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Environmental effects of yellowtail kingfish aquaculture in South Australia

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ABSTRACT: The culture of yellowtail kingfish *Seriola lalandi* in South Australia is one of the most rapidly growing aquaculture sectors in Australia. To date, there is a paucity of information on the environmental impacts of this industry, due to its relatively small size in comparison to other industry sectors. Here, we report on a study examining the response of a range of environmental variables to yellowtail kingfish aquaculture in Fitzgerald Bay, northern Spencer Gulf, South Australia. The clearest response occurred in several chemical variables, with an 81% increase in ammonia concentrations adjacent to cages relative to controls, increased sediment organic carbon (7 to 46%) and increased porewater phosphorus in the sediments (80 to 3077%). While there were also statistically significant effects on infaunal and epifaunal assemblages, results for both groups were equivocal because of high small-scale variability. For the infauna, assemblages at the sites furthest from cages tended to be similar to those adjacent to cages, while those at intermediate distances differed. No impacts were detected on phytoplankton, or on seagrasses (although the latter only occurred >250 m from any aquaculture cage). The lack of clear responses in the biotic datasets is probably related to the relatively low production levels (<2000 t yr^{-1} at the time of the study), although the responses in the chemical variables provide a valuable early warning as production increases.

KEY WORDS: Benthic impacts · Environmental impact · Seagrass disturbance · *Seriola lalandi* · Yellowtail kingfish · Water quality

INTRODUCTION

Aquaculture is growing rapidly worldwide as increasing demand for seafood conflicts with static production from the wild fisheries sector (Naylor et al. 2000). While this growth can have substantial economic and social benefits for rural communities, many of which, at least in Australia, are suffering from declines in employment in the traditional farming sector, careful consideration needs to be given to the potential ecological costs of aquaculture. Historically, most of the attention paid to environmental impacts of finfish aquaculture has focussed on salmonid farming in cold temperate waters, especially in Europe and North America (e.g. Brown et al. 1987, Gowen & Bradbury 1987, Frid & Mercer 1989). More recently, considerable attention has been paid to aquaculture in the warm temperate waters of the Mediterranean (Klaoudatos et al. 2006, Valle et al. 2007, Pitta et al. 2009, Piedecausa et al. 2010), and to other areas of the world, predominantly in the northern hemisphere (Holmer et al. 2003, Islam 2005, Lee et al. 2006). For example, in a recent review of benthic impacts of aquaculture, Kalantzi & Karakassis (2006) document 41 published studies, 30 of which were on salmonids, and an additional 7 were from the Mediterranean. In Australia, finfish aquaculture is concentrated in cold-temperate Tasmania (salmonids) and warm-temperate South Australia (southern bluefin tuna *Thunnus maccoyii*, and yellowtail kingfish *Seriola lalandi*). While southern bluefin tuna is by far the largest sector in South Australia, with a production of 9757 t (processed) worth $US163 million in 2007/08 (Econsearch 2009), growth is currently limited by a quota on the wild catch. The

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environmental impacts of this sector have been extensively studied (e.g. Fernandes et al. 2007, Tanner 2007, Tanner & Volkman 2009). The next largest sector, and the most rapidly growing, is yellowtail kingfish (YTK). It is thus important to develop an understanding of the range of potential environmental impacts that the aquaculture of this species could have before the industry reaches a size at which these impacts become pervasive and unsustainable.

In a recent meta-analysis of the impacts of aquaculture on water column nutrients, Sarà (2007a) showed large effect sizes of fish aquaculture in marine waters on ammonium and nitrite, where a large effect is the mean at impact sites being >0.8 SD higher than at control sites. No effect could be detected for nitrate or phosphate. Additionally, Sarà (2007b) detected effects on chlorophyll *a* (chl *a*), particulate organic nitrogen and phosphorus, total suspended matter, and bacterial abundance and productivity. Studies on benthic impacts have covered a much greater range of variables (at least 120; Kalantzi & Karakassis 2006), and vary with factors such as fish density, farm size, food conversion ratio, water depth, current speed and sediment mud content (Giles 2008).

