several PERMANOVAs. To estimate temporal variances independently of spatial interactions, the four temporal scales (Season, Month, Week, Day) were analysed separately for each area and beach using nested PERMANOVAs with all factors included as random effects. This provided four independent estimates of each temporal component of variation, one from each area on each beach. Likewise, the spatial components of variation were determined by doing separate nested PERMANOVAs comparing the densities of clams across patches and areas on each beach on each sampling day. This provided 16 independent estimates of the spatial components of variation on each beach. All PERMANOVAs for determining the components of variation were based on the euclidean distance measure with Type III (partial) sums-of-squares calculated using 9999 unrestricted permutations of the raw data. Any negative component values were treated as zero, eliminated from the analysis and the remaining variation components recalculated (Fletcher and Underwood, 2002). Each component directly estimated variability at each scale independent of the other scales.

A four factor PERMANOVA tested for differences in the size compositions of clams across days, weeks, months and seasons on each beach. For each analysis, the data were pooled across patches for each area, providing two replicate size compositions for each beach on each sampling day. This was done so that sample sizes were sufficient for subsequent analysis. The percent contribution of clams in each one mm length class was used to classify distributions using the Bray Curtis dissimilarity matrix with Type III (partial) sums-of-squares calculated using 9999 unrestricted permutations of the raw data. Again, the components of variation for each temporal and spatial scale were determined separately as described above, except that the temporal analyses was done at the level of beach (as areas were used as replicates for each day), and in the spatial analyses the scale of patch could not be included as the data for the four patches within each area were combined to provide a size composition for each area on each beach.

3. Results

3.1. Experiment 1: Tide effects

Tide stage had no significant global effect on the densities of clams sampled (PERMANOVA, df = 3, 18, MS = 202.59, Pseudo-F = 0.55, *P-Perm* = 0.65), as effects were dependent on the patch sampled on each beach (PERMANOVA, significant Patch x Tide interaction, df = 24, 320, MS = 247.33, Pseudo-F = 7.52, *P-Perm* < 0.001, Fig. 1). The pairwise comparisons identified that the densities of clams were significantly (*P-Perm* < 0.05) lower at high tide compared to all other tide stages across nine of the sixteen patches sampled; five patches on Smoky and four on Killick (Fig. 1). Densities were significantly (*P-Perm* < 0.05) lower at low tide across two patches, greatest at high tide in two patches and at low tide in one patch. No clear patterns were evident across other patches (Fig. 1).

The size composition of sampled populations differed significantly between beaches (PERMANOVA, df = 1, 3, MS = 3072.80, Pseudo-F = 11.11; *P-Perm* = 0.028), but they did not significantly differ among tides on each beach (PERMANOVA, df = 3, 3, MS = 185.95, Pseudo-F = 0.67; *P-Perm* > 0.755, Fig. 2).

3.2. Experiment 2: Temporal and spatial variability in densities

A total of 2839 and 3620 clams were sampled on Smoky and Killick beaches, respectively. The densities of clams differed significantly according to patch (i.e. the lowest spatio-temporal scale of sampling) on each beach (PERMANOVA; Smoky: df = 96, 640, MS = 96.57, Pseudo-F = 7.18, *P-Perm* < 0.001; Killick: df = 96, 640, MS = 93.72, Pseudo-F = 8.75, *P-Perm* < 0.001). There were no significant differences (*P-Perm* > 0.05) in densities of clams on either beach according to nested factors Day, Week, Month or Season and their interactions with Area (*P-Perm* > 0.05 in all cases; Figs. 3 and 4). Densities of clams differed significantly between areas on Smoky (PERMANOVA, df = 1, 1, MS = 91.44, Pseudo-F = 7802.80, *P-Perm* < 0.001), but not on Killick (df = 1, 1, MS = 441.05, Pseudo-F = 2.50, *P-Perm* > 0.05).

3.3. Experiment 2: Temporal components of variation in densities

Across both areas on both beaches, the components of temporal variation were greatest at the smallest scales of sampling; day and residual (i.e. among replicate samples) in three of the four analyses (Table 1). The temporal components of variation were greatest for season in Area 2 on Killick. Negative components of variation occurred for the factor week in one analysis, month in three analyses and season in two analyses. After these terms were eliminated and the analyses redone, day had the largest temporal component of variation (Table 1). Nevertheless, variation among replicate samples (i.e. spatial component) accounted for 54 to 98% of the total variation and was greater than any temporal component.

3.4. Experiment 2: Spatial components of variation in densities

The components of spatial variation were greatest at the smallest scales of sampling; residual (i.e. among replicate samples) and patch, and were least at the level of area across 15 of the 16 sampling days on each beach; the exception was day 13 on both beaches when area had the greatest component of variation (Table 2). The component of variation for area was negative on nine days on Smoky and eight days on Killick. In the subsequent re-analyses, patch had a negative value in 1 analysis on each beach, whereas in the other analyses the residual had a greater component of variation than patch, except for day 16 on Smoky Beach (Table 2). The average component of variation explained by the residual was 68% on Smoky and 63% on Killick (Table 2).

3.3. Experiment 2: Size compositions

The PERMANOVAs identified there were significant differences in the size compositions of clams across seasons (df = 1, 2, MS = 4357, Pseudo-F = 2.68, *P-Perm* < 0.01) and weeks (df = 4, 4, MS = 3263, Pseudo-F = 1.96, *P-Perm* < 0.05) on Smoky, and across months (df = 2, 4, MS = 2033, Pseudo-F = 1.44, *P-Perm* < 0.05) on Killick (Fig. 5). There were no significant differences in the size compositions of clams according to area on either beach (*P-Perm* > 0.05), but there was a significant area x week interaction (df = 4, 16, MS = 1910, Pseudo-F = 1.46, *P-Perm* < 0.05) on Killick.

On both beaches, the temporal components of variation were greatest for the residual (i.e. between the two replicate areas for each day on each beach), accounting for 91 and 89% of the variation on Smoky and Killick, respectively. Day had a negative component of variation on both beaches, whereas the components of variation for month and season on Smoky, and week on Killick, were also negative. These factors therefore did not contribute to variation in the data. Week accounted for 9% of the variation on Smoky, whereas season (7%) had a greater component of variation compared to month (4%) on Killick.

For each spatial analysis, the components of variation were greater for the residual than area, except for week 6 on Killick (Table 3). The residual included a spatial (patches pooled) and a temporal (the two sampling days were used as replicates for each week) component. Area had a negative component of variation in half the analyses; weeks 2, 3, 4 and 6 on Smoky, and weeks 3, 4, 5 and 8 on Killick.

4. Discussion

4.1. Tide

The fact that beach clams were most often less abundant in samples taken in the swash zone at high tide probably reflects their restricted shoreward (up-beach) tidal movements and distributions. Like other beach clams (Leber, 1982; Donn et al., 1986; Ellers 1995), *D. deltoides* can move up and down the beach with the tide to maintain their position in the swash zone; throughout sampling clams were often observed 'riding waves' up and down each beach with the incoming and outgoing tides, respectively. These clams may not move all the way to the upper swash limit at high tide, but remain at a lower height on the beach (e.g. mid tide level) therefore staying below the sampled swash zone at high tide, thus accounting for their lower densities in samples. This may particularly be the case during spring tides when tidal ranges are greatest and may be a behavioral mechanism to reduce the risk of being stranded high on the shore and not being inundated with water again until another spring tide. Although this could not be fully tested here due to limited sampling, clams were not present or densities were least at high tide across small (< 0.8 m; Day 1 Killick) and large (> 1.3 m; Day 2 Smoky) tide ranges. Swell and wave size may also be influential; for example, even though days 3 and 4 on Killick were consecutive clams displayed different patterns of occurrence at high tide. A 3 m swell was present on Day 3 (clams not present) but this abated to 1.5 m on Day 4 (mean clam density = 19.3 and 8.3 per sample/patch). Across both beaches, swell size was generally between 1 and 1.5 m on the other sampling days. As reported for other beach clams (Donn et al., 1986), *D. deltoides* do not always move all the way down to the lower tide limit with the outgoing tide, but remain buried in the damp (and ultimately dry) sand above the low tide swash zone, generally around the mid-tide level (unpublished data). Such behaviour could possibly explain why on two occasions the densities of sampled clams in the swash zone were least at low tide. These buried clams would be inundated with water each tidal cycle and at little risk of being left stranded without inundation for an extended time. Not only tidal range, but the distance of the intertidal and swash zone (as determined by beach morphology, tide range and wave conditions) are important in determining the extent of tidal movements of clams (and other motile beach fauna) and their upper distributions on beaches (Mikkelsen, 1981; Defeo and McLachlan, 2005; Scapini, 2014). The distance of the intertidal zone on local beaches can typically vary between 20 and 80 m width (unpublished data; Short, 2007).

Based on the size composition data, all size classes of sampled clams (10-70 cm SL) were similarly represented across all tide stages, suggesting they all have the same across-beach distributions and undertake similar tidal movements. Elsewhere, different sized clams can inhabit different vertical levels (horizontal zones) on beaches (Leber, 1982; Donn, 1990). The across-beach distribution of small (< 10 mm SL) *D. deltoides* needs to be examined. Further research is also required to determine what processes drive the extent of the across-beach tidal movements of *D. deltoides*, as well as their movements along beaches (Dugan and McLachlan, 1999) and the potential interactions between these dimensions. Ideally, this would involve tracking the movements of individual clams in situ over different tidal cycles and ranges across areas of beaches with different topographies.

The data indicated it would be unwise for assessment purposes to sample beach clams in the swash zone at high tide due to their inconsistent occurrence. Nevertheless, quantitative sampling could be done across the other tide stages, as the densities (in most cases) and size compositions of clams did not significantly differ between low and mid tides on each beach. Thus, future sampling of clams in the swash zone could be done over a six-hour window, three hours either side of low tide. This would make it logistical feasible to access, travel (by vehicle) and sample several locations along a beach, and then exit a beach prior to high tide. Moreover, many local ocean beaches have limited vehicle access at high tide due to a combination of their topographies, narrow widths and the high tide water level often reaching the base of the foreshore dunes, particularly during spring tides and large swell.

4.2. Time and space

Densities of *D. deltoides* inhabiting the swash zone were inherently variable, fluctuating greatly in time and space as reported for other species of clams (McLachlan et al., 1996; Denadai et al., 2005) and benthic invertebrates in general (see Introduction). Importantly however, the components of variation were consistently greatest at the smallest temporal and spatial scales examined; i.e. among days and among replicate samples within a patch on a beach. The latter result was unlikely an artifact of sample size; the standard area (and quantity of sediment) sampled per replicate was similar to other common corers and quadrats used to sample beach fauna (James and Fairweather, 1995; Defeo and Rueda, 2002). Whilst some small-scale temporal and spatial variation could be the result of stochastic events, given its prevalence, it was most likely a consequence of clams responding to the dynamic nature and concomitant small-scale heterogeneity of the physical and biogenic features of the swash zone habitat.

Swash zone beachscapes constantly change in a heterogeneous manner along and across beaches due to the complex interactions of variable hydrographic and terrestrial processes that operate at large and small scales (Wright and Short, 1984; Elfrink and Baldock, 2002; Masselink and Puleo, 2006). Waves, currents and winds continuously interact to redistribute and reshape the swash sedimentary environment in which clams bury, creating a mosaic of swash beachscapes comprising large and fine-scale heterogeneous beach morphologies that typically have different sedimentary (sediment size, composition, porosity, compactness, skewness), topographic (berms, steps, cusps, bays) and water flow (strength, direction, turbulence and depth of swash and backwash) properties. Moreover, local hydrographic features, such as wave size, period and direction, alongshore currents and the positioning and impacts of surf zone features such as bars and rips, and their interactions with beach sedimentary topographic features such as cusps and bays produce variable water distributions and circulations in the swash zone. Consequently, the supply and delivery of planktonic food within the swash zone would also vary greatly over small and large temporal and spatial scales (Talbot et al., 1990; Odebrecht et al., 2014).

Spatial variation among replicate samples within a patch (i.e. residual variance) was mostly greatest in all density analyses. The behaviour of clams and their ability to move rapidly over short distances and periods of time by utilizing waves and currents (Leber, 1982; Ellers, 1995) could explain some of the patchiness among replicate samples. Notably, many clams were at times caught in some replicate samples within a patch, but few or no individuals were caught in the other adjacent samples (< 20 m apart). Such micro-patchiness could be a consequence of fine-tuned, small-scale responses of clams to the variable and dynamic swash zone habitat. For example, clams may aggregate in small areas (micro-patches) where sediments are optimal for burying or similarly water movements for feeding (Levinton, 1991). Moreover, such areas and conditions may vary considerably over small spatial and temporal scales within the swash zone (McLachlan and Hesp, 1984), as found in other sedimentary habitats (Hogue and Miller, 1981; McArdle and Blackwell, 1989; Sun et al., 1993; Chapman et al., 2010; Gingold et al., 2011). Small-scale within patch variability in hydrographic and sedimentary conditions were not determined here to assess such hypotheses. A greater understanding of the responses and movements of clams to small-scale, and potentially subtle, changes in the abiotic and biotic features of the swash habitat are required to isolate the ecological processes that drive such observed patchiness.

Variability in densities of clams was greater among days within a week than among weeks in each month, months in each season or between the two seasons. As argued above, this was potentially due to the strong influences of a concert of local processes acting at the smallest scales examined. Although conditions such as swell direction and size often differed between the two days sampled each week, it was not possible here to isolate the mechanisms responsible for driving the observed small-scale temporal variability. This day-to-day variation in clam densities does not negate the importance of larger temporal scales (e.g. seasons, annual) on structuring clam populations; indeed many studies have documented seasonal and annual changes in clam distributions and densities along and across beaches (Lima et al., 2000; Laudien et al., 2003). Similarly, spatial variation was generally greater within and among patches in an area and least at the level of area. Again, this does not mean different areas or sections along a beach do not impact densities and are not important. Rather, the areas sampled were strategically located towards the middle of each beach in relatively homogeneous beachscapes devoid of large topographic features, and there may well be broader alongshore differences in the distributions and densities of clams; for example between different ends of a beach (Donn, 1987; Dugan and McLachlan, 1996). Regardless, any sampling across larger temporal and spatial scales must include adequate and appropriate replication that accounts for within-scale variation (James and Fairweather, 1996).

Small-scale variation in the densities of clams from day-to-day and patch-to-patch could potentially confound any larger temporal and spatial scale pattern (e.g. monthly, seasonal; between beaches) if not properly accounted for in sampling designs (Hurlburt, 1984). Future long-term and broader-scale ecological and assessment-based sampling will require considerable replication across small temporal and spatial scales. For example, to avoid potential confounding, determination of monthly or seasonal patterns will require inclusion of replicate times (days) of sampling within each temporal scale of interest (Morrisey et al., 1992b). Similarly at each time of sampling, several replicate patches (or sites) spread along a beach (or the scale or features of interest) must be sampled (James and Fairweather, 1996). Because small-scale spatial variation among replicate samples was consistently large compared to temporal variation, this suggests that if sampling of several patches or locations across or among beaches is spread across several days, then the resulting pattern would most likely be a consequence of spatial variation, as opposed to temporal variation. This could assist in defining future sampling strategies for the study clam species, but such patterns may not be the same across different taxa and systems.

In contrast to densities, variations in the size compositions of clams were not greatest at the smallest temporal scales examined, indicating they were relatively stable over small temporal scales and that the ecological processes influencing the size compositions of clams operate at different temporal scales to that of densities. The analyses identified that day contributed little to the components of variation, which further suggested that all sizes of sampled clams responded in a similar way to small temporal changes in the swash habitat. For the other scales the results were inconsistent, with each scale having varying degrees of importance depending on the beach. For example, on Smoky most variation in size compositions occurred between weeks within each month, particularly in Season 2, whereas on Killick there was little variation between weeks with most being observed between months within each season and between seasons. The variation on Smoky in Season 2 was mostly due to the presence or absence of small (< 20 mm) clams in samples. Such variable responses are not uncommon, often making it difficult to determine appropriate temporal scales of sampling (Olabarria and Chapman, 2002; Chapman et al., 2010). Nevertheless, assessments of seasonal or longer-term changes in size compositions will require replicate times (preferably weeks) be sampled within each temporal period. Here, each week comprised 192 replicate samples (48 in each of two areas on each of two days) and in all but one-week (Smoky week 5), at least 200 and up to 750 clams were sampled. Similar levels of sampling will probably be required in future studies to provide sufficient size composition data across the relevant temporal scale of interest.

Like densities, variation in size compositions of clams was greatest among replicate samples, which in this case was the four patches sampled each day in each area. Thus, to account for such spatial variability it would be advisable that clams be sampled across several sites along a beach on each sampling occasion. Here, too few clams were caught in each individual replicate sample to assess variation within each patch. Nevertheless, given the results for densities it could be argued that such variation could be large. Different sized clams could be distributed in different ways throughout the swash zone as a result of small-scale habitat heterogeneity and ecological processes operating across small spatial scales, and this needs closer examination.

4.3. Conclusions

As identified for a range of benthic organisms in other systems (Fraschetti et al., 2005; Chapman et al., 2010), ecological processes operating at small temporal and spatial scales appear important in structuring the distributions and densities of clams along ocean sandy beaches. This could equally be the case for other ocean beach fauna (Gimenez and Yannicelli, 2000; Cook et al., 2015). Thus, small-scale habitat and environmental heterogeneity could potentially be equally or even more important than large-scale physical processes in structuring ocean beach assemblages (Defeo and McLachlan, 2005). Consequently, small-scale variation should not be ignored, but rather incorporated into sampling strategies to determine what local processes operating at small scales influence ocean beach ecology.

Ocean sandy beaches and the organisms that inhabit them are under considerable risk and future sampling to assess natural disturbances (such as storms) and anthropogenic impacts (such as beach nourishment and erosion mitigation programs, fishing and climate-induced environmental changes) on beach fauna need to consider organism responses at small temporal and spatial scales. Indeed, it is at these small local scales that impacts may be most manifest and detectable (Pik et al., 2002; Chapman et al., 2010). Different components of ocean beach assemblages may respond in different ways to ecological processes across vastly different scales of time and space (Defeo and McLachlan, 2005). Knowledge of such relationships will help further our understanding of ocean beach ecology and potential threats.

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Figure 1. Mean (+ SE) density of *Donax deltoides* sampled at each tide stage across two patches on each of four sampling days on Smoky and Killick beaches as part of Experiment 1.

Figure 1

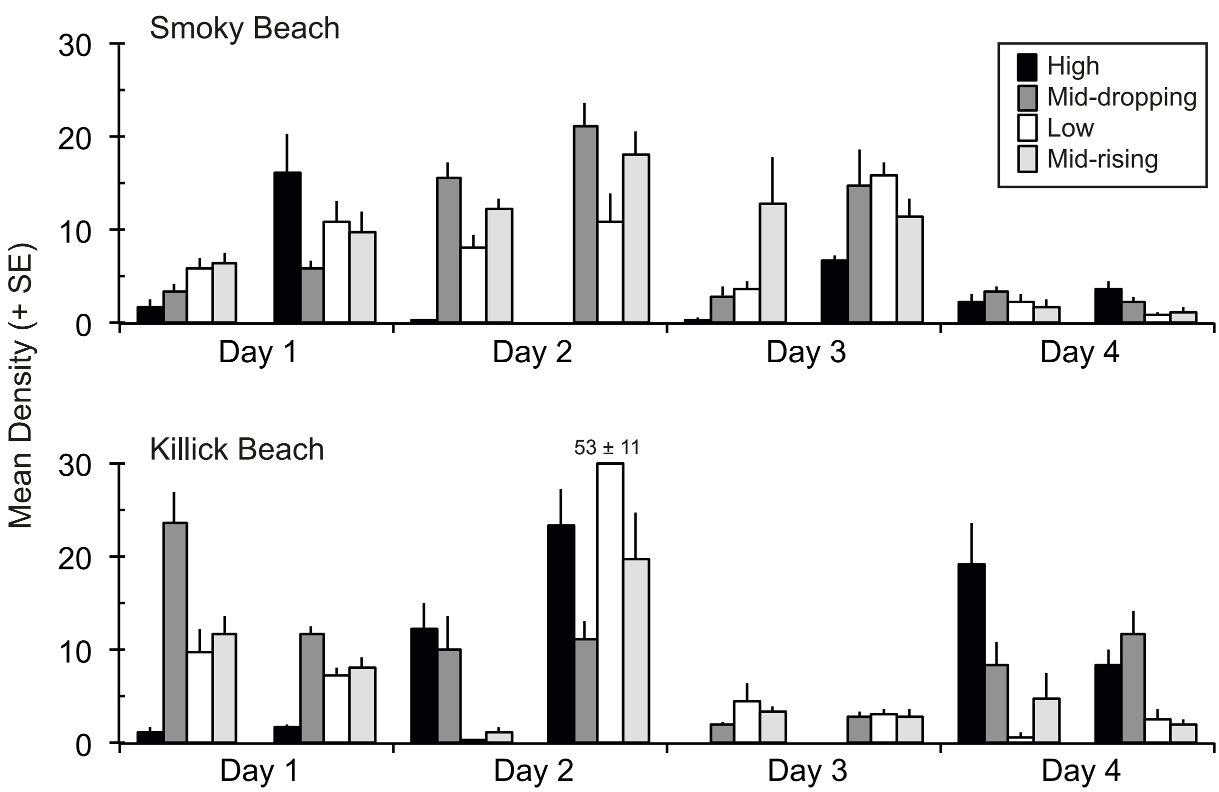

Figure 2. Size compositions of *Donax deltoides* sampled across each tide phase on Smoky and Killick beaches as part of Experiment 1. Data combined across patches and sampling days for each beach.

Figure 2

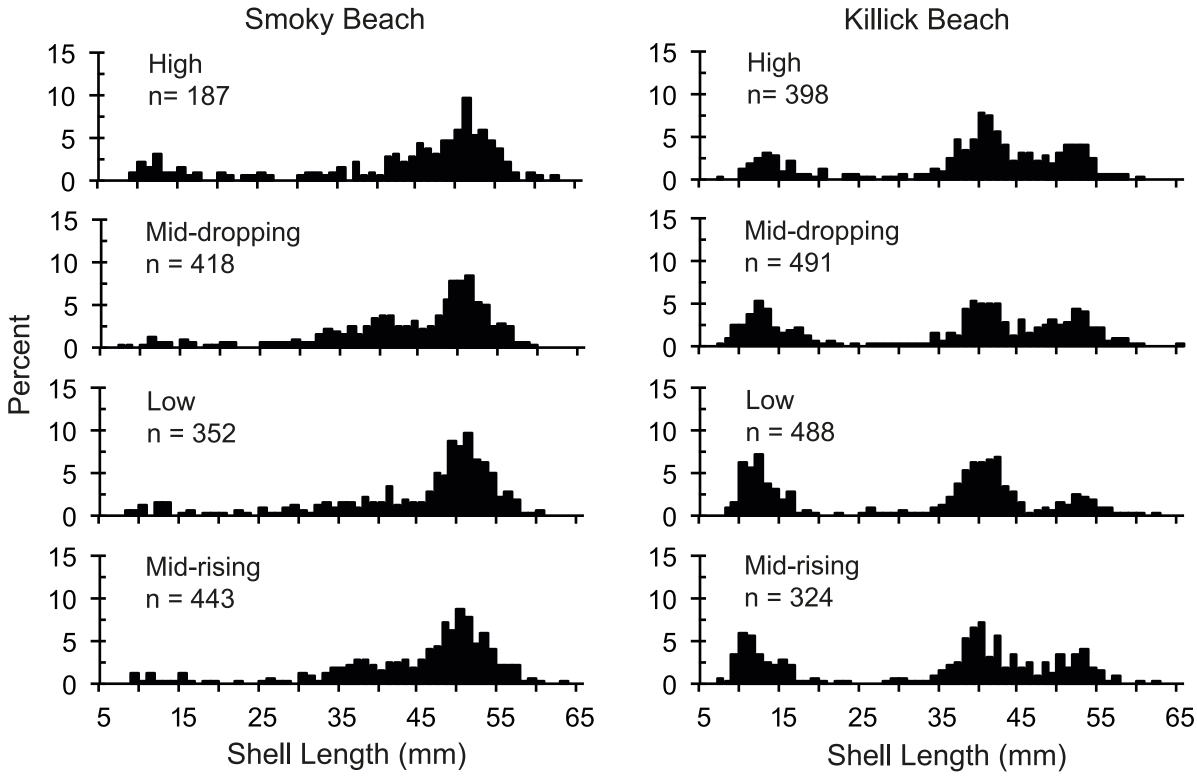

Figure 3. Mean (+ SE) density of *Donax deltoides* sampled across each spatio-temporal scale on Smoky Beach as part of Experiment 2.