In the present paper, we examine the impacts of YTK farming in Fitzgerald Bay on a range of biological and chemical components of the ecosystem. Both the water column (chemistry and phytoplankton) and the benthos (sediment chemistry, infauna, epifauna and seagrass) were examined. Primarily, we used a multivariate approach, although some parameters required univariate analyses.

MATERIALS AND METHODS

Study site and farm management. Yellowtail kingfish are primarily farmed in South Australia in Fitzgerald Bay, in northern Spencer Gulf, and at Arno Bay and Boston Bay, further south. Production reached slightly over 2000 t yr^{-1} in 2007/8 (Econsearch 2009), and was substantially less when the present study was undertaken, although actual figures are not available. There are a total of five 20 ha leases in Fitzgerald Bay, which was the focus of this study, with each lease supporting 2 to 9 cages, depending on the time of year. These leases are in ~10 to 20 m water depth, although actual cages tend to be located in the deeper parts of each lease, and within 1 to 2 km of the shore. Average current speeds are on the order of 15 cm s^{-1}, which is relatively high compared to many aquaculture sites (Giles 2008), and flow is predominantly north–south under the influence of the semi-diurnal tides (J. Tanner unpubl. data). Water temperature ranges from ~12 to 28°C, and the area is exposed to the southerly winds that predominate through most of the year, although it is sheltered from the northerlies which occur more frequently in winter. Sediments are dominated by silt (30 ± 4.6 (SE)%, n = 8), with high proportions of very fine and fine sand (25 ± 2.6 % and 20.5 ± 1.9 %, respectively). Sedimentation rates at the edges of the cages were measured as 79 to 83 g m^{-2} d^{-1}, and were significantly higher than the background rates of ~65 g m^{-2} d^{-1}, which occurred as close as 30 m to the cages (Fernandes & Tanner 2008).

Typically, approximately 10 000 to 15 000 fish are stocked into nursery cages in about October. These are kept for ~2 yr, during which time they undergo 2 gradings for size, when they are transferred between cages. Harvest size is ~3 to 3.5 kg, with a fish density at harvest of ~13 kg m^{-3}. The feed conversion ratio (FCR) at the time of the study was ~3:1 (Fernandes & Tanner 2008), which is high relative to many other studies (e.g in a review of 64 studies of benthic impacts, the highest FCR for the 20 studies that reported this value was 2.3:1; Giles 2008). A typical cage is 25 m in diameter, and 6 m deep. Standard husbandry procedures include the changing of nets on the sea cages to remove fouling, and freshwater bathing of fish to remove ectoparasites. There are strict regulations in place to limit the use of chemicals, such as antibiotics and other medications, and the area in which the sea cages are kept must undergo a fallowing period to allow the benthos to recover. Typical details on stocking and feeding regimes for individual cages are given in Fernandes & Tanner (2008).

Sampling strategy. The basic sampling design was a 2-way crossed design, with the factors Distance and Location. However, different variables leant themselves to different elaborations of this design, logistical issues sometimes interfered with adhering to the design, and the data presented come from several different studies, so there are differences between variables in the experimental design. In the basic design, Distance was treated as a factor, with sampling conducted at lease sites and control sites (>1 km from any lease). In the gradient design, Distance was treated as a covariate, with sampling occurring along a transect radiating out from either a lease or a cage. The 2 leases sampled for most variables were in the north-east and south-east of Fitzgerald Bay, and so the factor Location refers to north and south. Sites were nested within the Distance by Location interaction unless otherwise specified.

Water column nutrients: Sampling was conducted on 11 November 2004 using the basic design with Distance as a factor. Within each lease, samples were taken from <10 m downstream of 3 cages (sites) which were ~100 m apart; 3 sites in each control area having a similar separation were also sampled. At each site, a

Niskin bottle was used to collect 2 water samples for nutrient analysis from 2 m below the water surface. Each sample was filtered (0.45 µm) immediately after collection and placed on ice.

Water column chl a and phytoplankton: In 2004, sampling was conducted during the same period and using the same collection methods and sites as for water column nutrients. From 16 to 18 August 2005, 2 of the same sites were sampled in each lease and control area, with samples taken from just above the sea-floor (1 m) as well as just below the surface; some sites were not sampled due to deteriorating weather. Three water samples of 1250 ml were collected for each of chl *a* and phytoplankton at each site. Chl *a* samples were immediately placed on ice, while phytoplankton samples were kept in the dark.