Figure 3

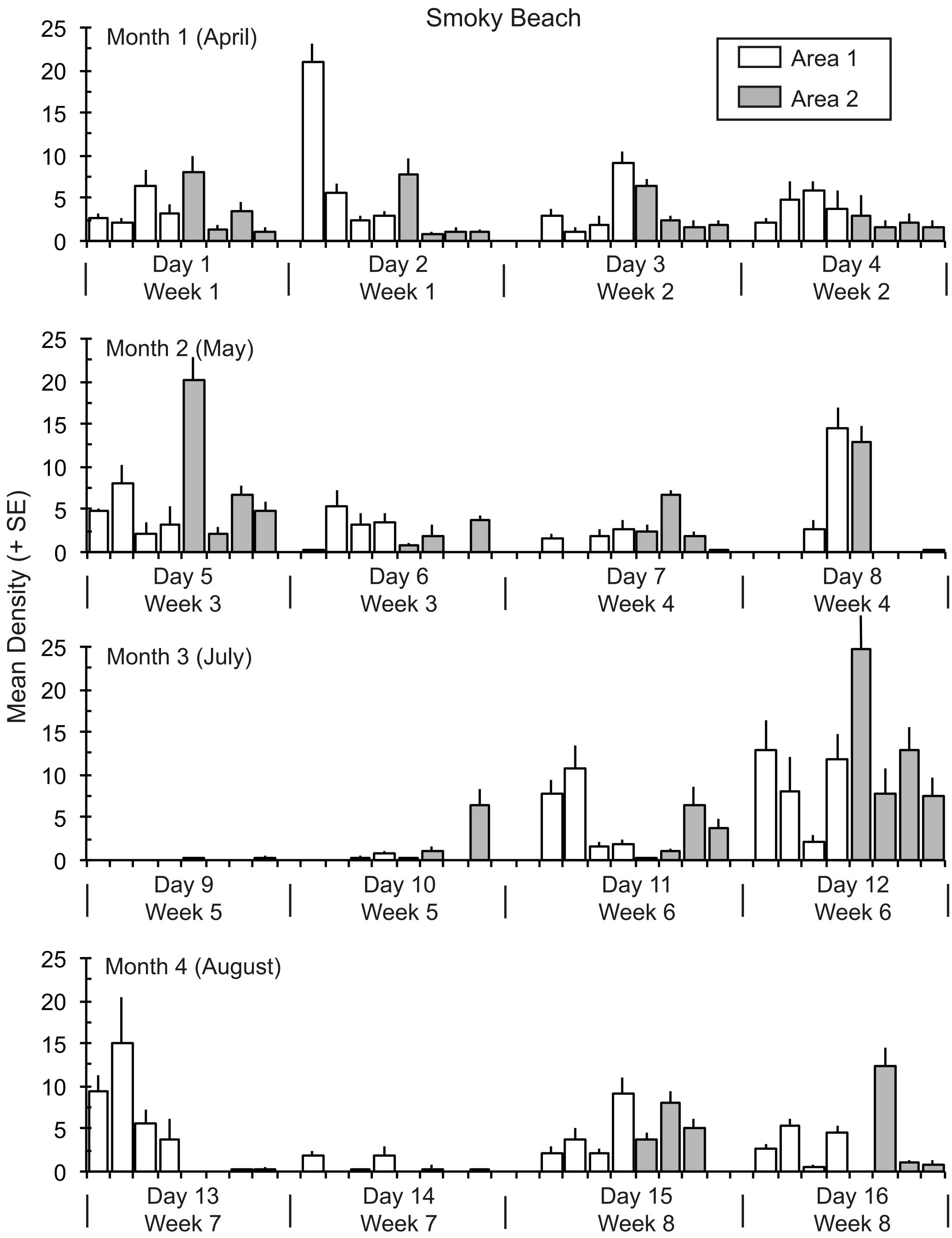

Figure 4. Mean (+ SE) density of *Donax deltoides* sampled across each spatio-temporal scale on Killick Beach as part of Experiment 2.

Figure 4

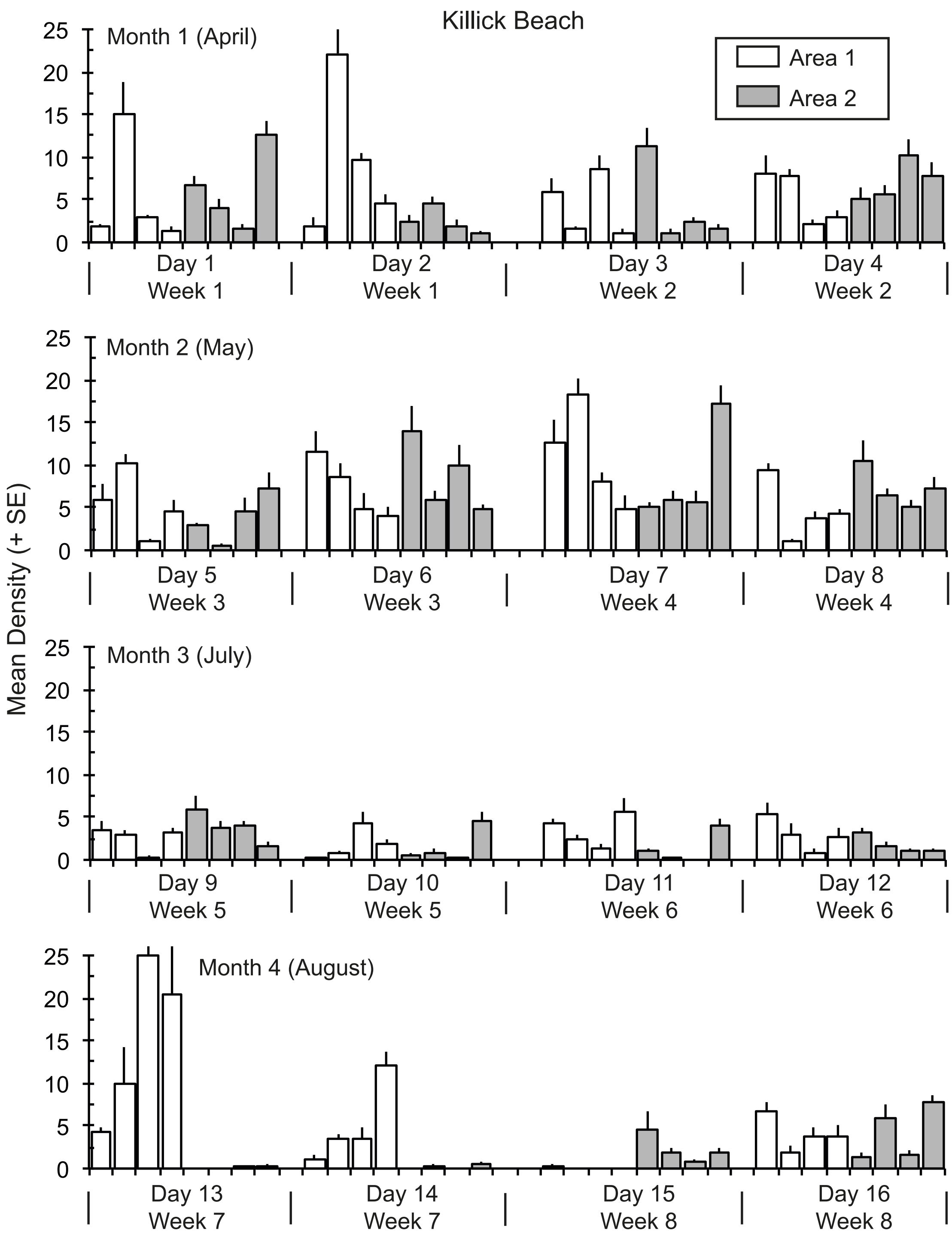

Figure 5. Size compositions of *Donax deltoides* sampled each week on Smoky and Killick beaches as part of Experiment 2. Data combined across patches, areas and the two sampling days in each week for each beach.

Figure 5

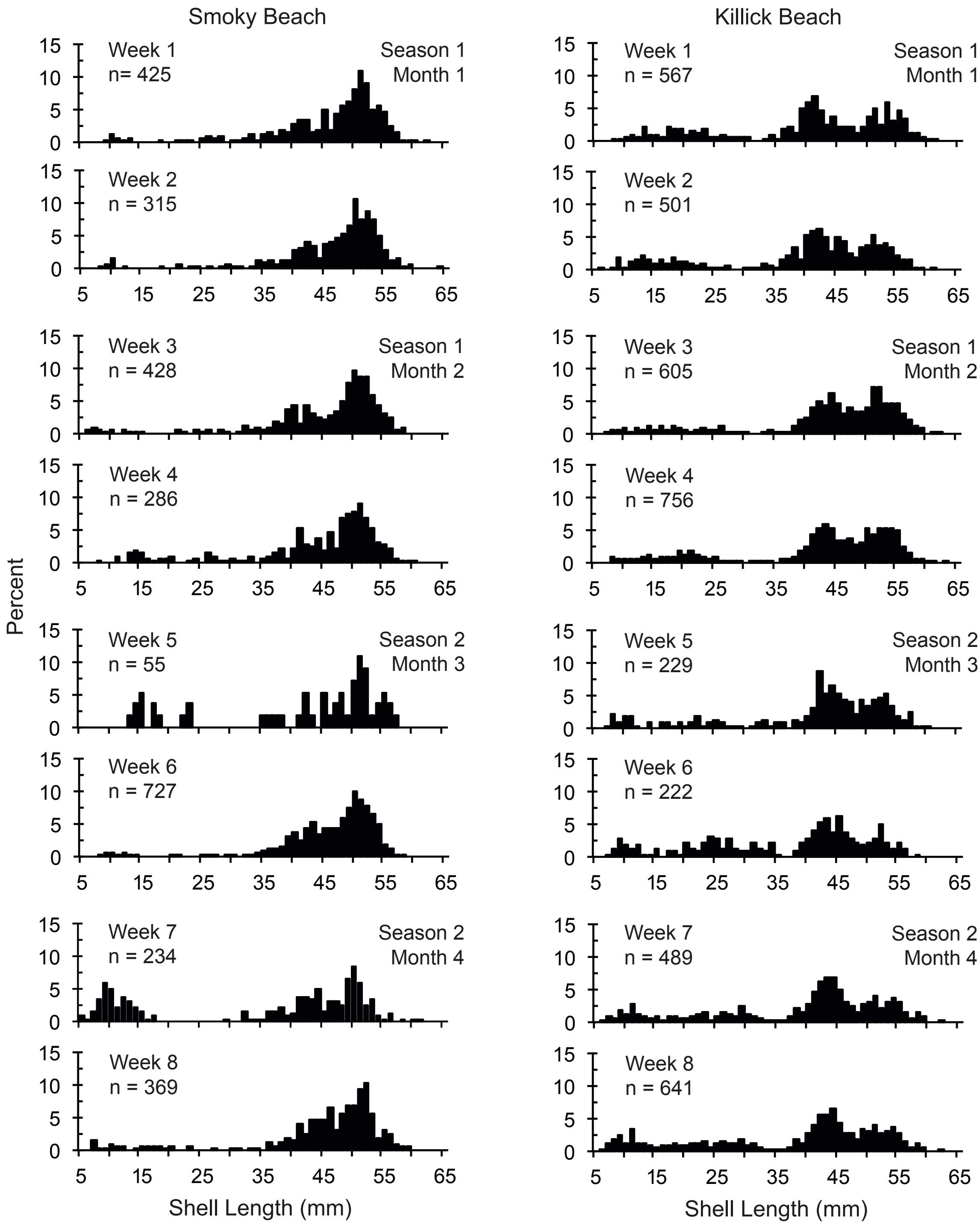

Table 1. Components of variation of densities of *Donax deltoides* for each temporal scale (Season, Month, Week, Day) examined across two separate areas on Smoky and Killick beaches determined from hierarchical PERMANOVAs. The percentage of the total variation given by the Residual in each analysis is provided.

	Smoky Beach		Killick Beach	
	Area 1	Area 2	Area 1	Area 2
Season	0	0	1.01	8.19
Month	0	0	0	0.17
Week	0	1.24	0.11	0.11
Day	0.43	1.50	5.23	2.96
Residual	28.75	32.86	36.13	13.69
% Residual	98.5	92.3	85.0	54.5

Table 2. Components of variation of densities of *Donax deltoides* for each spatial scale (Area, Patch, Residual) examined across the sixteen sampling days on Smoky and Killick beaches determined from hierarchical PERMANOVAs. The percentage of the total variation given by the Residual in each analysis is provided.

	Smoky Beach				Killick Beach			
Day	Area	Patch	Residual	% Residual	Area	Patch	Residual	% Residual
1	0	2.87	9.50	77	0	7.60	32.46	81
2	3.34	42.16	8.15	15	14.42	39.99	8.96	14
3	0	3.23	7.40	70	0	11.68	12.16	51
4	1.91	0.73	4.05	61	0	0	17.82	100
5	0	13.18	33.22	72	0	0.64	16.64	96
6	0.16	2.89	5.78	65	0	10.52	22.63	68
7	0	0.20	5.82	97	0	2.12	39.96	95
8	0	10.66	33.28	76	1.74	7.25	8.23	48
9	0	0.00	0.05	87	0.20	1.99	4.48	67
10	0.11	4.01	3.46	46	0	1.47	4.21	74
11	0	0	24.48	100	1.27	3.05	3.19	42
12	0	19.40	110.06	85	0.22	1.44	4.77	74
13	31.53	7.75	30.13	43	98.70	40.93	40.76	23
14	0.26	0.48	0.88	54	8.83	10.64	4.57	19
15	0	0.34	14.83	98	1.92	0.50	5.00	67
16	0	13.31	9.28	41	0	0.58	10.36	95

Table 3. Components of variation (%) of size compositions of *Donax deltoides* for the two spatial scales examined across Smoky and Killick beaches determined from PERMANOVAs.

	Smoky Beach		Killick Beach	
Week	Area	Residual	Area	Residual
1	39.7	60.3	28.2	71.8
2	0	100	14.7	85.3
3	0	100	0	100
4	0	100	0	100
5	1	99	0	100
6	0	100	53.2	46.8
7	4.3	95.7	31.4	68.6
8	16.6	83.4	0	100

Appendix 4.

Gray CA (2016) Assessment of spatial fishing closures on beach clams. ***Global Ecology and Conservation*** **5, 108-117. (doi:10.1016/gecco.2015.12.002)**

Assessment of spatial fishing closures on beach clams

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ABSTRACT

Spatial fishing closures are typically implemented for conservation and fisheries benefits, but the effects of such initiatives are often not tested. This study examined whether the densities and size compositions of beach clams differed between commercially fished and non-fished zones on beaches. Sampling of clams was stratified across two habitats (swash and dry sand) on two commercially fished beaches, before and during (early and late) the 6-month harvesting period. Two beaches that had no commercial fishing were also sampled the same way and acted as external controls. Differences in densities, but not size compositions, of clams were evident between zones on the commercially fished and control beaches, but they were mostly apparent only across short (day and week) periods before, early and late harvesting, and thus were most likely pulse responses of clams to stochastic, non-fishing related events that acted independently across the different zones on each beach. The potential movements of clams along and across beaches as well as current restrictions on commercial fishing probably dampened detection of longer-term fishing-related impacts and demographic differences in clams between commercially fished and non-fished zones. Direct fishing-related impacts on clams may only be discernable in the immediate vicinity of, and persist for a short period following, an actual fishing event on a beach. Nevertheless, the zones closed to commercial fishing may provide valuable protection to a portion of clams on each beach and alleviate beach-wide harvesting impacts. The broader value of these closed fishing zones requires knowledge of the impacts of fishing on other beach organisms and ecosystem functioning. Further experimentation that tests other aspects of management arrangements of beach clams may help determine their global applicability for sustainable harvesting, and contribute to the overall conservation management of sandy beach ecosystems.

1. Introduction

Spatial closures to fishing are increasingly being incorporated into conservation and fisheries management strategies as a means to provide protection to wild populations of aquatic organisms from human exploitation (Lubchenco et al., 2003; Botsford et al., 2009; Lester et al., 2009). The most notable examples of such measures are no-take marine protected areas and reserves, which compared to openly fished areas have in many instances been shown to enhance the densities and sizes of organisms as well as aquatic biodiversity (Lester et al., 2009; Scieberras et al., 2013). Much of this evidence has been based on work done on fishes and invertebrates inhabiting shallow coastal rocky reefs (Barrett et al., 2007;-Di Franco et al., 2009; Edgar and Barrett, 2012; Guidetti et al., 2014). Few studies have examined the effects of such management initiatives on the fauna inhabiting beaches.

Sandy beaches are the most common type of shoreline bordering the world's oceans and among the most dynamic, but threatened, habitats worldwide (McLachlan and Brown, 2006; Schlacher et al., 2008; Defeo et al., 2009). Ocean beaches are culturally valuable and of high socio-economic importance as they provide extensive ecosystem services to humans (Schlacher et al., 2008; Defeo et al., 2009). Even so, many such beaches support diverse assemblages of benthic invertebrates and other fauna (McLachlan and Brown, 2006; Defeo and McLachlan, 2013). Beach clams (burrowing bivalve molluscs) often dominate the macrofaunal biomass of shallow subtidal and intertidal zones and contribute greatly to the ecology of high-energy ocean beach ecosystems in tropical and temperate regions (McLachlan et al., 1996; Defeo and McLachlan, 2013). Beach clams are also widely harvested for human consumption and bait (McLachlan et al., 1996; Defeo, 2003), but like many exploited benthic invertebrates (Anderson et al., 2011) population declines have been observed in several species (McLachlan et al., 1996; Defeo, 2003; Ortega et al., 2012). Various management initiatives to conserve beach clam populations have been implemented, including closed areas and times to harvesting, and quotas (Castilla and Defeo, 2001; Defeo, 2003). Rarely, however, has the success of such strategies been evaluated in an experimental manner (Walters and Holling, 1990; Underwood, 1995), thus limiting their global applicability for sustainable resource management.

The beach clam *Donax deltoides* (Lamarck, 1818) supports significant fisheries throughout eastern and southern Australia, but in recent years there have been notable declines in population levels across its distribution, the causes of which have not been fully identified (Ferguson and Ward, 2014; Gray et al., 2014). In response to these declines and to appease social conflicts between commercial harvesters and other beach user groups, some east Australian beaches were zoned into fished and non-fished sections to commercial beach clam harvesting in 2010. Further to this, a six-month temporal closure to commercial harvesting was implemented across beaches in 2012 along with the introduction of a minimum shell length (SL) of 45 mm and a 40 kg per-day trip limit. Across all beaches, recreational and indigenous fishers are permitted to catch clams year round, but since 2010 they have not been permitted to remove clams from beaches due to toxin concerns and they can only be retained and used in-situ as bait. The presumed current total harvest from these two sectors is therefore considered low and may be < 5% the total annual commercial harvest (Murray-Jones and Steffe, 2000). The harvesting of clams by all sectors is restricted to digging by hand. The impacts of these management arrangements on beach clams and in particular the potential value of the spatial closures to commercial fishing have not been assessed, and are the subject of investigation here.

The overall goal of this study was to evaluate the potential effects of within-beach spatial closures to commercial harvesting on beach clams. This was done by quantitatively sampling clams across two habitats in the commercially fished and non-fished zones on two beaches, before, early and late harvesting. This was done to specifically test the hypothesis that changes in the densities and size compositions of clams from before to during harvesting would differ between the commercially fished and non-fished zones on beaches. Because the potential impacts of commercial harvesting of clams may not be limited to just the fished zones on beaches but also the non-fished zones, two delineated zones across two non-commercially fished beaches were also sampled in the same way and acted as external controls, thus providing a before versus after, control versus impact (BACI) type assessment.

2. Methods

2.1. Experimental design and sampling

The two commercially fished study beaches were South Ballina (-28.95, 153.51; 30 km long) and Stockton (-32.80, 151.88; 32 km), with the northern 5 and 3 km of each beach, respectively, being closed to commercial fishing. The sampling of the commercially fished zone on each beach was limited to a 6 km section where commercial fishing effort is most concentrated, and immediately abutted the non-fished zone. A total of 6 commercial fishers reported harvesting clams on each beach throughout the study period. The two non-commercially fished control beaches were Sandon (-29.64, 153.32; 7.3 km) and Illaroo (-29.72, 153.30; 9.2 km) and each of these beaches were split into two zones (north and south) of similar size to simulate the management zoning of the commercially fished beaches. All beaches are characteristically fronted by bar and rip systems and exposed to a wide range of ocean conditions (Short, 2007).

Sampling of clams was stratified temporally across three distinct periods, before and during the six-month austral winter-spring (1 June to 30 November) commercial harvesting season for clams in 2013. This was 3 years after the spatial fishing closures, and 1 year after the six-month temporal fishing closure and the size and trip limit restrictions were implemented. The length of each sampling period and the interval between consecutive sampling periods was six weeks. The 'Before' sampling was in April/May when all beaches were totally closed to commercial clam harvesting, the 'Early' harvesting in July/August and 'Late' harvesting in October/November, with sampling beginning 6 and 18 weeks, respectively, after the commencement of the harvesting season. In each of these three periods, sampling was further stratified across two randomly selected days in each of three randomly selected weeks to account for short-term variability in clam densities (Gray, 2016).

Sampling was also stratified spatially across two habitats, the swash zone and the dry sand belt typically located 10 to 30 m above the swash zone at low tide. To account for small-scale spatial variability (Gray, 2016), on each sampling day, four locations in the swash zone and another four locations in the dry sand clam belt were selected at random within each commercially fished and non-fished zone on each commercially fished beach, and in each simulated zone on each control beach. At each of these locations, six replicate samples were taken so that a total of 96 samples were collected each day of sampling on each beach. Sampling was done during daytime within 3 hours either side of low tide (Gray, 2016). It took approximately 4 hours to complete sampling each day and the order in which each zone was sampled was randomly determined each day.

Different sampling gears and methods were used to sample clams in each habitat. Clams in the swash zone were sampled by finger digging for 30 sec a small area (average diameter 57 cm, depth 18 cm) of sand and scooping it into a net that had 12 mm mesh hung on a frame measuring 35 x 21cm (Gray et al., 2014). Clams in the dry sand were sampled by excavating sand to a depth of 20 cm within a square box quadrat that had 22 cm sides (James and Fairweather, 1995). The excavated sand was sieved through a net with 6 mm mesh. All clams collected in each replicate sample were counted and measured for shell length (SL, mm) and operational information including time of sampling and beach and sea conditions were recorded.

2.2. Data Analyses

For each beach, differences between zones in the densities of clams across the 3 harvesting periods were tested using five-factor nonparametric permutational analyses of variance (PERMANOVA; Anderson, 2001). The analytical design had the factors: Zone (fixed), Period (fixed), Week (nested in period – random), Day (nested in week and period - random), Site (nested in zone, day, week and period – random). Separate analyses were done for each habitat (swash and dry sand) on each beach because they were sampled in different ways. Separate analyses were done on the densities of total, legal (≥ 45 mm SL) and sublegal (< 45 mm SL) sized clams. Each univariate analysis was based on the Euclidean distance measure and Type III (partial) sums of squares were calculated using 999 unrestricted permutations of the raw data. The proportion of variation attributable to each factor and interaction in each model was calculated to aid interpretation of the results. All negative variation component values were treated as zero, eliminated from the analysis and the remaining variation components recalculated (Fletcher and Underwood, 2002). Each component directly estimated variability for each term independent of the other terms. All analyses were done using the PRIMER 6 and PERMANOVA^{+} programs (Anderson et al., 2008).

PERMANOVA was also used to test whether the size compositions of sampled populations of clams differed between zones and periods. The proportion of clams in each 5 mm SL class was used to classify samples. Because, clams were not sampled in large densities across all replicate locations and days within each week, the two samples taken in each week were pooled for each zone separately to provide a total of three replicate size compositions at the level of week for each sampling period and zone. Thus, the two factor analytical design for each analysis was: Zone and Period (both fixed). Separate analyses were done for the dry and swash habitats on each beach and each analysis was based on the Bray Curtis dissimilarity matrix with Type III (partial) sums-of-squares calculated using 999 unrestricted permutations of residuals under a reduced model.

3. Results

3.1 Densities of clams

For the two commercially fished beaches there was no significant zone x period effect in any analysis (PERMANOVA, *P-perm* > 0.05, Table 1), indicating there were no detectable large-scale effects of commercial harvesting on densities of clams from before to during (early and late) the harvesting season. The densities of total and legal clams in the dry on South Ballina differed significantly between zones, but these were consistent across periods (Table 1, Fig. 1). Potential short-term fishing effects were identified as: (1) significant zone x week interactions in the densities of clams in the swash on South Ballina, and (2) significant zone x day interactions for the densities of clams in the dry habitat across both habitats (except for sublegal clams in the swash) on Stockton and in the dry on South Ballina (Table 1). Although most pairwise comparisons were limited in power (low number of available permutations) to detect specific differences spatio-temporal differences in densities, they identified that: (1) clams in the swash habitat occurred in lower densities in the non-fished zone on South Ballina across weeks 1 and 7 (Fig. 1); (2) densities of total and legal clams were significantly lower in the non-fished zone in the dry habitat on South Ballina on days 2, 8, 9 and 12, and Stockton on day 7, as well as in the swash on Stockton on days 2, 12 and 13 (Fig. 1), and (3) densities of total and legal clams were significantly lower in the commercially fished zone in the dry habitat on Stockton on days 4 and 17 and in the swash on day 1 (Fig. 1).

For the control beaches, the densities of total and legal clams in the dry were greater in the northern zone on Sandon throughout sampling (PERMANOVA, *P-perm* < 0.05, Table 1, Fig. 2). There were significant zone x period interactions for total and sublegal clams and zone x day interactions for total and legal clams in the swash on Sandon (Table 1). The densities of total and legal clams also differed according to the zone x day interaction in the swash and sublegal clams in the dry on Sandon. The pairwise comparisons identified that densities of total and legal clams in the swash were significantly greater in the northern zone on days 14, 15 and 17 (Fig. 2). There were no significant zone or zone x time interactions for any density parameter in either habitat on Illaroo (Table 1, Fig. 2).

The densities of clams across some beaches also significantly differed according to harvesting period, but these were consistent across zones (i.e. non-significant zone x period interactions, Table 1). The pairwise tests indicated that densities of total clams were significantly lower in the: (1) late harvest period than in the before and early periods in the dry on South Ballina (Fig. 1), and (2) early compared to the before and late periods in the swash on Illaroo (Fig. 2).

The densities of clams across both habitats and all four beaches consistently differed in a significant manner at the smallest scale of sampling (i.e. across sites sampled each day in each zone, Table 1). Moreover, the components of variation in all density analyses were greatest in 22 of 24 analyses for the residual (i.e. among replicate samples), accounting for 30 to 71 % of total variation in each analysis. The factor site contributed the second largest component of variation in 14 of 24 analyses (4-36%). These combined results highlight the dominance of small-scale spatio-temporal patchiness in clams across all beaches. The contribution to total variation was also high for Period on South Ballina (6-20%) and Illaroo (9-15%), and for week on South Ballina (5-37%) and Stockton (9-24%). Zone contributed < 2% of variation across all beaches except Sandon where it accounted for 6 to 11% of variation.

3.2 Size compositions of clams

Across both habitats there were no significant differences in the size compositions of clams between zones on the commercially fished or the control beaches, before or during the harvesting season (PERMANOVA, $P > 0.05$, Table 2, Fig. 3 and 4). In contrast, significant differences in size compositions were evident among some sampling periods, but these were the same across the two zones on each beach (i.e. non-significant Zone x Period interactions, Table 2). These combined results indicated that commercial fishing did not significantly impact size compositions of clams.

There was a general trend across both habitats on each beach for a greater proportion of small (5-20 mm) clams in size compositions in the early and late harvest periods compared to before harvesting (Fig. 3 and 4). The predominant exception being South Ballina, where small clams were only apparent late harvesting. On each beach, the size compositions of clams were generally similar across both habitats within each sampling period.

4. Discussion

For the commercially fished beaches, there were no significant zone-related differences in densities of clams in either the swash or dry habitats greater than the level of week, indicating that the potential effects of commercial harvesting on clam densities were highly variable and ephemeral, being dependent on the particular day or week sampled. This occurred even though during the 6-month fishing season 10500 and 10200 kg of clams were harvested from South Ballina and Stockton beaches, respectively. It is unlikely that the harvesting of clams by recreational and indigenous fishers impacted the results obtained here. Relatively few non-commercial fishers were observed harvesting clams in either the commercially fished or non-fished zones throughout the study, and although their total harvests are unknown, it probably was considerably less than reported total commercial harvests across each beach (Murray-Jones and Steffe, 2000). It is also unlikely that commercial harvesters extensively worked the non-fished zones on either beach due to a combination of strong industry codes, local community awareness and fisheries compliance.

Significant spatio-temporal interactions (zone x period and zone x day) in clam densities were also evident in the swash habitat across one control beach: Sandon. Moreover, densities of clams in the dry habitat on Sandon differed between zones in a consistent manner across all periods. These combined results demonstrate that densities of clams can naturally fluctuate between designated zones or sections along beaches across short (days) and longer (months) temporal scales. Whilst the reasons for this are not apparent here, it means that the observed zone x time interactions on the commercially fished beaches may have been the result of natural processes unrelated to commercial fishing. Delineating between these alternative hypotheses is difficult, especially given that densities of clams across both habitats were often lower in the non-fished zones than the commercially fished zones, and that such interactions occurred before and during harvesting. Many such interactions, therefore, were probably pulse responses (Underwood, 1989) of clams to natural small-scale stochastic processes, such as local changes in wave or beach conditions, operating independently in the different zones along each beach (McLachlan and Hesp, 1984).

In contrast to densities, the size compositions of clams across both habitats did not significantly differ between zones on either the fished or control beaches, but this was only assessed at the scale of period due to sample size considerations. Although greater proportions of sublegal clams were present in the early and late compared to the before harvest period, this was consistent across both zones on each beach and due to the recruitment of small (10-20 mm SL) clams, and not the result of truncation of larger animals (i.e. due to harvesting). This austral winter/spring timing of recruitment of small clams concurs with reported spawning periods (Ferguson and Ward, 2014).

Current restrictions on commercial fishing may have dampened the detection of longer-term harvesting impacts and concomitant differences between management zones in densities and sizes of clams. Commercial fishers can often harvest 40 kg of clams across a relatively small stretch of swash habitat (< 100 m), and from a small area (< 50 m^2) of dry habitat, in < 1 hour (unpublished data). Thus, the immediate (and cumulative total fishing season) environmental footprint left by this scale of harvesting may not be broad enough to be detected across large spatio-temporal scales. It is hypothesized that under current management arrangements, fishing-related impacts may only manifest across local spatial and temporal scales on a beach. For example, the few instances when clam densities were lower in the commercially fished zone (e.g. days 14 and 17 in the dry on Stockton) may have resulted from commercial fishers (coincidentally) harvesting clams at the actual sampling sites (as opposed to the general vicinity) shortly before (e.g. previous tide or day) sampling occurred. Whilst this cannot be tested here, sampling across small scales immediately before and after actual harvesting events could potential identify the extent and longevity of localized fishing-related impacts on clams (Carvalho et al., 2013).

Beach clams are mobile organisms that actively move along and across beaches depending on ocean and beach conditions (Leber, 1982; Ellers, 1995; Dugan and McLachlan, 1999), with large seas and storm events also potentially redistributing clams across each spatial dimension. It is reasonable to assume that individual clams may have actively migrated between management zones on each fished beach, thereby masking detection of any potential longer-term zone-related differences in densities and size compositions. Quantifying the extent and the mechanisms that drive the translocation of clams along and across beaches is an important avenue of research that will assist in determining the value of closed fishing zones and their potential conservation benefits.

The lack of significant longer-term impacts (i.e. zone x period) on the densities and sizes of clams does not imply commercial harvesting at the levels reported here has no impact on populations, and similarly that the zones closed to commercial fishing do not provide conservation benefits to clams. Indeed, like other no- and partial-take fishing closures, the non-fished zones may provide necessary (albeit even temporary) refuge to a proportion of the total clam population on each beach, potentially alleviating population-wide impacts of commercial harvesting (Gell and Roberts, 2003; Botsford et al., 2009). The non-fished zones sampled here represented < 20% of available habitat on each beach, and more research is required to determine whether this is adequate for sustainability purposes (Halpern, 2003). However, many east Australian beaches are now totally closed to clam harvesting and the total area that is protected across all beaches, as well as the actual area on beaches that fishers actively utilize for harvesting, needs to be considered in the overall management of the resource. In particular, the broader ecological effects of alternative levels of protective areas and harvesting on total reproductive output (Brazeiro and Defeo, 1999) and linkages with recruitment need to be assessed. Moreover, the potential impacts of clam harvesting on other beach organisms and ecosystem functioning is required for a more holistic approach to protective zoning strategies and the overall management of sandy beaches.

The zoning of beaches into commercially fished and non-fished zones was in part implemented to alleviate social conflicts among different beach user groups. Although the success of this objective was not assessed here, it needs to be addressed as it is imperative that management initiatives whether they are for social, economic or biological reasons be tested (Underwood, 1995). Such knowledge will ultimately help determine the broader applicability of such strategies for managing social issues on sandy beaches (Charles and Wilson, 2009; McLachlan et al., 2013).

This study was the first experiment to test for the effects of an implemented fisheries spatial management strategy on beach clams in eastern Australia, and among the few done globally (Defeo et al., 2009). In doing so, it highlighted the difficulties in determining potential impacts of fishing and the subsequent value of spatial fishing closures on clams, particularly given the current levels of commercial harvests in the study fishery. Nevertheless, the study demonstrated the necessity of incorporating appropriate external controls (i.e. the non-fished beaches) in evaluating management strategies, as well as the value of hierarchical sampling schemes that allow for response measures of potential impacts across different spatial and temporal scales. Such measures are not only important for management, but for assessing the relevant scales of ecological processes and their subsequent influences on assemblages (Underwood et al., 2000; Gray, 2016). Along with a greater understanding of the spatial dynamics and connectivity of clams along and among beaches, further experimental studies are required to evaluate other aspects of management initiatives concerning clam fisheries, including temporal closures, quotas and legal size restrictions. Such information will help assess the global applicability of alternative management initiatives for sustainable clam harvesting, and contribute towards the greater conservation and management of sandy beach ecosystems.

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Figure 1. Mean (+ SE) densities of *Donax deltoides* sampled in the swash and dry habitats across the commercially fished and non-fished zones on South Ballina and Stockton beaches before, early and late harvesting 2013.

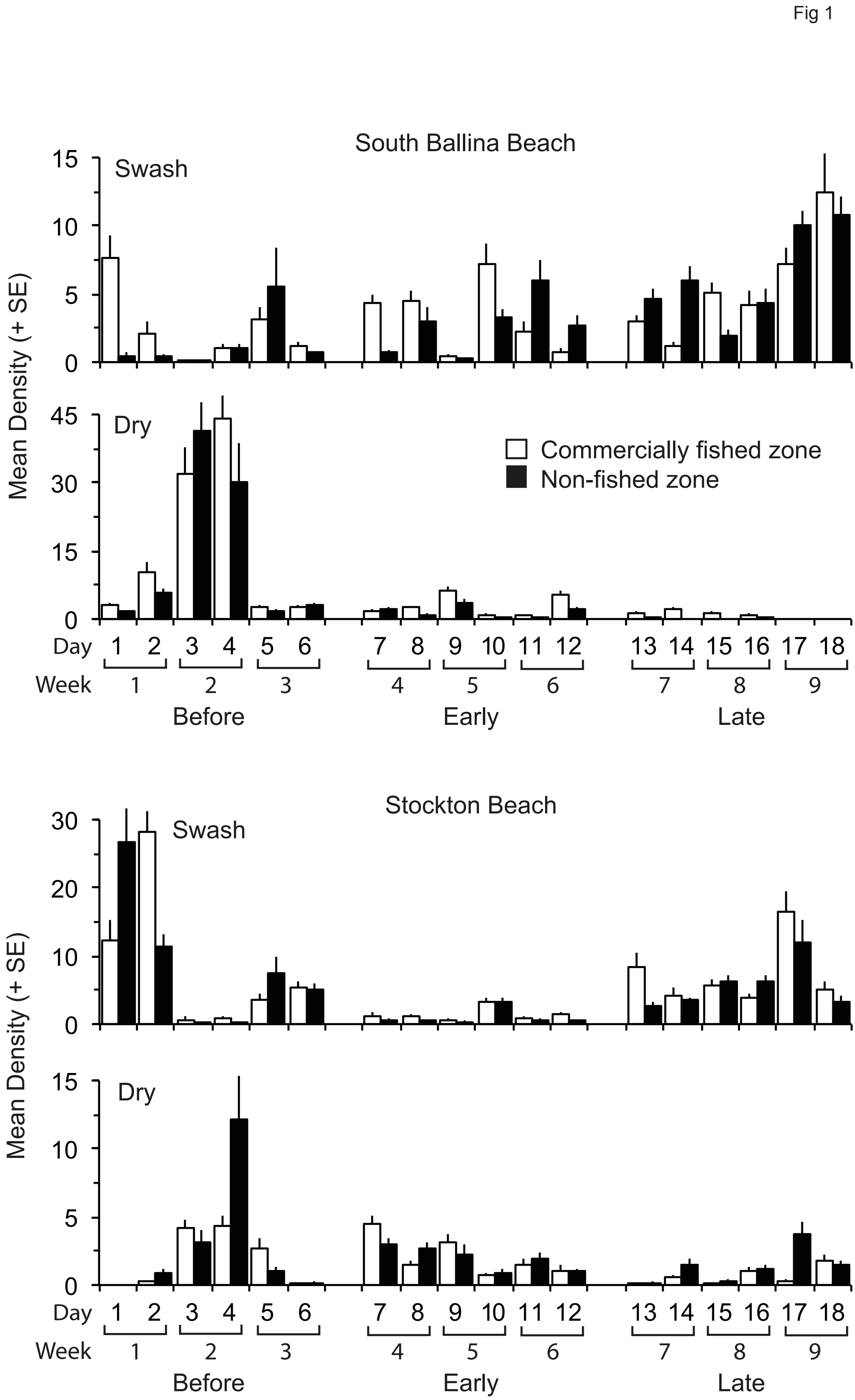

Figure 2. Mean (+ SE) densities of *Donax deltoides* sampled in the swash and dry habitats across the two non-fished zones on Sandon and Illaroo beaches before, early and late harvesting 2013. NS = not sampled.

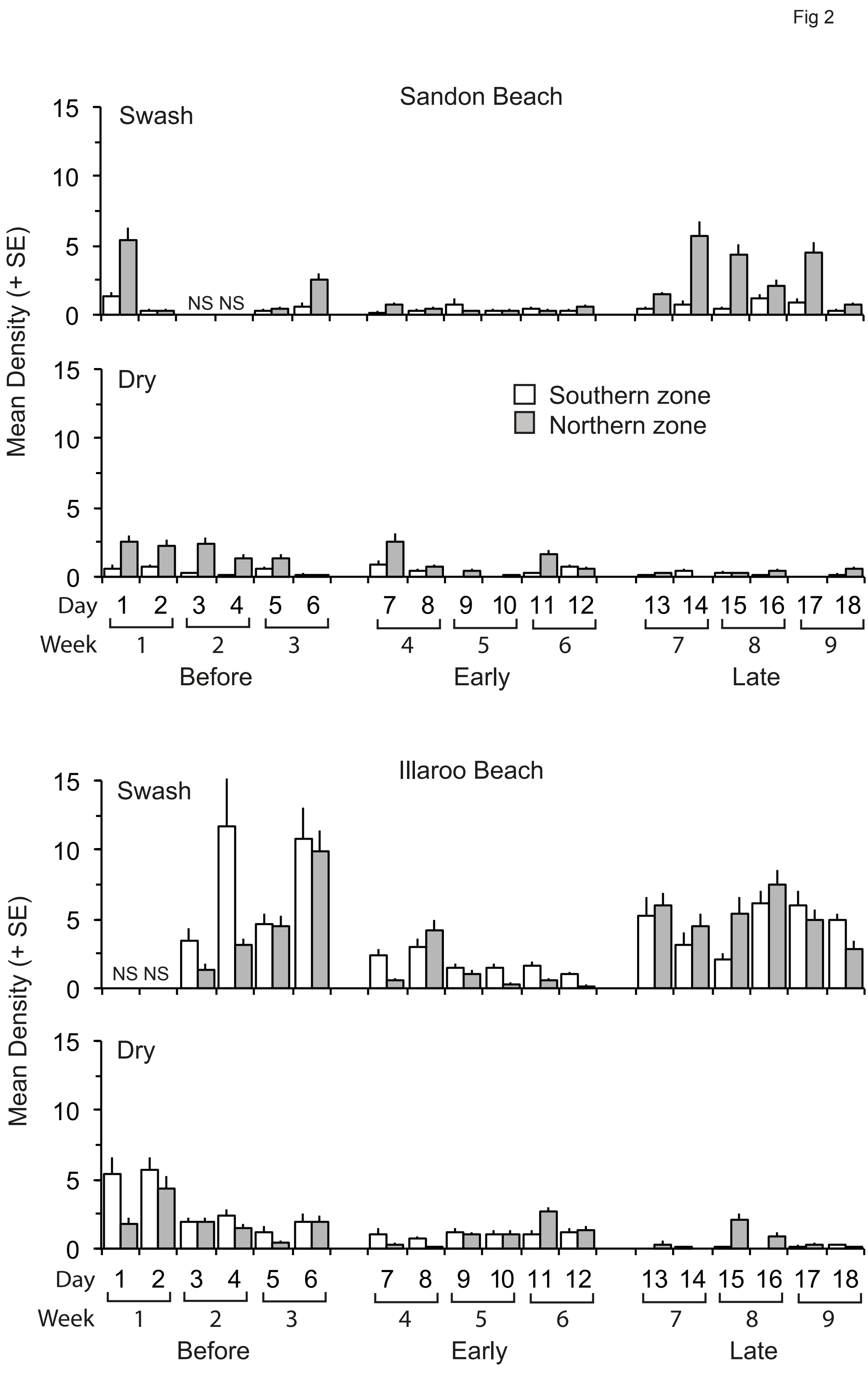

Figure 3. Size compositions of *Donax deltoides* sampled in the swash and dry habitats across the commercially fished and non-fished zones of South Ballina and Stockton beaches before, early and late harvesting 2013. Sample sizes are shown on each graph. Shading as in Fig. 2; CF = commercially fished zone, NF = non-fished zone.

Fig. 3

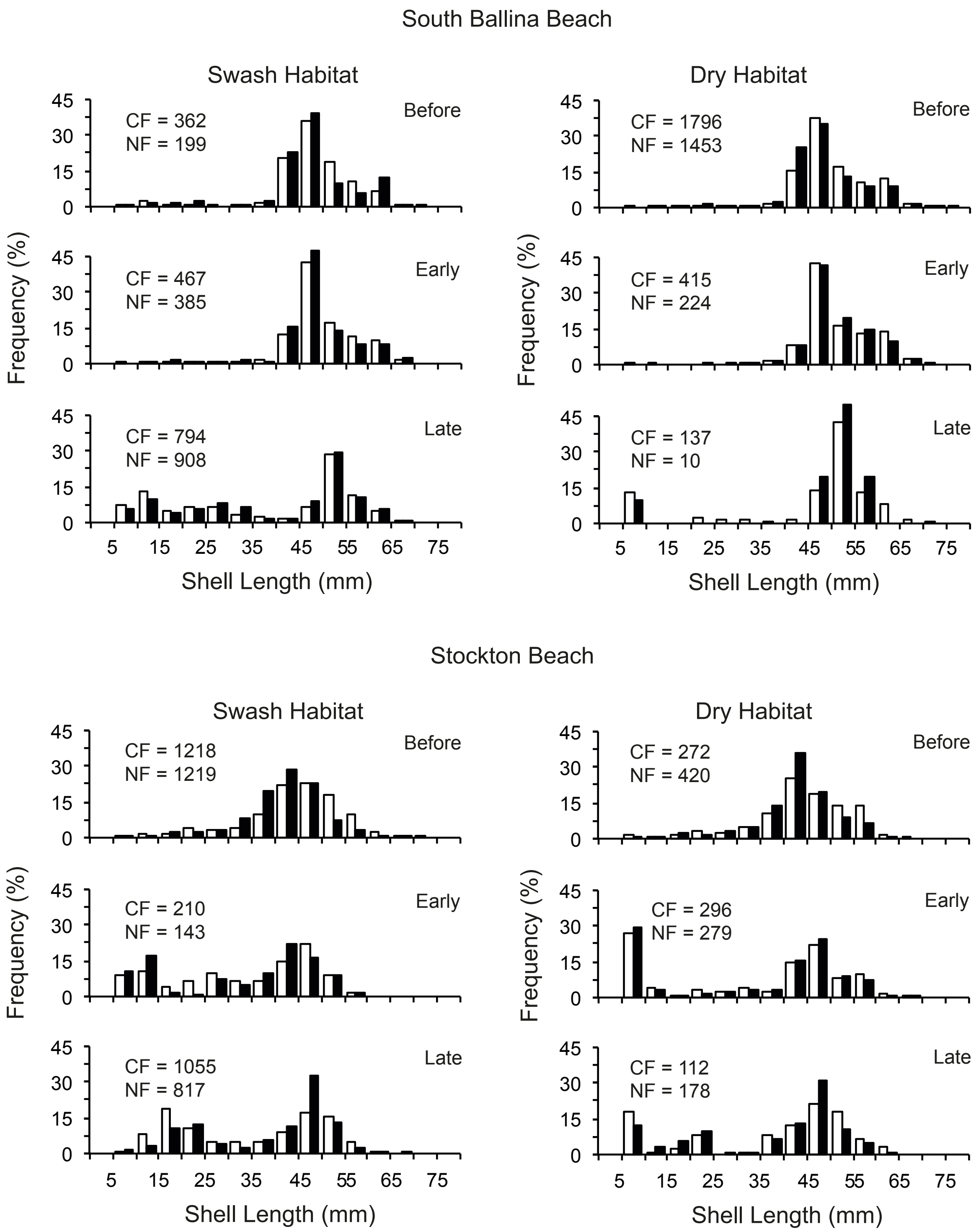

Figure 4. Size compositions of *Donax deltoides* sampled in the swash and dry habitats across the two non-fished zones of Sandon and Illaroo beaches before, early and late harvesting 2013. Sample sizes are shown on each graph. Shading as in Fig. 3; S = southern zone, N = northern zone.

Fig. 4

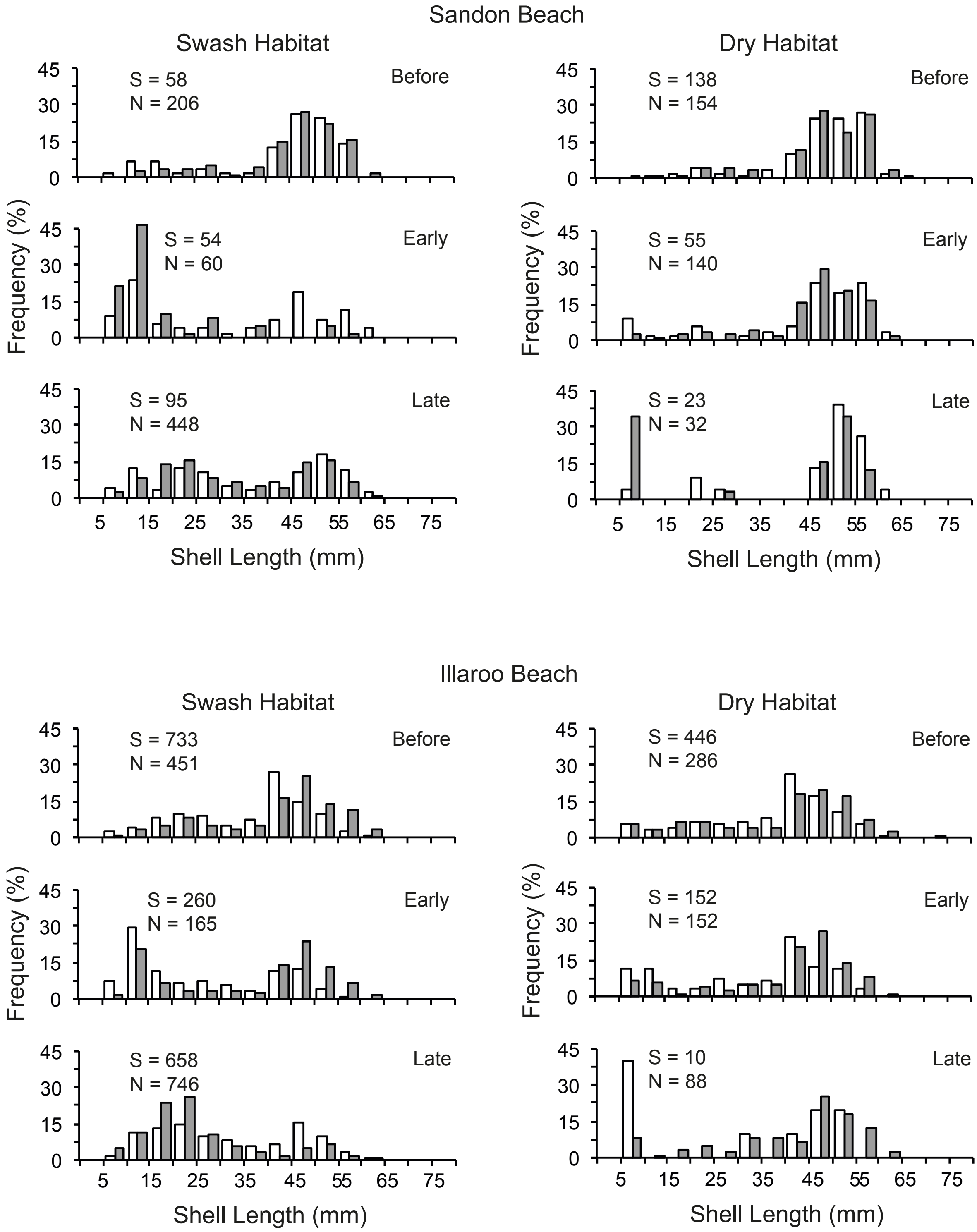

Table 1
Results of PERMANOVAs comparing densities of total, legal and sublegal sized clams across commercially fished and non-fished zones on South Ballina and Stockton beaches, and across zones on the control beaches of Sandon and Illaroo. Clams. * = Pseudo-F < 0.5, ** < 0.01, *** < 0.001, ns > 0.05. Bold and shaded terms are those that if significant might signify a possible effect of management zoning and fishing impact.

		Swash Habitat				Dry Habitat		
Commercially-fished beaches	df	Total	Legal	Sublegal	df	Total	Legal	Sublegal
South Ballina Beach								
Zone	1	ns	ns	ns	1	**	**	ns
Period	2	**	ns	*	2	**	*	**
Week(Period)	6	ns	ns	ns	6	*	*	*
Zone x Period	2	ns	ns	ns	2	ns	ns	ns
Day(Week(Period))	9	ns	ns	ns	9	*	**	ns
Zone x Week(Period)	6	ns	*	**	6	ns	ns	ns
Zone x Day(Week(Period))	9	ns	ns	ns	9	***	**	**
Site(Zone x Day(Week(Period)))	108	***	***	***	108	***	***	***
Residual	720				720			
Stockton Beach								
Zone	1	ns	ns	ns	1	ns	ns	ns
Period	2	ns	ns	ns	2	ns	ns	ns
Week(Period)	6	**	*	**	6	**	*	*
Zone x Period	2	ns	ns	ns	2	ns	ns	ns
Day(Week(Period))	9	ns	***	ns	9	**	**	ns
Zone x Week(Period)	6	ns	ns	ns	6	ns	ns	ns
Zone x Day(Week(Period))	9	***	**	**	9	*	**	ns
Site(Zone x Day(Week(Period)))	108	***	***	***	108	***	***	***
Residual	720				720			
		Swash Habitat				Dry Habitat		
Control beaches	df	Total	Legal	Sublegal	df	Total	Legal	Sublegal
Sandon Beach								
Zone	1	**	*	***	1	**	**	**
Period	2	*	ns	**	2	ns	ns	ns
Week(Period)	5	ns	ns	ns	6	ns	ns	ns
Zone x Period	2	**	ns	**	2	ns	ns	ns
Day(Week(Period))	8	***	***	*	9	*	*	*
Zone x Week(Period)	5	ns	ns	ns	6	ns	ns	ns
Zone x Day(Week(Period))	8	***	***	ns	9	ns	ns	*
Site(Zone x Day(Week(Period)))	96	***	***	***	108	***	***	*
Residual	640				720			
Illaroo Beach								
Zone	1	ns	ns	ns	1	ns	ns	ns
Period	2	**	**	**	2	*	*	*
Week(Period)	5	ns	ns	ns	6	*	ns	**
Zone x Period	2	ns	ns	ns	2	ns	ns	ns
Day(Week(Period))	8	**	**	**	9	ns	ns	ns
Zone x Week(Period)	5	ns	ns	ns	6	ns	ns	ns
Zone x Day(Week(Period))	8	ns	ns	ns	9	ns	ns	ns
Site(Zone x Day(Week(Period)))	96	***	***	***	108	***	***	***
Residual	640				720			

Table 2

Summary results of PERMANOVA and subsequent pairwise tests comparing the size compositions of clams across zones and harvest periods in the swash and dry habitats on each fished and non-fished beach. ** = Pseudo-F < 0.01, ns = Pseudo-F > 0.05, df = degrees of freedom, B = before, E = early, L = late harvest

		Commercially fished beaches			
		South Ballina	South Ballina	Stockton	Stockton
Source of Variation	df	Swash	Dry	Swash	Dry
Zone	1, 12	ns	ns	ns	ns
Period	2, 12	***	**	ns	**
Zone x Period	2, 12	ns	ns	ns	ns
Pairwise Period		B=E, B≠L, E≠L	B≠E, B≠L, E≠L		B≠E, B≠L, E=L

		Control beaches			
		Sandon	Sandon	Illaroo	Illaroo
Source of Variation	df	Swash	Dry	Swash	Dry
Zone	1, 12	ns	ns	ns	ns
Period	2, 12	**	ns	**	ns
Zone x Period	2, 12	ns	ns	ns	ns
Pairwise Period		B≠E, B=L, E≠L		B≠E, B≠L, E≠L	

Appendix 5.