Infauna: Sediment cores were sampled using the gradient design in August/September 2004. Seven (sometimes 8) replicate samples were taken using a HAPS corer (KC Denmark A/S) at each of 0, 20, 50, 100 and 1000 m along transects radiating out from 2 cages (sites) in each of 2 leases. Each core had a diameter of 67 mm, and was taken to a depth of 10 cm. Cores were extruded from the barrel and preserved in Bennett's solution.

Epifauna: Epifaunal assemblages were assessed in 2 separate remote video surveys. In the first, 100 m transects were filmed radiating out in the 4 cardinal directions from the edge of each of 2 lease and 2 control sites. For each transect, a digital video camera was lowered to approximately 0.2 to 0.5 m above the sea-floor and the substrate was filmed while the boat motored slowly along the length of the transect. A GPS was used to record the location where the image first became clear, and the distance from this was monitored to ensure that each transect was 100 m long. The location of the camera relative to the sea-floor, and quality of the footage, were monitored via a live feed to a surface monitor. These transects were conducted from 21 June to 25 June 2004.

The second survey followed the first, with the exception that lease site transects radiated out from the cages (rather than from the perimeter of the leases), and there were no control site transects. Two cages in each of 3 lease sites were surveyed, with transects running roughly north and south from each cage. No transect ended <100 m from an adjacent cage. These transects were filmed on 28 November 2005.

Seagrass: As seagrasses only occur in shallow water (maximum depth ~5 m), and thus do not occur within aquaculture leases, the sampling design for seagrass was somewhat different to the other variables. Samples of *Posidonia australis* were collected by divers on 28 and 29 November 2005 at 4 to 5 m water depth, adjacent to leases and in a control area in the north of the bay >1 km from any lease. To assess bay-wide effects, samples were collected from a thrid area, 3 to 5 km north of the bay in a region not known to be directly influenced by anthropogenic nutrient inputs (i.e. lacking aquaculture activity, coastal development and riverine runoff). In each of these 3 areas, 10 replicate quadrats (25 × 25 cm) at each of 2 sites were harvested of aboveground biomass, which was then frozen prior to later analysis. The impact sites were located as close as possible to active YTK leases (within 250 m of a lease boundary).

Sediments: Sediment samples were collected in May 2005 according to the basic sampling design. Cage sites were immediately adjacent to the edge of the sampled cage. Sediments were collected by divers using 73 mm (inner diameter) PVC tubes. The overlying water in the tube was carefully discarded to minimise surface disturbance and the sediment extruded onto a clean stainless steel table. Four cores were used for the analysis of total nitrogen (N), organic carbon (OC) and total phosphorus (P). The top layer (0 to 1 cm) of each core was sliced, transferred into a pre-combusted glass jar and stored frozen (–30°C). Two cores were collected for the determination of ammonia and phosphate in porewaters. The top layer (0 to 2 cm) of each core was sliced, transferred into a centrifuge tube and stored in ice before transfer to the laboratory.

Laboratory analysis. ***Water column nutrients:*** On return to shore, samples were frozen, and then sent to the Water Studies Centre at Monash University for analysis. Each sample was analysed for total P, total N, nitrate + nitrite and ammonia ($NH_4^+ + NH_3$) using flow injection analysis on a QuikChem 8000 automated ion analyser (Lachat Instruments). Organic nitrogen was calculated by subtraction.

Water column chl a and phytoplankton: On return to shore, 1000 ml of each chl *a* sample was filtered under vacuum through a 0.7 µm glass fibre filter (MFS GF-75). Filters were then stored in liquid nitrogen until analysis (Mantoura et al. 1997). Chl *a* analysis was based on the methods developed in Golterman et al. (1978). Filters were transferred to test tubes to which 5 ml of methanol was added, and then refrigerated for 24 h to facilitate extraction. Chl *a* analysis was carried out on a Helios gamma spectrophotometer (Thermo Scientific). Phytoplankton samples were preserved with 5 ml of Lugol's solution, prior to being sent to Microalgal Services (Ormond, Victoria) for enumeration of each species present (community composition).