Gray CA (2016) Effects of fishing and fishing closures on beach clams: experimental evaluation across commercially fished and non-fished beaches before and during harvesting. *PLoS One* 11, e0146122. (doi:10.1371/journal.pone.0416122)

Effects of fishing and fishing closures on beach clams: experimental evaluation across commercially fished and non-fished beaches before and during harvesting

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Running title: Gray: Fishing impacts on beach clams

KEY WORDS: Management evaluation; Harvest strategy; Human impact; Population Assessment; Exploitation; Fishery closure; Bivalve mollusc; *Donax deltoides*

ABSTRACT: Management responses to reconcile declining fisheries typically include closed areas and times to fishing. This study evaluated this management strategy by testing the hypothesis that changes in the densities and sizes of the beach clam *Donax deltoides* from before to during harvesting would differ between commercially fished and non-fished beaches. Sampling was spatially stratified across eight sites within the swash and dry sand habitats on each of two commercially fished and two non-fished beaches, and temporally stratified across six days in each of three six-week blocks; before, early and late harvesting. Small-scale spatio-temporal variability in the densities and sizes of clams was prevalent across both habitats and the components of variation were generally greatest at the lowest levels examined. Despite this, differences in the densities and sizes of clams among individual beaches were evident, but there were no global differences across the commercially fished versus non-fished beaches. There was no evidence of reduced densities or truncated size compositions of clams on fished beaches, which contrasts reports of greater densities and sizes of organisms in protected areas. This was probably due to a combination of factors, including the current low levels of commercial harvests, the movements and other local scale responses of clams to ecological processes acting independently across individual beaches. The results identify the difficulties in detecting fishing-related impacts against natural levels of variability in clam populations. Nevertheless, continued experimental studies that test alternative management arrangements may help refine and determine the most suitable strategies for the sustainable harvesting of beach clams, ultimately enhancing the management of sandy beaches.

1. Introduction

Fishing has had detrimental impacts on wild populations and assemblages of aquatic organisms of various phyla across a spectrum of habitats throughout the world (Dayton et al. 1995, Pauly et al. 1998, Stevens et al. 2000, Jackson et al. 2001). In particular, many harvested species have experienced substantial population declines as well as changes in demographic characteristics such as truncation of size and age composition, reduced sizes and ages at reproduction, altered growth rates and mortality schedules (Trippel 1995, Rochet 1998, Jennings et al. 1999, Bianchi et al. 2000, Sharpe & Hendry 2009, Enberg et al. 2012). The impacts and responses of organisms to fishing can vary considerably depending on the type, intensity and history of fishing activities, as well as the life history characteristics and resilience of individual species and populations (Jennings et al. 1998, Marty et al. 2014).

Initiatives to reconcile the effects of fishing and provide greater protection to wild organisms and habitats include areas and times either fully or partially closed to fishing (Lubchenco et al. 2003, Lester et al. 2009), fishing gear restrictions and modifications (Broadhurst 2000), catch and bycatch quotas and size and bag limits (Cooke & Cowx 2006). Several such measures have been shown to be effective across different fisheries and landscapes. For example, no-take fishing areas can restore densities and size compositions of harvested species, and help maintain ecosystem biodiversity and functioning (Barrett et al. 2007, Russ et al. 2008, Lester et al. 2009, Edgar & Barrett 2012). Similarly, modifications to fishing gears can reduce levels of catches of unwanted species as well as damage to habitats (Broadhurst et al. 2014). Nevertheless, in many cases the effects of implemented management arrangements have not been tested. Ideally, the success or failure of such management measures should be evaluated experimentally as part of an adaptive management regime (Walters & Holling 1990, Underwood 1995).

Beach clams (Bivalvia: Donacidae, Mesodermatidae, Veneridae) are harvested for food and bait on sandy beaches worldwide (McLachlan et al. 1996, Defeo 2003), but because they primarily inhabit the intertidal and shallow subtidal they are easily accessible and relatively simple and cheap to harvest, making them readily susceptible to over exploitation (Defeo 2003). Indeed, populations of several species have over relatively short periods of time been depleted (McLachlan et al. 1996, Defeo 2003), a trend observed for many other exploited invertebrates (Anderson et al. 2011). This scenario could also be true for the Australian beach clam *Donax deltoides* (Ferguson & Ward 2014, Gray et al. 2014). For example, in the state of New South Wales (NSW) alone, following the developmental phase of the fishery in the 1950s total reported commercial landings of *D. deltoides* increased to peak at 670,000 kilograms (kg) in 2001, after which it fell (along with commercial catch-per-unit-effort) to only 9,000 kg in 2011, despite increasing product prices and markets (Rowling et al. 2010). Throughout this time, recreational and indigenous catches were unrestricted and unchecked but were probably large across many beaches (Murray-Jones & Steffe 2000, Henry & Lyle 2003). Although the reasons for the rapid decline in commercial catches and catch rates are unclear, fishing was probably a contributing factor (Ferguson & Ward 2014).

Management responses to declining beach clam fisheries have usually included closed areas and times to fishing, trip and size limits and had varying degrees of success (i.e. when actually tested) (McLachlan et al. 1996, Brazeiro & Defeo 1999, Defeo 2003). In response to the fall in commercial catches and other reported broader population declines of *D. deltoides*, several management initiatives designed to reduce fishing effort and harvest and stabilize the fishery, and therefore halt further population declines, were introduced to the NSW fishery in 2012. The strategy incorporated a six-month total commercial fishing closure, spatially explicit commercial fishing closures of whole beaches and specific zones along particular beaches, a maximum daily catch quota of 40 kg per-commercial fisher, as well as the introduction of a minimum legal size limit (45 mm shell length, SL). Recreational and indigenous fishers remain permitted to catch clams year round across all beaches, but due to concerns over toxins they can now only use clams as bait in-situ and cannot remove them from beaches. Because of this, the combined harvest from these two sectors is estimated to be small and may be as little as 2% the commercial harvest (Murray-Jones & Steffe 2000, Rowling et al. 2010). The harvesting of clams by all sectors is restricted to digging by hand, with no mechanical apparatus permitted.

This study was done in response to the above management arrangements being implemented in the NSW commercial beach clam fishery and the first to examine the potential impacts of fishing on beach clams by comparing populations across beaches open and closed to commercial fishing, both before and during the harvesting season. The specific hypothesis tested was that changes in the densities and size compositions of *D. deltoides* from before to during (early and late) the six month harvesting season would differ between commercially fished and non-fished beaches. It was predicted that the densities of clams would decline and their size compositions become truncated throughout the fishing season on commercially fished compared to non-fished beaches. The results are discussed in terms of identifying fishing impacts, assessing management strategies and sandy beach ecology.

2. Materials and Methods

2.1. Experimental design and sampling

This study was done across four high-energy ocean sandy beaches in eastern Australia: Ten Mile (length - 28.5 km), Sandon (7.3 km), Illaroo (9.2 km) and Smoky (16.0 km) (Fig 1). Each beach is enclosed between rocky headlands, fronted by rip-dominated bar systems and exposed to seas from the north, east and south directions (Short 2007). Sandon and Illaroo have a history of sporadic commercial harvesting of clams and were closed to harvesting throughout the study. In contrast, both Ten Mile and Smoky have a strong history of continuous commercial harvesting for clams and were open to commercial clam harvesting between 1 June and 30 November.

Sampling of clams on each beach was stratified temporally across three discrete periods, before and during the six-month commercial harvesting season in 2013. The length of each sampling period and the interval between consecutive sampling periods was 6 weeks. Period 1 (before) was in April/ May when all beaches were totally closed to commercial clam harvesting. Period 2 (early harvesting) was in July/August and Period 3 (late harvesting) was in October/ November with sampling beginning 6 and 18 weeks, respectively, after the commencement of harvesting on 1 June 2013.

Clams were sampled across two habitats, the swash zone and the dry sand belt typically located between 10 and 30 m above the low-tide swash zone level on each beach. In each of the three periods, sampling was done across two randomly selected days in each of three randomly selected weeks, except for the swash habitat in Period 1 when only four days (two weeks) were sampled. On each sampling day, eight sites in the swash zone and another eight sights in the identified clam belt in the dry sand were selected at random on each beach. At each of these locations, six replicate samples were taken using the appropriate sampling gear. A total of 96 samples were therefore collected each day of sampling on each beach. Sampling was done during daytime within 3 hours either side of low tide (Gray et al. 2014) and it took approximately 4 hours to complete sampling each day.

Different sampling methods were used to sample clams in each habitat. Clams in the swash zone were sampled by finger digging for 30 sec a small area (average diameter 57 cm, depth 18 cm) of sand and scooping it into a net that had 12 mm mesh hung on a frame measuring 35 x 21cm (Gray et al. 2014). Clams in the dry sand were sampled by excavating sand to a depth of 20 cm within a square box quadrat that had 22 cm sides (James & Fairweather 1995), after which the excavated sand was sieved through a bag with 6 mm mesh. All clams collected in each replicate sample were counted and measured for shell length (SL, mm) and operational information including time of sampling and beach and sea conditions were recorded.

2.2. Data Analyses

Differences in the densities of total, legal and sublegal sized clams between commercially fished and non-fished beaches, before and during the harvesting season were tested using PERMANOVA (Anderson 2001). The specific model included the factors: Beach Management Type (i.e. commercially fished v non-fished - fixed), Beach (nested in Beach Management Type - random), Period (Before, Early & Late harvest - fixed), Day (nested in Period and Beach - random), Site (nested in Beach, Day and Period– random). Separate analyses were done for each habitat (swash and dry sand) because they were sampled in different ways. Each analysis was based on the Euclidean distance measure and Type III (partial) sums of squares were calculated using 999 unrestricted permutations of the raw data. All analyses were done using the PRIMER 6 - PERMANOVA$^+$ program (Anderson et al. 2008).

PERMANOVA was also used to test whether the size compositions of sampled populations of clams differed between commercially fished and non-fished beaches and across sampling periods. The proportion of clams in each 5 mm size class was used to classify samples. Because, *D. deltoides* were not always sampled in large densities at each site on each sampling day, data were pooled across all eight sites sampled on each day on each beach. Days when no clams were caught were omitted from the analyses. The total size composition data for each day were then used as replicates for each period so that the analytical design for each analysis was: Beach Management Type (commercially fished v non-fished - fixed), Beach (nested in Beach Management Type – random) and Period (fixed). Separate analyses were done for the swash and dry habitats on each beach and each analysis was based on the Bray Curtis dissimilarity matrix with Type III (partial) sums of squares calculated using 999 unrestricted permutations of residuals under a reduced model.

The proportion of variation attributable to each factor and interaction in each PERMANOVA model was calculated to aid interpretation of results (Anderson et al. 2008). All negative variation component values were treated as zero, eliminated from the analysis and the remaining variation components recalculated (Fletcher & Underwood 2002). Each component directly estimated variability for each term independent of the other terms.

Fishing and other anthropogenic perturbations can potentially affect the levels and patterns of variability in the densities of organisms (Warwick & Clarke 1993, Anderson et al. 2008). This was investigated here by examining if the components of variation of the densities of clams changed from before to during harvesting in different ways on the fished versus the non-fished beaches. For each habitat and beach the components of variation were determined separately for each period using a 2-factor PERMANOVA with Day (random) and Site (nested in day - random). Each analysis was based on the Euclidean distance measure and Type III (partial) sums of squares were calculated using 999 unrestricted permutations of the raw data.

3. Results

3.1. Densities of clams

There was no global effect of Beach Management Type on densities of clams (PERMANOVA, $P > 0.05$ in all 6 analyses, Table 1, Figs 2 & 3). The number of legal clams sampled in the dry differed significantly according to the Beach Management Type x Period interaction (Table 1), but the pairwise tests could not distinguish significant differences among groups due to the low number (3) of available permutations for each pairwise comparison. Nevertheless, the data in Fig 3 indicate that a greater number of clams were sampled across the fished beaches compared to the non-fished beaches early harvesting, whereas no such patterns were apparent before or late harvesting. The densities of total and sublegal clams in the swash and sublegal clams in the dry significantly differed according the factor Beach and the Beach x Period interaction (PERMANOVA, $P < 0.01$ in all cases, Table 1). The significant interaction term indicated that changes between periods for these particular indices were not the always same for both beaches within each beach management type. For example, the pairwise tests identified that for the commercially fished beaches, the densities of total clams in the swash on Smoky did not significantly differ between periods, whereas on Ten Mile densities during late harvest were significantly greater than early harvest, but not the before period (Fig 2). For the non-fished beaches, total densities in the swash were significantly lower early harvest than before or late harvest on Illaroo, but only less than late harvest on Sandon (Fig 2). Similarly, the densities of sublegal clams in the dry did not significantly differ between periods on Ten Mile, but were significantly greater late harvest than in either before or early harvest on Smoky.

Densities of total, sublegal and legal sized clams consistently differed significantly according to the factors Site and Day (PERMANOVA, $P < 0.01$ in all 12 analyses, Table 1). These results demonstrate there was significant variability in densities from site-to-site on each beach on each sampling day, as well as among individual sampling days within each period on each beach (Figs 2 & 3).

In all global analyses the components of variation were consistently greatest for the residual, accounting for 35 to 49% of total variation (Table 1). This identified that in both habitats there was considerable small-scale variability in densities of clams among replicate samples taken at each site on each sampling day. Likewise, across both habitats there was much variation (21 to 32% of total variation) among the individual sites sampled each day on each beach, and except for sublegal clams in the swash, variation was also generally high for the factor Day (2 to 20%) denoting the level of variability among replicate sampling days within each period. Beach Management Type explained less than 3% of total variation in each analysis (Table 1).

When the components of variation of densities of clams were determined for each individual sampling period, different patterns were evident for sublegal clams compared to total and legal clams (Table 2). Sublegal clams consistently displayed across both habitats on each beach (except Sandon, dry habitat, late harvest) greatest variation at the level of residual (i.e. among replicate samples at each site) (Table 2). This was also true for legal clams in the swash on Ten Mile and Sandon and in the dry on Sandon and Illaroo, and for total clams in the swash on Ten Mile and Illaroo and in the dry on Sandon (Table 2). Elsewhere, however, the components of variation were greatest across days (e.g. Ten Mile, Dry, Before; Smoky, Dry, Early) and sites (e.g. Smoky, Swash, Before and Early). Nevertheless, there were no consistent changes in the patterns of variability in total and legal clams from before to during harvesting across the fished and non-fished beaches.

3.2. Sizes of clams

PERMANOVA identified that across both habitats there were significant differences in the size compositions of clams according to the Beach x Period interaction ($P < 0.05$ in both cases), but there were no significant effects due to Beach Management Type or its interaction with Period ($P > 0.05$ in all cases, Table 3). The pairwise tests indicted that in both habitats the size compositions of clams on Sandon and Illaroo (non-fished beaches) differed between each period (Fig 4 & 5). For the two commercially fished beaches, the patterns were more complex; across both habitats the size compositions of clams on Ten Mile differed between early and late harvest, whereas on Smokey the before period differed to early and late harvest (which did not differ).

In general, across all beaches two sizes classes of clams (10-25 and 40-60 mm SL) were prevalent in samples taken in the swash, whereas the smaller size class was less prevalent in the dry (Figs 4 & 5). In general, greater proportions of small juveniles (< 25 mm SL) were present across all beaches, particularly in the swash habitat, in the early and late harvesting periods than before harvesting (Figs 4 & 5). The notable exception to this was the dry habitat on Smoky and Illaroo. The length composition of clams on Smoky was truncated at 55 mm, with few clams > 55 mm SL were sampled across all three periods in either habitat on Smoky.

More than 50% of the components of variation in size composition were attributed to the residual (replicate sample days within each period) in each analysis (Table 3). This may have been driven by the days when few clams were caught in a particular habitat. Little variation was attributed to Beach Management Type or the Beach Management Type x Period interaction in either habitat (Table 3).

4. Discussion

Populations of clams on all four beaches were inherently variable across both habitats with significant differences in densities consistently occurring across individual sites sampled each day, as well as among days sampled within each period on each beach. Moreover, the components of variation in the global analyses were consistently greatest across the smallest spatial scale sampled; among replicate samples taken at each site on each day and they were also generally high for the factors Site and Day. These results exemplify the need for future assessments of beach clams to adequately account for small-scale variability in sampling strategies to avoid potential confounding of larger scale comparisons (see also Gray 2016a). Small-scale spatial and temporal variability is not uncommon in benthic assemblages (Morrisey et al. 1992, Fraschetti et al. 2005, Chapman et al. 2010) and was expected; previous sampling over a hierarchy of scales identified that variability in the densities of clams was consistently greatest across the smallest spatial and temporal scales examined (Gray 2016a). The ecological processes driving such small-scale variability require determination using appropriate sampling strategies and experimentation.

Despite the prevalence of small-scale variability, some differences in the densities and sizes of clams among individual beaches were evident, but there were no global differences across the commercially fished versus non-fished beaches. This contrasted expectations and that often observed between protected versus non- and partially-protected areas in other systems (Russ et al. 2008, Franco et al. 2009, Butcher et al. 2014). Fishing closure effects on organisms, including beach clams, can be rapid and manifest within 1-3 years (Halpern & Warner 2002, Defeo 2003), with this study commencing 1-year post management implementation. Nevertheless, there were no detectable reductions in densities and truncation of size compositions of clams on fished compared to non-fished beaches throughout harvesting even though during the study 1,700 and 3,200 kg of clams were reportedly harvested from Ten Mile and Smoky beaches, respectively. In a concurrent study, no significant differences were detected in the densities and sizes of clams occurring in commercially fished (open) versus non-fished (closed) zones across other beaches (Gray 2016b). These combined results highlight the difficulties in determining the potential effects of current fishing levels and management strategies on *D. deltoides*. Nevertheless, similar experimental evaluation is required to test the general applicability of closed areas and times for managing the sustainable harvesting of beach clam resources elsewhere. Similarly, as advocated by Defeo (2003), the temporal rotation of harvesting of clams across different zones along beaches as well as among beaches requires further investigation.

The lack of fishing-related effects in the current study was unlikely due to any potential confounding of recreational and indigenous harvesting of clams across open and closed beaches, or zones on beaches (Gray 2016b). Very few non-commercial fishers were observed collecting clams during sampling and the current levels of harvesting from these two sectors is considered to be very low. Moreover, the low levels of total commercial catches and the daily trip limit of 40 kg per-fisher may have dampened the potential manifestations of fishing on populations. Potentially, this could be tested by allowing different harvest levels across different beaches as part of a controlled experiment. This could assist in determining the most appropriate total quotas and daily trip limits for the sustainable harvesting of the species as well as the commercial viability of the fishery.

The harvesting of clams does not occur evenly along and across beaches as it is dependent on clam aggregations and ease of access (unpublished data, Defeo 2003) and thus any potential effects of harvesting on the densities and sizes of clams may be manifest only across small spatial scales in the immediate vicinity of actual harvesting (i.e. digging). Moreover, areas of fishing intensity on beaches often vary with time and any potential impacts of harvesting may persist for only a small temporal period (e.g. single tidal phase) (Gray 2016b). The active and passive movements of clams along and across (Leber 1982, Ellers 1995, McLachlan et al. 1995, Dugan & McLachlan 1999), and potentially between beaches may further mitigate, or confound detection of, any effects of fishing on populations at the level of beaches or zones within beaches. Knowledge of such movements and relationships with ecological processes could help clarify potential harvesting-related impacts on clams.

There were no detectable or observable effects of commercial harvesting on the size composition of clams across beaches from before to early or late during the harvesting season. The only notable difference among beaches was the absence of large clams > 55 mm SL across both habitats on Smoky, but this was evident before, early and late harvesting. Whilst this particular feature could be an artefact of historic fishing activities, it could also be due to differing growth and mortality schedules of clams on Smoky compared to the other beaches studied. Spatially explicit differential growth rates and concomitant size compositions of other clams and benthic molluscs, such as abalone and mussels, are common (Peterson & Beal 1989, McShane & Naylor 1995, Fiori & Defeo 2006, Blanchette et al. 2007, Saunders et al. 2009). An understanding of the spatio-temporal levels of plasticity in the growth and longevity of clams and their potential relationships with biotic and abiotic processes of the beach environment could assist in determining potential impacts of fishing on populations as well as the resilience and responses of clams to differing levels of harvesting.

Two distinct size classes (5-25 and 40-60 mm SL) of clams were evident across both habitats and all beaches, but they occurred in differing proportions spatially and temporally. In general, across both habitats on the fished and non-fished beaches greater proportions of smaller clams were evident in the size compositions during early and late harvesting compared to the before period, indicating recruitment had taken place. This timing concurs with the predominant austral winter/spring spawning of the species (Ferguson & Ward 2014). The smaller size class was also generally more prevalent in the swash compared to the dry habitat, suggesting small clams may primarily be distributed in the lower zones of beaches, as reported for other clam species (Leber 1982, Donn 1990), and this could potentially be a mechanism to reduce predation and resource competition (Defeo & McLachlan 2005).

The overall lack of differences in the densities and sizes of clams between the commercially fished and non-fished beaches would have also been confounded by clams responding to a suite of ecological processes and natural perturbations operating independently on each individual beach. For example, abiotic factors such as beach profiles, wave conditions and storm events (McLachlan & Hesp 1984, McLachlan et al. 1995), in combination with biotic processes such as levels of predation, quantity and quality of food resources and competitive interactions among fauna (McLachlan 1998, Mikkelsen 1981, Brazeiro & Defeo 1999, Defeo & McLachlan 2005), could all affect the dynamics of clam populations on each individual beach in different ways. Unfortunately, the ecology of *D. deltoides* has been little studied to help unravel such complexities and potential relationships with harvesting. Sampling clams across beaches with different management arrangements over several years may be necessary to ascertain the potential effects of fishing on populations as opposed to natural environmental processes (Defeo & de Alava 1995, Ortega et al. 2012).

Although there were no identifiable effects on clams of commercial harvesting here, harvesting could be having more widespread effects across the entire stock, as reported for other exploited species (McLachlan et al. 1996, Defeo 2003). Indeed, this could have particularly been the case with the previous unlimited harvesting of clams by all sectors across many beaches, which may have impacted total reproductive output and concomitant levels of recruitment and population replenishment across non-fished as well as fished beaches. A strong genetic connectivity exists among clam populations along eastern Australia, suggesting high exchange of larvae among beaches (Murray-Jones & Ayre 1997). However, stock-recruitment relationships and the source-sink dynamics of larvae are unknown, and are thus potential avenues of future research. The current management restrictions on clam harvesting may potentially allow populations to rebuild, but this could take some time to manifest. Unfortunately, long-term closures to clam harvesting elsewhere have in some cases had minimal impacts on restoring populations due to other environmental perturbations, namely climate variability and associated effects on recruitment pulses and large-scale mortalities (McLachlan et al. 1996, Defeo 2003, Ortega et al. 2012).

Future field-based studies that test alternative management arrangements may help refine and determine the most suitable harvesting strategies for this particular species and assist in developing appropriate strategies for the sustainable harvesting of beach clams elsewhere. The potential impacts of commercial harvesting of clams on sublegal conspecifics, other organisms and the broader beach ecosystem have not been examined, but they need to be considered for the holistic management of sandy beaches.