Infauna: Each sample was sieved on a 1 mm mesh, and then sorted to extract all infauna remaining on the sieve. Taxa were identified and enumerated to the lowest taxonomic level possible, generally family.

Taxonomic classification at the family level is often used for infauna, as they are time-consuming to identify, are poorly known in many areas (including Australia, where it is estimated that ~90% of species remain undescribed; Butler et al. 2010), and species level differences are often related to minor variation in extraneous environmental variables rather than being a result of important ecological differences (e.g. Somerfield & Clarke 1995, Bevilacqua et al. 2009).

Epifauna: Video footage was analysed to determine the identities of the species present along each transect. The abundance of each taxon was recorded for a central strip of the transect approximately 0.5 m wide. Sessile unitary organisms were recorded as number of individuals, whereas clonal species such as seagrass and algae, and substrates such as sand and rubble, were recorded as percent cover. To examine how epifaunal assemblages change with distance from the edge of the lease, transects were divided into 25 m segments, with abundance and cover recorded separately for each segment. Abundance was based on total numbers of individuals in each segment, while cover was mean percent cover recorded in each of 10 non-overlapping frames. Taxa were often only identifiable to morphological grouping or genus, rather than species. For the second survey, abundance was also measured for the 10 non-overlapping frames, rather than for the entire length of each segment.

Seagrass: The total number of leaves and shoots within each quadrat was counted. The maximum leaf length and width were also measured. Ten intact shoots were then haphazardly selected, and the longest leaf from each removed to determine epiphyte loading. These leaves were dried at 60°C for 48 h and weighed. All 10 leaves from each quadrat were then placed in 5% HCl, rinsed in freshwater, and scraped to remove all epiphytes. Leaves were then redried at 60°C for another 48 h and reweighed to determine epiphyte-free dry weight. A further 10 leaves with low epiphyte loading were randomly selected to determine carbon (C) and N content. Each leaf was scraped clean on aluminium foil which was cleaned with methanol, and placed in a glass jar that had been muffled at 450°C overnight and then weighed. Each jar was then reweighed to obtain wet leaf biomass and frozen prior to CN analysis. Samples were freeze-dried overnight and then ground to a fine powder. An aliquot (150 mg) was then analysed for %C and %N on a LECO Truspec CNS elemental analyser. The remainder of the sample was dried at 60°C for 48 h and weighed to obtain total dry weight (seagrass plus epiphytes) for each quadrat.

Sediments: For the analysis of total N, OC and total P, samples were freeze-dried, sieved to 500 µm to remove large shell fragments, and homogenized with a mortar and pestle. Total N, OC and their stable isotopes ($^{15}N/^{14}N$ and $^{13}C/^{12}C$) were analysed by CSIRO Land & Water (Adelaide) with continuous flow stable isotope ratio mass spectrometry (CF IRMS) using a Europa Scientific ANCA-SL elemental analyser coupled to a Sercon Geo 20-20 mass spectrometer. Aliquots for carbon analysis were pre-treated with 1 N hydrochloric acid to remove carbonates, rinsed with MilliQ water to remove hygroscopic salts and oven dried at 40°C using a method modified from Fernandes & Krull (2008). For P, samples were digested with aqua regia (1:3 HNO_3:HCl mixture) (Standards Australia 1997) and the extracts analysed in a Varian Vista Axial Inductively Coupled Plasma Atomic Emission Spectrometer (ICP-AES). OC concentrations were corrected for carbonate content, determined with a pressure transducer after acidification of aliquots with hydrochloric acid. Concentrations are reported as a percentage of total dry sediment. Natural isotopic abundances for C and N are reported as $\delta^{13}C$ and $\delta^{15}N$ (Peterson & Fry 1987), referenced to internationally accepted standards (C in Pee Dee Belemnite limestone and N in air).

For ammonia and phosphate in porewaters, samples were centrifuged at 1400 × *g* for 10 min, the supernatant filtered (0.45 µm) and stored frozen (–30°C). Ammonium and phosphate were determined spectrophotometrically by flow injection analysis in a QuickChem 8000 automated ion analyser.