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Fig 1. Map showing the location of the four study beaches on the east coast of Australia

Fig 1

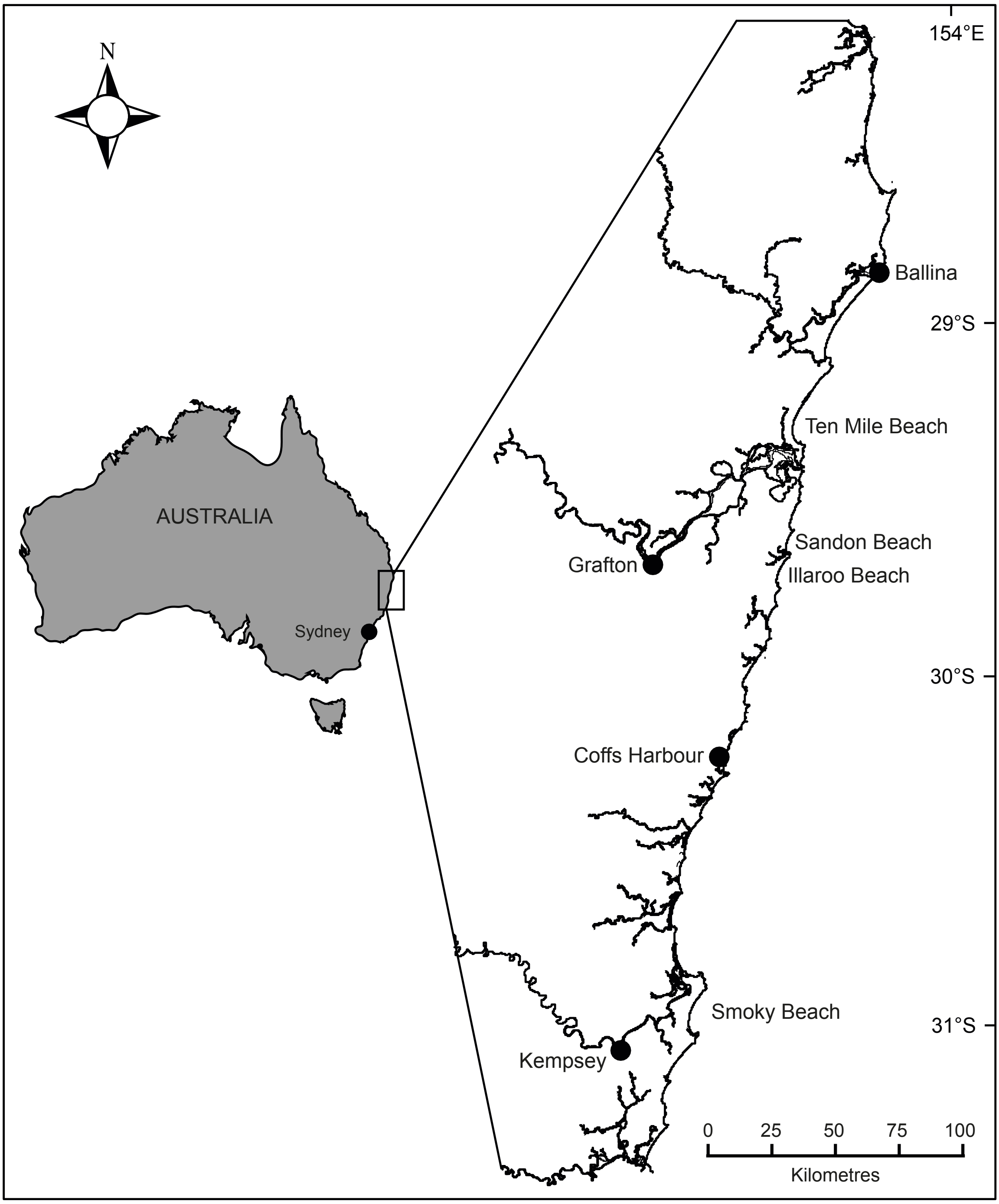

Fig 2. Mean (+ SE) density of *Donax deltoides* sampled in the swash habitat on each of six days before, early and late harvesting across the two commercially fished and non-fished beaches

Fig 2

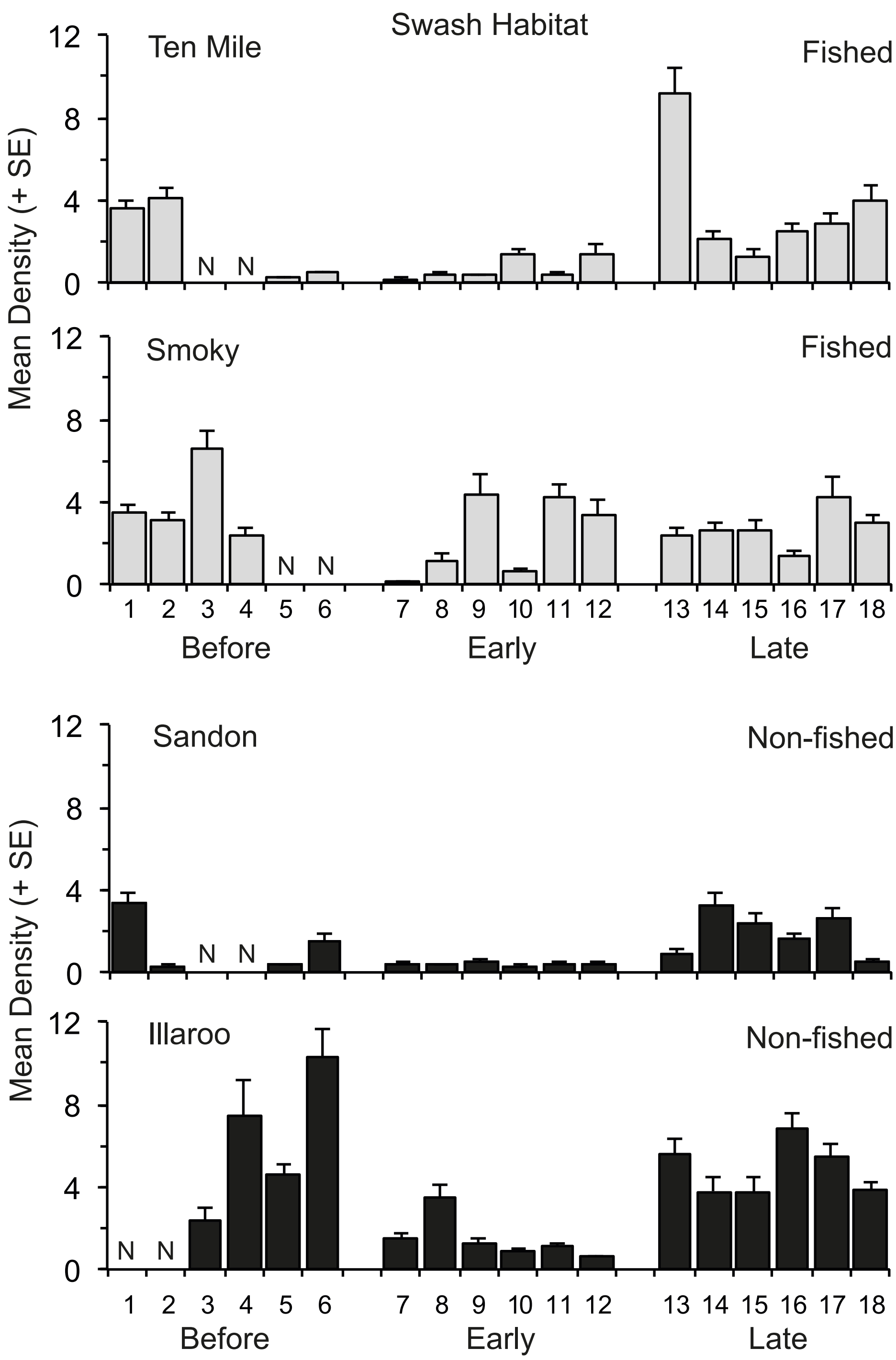

Fig 3. Mean (+ SE) density of *Donax deltoides* sampled in the dry habitat on each of six days before, early and late harvesting across the two commercially fished and non-fished beaches

Fig. 3

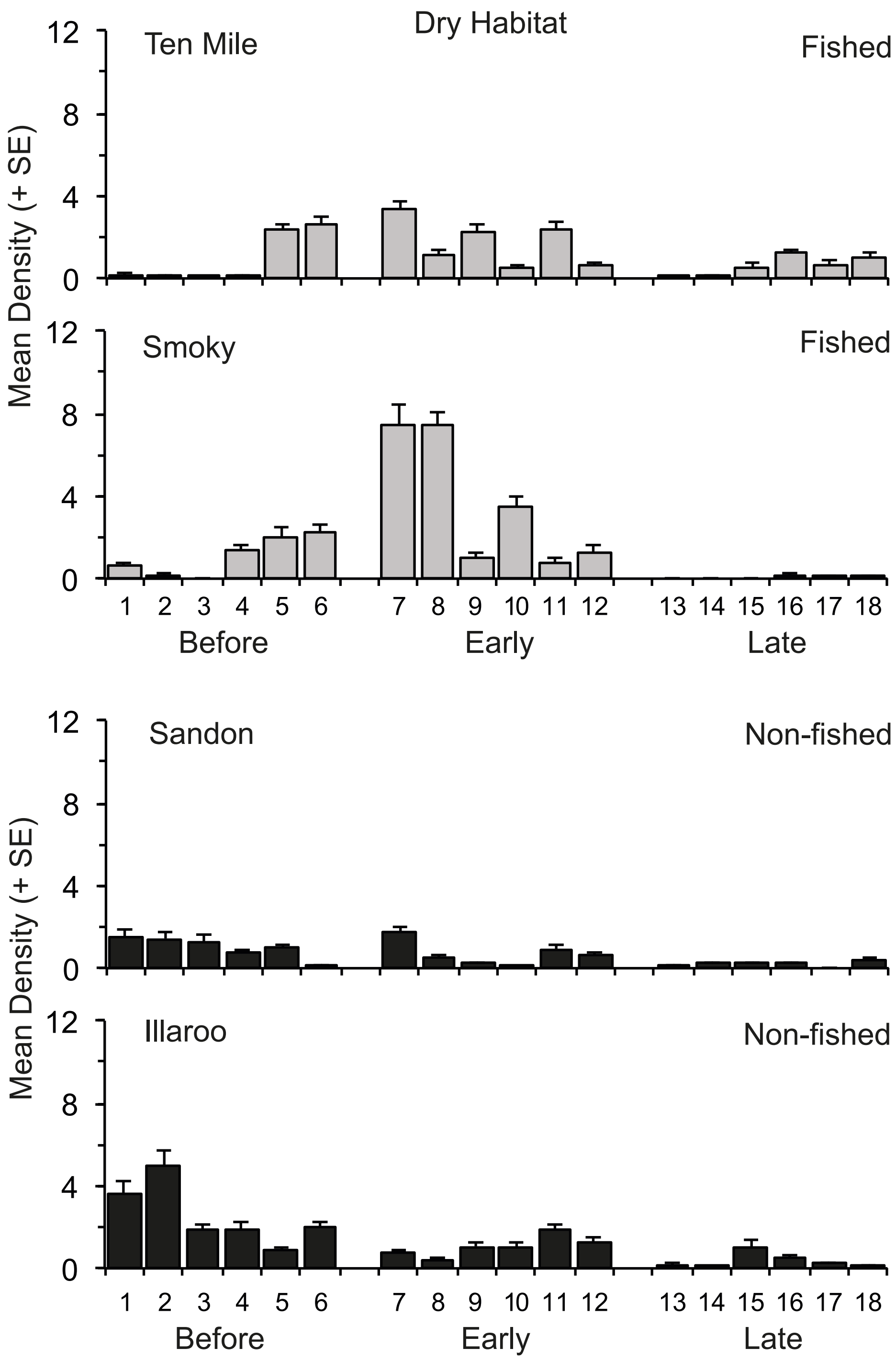

Fig 4. Size compositions of *Donax deltoides* sampled in the swash habitat before, early and late harvesting across the two commercially fished and non-fished beaches

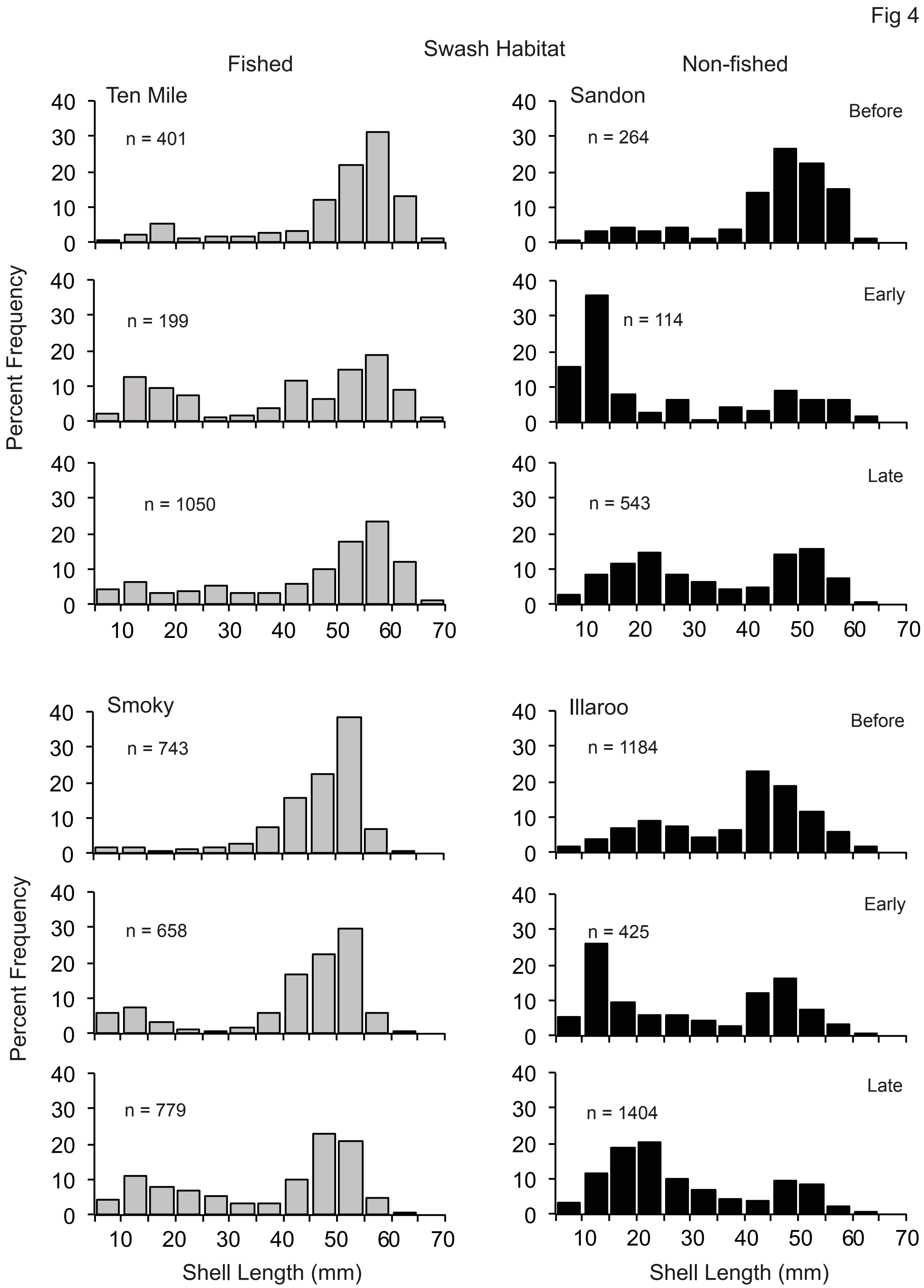

Fig 5. Size compositions of *Donax deltoides* sampled in the dry habitat before, early and late harvesting across the two commercially fished and non-fished beaches

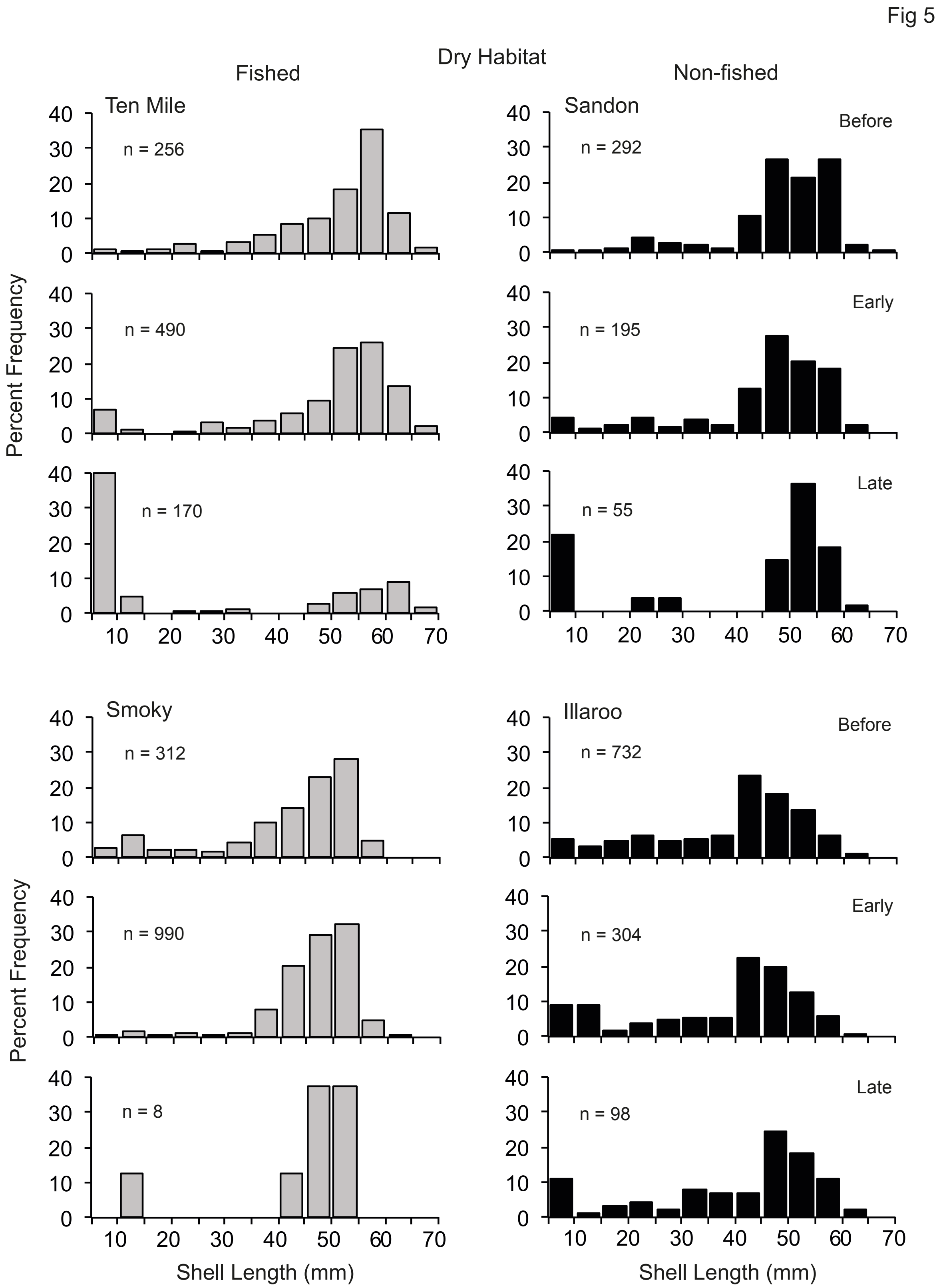

Table 1. Results of PERMANOVAs comparing the densities of total, legal and sublegal clams across commercially fished and non-fished beaches before, early and late harvesting. For each analysis, the percentage component of variation (CoV%) is given for each factor in the model. P(perm) values in bold are significant (p < 0.05). Shaded terms are those that if significant might signify a possible effect of management zoning.

Swash Habitat		Total clams				Legal clams				Sublegal clams			
Source	df	MS	Pseudo-F	P(perm)	CoV%	MS	Pseudo-F	P(perm)	CoV%	MS	Pseudo-F	P(perm)	CoV%
Beach Management Type	1	16.254	0.009	0.667	0.0	362.520	3.523	0.345	2.6	532.300	0.530	0.649	0.0
Period	2	1597.100	4.484	0.096	5.9	440.950	3.892	0.127	4.8	468.030	3.755	0.122	3.9
Beach(BMT)	2	1737.100	12.708	**0.001**	10.4	102.890	1.690	0.200	0.8	1003.600	32.049	**0.001**	14.9
BMT x Period	2	108.340	0.304	0.778	0.0	14.477	0.128	0.858	0.0	132.970	1.067	0.418	0.2
Beach(BMT) x Period	4	356.210	2.606	**0.045**	4.2	113.310	1.861	0.139	3.1	124.640	3.981	**0.006**	4.2
Day(Beach(BMT)xPeriod)	52	136.700	3.133	**0.001**	9.3	60.891	4.482	**0.001**	**14.6**	31.313	1.498	**0.018**	2.5
Site(Day(Beach(BMT)xPer	448	43.632	4.925	**0.001**	27.8	13.584	4.164	**0.001**	**25.6**	20.903	5.624	**0.001**	32.4
Residual	2560	8.860			42.5	3.262			48.4	3.717			42.1
Total	3071												

Dry Habitat		Total clams				Legal clams				Sublegal clams			
Source	df	MS	Pseudo-F	P(perm)	CV%	MS	Pseudo-F	P(perm)	CV%	MS	Pseudo-F	P(perm)	CV%
Beach Management Type	1	87.529	0.598	0.646	0.0	102.090	6.571	0.336	1.9	0.560	0.006	1.000	0.0
Period	2	644.650	4.669	0.106	7.1	289.060	9.512	**0.010**	**8.4**	77.063	1.449	0.300	1.3
Beach(BMT)	2	146.330	2.052	0.170	1.4	15.538	0.507	0.636	0.0	89.433	8.104	**0.001**	5.8
BMT x Period	2	464.450	3.364	0.117	9.1	222.690	7.328	**0.021**	**12.5**	61.783	1.162	0.385	1.0
Beach(BMT) x Period	4	138.080	1.937	0.112	3.7	30.389	0.992	0.447	0.0	53.190	4.820	**0.001**	9.4
Day(Beach(BMT)xPeriod)	60	71.298	6.624	**0.001**	20.2	30.633	6.931	**0.001**	**20.4**	11.036	3.915	**0.001**	11.0
Site(Day(Beach(BMT)xPer	504	10.764	4.838	**0.001**	22.8	4.420	4.722	**0.001**	**21.7**	2.819	3.658	**0.001**	21.9
Residual	2880	2.225			35.7	0.936			35.0	0.771			49.5
Total	3455												

Table 2. The percentage contribution of each factor to the total components of variation determined by PERMANOVA for the densities of total, legal and sublegal clams in each sampling period across the swash and dry habitats on the two commercially fished and non-fished beaches. The factor having the greatest component of variation is marked in bold.

		Swash Habitat			Dry Habitat		
		Before	Early	Late	Before	Early	Late
Commercially fished - Total clams							
Ten Mile	Day	39.7	12.4	23.2	**40.8**	17.5	10.9
	Site (Day)	16.2	22.0	19.3	23.3	37.1	34.4
	Residual	**44.1**	**65.6**	**57.5**	35.9	**45.4**	**54.7**
Smoky	Day	9.3	12.3	2.4	14.4	**40.7**	0.9
	Site (Day)	**55.9**	**47.1**	20.0	**45.2**	20.9	12.3
	Residual	34.8	40.6	**77.6**	40.4	38.5	**86.8**
Non-fished - Total clams							
Minnie	Day	9.6	17.7	0.9	14.4	6.0	5.2
	Site (Day)	36.9	27.8	40.9	33.1	27.7	**48.4**
	Residual	**53.5**	**54.5**	**58.2**	**52.4**	**66.3**	46.3
Sandon	Day	19.7	0.0	6.7	3.0	13.7	1.3
	Site (Day)	**44.7**	17.8	40.8	42.2	24.0	12.7
	Residual	35.5	**82.2**	**52.5**	**54.8**	**62.3**	**86.0**
Commercially fished - Legal clams							
Ten Mile	Day	35.2	6.5	24.8	**41.2**	12.3	9.5
	Site (Day)	18.8	28.7	16.0	25.6	38.3	**48.6**
	Residual	**46.0**	**64.8**	**59.3**	33.2	**49.4**	41.9
Smoky	Day	5.1	12.0	4.9	19.4	**42.2**	0.9
	Site (Day)	**59.5**	**52.1**	13.3	30.2	22.2	4.4
	Residual	35.4	36.0	**81.8**	**50.4**	35.7	**94.8**
Non-fished - Legal clams							
Minnie	Day	17.0	9.3	1.8	9.3	3.0	8.7
	Site (Day)	16.7	36.1	**51.8**	27.4	18.7	44.3
	Residual	**66.2**	**54.6**	46.4	**63.3**	**78.4**	**47.0**
Sandon	Day	23.1	0.0	10.7	5.1	12.9	0.0
	Site (Day)	38.3	31.9	21.2	41.0	25.4	13.0
	Residual	**38.5**	**68.1**	**68.1**	**54.0**	**61.8**	**87.0**
Commercially fished - Sublegal clams							
Ten Mile	Day	13.8	9.4	11.2	13.7	14.4	15.1
	Site (Day)	4.3	7.5	37.8	16.6	14.5	14.9
	Residual	**81.9**	**83.2**	**51.0**	**69.7**	**71.0**	**69.9**
Smoky	Day	9.1	8.9	0.0	3.9	22.9	0.0
	Site (Day)	26.5	39.6	24.7	41.3	22.8	0.0
	Residual	**64.4**	**51.6**	**75.3**	**54.8**	**54.4**	**100.0**
Non-fished - Sublegal clams							
Minnie	Day	3.1	9.6	0.1	12.2	2.6	0.0
	Site (Day)	47.4	23.1	44.2	31.2	34.0	27.1
	Residual	**49.5**	**67.3**	**55.7**	**56.7**	**63.4**	**72.9**
Sandon	Day	2.6	0.0	1.0	0.1	8.2	4.2
	Site (Day)	36.6	14.5	**50.0**	13.9	9.7	20.1
	Residual	**60.8**	**85.5**	49.1	**85.9**	**82.1**	**75.7**

Table 3. Results of PERMANOVAs comparing the size compositions of clams across commercially fished and non-fished beaches before, early and late harvesting. For each analysis, the percentage component of variation (CoV%) is given for each factor in the model. P(perm) values in bold are significant ($p < 0.05$). Shaded terms are those that if significant might signify a possible effect of management zoning.

	Swash Habitat					Dry Habitat				
Source	df	MS	Pseudo-F	P(perm)	CoV%	df	MS	Pseudo-F	P(perm)	CoV%
Beach Management Type	1	7670.2	0.749	0.664	0.0	1	5760.1	0.714	0.837	0.0
Period	2	11852.0	4.936	**0.012**	17.3	2	12795.0	3.070	**0.039**	12.5
Beach(BMT)	2	10244.0	7.402	**0.001**	22.1	2	8099.6	4.006	**0.001**	11.7
BMT x Period	2	2215.3	0.923	0.519	0.0	2	4197.9	1.007	0.440	0.1
Beach(BMT) x Period	4	2401.2	1.735	**0.020**	7.4	4	4183.1	2.069	**0.001**	12.4
Residual	52	1384.1			53.2	55	2022.1			63.3
Total	63					66				

Appendix 6.

Integration of local fisher knowledge into a fishery-independent sampling strategy: a pilot assessment

Integration of local fisher knowledge in a fishery-independent sampling strategy: a pilot assessment

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Donax deltoides

1. Introduction

Fishery-independent sampling strategies are increasingly being used to monitor and assess aquatic populations of harvested organisms across a plethora of habitats throughout the world (Gunderson, 1993; Pennington and Stromme, 1998). Such strategies are often preferred over fishery-dependent data sources as they can be done consistently where fishing does and does not occur, sample a wider size range of organisms of different life history stages rather than just those retained in commercial fishing gears, and importantly are not plagued with the many biases often associated with fishery-dependent sampling strategies (Rotherham et al., 2007). Standardized research surveys are becoming particularly important in comparing populations of fish and invertebrates across areas of differing management arrangements, such as fished versus non-fished (marine protected) areas (Gray et al., 2014; Kelaher et al., 2014; Gray, 2016a).

Fishery-independent methodologies specifically rely on the basic design principals of strategic stratified randomised sampling using standardized gears and practices that can be replicated across space and time (Gunderson, 1993). Nevertheless, the sampling designs of such programs are often criticized by fishing industry representatives, as they often do not take into account places and times where organisms are reportedly most abundant (i.e. where fishing is often most concentrated) and include non-representative fishing areas. This has led in some instances to industry representatives having little faith in the data obtained in some assessment programs and subsequent management actions. Consequently, there has been much advocacy for greater engagement and more direct involvement of industry in sampling programs, particularly in small-scale fisheries (Defeo et al., 2014). It may be possible to integrate fisher's knowledge of organism distributions and abundances in sampling designs by accommodating their preferred sampling locations as part of a larger sampling strategy.

This pilot study was done as part of developing a collaborative strategy that included greater involvement and cooperation of industry in surveying and assessing beach clams harvested in a small-scale fishery in eastern Australia. Specifically, this experiment examined whether the densities and size compositions of beach clams differed between standard survey sites and those chosen by industry representatives. The study had two main goals: 1) examine if the densities and size composition of clams differed between standard survey and fisher chosen sampling sites, and 2) explore if fisher chosen sites could be incorporated into a future monitoring strategy. It was hypothesized that densities and sizes of clams at fisher chosen sites would be greater than at standard survey sites. This small test case could serve as a model for development of similar arrangements elsewhere and in other fisheries more globally.

2. Methods

2.1. Sampling

Sampling for this pilot experiment was done across a 4 km section of Lighthouse Beach (-31.519; 152.882) where commercial fishers actively harvested clams throughout the 2015 fishing season. Within this 4 km section, there were four standard survey sampling locations separated at intervals of 1 km (0.5, 1.5, 2.5, 3.5 km). At each of these sites, clams were sampled in the swash and dry sand habitats (as per previous protocols; Gray, 2016a). Within the same 4 km section, a commercial fishing representative who had actively fished the beach for the past three months chose four sites to sample clams in the swash and another four sites to sample clams in the dry sand. Sampling was done over two specific periods, firstly in September and again in October 2015. The standard sites remained the same in each period, whereas the fisher chosen sites differed in each sampling period. Note that the fisher only chose three sites in each habitat in October. Each site was sampled twice over 3-days in each of the two periods to incorporate small-scale variability in clam densities (Gray, 2016a).

The dry sand samples were done two hours either side of low tide whereas the swash samples were done on the rising tide up to 2 hours before high tide. The latter was done as commercial fishers claimed that clams occur in greater densities during the rising tide, mimicking their harvesting activities. Clams in the swash zone were sampled by finger digging for 30 sec a small area (average diameter 57 cm, depth 18 cm) of sand and scooping it into a net that had 12 mm mesh hung on a frame measuring 35 x 21cm (Gray et al., 2014). Clams in the dry sand were sampled by excavating sand to a depth of 20 cm within a square box quadrat that had 32 cm sides (James and Fairweather, 1995). Because of large quantities of shell fragments and small pebbles along the beach, the excavated sand was sieved through a net with 12 mm mesh. All clams collected in each replicate sample were counted and measured for shell length (SL, mm) using digital calipers. Operational information, including time of sampling and beach and sea conditions, was recorded.

2.2. Analyses

Nonparametric permutational analyses of variance (PERMANOVA; Anderson, 2001) were used to test if densities of clams differed between sampling strategies (i.e. standard versus fisher-chosen) and if these were consistent across replicated sites and sampling times. Separate analyses were done for each habitat (swash and dry sand) because they were sampled in different ways and also as the habitats were not adjacent at the fisher chosen locations. Each univariate analysis was based on the Euclidean distance measure and Type III (partial) sums of squares were calculated using 999 unrestricted permutations of the raw data. The proportion of variation attributable to each factor and interaction in each model was calculated to aid interpretation of the results. All analyses were done using the PRIMER 6-PERMANOVA^{+} program (Anderson et al., 2008).

Kolmogorov-Smirnov tests were used to test if the size compositions of clams differed between the standard and fisher chosen sites (data pooled across sites) for each habitat and sampling period.

3. Results

3.1. Sampling site characteristics

In September the fisher-chosen swash sites were all located in the northern 2 km of the study zone and characterized by flat sections of beach that had wide intertidal zones and hard compacted sediment (Table 1). This was also the case for one site in October, but the other two were located in the southern section of the study zone where the beach profile was steeper and sand less compact. All fisher chosen dry sites were associated with steeper sections of beach in the southern 2 km of the study zone. Two of the standard sites were located in each of the flat and steep sections of the beach.

3.2. Densities of clams

In September, the densities of clams significantly differed between the standard and fisher chosen sites in the dry habitat across both sample days, and in the swash on the 1st sampling day (PERMANOVA, Table 2, Fig. 1). There were no significant differences in densities of clams between sampling strategies across either habitat in October. Across both habitats and times, small-scale variability was prevalent with densities significantly varying according to sample day and individual sites within each strategy (Table 2, Fig. 1).

3.3. Sizes of clams

Across both habitats, the size compositions of clams significantly differed between the standard and fisher locations (data pooled across all sites) in September (K-S tests, $P < 0.05$), but not in October (K-S tests, $P > 0.05$, Fig. 2). A greater proportion of clams < 45 mm SL were sampled across the standard compared to the fisher chosen sites in the swash, but the opposite was evident in the dry (Fig. 2). The size compositions of clams were dominated by two size distinct classes (45 and

55 mm SL) across both habitats in October. This was also observed across the fisher chosen dry habitat sites in September.

4. Discussion

As hypothesized the densities and sizes of clams varied according to sampling strategy, but this was only evident in the dry and across one sampling day (densities only) in the swash in September. This sampling period corresponded with clams being highly aggregated in small areas, with commercial fishers having a strong knowledge of the location of such aggregations across both habitats. Clams had displayed relatively stable distributions across both habitats for approximately two months prior to this actual experiment. In contrast, sampling in October was done approximately 3 weeks after a large storm event (4 m swell) that modified the morphology of the beach and redistributed clams across and along the section of beach commercial fishers had previously been operating. Consequently, clams were more dispersed, less dense and aggregations less pronounced and commercial fishers had greater difficulty identifying suitable aggregations for sampling. Essentially, beach conditions and clam distributions had not settled and accordingly, fishers had little harvesting history of where clams were aggregated. This resulted in there being no significant differences in estimated densities or size compositions of clams between the standard and fisher-chosen locations during this sampling.

Differences in the size compositions of clams in the swash in September may have been due to the fishers choosing areas where larger (commercial sized, > 45 mm SL) clams predominated. Hence, the smaller representation of sublegal (< 45 mm SL) clams in these swash samples compared to the standard sites. Such differences could impact assessments, especially if the fisher-chosen samples were considered in isolation from the standard samples. This difference in size composition was not evident in October when clams were more dispersed, or in the dry habitat across both sampling periods. The size compositions of sampled clams across the standard sites and the fisher-chosen dry habitat sites remained similar across both sampling periods.

The fisher-chosen sites were generally concentrated across smaller sections of the beach (study zone) and were dependent on beach morphology (especially in September), potentially explaining some observed differences in densities and sizes of clams between sampling strategies. The areas that the fishers chose to sample clams in the swash in September were within the northern 2 km of the study zone and adjacent to flat sections of beach that had a wide intertidal zone, gentler slopes and more compacted beach material. At such sites few clams were present in the dry habitat. In October, only one fisher chosen swash site was located in this area whilst the other two were located in the southern end of the study zone and adjacent to steeper beach profiles. Two standard sampling sites (S1 and S2) were located in the northern flat section of the study zone, but still they produced lower average densities than the actual fisher-chosen sites in this area in September. Contrasting this, the standard S4 site was located between the fisher chosen F10 and F11 sites and displayed similar levels of clam densities in October.

In contrast and across both periods, the fisher-chosen dry habitat sites corresponded with steeper sections of the beach where clams were deposited along ridges (approximately 2 hours inundation above low tide), and were specific habitat features that commercial fishers had been harvesting for several months. The standard S3 and S4 sampling sites were also located along these steeper sections of the beach. The observer spatial and temporal variability in clam densities among these sites may be the result of local-scale differences in beach topography (Gray, 2016a).

5. Conclusions

Commercial fishers have a unique experience and knowledge in identifying clam aggregations and harvesting 'hotspots' across and along beaches and this was most evident in the September sampling. Clearly, this demonstrated that densities and sizes of calms were appropriate for commercial harvesting. However, this was not extended to the October sampling, highlighting the dynamic nature of clam populations and consequent sample variability. Small-scale spatial and temporal variability in clam densities was prevalent across both habitats and sampling strategies, further demonstrating the importance of capturing such small scales of sampling in any longer-term sampling programs (Gray, 2016a).

Future surveys of clams could involve a collaborative approach where standard scientific surveys are done across entire beaches along with a set of fisher chosen locations. Along with standard fishery-dependent data (e.g. logbooks), this may provide fishers a greater opportunity to provide input into resource and fishery assessments. It could also help unravel relationships between CPUE and actual stock abundance that typically plague aggregation-based fisheries. Such collaborative sampling could involve commercial, recreational and indigenous fishers depending on the beach and management arrangements. Such an approach may help strengthen cooperation among industry, scientists and managers and assist in the uptake of fishery-independent survey data in assessments and management deliberations. This pilot study is a small step in developing such co-management successes.

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Fig. 1. Mean (+ 1 SE) densities of clams sampled in the swash and dry habitats across each of the scientific and fisher-chosen sites in each sampling period.

FIGURE 1

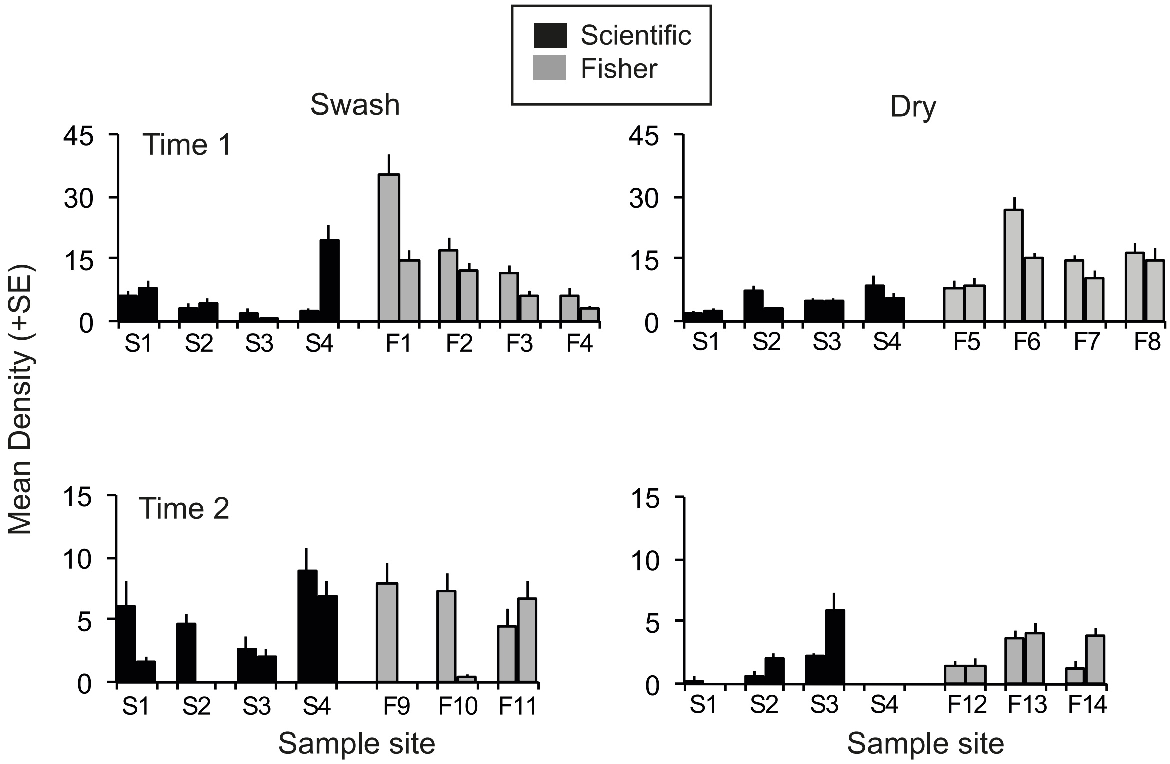

Fig. 2. Size compositions of clams sampled in the swash and dry habitats across the scientific and fisher-chosen sites in each sampling period.

FIGURE 2

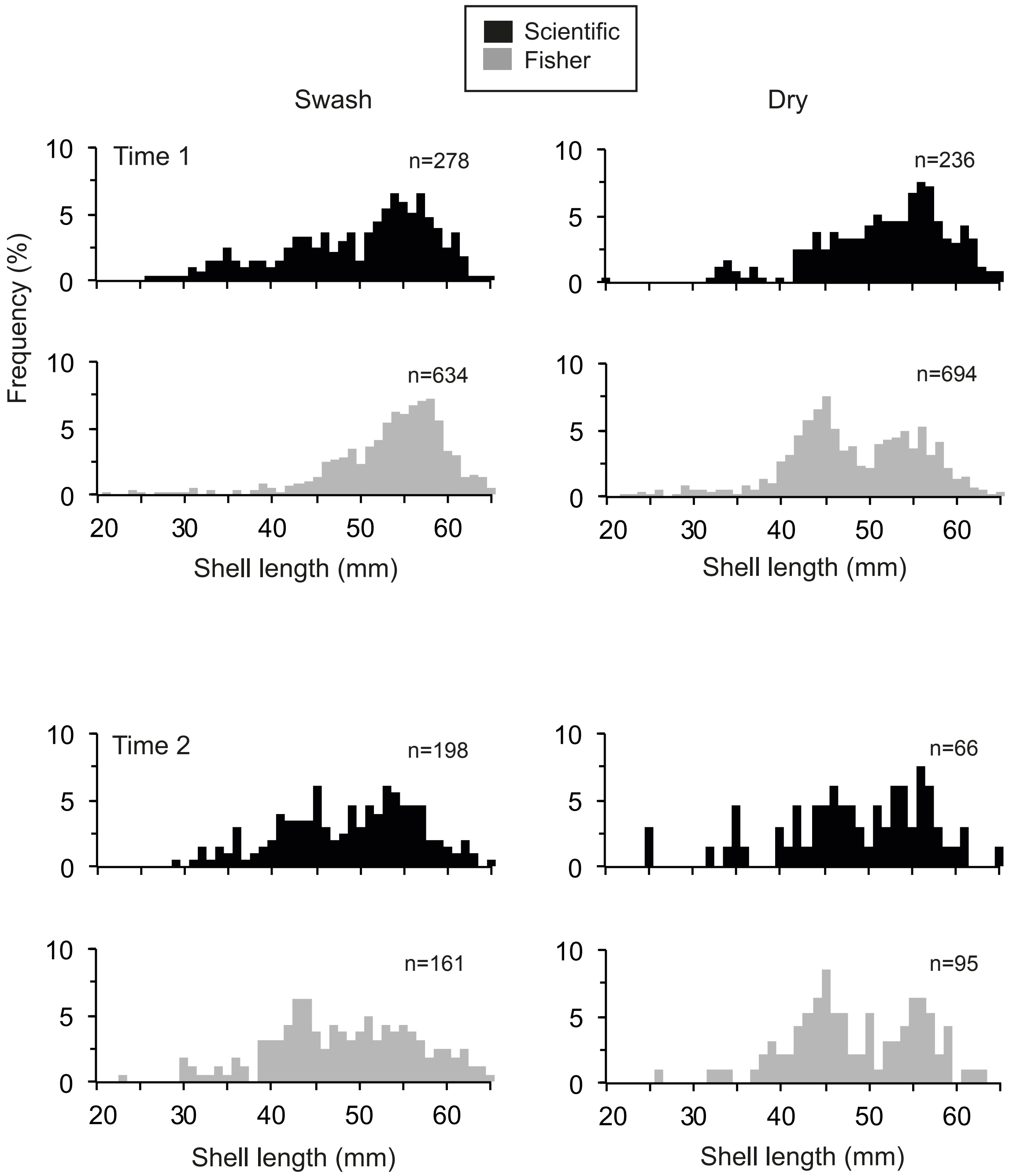

Table 1.

Table 2. Results of PERMANOVA comparing densities of clams between scientific and fisher chosen sites across the swash and dry habitats in each sampling period.

		Swash			Dry		
Source	df	MS	Pseudo-F	P(perm)	MS	Pseudo-F	P(perm)
Time 1							
Day	1	86.260	0.416	0.601	297.920	7.515	**0.035**
Choice	1	1342.500	0.955	0.503	1578.600	6.261	**0.022**
Site(Choice)	6	576.730	2.779	0.114	172.620	4.354	**0.042**
Day x Choice	1	1046.800	5.043	**0.050**	80.016	2.018	0.235
Day x Site(Choice)	6	207.570	9.825	**0.001**	39.644	1.812	0.119
Residual	80	21.127			21.879		
Total	95						
Source	df	MS	Pseudo-F	P(perm)	MS	Pseudo-F	P(perm)
Time 2							
Day	1	264.140	6.129	0.060	22.921	2.814	0.164
Choice	1	2.480	0.742	0.636	32.861	1.307	0.385
Site(Choice)	5	54.461	1.264	0.441	31.128	3.822	0.087
Day x Choice	1	7.000	0.162	0.679	0.254	0.031	0.865
Day x Site(Choice)	5	43.100	4.977	**0.001**	8.144	3.936	**0.006**
Residual	70	8.660			2.069		
Total	83						

Appendix 7.

Evaluation of fishery-dependent sampling strategies for monitoring a small-scale beach clam fishery

Evaluation of fishery-dependent sampling strategies for monitoring a small-scale beach clam fishery

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Observer

CPUE

Data-poor

Donax deltoides

Abstract

This study comparatively examined industry logbooks, and beach- and port-based fishery-dependent data sources for monitoring and assessing catch, effort, catch-per-unit-of-effort (CPUE) and size compositions of beach clams in a small-scale fishery in eastern Australia. The study was done across the six-month fishing season and encompassed two management regions, serving a model for elsewhere. In general, values of catch, effort and CPUE did not differ significantly between logbooks and beach sampling, and spatial and temporal trends in examined indices were similar across both data sources. Beach sampling captured additional data that included the partitioning of fishing effort into search and dig time, and also the number and location of sites fished each day, which could be useful in unravelling CPUE-clam density relationships and potential fishing impacts, and assist in spatial management of fishing across beaches. These data could be future sourced from industry and provided on modified logbooks. Compared to port sampling, the beach-based size composition data appeared to be biased and influenced by fisher behaviour. Cost-effective future monitoring of the fishery could be done using a combination of logbooks for catch and effort that includes strategic periodic validations using beach-observers, and port sampling for size compositions. The success of such a strategy is reliant on strong fisher cooperation that requires open co-management arrangements. Future assessments of the beach clam resource need to account for inherent differences in populations across individual beaches, including non-fished (control) beaches.

1. Introduction

Fishery-dependent sampling strategies such as industry logbooks, and boat- and port-based sampling of catches are commonly used as a primary means to monitor and assess many commercial fisheries and harvested species (Ricker, 1975; Hilborn and Walters, 1992). The data collected in such sampling schemes are typically used to assess trends across time and space in catches, catch-per-unit-of-effort (CPUE), and sizes and ages of organisms. Ideally, the appropriateness, cost-benefits and inherent biases associated with each data collection scheme need to be understood and assessed prior to long-term implementation. This is particularly pertinent to the many small-scale, low monetary valued (and typically data poor) fisheries for which all but the basic monitoring is often logistically and cost prohibitive, and any form of cost recovery limited.

For many such fisheries, the only (if any) data available are the landed catch information supplied by harvesters and this is often of limited value for population and fisheries assessments and management (Castilla and Defeo, 2005; Salas et al., 2007; Defeo et al., 2014). Typical of such scenarios are various fisheries for beach clams (Bivalvia: Donacidae, Mesodermatidae, Veneridae). Beach clams are harvested for food and bait on sandy beaches worldwide, but many populations have been depleted due to over exploitation, primarily because they are easily accessible to many people, simple and cheap to harvest, and harvesting has been largely unregulated (McLachlan et al., 1996; Defeo, 2003). Beach ecosystems are also subject to much other anthropogenic and environmental stressors, further impacting clams (Defeo et al., 2009; Ortega et al., 2012). Determination of appropriate management arrangements in such fisheries is difficult, and in many cases the data streams and data collection strategies for long-term monitoring and management have not been assessed.

This study comparatively evaluated alternative fishery-dependent data sources for monitoring a small-scale beach clam (*Donax deltoides*) fishery in eastern Australia. The species has a substantial history of indigenous, and more recently recreational and commercial, exploitation (Ferguson et al., 2015). Commercially exploited populations have displayed considerable fluctuations in production, with notable declines and recent management interventions across the different management jurisdictions (Gray et al., 2014; Gray, 2016a,b; Ferguson et al., 2015). Clam harvesters that participate in the New South Wales (NSW) commercial fishery currently provide catch and effort information via logbooks, but other fishery-dependent data streams such as beach and port-based sampling of catches, which can also provide data on size compositions of catches, warrant investigation as potential data sources for long-term monitoring. Such strategies have been successfully implemented in other regional small-scale fisheries (Gray, 2008).

Specifically, catch, effort and CPUE data supplied by industry logbooks were compared with those obtained from beach-based sampling, and similarly size composition data of catches from port- and beach-based samples were contrasted. This was done across two regions to test the generality of results and was part of a broader study to assess both fishery-dependent and –independent (Gray et al., 2014; Gray, 2016a,b) strategies to monitor and assess beach clams. This study serves as a model for other small-scale fisheries and the alternative fishery-dependent data sources are discussed in terms of their value and cost-efficiencies for future population and fishery monitoring and assessment.

2. Materials and Methods

2.1. Fishery overview

The commercial fishery for beach clams in NSW developed throughout the 1950s and through a period of unrestricted fishing regulations, the numbers of clam harvesters and beaches accessed increased till total production peaked at 670,000 kilograms (kg) in 2001. This was followed by a sharp decline in commercial landings to 9,000 kg in 2011, despite increasing product prices and markets (Rowling et al., 2010). Recreational and indigenous catches throughout this period were also unrestricted and unchecked, and were probably large across many beaches (Murray-Jones and Steffe, 2000; Henry and Lyle, 2003). Although the reasons for the rapid decline in commercial catches remain unclear and potentially related to beach conditions and environmental variability, unrestricted harvesting probably contributed (Ferguson and Ward, 2014).

In response to the decade of decline in commercial catches of clams, several management initiatives designed to substantially reduce commercial fishing effort and harvest, and therefore halt further population declines were introduced to the NSW fishery in 2012. These included a six-month total commercial fishing closure, spatially explicit commercial fishing closures of whole beaches and specific zones along particular beaches, a maximum daily catch quota of 40 kg per-commercial fisher, and a minimum legal size limit (45 mm shell length, SL). Concomitant restrictions to recreational and indigenous fishers were also introduced, but these were primarily in response to concerns over human health issues associated with bio-toxins. These latter two groups can still harvest clams year-round across most beaches, but this is limited to 50 clams per day for immediate in-situ bait use only (unless for specific indigenous cultural events). The current combined harvest from these two sectors is therefore considered to be much smaller than the commercial harvest (Murray-Jones and Steffe, 2000; Rowling et al., 2010). The harvesting of clams by all sectors is restricted to digging by hand, with no mechanical apparatus permitted.

The NSW commercial beach clam fishery is presently compartmentalised into seven designated regions, with clam harvesters being able to access specified beaches within each region. However, the size, numbers of permitted fishing beaches and commercial harvesters and hence commercial production, differ greatly among regions. Currently, there are 76 licence endorsements to harvest clams and the current value of the fishery is approximately $AUD 2 million per annum (Rowling et al., 2010).

2.2. Sampling strategy

This study was done throughout the 2013 fishing season (1 June to 30 November) across two commercial beach clam fishing regions in NSW: Region 1 (latitude 29°00’-29°15’ S), which solely comprised one commercially fished beach (South Ballina, length 30 km), and Region 3 (latitude 29°45’-31°44’ S) that encompassed several commercially fished beaches (the main three being: Smoky, 16 km; Killick, 12 km; and Goolawah, 7 km). The field sampling and analyses were stratified at the spatial management level of Region, as opposed to individual beaches. This was also necessary for logistic purposes for the beach-based sampling component as it was not possible to accompany fishers across any predetermined beach as they often moved among beaches depending on beach and clam conditions. Three fishery-dependent data steams were assessed; industry logbooks, and beach- and port-based sampling of catches.

2.2.1. Industry logbooks

Commercial fishers are mandated to report to the NSW Government the beach and region, effort in hours spent harvesting (time on beach) and the total retained catch (kg) of clams for each days fishing each month.

2.2.2. Beach-based sampling

A scientific observer accompanied a commercial clam harvester on four (Region 1) or five (Region 3) randomly selected fishing trips (days) each month during the fishing season. The exception was in Region 1 when only one day was done in October due to the fishery being predominantly closed because of bio-toxins in water samples. For Region 3, observed trips covered the three predominantly harvested beaches: Smoky, Killick and Goolawah. On each trip the observer recorded the time spent searching and that actively digging clams (minutes), the number, location (GPS) and habitat (swash versus dry sand) of each harvesting event, and the total retained and discarded catch (kg) and sizes (shell length - SL, mm) of clams.

2.2.3. Port-based sampling

Retained clam catches were sampled for size composition on a weekly basis at a local cooperative in Region 3 throughout the study. The shell length (mm) of all clams in 1 kg subsamples of available catches (generally 2 or 3 per sampling day) were measured and pooled to form a monthly total size composition.