Data analysis. All data sets were analysed using some variation of ANOVA. Multivariate data sets (water column nutrients, phytoplankton, infauna, epifauna, seagrasses and sediments) were analysed using permutational multivariate analysis of variance (PERMANOVA; Anderson 2001) with the software PERMANOVA+ (Primer-E). PERMANOVA is a multivariate ANOVA technique that does not require the assumption of multivariate normality, and that allows different measures of distance to be used, thus making it much more suitable for most ecological data than standard parametric MANOVA. Physico-chemical and seagrass variables were analysed using Euclidean distances and range-standardised data. Phytoplankton, infauna and epifauna community composition were analysed using Bray-Curtis dissimilarities and 4th-root transformed data, as is standard for community analyses, to eliminate the effects of joint absences and downweigh the influence of highly abundant species, respectively (McCune & Grace 2002). We ran 4999 permutations of the residuals under a reduced model to calculate probability values in PERMANOVA. To examine the response of individual variables (and following any PERMANOVA where it was considered useful to assess individual variables after a significant Distance effect was found), univariate ANOVAs were conducted using the statistical package PASW Statistics (v. 18.0, SPSS). In all cases, Location (north/south)

was fixed, Site was nested within Location (when included), while Distance (cage/control) was fixed in the basic design, or a covariate (ANCOVA) in the gradient design.

Water column nutrients: Data analysis proceeded as described in the previous paragraph. Distance was treated as a fixed factor.

Water column chl a and phytoplankton: The influence of proximity to aquaculture cages on chl *a* was determined using univariate ANOVA. As only surface samples were collected in 2004, and poor weather meant that only surface samples were collected from the northern farm and reference sites in 2005, a full analysis incorporating all factors could not be conducted. Instead, the data were analysed in 2 subsets. The first subset considered only the surface samples, and proceeded as above with Month as an additional random factor crossed with Distance, Location and Site. The second analysis involved only the 2005 samples with Depth as an additional fixed factor crossed with Distance, Location and Site. Phytoplankton composition was analysed in a similar fashion using PERMANOVA.

Infauna: Infauna community composition was analysed using PERMANOVA as described in the first paragraph of 'Data analysis' for the gradient design. The responses of total infaunal abundance, and taxonomic richness (number of taxa), were analysed using ANCOVA. Both variables were natural log-transformed prior to analysis to meet the assumption of normality, and both met the assumption of homogeneity of variances after transformation (based on Levene's test). To test the assumption of equality of slopes for ANCOVA, initial analyses included terms for the interaction between Distance and Lease and Distance and Cage. If these terms were non-significant ($p > 0.05$), then the analysis proceeded.

Epifauna: PERMANOVA was used as described in the first paragraph of 'Data analysis', but with an additional fixed factor of position along each transect. The 4 directions were treated as replicates in the first analysis, but Direction was included as a fixed factor in the second. As 3 leases were included in the second survey, Lease was included as a random factor in place of Location.

Seagrass: PERMANOVA was used, with 3 levels of Distance: adjacent to leases, within bay controls, and outside bay controls. All data were averaged at the quadrat level.

Sediments: As measurements were made on different samples, separate PERMANOVAs were conducted for sediment composition (%C, %N, %P, $\delta^{13}C$ and $\delta^{15}N$) and porewater nutrients as described in the first paragraph of 'Data analysis'. Distance was treated as a fixed factor.

Table 1. Variation in individual water column N concentrations (µM) at Fitzgerald Bay, South Australia, as a function of location (north lease, south lease) and proximity to finfish aquaculture cages (control versus lease). Data are means ± SE

Site	Organic nitrogen	Ammonia ($NH_4^+ + NH_3$)	Nitrate + nitrite
North lease	6.30 ± 0.18	2.36 ± 0.26	0.27 ± 0.011
North control	6.42 ± 0.26	1.27 ± 0.12	0.29 ± 0
South lease	7.33 ± 0.33	1.31 ± 0.30	0.29 ± 0
South control	6.79 ± 0.95	0.76 ± 0.08	0.31 ± 0.015