2.3. Analyses

Permutational analyses of variance (PERMANOVA, Anderson et al., 2008) were used to test for differences between data collection strategies. Three factorial analyses with the levels Data source (i.e. logbook v beach), Region and Month were used to test for differences in fishing effort (hours per day) and CPUE (catch-per-day and catch-per-hour/day). A two factor design tested for differences in size compositions of catches between the port- and beach-based samples (Factors: Data and Month) obtained in Region 3, and between the beach-based samples obtained across both regions (Factors: Region and Month). Two factorial PERMANOVA were also used to test for differences between regions and across months in the time spent searching and digging for clams, the number of locations (patches) of clams fished per day, and the ratio of retained to discarded clams in observed (beach-based) catches. All factors were considered fixed and each univariate analysis was based on the Euclidean distance measure, with Type III (partial) sums of squares calculated using 999 unrestricted permutations of the raw data.

Estimates of total (+SE) retained clam catches based on the beach sampling data were determined by multiplying the mean catches per observed trip with the number of reported fishing trips on logbooks for each month and region (Liggins and Kennelly, 1996; Gray et al. 2001).

Linear (alongshore) distances were calculated among fishing events based on the GPS coordinates taken during beach sampling. The maximum distance between fishing events and the proportion of events that occurred within a set of distances across South Ballina, Smoky and Killick beaches was determined.

3. Results

3.1. Catch, effort and CPUE: Logbook versus beach sampling

Throughout the fishing season, the logbook data identified that a total of 4 and 13 commercial fishers reported harvesting clams on a total of 272 and 888 days in regions 1 and 3, respectively. In region 3, a total of 493, 223 and 88 fishing days were reported for Smoky, Killick and Goolawah beaches respectively, with a further 84 trips reported across another four beaches (Stuarts, North Port Macquarie, Lighthouse and Dunbogan). Beach samples comprised 18 fishing days on Killick, 9 on Smoky and 3 on Goolawah and covered 7.7% and 3.4% of total logbook reported days fished in each of regions 1 and 3, but the level varied among months within each region (Region 1: 5.8 to 12.9% excluding October; Region 3: 2.1 to 13.9%). Across both regions, the reported number of days fished (i.e. trips) per month was greatest in July and August, after which it declined, particularly in Region 3 (Fig. 1B). The number of days fished per month ranged from 6 to 69 days in Region 1 and 36 to 249 days in Region 3.

On logbooks, the commercial clam harvesters reported retaining a season cumulative total of 10,400 and 30,200 kg of clams in regions 1 and 3 respectively. For Region 3, a total 17,800, 7,300 and 3,000 kg of clams were harvested from Smoky, Killick and Goolawah beaches respectively, with a further 2300 kg reported taken from the other four beaches. In comparison to logbooks, the estimated total retained catch based on the beach sampling data was slightly higher for each region at 11,200 (±150 SE) and 32,500 (±1100 SE) kg. The beach-based catch estimates for each month were also slightly higher than reported on logbooks, the notable exceptions being October and November in Region 3 (Fig. 1A). Nevertheless, temporal trends in total catches within each region were similar across both data streams, being greatest in July and August across both regions. These trends mirrored logbook reported total fishing effort in days fished (Fig 1A,B).

The number of hours per day spent harvesting (combined searching and digging) clams did not significantly differ between the logbook and beach data (PERMANOVA, P(perm) > 0.05, Table 1), but it did significantly (P(perm) < 0.05) differ between regions, with both logbook and observed effort being consistently greater in Region 3 (Fig. 1D). The analysis did not detect any significant differences among months within either region in the time spent harvesting clams.

Estimated CPUE (both catch-per-day and catch-per-hour/day) did not significantly differ between the logbook and beach-based data (PERMANOVA, P(perm) > 0.05, Table 1, Fig. 1C,E). However, the catch-per-hour data significantly differed according to the Data x Region interaction (Table 2); catch-per-hour was generally greater in logbooks compared to beach observations in Region 3, but not in Region 1 where it fluctuated among months (Fig. 1E). Moreover, both the logbook and beach data for both CPUE indices (catch per day and catch per hour/day) did not significantly differ among months in Region 1, but they were significantly less in October and November than the previous months in Region 3 (Fig. 1C,E). Notably, temporal and spatial trends in catch, effort and CPUE were mostly consistent across the two data streams.

3.2. Data obtained solely by beach sampling

The beach sampling also collected additional fishing-related data not reported on logbooks. The partitioning of fishing effort into actual search and dig time identified that on observed days, the time spent searching for suitable fishing locations was highly variable and did not significantly differ between regions or among months (PERMANOVA, P(perm) > 0.05 for both factors and their interaction), with average (+1 SE) observed search time being 29.5 (7.2) and 45.5 (7.0) minutes for regions 1 and 3 respectively (Fig. 2A). In contrast, the actual time spent digging significantly differed between regions (PERMANOVA, df = 1, 39, Pseudo-F = 20.30, P(perm) = 0.001), but not months (PERMANOVA, P(perm) > 0.05) (Fig. 2B). The average observed digging time was 85.7 (6.1) and 139.7 (8.4) minutes in regions 1 and 3 respectively. There was no significant relationship between the time spent searching and digging for either region (Region 1: R = 0.27, P > 0.05; Region 3: R = -0.31, P > 0.05).

The beach sampling data further identified that the number of sites (or patches of clams) fished on a day varied from 1 to 9 (Fig. 2C), with no significant differences evident across regions or months (PERMANOVA, P(perm) > 0.05 for both factors and their interaction). The mean (+ SE) observed number of sites fished across all months was 2.7 (0.47) and 3.2 (0.32) for regions 1 and 3 respectively. Approximately 50 and 80% of observed harvesting events in regions 1 and 3 occurred in the swash zone compared to the dry sand habitat. The location data identified that harvesting was generally confined to a small area of each beach, with the maximum distance between observed fishing events being 4, 4 and 7 km across South Ballina, Smoky and Killick beaches respectively. Further, 64, 88, and 37% of catches occurred within a linear distance of 2 km along each of these beaches.

For observed trips, the proportion of sublegal clams gathered but then returned to the substratum upon sorting of catches was low, with the mean (+SE) ratio of legal to sublegal clams being 1:0.11 (0.015) and 1:0.14 (0.022) in regions 1 and 3 respectively. The ratio of legal to sublegal clams did not significantly differ according to either region or month, or their interaction (PERMANOVA, P> 0.05 in all cases).

3.2. Size compositions: Port- versus beach-based sampling

The size composition of clams significantly differed between the port- and beach-based samples (PERMANOVA, df = 1,10, MS = 2695, Pseudo-F = 11.04, P(perm) = 0.007; Fig. 3). Across all months except October, the beach samples contained a greater proportion of clams 45-50 mm SL than the port samples, whereas the opposite was evident for clams > 50 mm SL.

The size compositions of clams determined by beach sampling differed significantly between regions 1 and 3 (PERMANOVA, df = 1, 10, MS = 734, Pseudo-F = 3.10, P(perm) = 0.025) and these were consistent across months (Fig. 3). A greater proportion of clams > 55 mm SL were present in samples in Region 1 compared to Region 3.

4. Discussion

Industry logbook data could form the primary data source for future monitoring and assessment of catch and effort in this study beach clam fishery. Even though there were some differences between the logbook and beach-based data in the actual values of some indices (e.g. catch-per-hour in Region 3), both data streams generally displayed similar spatial and temporal trends in catch, effort and CPUE. The beach-based data verified the legitimacy of the logbook catch and effort data, and thus its potential value as a data source in monitoring this fishery.

Nevertheless, caution must be exercised in interpreting CPUE in this fishery. For example, the CPUE index of catch-per-day would not be an appropriate measure for monitoring the relative densities of clams due to the imposed 40 kg daily trip limit. Notably in Region 1 where CPUE remained stable across the entire season, 85% of logbook and 100% of beach observed trips reported attaining the maximum 40 kg of clams per day. Under this management arrangement, use of this index may produce hyper-stable CPUE rates that may not reflect actual clam densities either at the local within-beach, or broader beach- or region-wide, scale (Hilborn and Walters, 1992).

Although the CPUE index of catch-per-hour may provide a better measure of clam densities in this fishery, it still has potential problems. The fishing of clam aggregations, particularly across small spatial scales as observed here, can produce CPUE values that may only reflect densities at very local (areas fished) scales and not across an entire beach or region, as reported for similar aggregation-based abalone fisheries across reefs (Prince, 1992). Further, in other fisheries that target specific aggregations across small scales compared to the entire stock distribution, local CPUE has remained stable even under declining stock abundances as a result of fishers finding enough local aggregations to fish (Rose and Kulka, 1999). Consequently, whilst fishers continue to find suitable clam aggregations to harvest across small-scales on a beach, CPUE may neither change nor reflect clam densities. Indeed, changes in CPUE may only be evident when the numbers of aggregations drastically decline or dissipate as clams become less patchy and dense. This may have occurred in October and November in Region 3. However, across these two months the time observed searching and digging was similar to the previous four months, but there was an increase in the number of locations fished from September through November. This, however, could potentially be an artefact of fishers altering their behaviour in the presence of an observer, especially when clam densities (and aggregations) may have been low. Further, during this period catch-per-day of beach data was lower than reported on logbooks, which contrasted the previous four months.

Beach sampling provided some unique data, including the partitioning of fishing effort into search and dig time and the number, habitat and location of fishing events that may be helpful in future assessments and management. Incorporation of actual search and dig times and the numbers of locations fished may help unravel complex relationships between fishery CPUE and clam densities across different spatial scales. This aspect of fishing behaviour is an important avenue of research that may help assess potential effects of fishing on clams.

The quantification of areas and habitats fished across beaches could help determine the extent of any potential impacts of clam harvesting on beach ecosystems, as well as mitigate concerns from other beach user groups about over exploitation and fishing-related impacts on beaches. Here, fishers concentrated their fishing activities into a relatively small area on each beach where clams were primarily aggregated and local densities high. When possible, fishers also chose to fish areas away from local beach access points where other beach users typically congregated, to potentially avoid social conflicts. Knowledge of the proportion of available habitat clam harvesters actually utilise across beaches could also be incorporated into fisheries and conservation management strategies concerning protected and closed fishing areas (Gray, 2016b,c). Such initiatives need to be considered in the broader ecosystem-wide context of managing sandy beaches (Defeo et al., 2009).

Concentrated fishing as observed here could lead to localised depletions of clams on beaches. However, the potential mixing and movements of clams along and across beaches (Leber, 1982; Dugan and McLachlan, 1999) may preclude, or at least mitigate or mask such effects. A better understanding of the ecological processes that drive movements, spatial aggregations and small-scale patchiness of clams across beaches is required to further understand the potential impacts of fishing on clams (Gray, 2016a).

Future collection of these additional data in this fishery could be incorporated into the industry logbooks. Clam harvesters could partition their fishing effort into actual search and dig times, and if logbooks were modified to include beach maps with spatial grids, then fishers could report the area (e.g. grid code) and habitat of each days harvesting along with the number of locations (or patches of clams) fished. Sourcing this extra information from industry via logbooks would be a cheaper option and more cost-effective than employing independent persons to obtain this data across beaches. This is equally important whether industry or government bears the costs of such programs. Regardless, the value of industry supplying such extra data would have to be clearly demonstrated and delivered to fishers for cooperation. At present, different clam fishers interpret and report 'fishing time' on logbooks in different ways. There is a need to better educate fishers on what they should be reporting for greater standardisation of data within and across regions. This would be best done in a co-management arrangement, where fishers take greater ownership in managing the fishery and thus greater value in the data they report (Defeo et al., 2014).

Future monitoring of size compositions of retained catches would be best based at cooperatives and not the beach. The fact that the size compositions of clams differed between the port- and beach-based sampling strategies was probably due to fishers altering their fishing behaviours in the presence of an observer. This is a common issue in observer-based studies that needs to be examined within each program (Liggins et al., 1997; Faunce and Barbeaux, 2011). Here, clam harvesters probably concentrated on gathering their 40 kg catch limit in the shortest possible time (thereby maximising observed CPUE), and in doing so were most likely less selective in the sizes of clams they retained in the presence of an observer.

In addition to alleviating any possible sampling biases, port- as opposed to beach-based sampling would be cheaper and overall more cost-effective in obtaining size composition data across broader spatial scales for assessment purposes. Several independent catches from different harvesters and potentially different beaches (as in Region 3) could be sampled on any given day at a cooperative, whereas only one sample (but potentially more) could be obtained on the beach. Moreover, beach sampling potentially interferes with clam fishers operations and imposes on their time (e.g. fishers do not want to wait around at the end of a trip for their catches to be measured), making it in some instances logistically difficult to accomplish. The size compositions of clams can differ markedly among beaches within a fishing region (Gray et al., 2014; Gray, 2016a,b), and therefore it would be most pertinent and strategic to capture such data at the level of beach.

Future monitoring and assessment of the beach clam fishery should be beach- and not region-based. Clam populations are highly dynamic and variable among individual beaches (Gray, 2016a,b) and region-based assessments could be confounded depending on what mixtures of beaches are sampled. Moreover, in some regions fishers harvest clams across several beaches, and individual beaches within a region can be open or closed to fishing depending on clam and beach conditions within and among fishing seasons. It is therefore important to understand the dynamics of clam populations at the level of individual beach, including non-fished beaches.

5. Conclusions

Comparative studies as done here and elsewhere (e.g. Lunn and Dearden, 2006) can help clarify and evaluate the appropriateness and cost-benefits of alternative data sources that are necessary for building long-term fishery monitoring strategies. Cost-effective monitoring in this beach clam fishery could be based on logbooks for catch, effort and CPUE and port sampling for size composition. A long-term strategy could involve a periodic (e.g. every 3-5 years) validation program that utilises beach sampling to 'truth' industry logbook data. This may help mitigate any criticism regarding the reliability of industry-reported data for assessment and management purposes. Such a strategy, however, relies on good industry-management relations. Nevertheless, fishery-dependent data sources are only available for beaches that are commercially fished and since standardised protocols for sampling beach clams have been developed and tested (Gray et al., 2014; Gray, 2016a,b), their incorporation into a long-term strategy for population assessments needs consideration.

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Figure Captions

Fig. 1. Total reported catch and effort, estimated total catch and mean (+ SE) effort and CPUE data determined by logbook and beach sampling for each month across study regions 1 and 3.

FIGURE 1

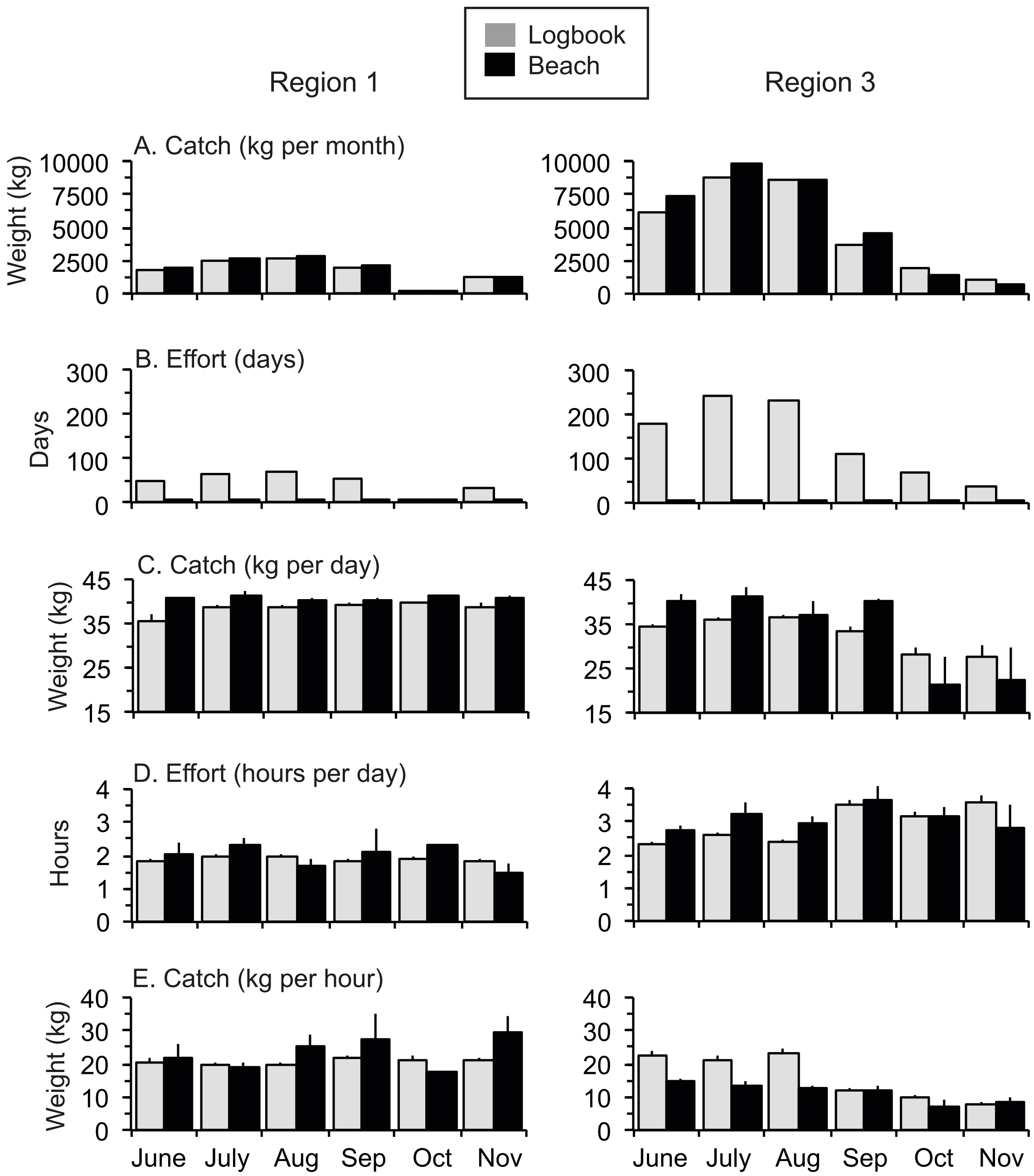

Fig. 2. Mean (+ SE) time spent searching and digging beach clams, and the number of locations fished per day for each month across study regions 1 and 3 determined by beach sampling.

FIGURE 2

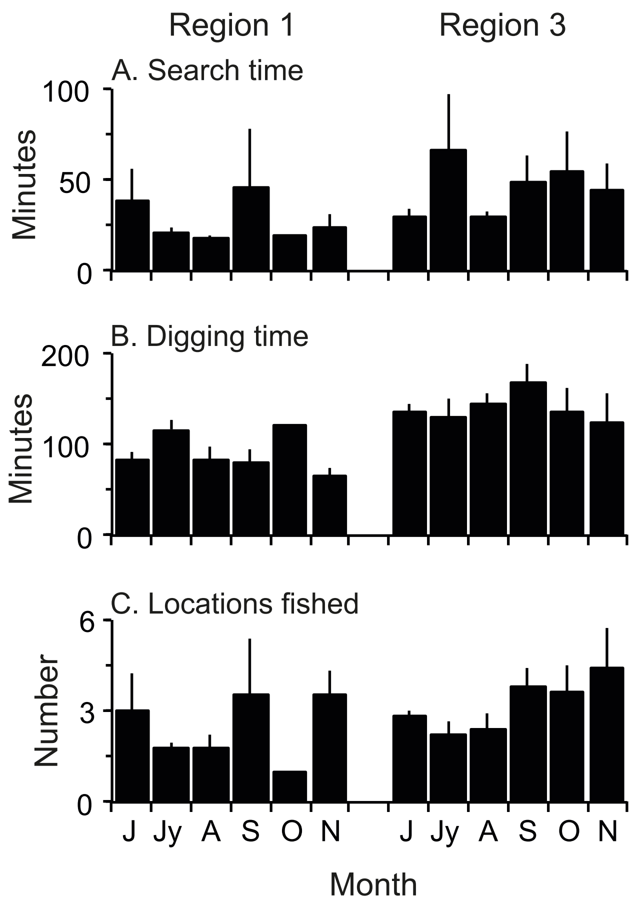

Fig. 3. Size composition data of retained catches of beach clams as determined by beach sampling across regions 1 and 3 and port sampling in Region 3. n denotes sample size.

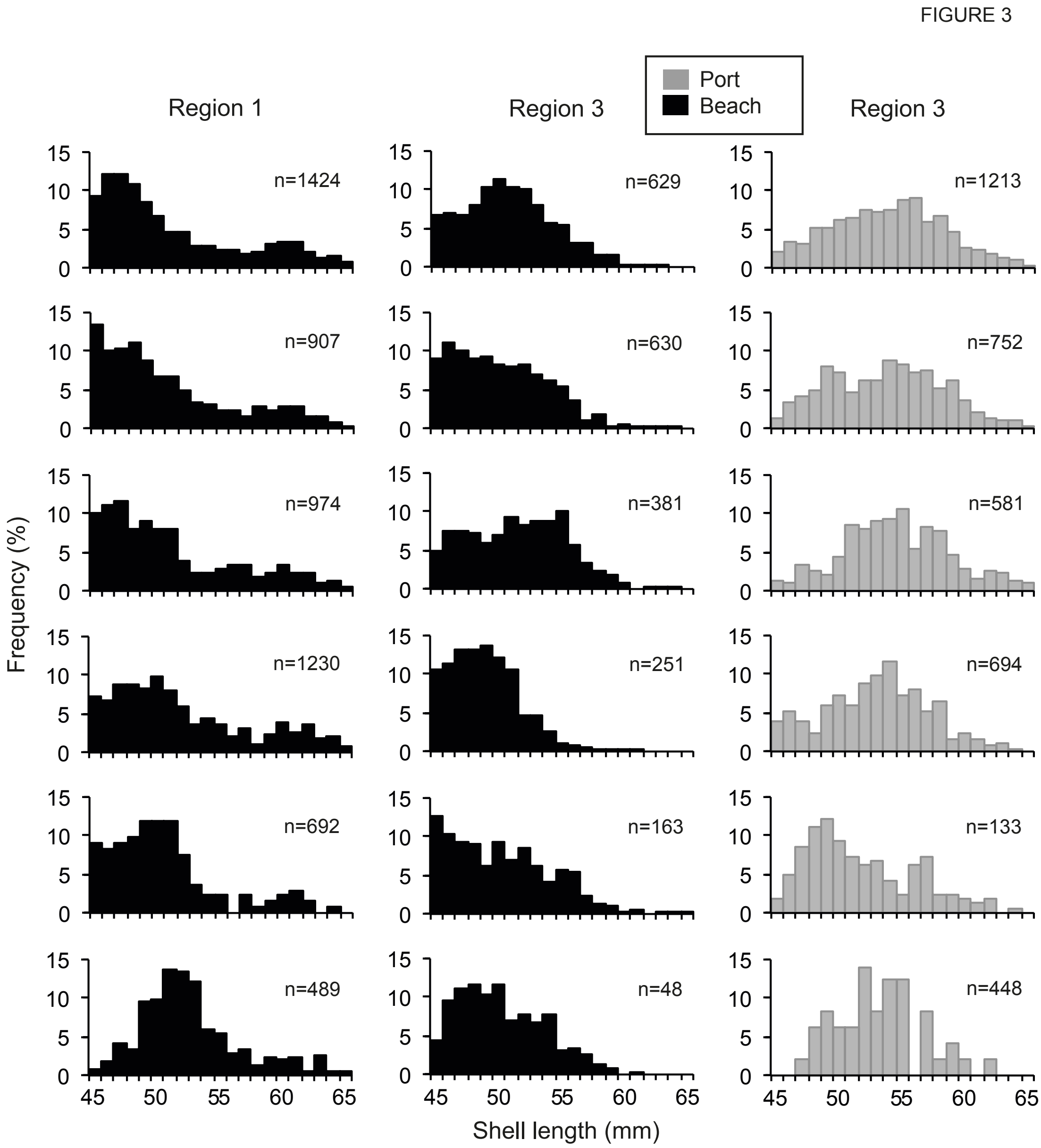

Table 1. PERMANOVA results comparing fishing effort, catch-per-hour and catch-per-day between logbook and beach data.

		Effort (Hours)			Catch (kg per hour)			Catch (kg per day)		
Source	df	MS	Pseudo-F	P(perm)	MS	Pseudo-F	P(perm)	MS	Pseudo-F	P(perm)
Data Source	1	0.578	0.383	0.536	50.128	0.374	0.546	112.160	1.615	0.195
Region	1	42.528	28.149	**0.001**	2644.900	19.743	**0.001**	1543.000	22.223	**0.001**
Month	5	1.468	0.972	0.416	106.400	0.794	0.583	237.600	3.422	**0.014**
Data x Region	1	0.051	0.034	0.867	536.020	4.001	**0.038**	18.101	0.261	0.618
Data x Month	5	1.083	0.717	0.594	103.200	0.770	0.564	69.723	1.004	0.363
Region x Month	5	1.446	0.957	0.424	306.180	2.286	**0.030**	283.470	4.083	**0.005**
Data x Region x Month	5	0.445	0.294	0.909	42.086	0.314	0.916	49.850	0.718	0.572
Rersidual	1173	1.5108			133.97			69.434		
Total	1196									

Appendix 8.

Gray, CA. Assessment of the across-beach distributions of the beach clam, *Donax deltoides*

Assessment of the across-beach distributions of the beach clam, *Donax deltoides*

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1. Introduction

A general paradigm in ocean beach ecology is that the distributions and abundances of organisms are primarily determined by abiotic factors (McLachlan et al., 1993; Defeo and McLachlan, 2005, 2013). Exposed ocean beaches are considered environmentally harsh and dynamic habitats and the fauna inhabiting them often display distinct shore-parallel and shore-perpendicular distributions that are associated with particular beach topographic features (McLachlan and Hesp, 1984; McLachlan and Jaramillo, 1995; Defeo and McLachlan, 2013).

Mobile fauna such as beach clams can actively alter their along- and across-shore distributions in response to environmental conditions such as wave and swell direction and strength (Leber, 1982; Dugan and McLachlan, 1999; Scapini, 2014). For example, beach clams have been observed riding the swash up and down beaches in response to abiotic and biotic variables and different species and sized clams can inhabit different zones across beaches (Leber, 1982; Ellers, 1995). Knowledge of the distributions of clams and how they vary in space and time and with size (life history stage) is required for designing resource surveys, determining population sizes and assessing anthropogenic impacts such as harvesting (Defeo and Rueda, 2002; Defeo et al., 2009).

The sampling reported here was done to identify the across-beach distributions of the beach clam *Donax deltoides* and investigate whether this differed according to sections of beaches with different topographic features and profiles. It further examined whether clam distributions varied according to lunar phase and size of organism. Sampling was done across three beaches to test for generality of results.

2. Methods

Sampling was done across three high-energy beaches open to commercial fishing: Lighthouse, North Port Macquarie and Goolawah. Preliminary sampling also was done across Killick and Warilla beaches but low clam densities at the time of sampling precluded further sampling from occurring.

Sampling of Lighthouse and North Port Macquarie occurred between 5 November and 12 December 2014 whereas Goolawah was sampled between 15 and 18 June 2015. Sampling of Lighthouse and North Port Macquarie was stratified by lunar phase, with sampling occurring 3 days around each quarter of the moon over six weeks. Sampling across each beach was further stratified according to the predominant topographic features and beach profiles present; primarily flat sections, and steeper berms and adjacent swales when present. Across Lighthouse Beach flat sections and steeper sections of the beach where an alongshore berm was present were sampled during the 1st three sampling times. After this time, swales that interspersed the alongshore berm had developed that produced cusp-like features, and these swales were included as a sampling feature in subsequent periods and across North Port Macquarie Beach. Across each beach, two such topographic features were randomly chosen and examined each sampling time.

The sampling of Goolawah Beach was spatially stratified to incorporate sections at either end and in the middle of the beach. Seven sites spread across the beach were each sampled at least twice over the four sampling days to account for small-scale variability (Gray, 2016a).

At each sampling site (feature), transects perpendicular to the shoreline from the previous high tide mark to the edge of the swash zone were made and the presence of clams at approximately 1 m intervals along each transect was determined. This sampling identified the upper and lower levels of the dry clam belt (when present). Six replicate samples using a box quadrat with 32 cm sides (James and Fairweather, 1995) were then used to quantitatively sample clams for density and size composition within each identified belt. The swash sampler was used to sample clams in the swash zone (Gray et al., 2014).

All clams collected in each replicate quadrat or swash sample were counted and measured for shell length (SL, mm) using digital calipers. Across each transect, the distance from the preceding high tide mark to the upper and lower level of the dry clam belt, and to the shoreward edge of the swash zone was measured (m). The slope of the beach was determined using a digital level and this was done at the edge of the swash zone, within and above the clam belt (when present).

Low densities and sporadic occurrences of clams prevented meaningful statistical analyses of the data. Hence, the across-beach distributions of clams were described relative to beach features. The size compositions of clams in the dry and swash were examined when possible. This was done for each beach separately by combining the size composition data across all sites and times when clams were present across both the swash and dry habitats. The resulting size compositions were compared using Kolmogorov-Smirnov tests.

3. Results

3.1. Lighthouse Beach

Clams were sampled in the swash zone on all eight sampling occasions, but were only present in the dry habitat across four such occasions (Table 1). On the 1st sampling date the identified dry clam sand belt was 6-12 m wide and situated between 2-10 m above the landward edge of the swash zone at low tide. On the following day, the clam belt was 3-9 m wide and 4-13 m above the swash zone. A week later clams were present in the dry across two sites, approximately 4-11 m above the swash zone and the belt was 5-6 m wide. In the last sampling period, clams were present only in the dry across the flat section of the beach where they occurred in a belt 2-4 m wide and 2-6 m above the swash zone.

Non-quantitative sampling (19-21 November) of shallow gutters and sand banks adjacent to the swash zone identified that clams were present in such seaward habitats (Table 1). Note that this sampling could only be done during very calm conditions and that clams were hard to observe and sample and their densities could not be determined.

Low densities of clams precluded meaningful comparisons of size compositions between the dry and swash across each topographic feature and sampling period. Nevertheless, sizes of clams pooled across the 1st three sampling dates for each habitat type differed significantly between swash and dry habitat (KS test, $P > 0.05$) (Fig. 1). Most clams sampled were > 40 mm SL and few small clams (< 20 mm SL) were sampled across either habitat.

3.2. North Port Macquarie Beach

Clams occurred in the swash zone across each topographic feature in all four sampling periods (Table 2). In contrast, clams were encountered in the dry across all topographic features in the 1st period, and only across the flat section of the beach in the last period. This latter observation corresponded with low clam densities in the swash zone (Table 2). In the 1st period the dry clam belt was 2-9 m in width and located 6-12 m above the edge of the swash zone, depending on the topographic feature. In the last period the corresponding belt was 7-11 m wide and located 3-10 m above the edge of the swash zone.

In the last period clams in the flat section of the beach remained high with the dropping tide whereas adjacent to the berms and swales clams were observed moving down the beach with the outgoing tide. At low tide these clams were located at the seaward edge of the swash zone on the edge of a gutter that ran along the beach. No clams were collected in the shallow subtidal gutter. In the second and third periods clams were also observed across all sampled locations low in the swash zone about 15-20 seaward adjacent to the alongshore gutter. It could not be determined if clams were located in the gutter due to swell size.

The size composition of clams differed significantly between the swash and dry clam belt (K-S test, $P > 0.05$), with greater proportions of clams < 35 mm SL sampled in the swash habitat (Fig. 1).

3.3. Goolawah Beach

Clams were sampled in the swash and dry zones across all sampling days and sites (Table 3). Across several sites two identifiable and separate clam belts were evident (e.g. Site B Day 2, Site C days 2 and 4, Site F Day 1). These dry habitat belts were separated by 3-11 m depending on the site and day. The distance of the upper clam belt differed among locations and days. However, on Day 4 the upper clam belt occurred at a higher level on the beach and closer to the high tide mark than the preceding days, suggesting that clams had moved up the beach.

The KS test provided a significant ($P > 0.05$) difference in the size composition of clams sampled in the swash and dry clam belt. However, the low sample size for the swash may have caused this difference. Notably, small clams < 20 mm SL were sampled across both habitats (Fig. 2).

4. Discussion

Physical processes such as waves, currents and interactions with beach profiles are purported to drive the distributions of organisms, including clams, across exposed ocean beaches (McLachlan et al., 1995; Defeo and McLachlan, 2005). Here, there was no evidence that clams were distributed in different ways in a consistent manner across sections of beaches with different beach profiles, or along different sections of a beach exposed to different physical forcing. Clams displayed flexible across-beach distributions in space and time that were not predictable according to beach profile, alongshore position on a beach and measured environmental conditions. When clams did inhabit the dry sand, they occurred at varying distances between the high tide level and the swash zone. It was not possible here, however, to quantify the processes responsible for driving such variability in the across-beach distributions of clams.

Small-scale variability in the distributions and densities of clams, along with the width and location of the dry sand clam belt between the high tide level and the swash zone was evident. The across-beach distributions of clams differed between subsequent days on Lighthouse and Goolawah beaches, and were not consistent along different sections of each beach or for particular topographic features between beaches. Moreover, on many occasions clams only occurred in the swash zone and were not present in the dry sand. This extended for four weeks on Lighthouse and North Port Macquarie beaches, which suggests some ecological process may be concordant across large spatial and temporal scales.

Clams can actively alter their across-beach distributions by utilizing the swash and backwash (Leber, 1982). The across-beach tidal movements of clams may at times be synchronized across beaches. For example, on Day 4 of sampling on Goolawah Beach the upper clam belt at all sampled sites was situated much higher and closer to the high tide level than on the previous days. This suggested that clams across the entire beach moved up the beach in a concordant manner. Nevertheless, the actual distance of this upper clam belt from the previous high tide mark was variable and spatial differences in swash flow and strength may have contributed to such variability (McLachlan and Hesp, 1984).

The size compositions of clams differed according to habitat; however small sample sizes may have influenced some results and the pooling of data across sites and sampling dates may have confounded comparisons. Small (< 20 mm SL) clams were present in swash and dry habitats but a greater proportion was sampled in the swash habitat on both North Port Macquarie and Goolawah beaches. On occasions, many small clams fell through the 12 mm mesh used to sample the swash habitat and thus the presence of small clams in this habitat may have been underestimated. Nevertheless, this and other sampling reported in the preceding chapters suggest that more small clams inhabit the swash than the dry sand. This has generally been observed in other clam species (Mikkelsen, 1981; Leber, 1982; Denadai et al., 2005). It is recommended that a specific method be developed to quantitatively sample small (< 10 mm SL) clams across beaches.

Opportunistic and qualitative sampling of the alongshore shallow gutters and tidal sand banks situated approximately 20-50 m seaward of the swash zone on Lighthouse and North Port Macquarie beaches identified that at times clams were present in such habitats. However, the extent of their distributions and actual densities could not be quantified by hand digging and sampling of such habitats was limited to low tide when the swell was < 0.5 m, and it was possible to access such habitats. It was not logistically possible to sample clams in the active surf zone. Even in calm conditions, clams were hard to locate in these seaward habitats as when located they buried fast and deep escaping capture by digging. Nonetheless, indigenous fishers at times harvest clams in these habitats but the extent to which they do this has not been quantified. Knowledge of these activities could help identify when clams are present in such habitats.

This qualitative sampling identified that clams at times do occur seaward of the swash zone. It is not known, however, whether clams actively inhabited or were passively swept by waves and currents into these seaward habitats and whether this occurs in both calm and storm conditions. Sampling the surf zone environment will require novel gears and techniques that may still only be able to be used during calm conditions, thus limiting our understanding of the seaward distributions of clams. Moreover, such techniques need to be able to sample or identify clams buried deep in the sediment and not just at the surface.

In conclusion, sampling and monitoring strategies to assess clam populations on ocean beaches need to incorporate designs that include both the swash and dry habitats, such as those described in the preceding experiments (Gray, 2016b). Moreover, such sampling needs to be flexible to account for the variable across-beach distributions of clams. Identification of the position of the dry sand clam belt (when present) using a preliminary sampling methodology as done here is required so that broader-scale sampling among habitats can be stratified accordingly. Utilizing local fishers unique knowledge in identifying the across-beach distributions of clams could assist in this process and be a further way to engage industry in sampling and the uptake of research results.

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Figure 1. Size compositions of clams sampled in the swash and dry habitats across: A. Lighthouse Beach – data for each habitat pooled across the 1st 3 sampling dates when clams were present across both habitats, and B. North Port Macquarie Beach – data for each habitat pooled for 1st sampling date.

Figure 1

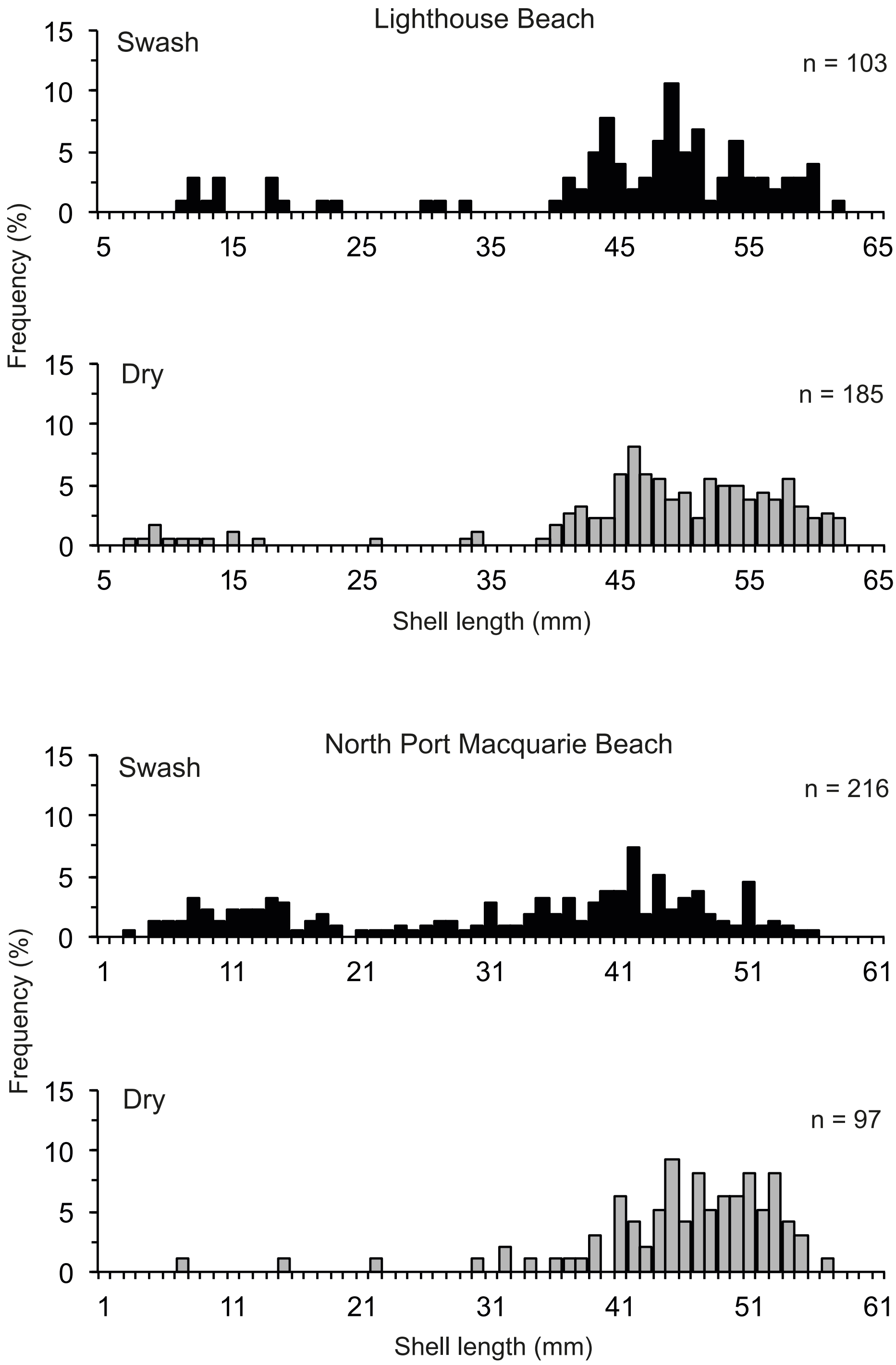

Figure 2. Size compositions of clams sampled in the swash and dry habitats across Goolawah Beach. Data for all sites and days pooled for each habitat.

Figure 2

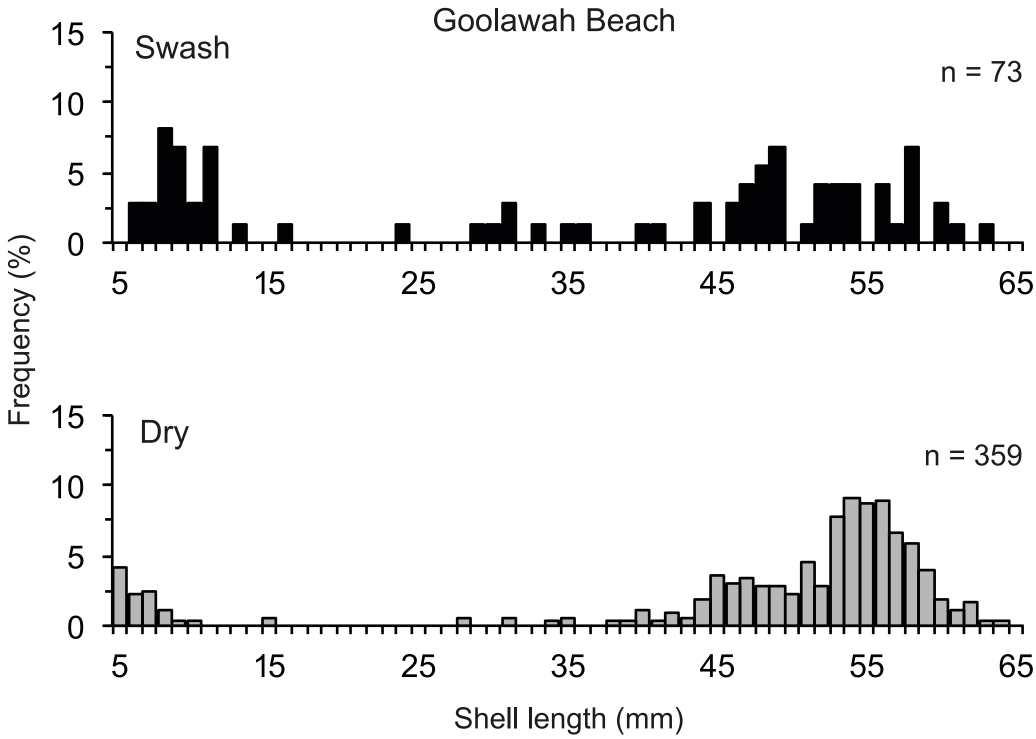

Table 1. Lighthouse Beach. Sumary of mean (+SE) densities of clams in the identified clams belt and the swash at each site and day sampled. The distance (m) between the hight tide mark to the top and bottom of the identified upper (HT-UCB) and lower (HT-LCB) dry sand clamd belt and the distance to the swash is given.

Profile	Swash	Swash	Clam Belt	Distance High Tide to (m)			Slope	Slope
	Mean	SE	Mean	SE	Swash	Clam Belt	Swash	Clam Belt
Last Quarter; Day 1								
Flat	0.0	0.0	1.3	0.6	44.0	24-30	2.1-2.3	2.2-2.7
Flat	0.0	0.0	2.7	0.6	37.0	23-30	2.1-2.2	2.2-2.6
Berm	1.7	0.6	3.0	0.4	36.0	16-28	3.4-3.6	4.3-6.3
Berm	4.3	0.8	5.0	0.7	32.0	21-30	4.1-4.9	7.0-7.2
Last Quarter; Day 2								
Flat	0.0	0.0	1.0	0.4	53.0	28-37	3.2-3.3	2.7-3.2
Flat	0.3	0.2	1.8	0.7	38.0	30-33	2.6-2.7	2.2-2.5
Berm	0.3	0.2	4.2	0.4	29.0	21-25	5.8-6.1	6.0-6.4
Berm	0.7	0.3	0.5	0.2	40.0	25-28	4.1-4.6	4.8-5.6
New Moon; Day 1								
Flat	0.7	0.5	6.3	1.6	31.0	14-19	2.8-3.1	3.1-3.6
Flat	0.0	0.0	0.0	0.0	35.0	NA	2.0-2.2	NA
Berm	2.2	0.3	0.2	0.2	37.0	27-33	4.8-5.9	6.2-6.9
Berm	2.7	0.6	0.0	0.0	29.0	NA	5.4-5.9	6.1-6.6
New Moon; Day 2								
Flat	3.0	1.7	0.0	0.0	32.0	NA	2.6-2.9	NA
Flat	0.7	0.4	0.0	0.0	31.0	NA	2.4-2.6	NA
Berm	1.5	0.5	0.0	0.0	18.0	NA	5.0-5.3	NA
Berm	1.8	0.7	0.0	0.0	21.0	NA	6.2-6.4	NA
First Quarter; Day 1								
Flat	4.5	1.9	0.0	0.0	30.0	NA	1.6-1.9	NA
Flat	1.3	0.2	0.0	0.0	42.0	NA	1.4-1.6	NA
Berm	5.8	1.8	0.0	0.0	19.0	NA	4.6-4.7	NA
Berm	0.0	0.0	0.0	0.0	26.0	NA	3.8-4.2	NA
Swale	6.0	1.2	0.0	0.0	19.0	NA	3.6-3.8	NA
Swale	2.0	0.8	0.0	0.0	28.0	NA	3.0-3.1	NA
First Quarter; Day 2								
Flat	1.3	0.5	0.0	0.0	42.0	NA	2.1-2.2	NA
Flat	8.2	1.1	0.0	0.0	34.0	NA	2.3-2.4	NA
Berm	1.3	0.7	0.0	0.0	23.0	NA	2.6-2.9	NA
Berm	2.7	0.4	0.0	0.0	20.0	NA	2.6-3.0	NA
Swale	0.7	0.3	0.0	0.0	28.0	NA	2.1-2.5	NA
Swale	1.8	0.7	0.0	0.0	25.0	NA	2.2-2.6	NA
Full Moon								
Flat	4.8	0.7	0.0	0.0	32.0	NA	2.3-2.6	NA
Flat	1.8	0.7	0.0	0.0	38.0	NA	2.1-2.4	NA
Berm	3.3	1.0	0.0	0.0	21.0	NA	3.9-4.4	NA
Berm	2.5	0.4	0.0	0.0	26.0	NA	4.1-4.3	NA
Swale	0.5	0.3	0.0	0.0	28.0	NA	3.8-3.9	NA
Swale	1.3	0.7	0.0	0.0	26.0	NA	3.3-3.8	NA
Last Quarter								
Flat	0.5	0.2	1.5	0.4	35.0	29-33	2.1-2.2	2.2-2.6
Flat	0.8	0.3	2.3	0.7	37.0	31-33	1.9-2.1	2.0-2.2
Berm	0.8	0.5	0.0	0.0	25.0	NA	2.3-2.5	NA
Berm	2.8	0.4	0.0	0.0	28.0	NA	2.1-2.3	NA
Swale	1.7	0.4	0.0	0.0	25.0	NA	2.2-2.4	NA
Swale	1.0	0.5	0.0	0.0	32.0	NA	2.1-2.3	NA
New Moon; large 4 m swell, 30+ SE wind, could not sample								

Table 2. North Port Macquarie Beach. Sumary of mean (+SE) densities of clams in the identified clams belt and the swash at each site and day sampled. The distance (m) between the hight tide mark to the top and bottom of the identified upper (HT-UCB) and lower (HT-LCB) dry sand clamd belt and the distance to the swash is given.

Profile	Swash Mean	Swash SE	Clam Belt Mean	Clam Belt SE	High Tide Swash (m)	High Tide Clam Belt (m)	Slope Swash	Slope Clam Belt
First Quarter; Day 2								
Flat	4.0	0.7	0.8	0.2	35.0	21-23	2.1-2.2	2.3-2.6
Flat	6.0	1.9	3.0	0.8	37.0	16-25	3.1-3.3	3.2-3.7
Berm	2.7	1.1	0.8	0.3	42.0	28-34	3.1-3.3	5.4-5.7
Berm	2.0	0.6	1.0	0.3	39.0	27-31	3.5-3.6	4.8-5.2
Swale	12.5	3.3	1.8	0.4	49.0	33-35	3.3-3.4	3.5-3.6
Swale	8.7	1.3	2.0	0.3	39.0	28-33	3.4-3.5	3.5-3.7
Full Moon								
Flat	3.0	0.9	0.0	0.0	38.0	NA	2.2-2.4	NA
Flat	9.0	3.9	0.0	0.0	45.0	NA	2.2-2.5	NA
Berm	5.7	1.4	0.0	0.0	39.0	NA	5.9-6.7	NA
Berm	3.0	0.7	0.0	0.0	56.0	NA	5.1-5.4	NA
Swale	1.0	0.3	0.0	0.0	38.0	NA	3.3-3.6	NA
Swale	1.5	0.4	0.0	0.0	45.0	NA	3.1-3.4	NA
Last Quarter								
Flat	17.0	5.0	0.0	0.0	42	NA	2.1-2.3	NA
Flat	0.8	0.4	0.0	0.0	40	NA	2.2-2.3	NA
Berm	3.7	1.2	0.0	0.0	34	NA	4.9-5.4	NA
Berm	10.0	2.9	0.0	0.0	31	NA	5.1-5.7	NA
Swale	8.2	0.9	0.0	0.0	39	NA	3.4-3.9	NA
Swale	3.2	0.6	0.0	0.0	36	NA	3.7-4.1	NA
New Moon								
Flat	0.2	0.2	3.5	0.9	33.0	19-30	1.8-2.0	3.5-3.6
Flat	0.0	0.0	3.2	0.7	36.0	19-26	2.1-2.7	3.2-3.5
Berm	3.2	0.7	0.0	0.0	28	NA	4.5-4.8	NA
Berm	7.2	1.5	0.0	0.0	26	NA	4.4-4.6	NA
Swale	6.5	1.1	0.0	0.0	31	NA	3.8-4.2	NA
Swale	5.7	1.0	0.0	0.0	33	NA	3.4-3.9	NA

Table 3. Goolawah Beach. Sumary of mean (+SE) densities of clams in the identified clams belt and the swash at each site and day sampled. The distance (m) between the hight tide mark to the top and bottom of the identified upper (HT-UCB) and lower (HT-LCB) dry sand clamd belt and the distance to the swash is given.

Location	Site	Day	Upper Clam Belt		Lower Clam Belt		Swash		HT-UCB	HT-LCB	HT-Swash
			Mean	SE	Mean	SE	Mean	SE			
North Corner	A	2	2.2	0.5			0.2	0.2	35-41		55
North Corner	A	3	1.3	0.4			0.2	0.2	47-52		75
North	B	1	2.8	0.9			0.3	0.2	15-21		
North	B	2	0.7	0.3	4.0	0.6	0.7	0.7	24-31	37-43	60
North	B	3	4.7	0.5			0.5	0.2	13-23		65
Mid	C	1	2.3	0.6			1.2	0.4	26-29		40
Mid	C	2	1.0	0.5	2.2	0.5	2.2	0.9	9-16	21-23	40
Mid	C	4	1.7	0.6	2.8	0.5	0.0	0.0	6-9	12-15	25
Mid	D	2	3.8	0.5			0.0	0.0	13-18		50
Mid	D	3	3.5	0.3			0.0	0.0	16-19		45
Mid	D	4	1.5	0.4			2.2	0.6	8-14		55
Mid	E	2	2.0	0.4	1.2	0.5	0.7	0.2	34-37	45-50	55
Mid	E	3	2.2	0.7			0.2	0.2	15-19		35
Mid	E	4	3.0	0.5			0.5	0.2	5-11		40
South	F	1	4.2	1.6	1.3	0.3	1.2	0.3	22-27	38-45	65
South	F	3	4.2	1.0			0.5	0.3	27-30		55
South	F	4	3.2	1.1			0.2	0.2	7-14		40
South Corner	G	2	1.2	0.2			0.0	0.0	19-27		50
South Corner	G	4	3.0	0.7			0.2	0.2	3-10		40

WildFish Research Reports:

Gray CA, Young CL, Johnson DD, Rotherham D (2015) Integrating fishery-independent and -dependent data for improved sustainability of fisheries resources and other aspects of biodiversity. FRDC Project 2008/012. ISBN 978-0-9941504-1-7

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