RESULTS

Water column nutrients

Water column nutrients varied significantly with Distance ($F_{1,8} = 7.14$, $p = 0.007$), and Location ($F_{1,8} = 7.22$, $p = 0.008$) but not with their interaction ($F_{1,8} = 0.93$, $p = 0.44$) or Site ($F_{8,12} = 1.09$, $p = 0.40$). Univariate analyses showed significant variation in ammonia levels associated with both Distance ($F_{1,8} = 15.5$, $p = 0.004$) and Location ($F_{1,8} = 14.2$, $p = 0.005$). Ammonia levels in the north of the bay were 75% higher than in the south, and adjacent to cages they were 81% higher than at control sites (Table 1). The measured concentrations of nitrate + nitrite varied between Sites ($F_{8,12} = 5.0$, $p = 0.007$), but not with Distance or Location ($F_{1,8} = 1.8$, $p = 0.22$ for both factors). Total organic N did not vary with any factor, and total P was below detectable levels (0.01 mg P l^{-1}) in all samples.

Water column chl *a* and phytoplankton

With the exception of Month ($F_{1,1} = 2025$, $p = 0.014$) for the surface samples, and Location ($F_{1,4.5} = 14.4$, $p = 0.016$) for the 2005 samples, chl *a* concentrations did not vary as a function of any of the factors included in the experimental design, either for the surface samples over both years or for the 2005 samples. Chl *a* levels were 30% higher in November 2004 (0.65 ± 0.03 (SE) µg l^{-1}) than in August 2005 (0.49 ± 0.046 µg l^{-1}).

Phytoplankton composition in 2005 was not affected by Distance ($F_{1,8} = 0.90$, $p = 0.53$). There was significant small-scale variability between Sites ($F_{4,28} = 1.75$, $p = 0.0004$). A similar result was found for the surface samples (Distance: $F_{1,9.4} = 0.42$, $p = 0.97$; Site: $F_{8,40} = 1.45$, $p = 0.005$), which also differed over time ($F_{1,4} = 42.8$, $p < 0.001$). This difference is due to a large suite of species being much more abundant in November 2004 than in August 2005, and only a few species being less abundant, as would be expected for a spring versus winter comparison.

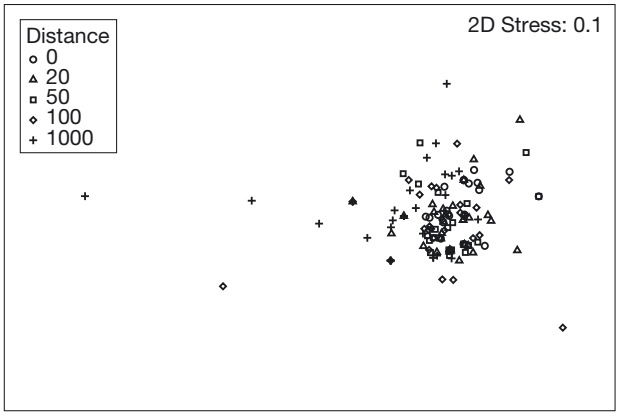

Fig. 1. Multidimensional scaling ordination plot for infauna showing extensive overlap between different distances (m) from the aquaculture cages

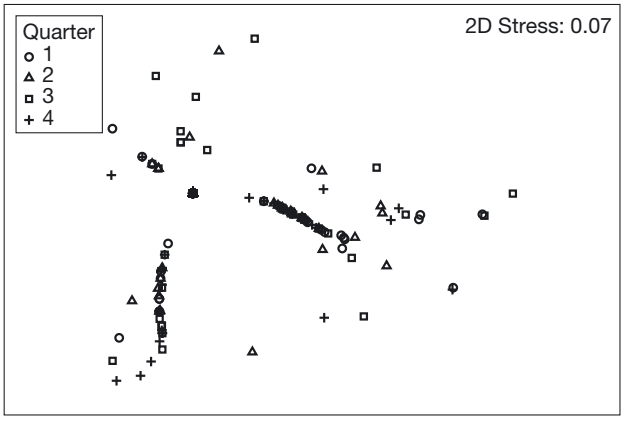

Fig. 2. Multidimensional scaling ordination plot for epifauna showing extensive overlap between different distances (Quarter 1 = adjacent to cages, Quarter 4 = 75 to 100 m from cages)

Infauna

There was a highly significant effect of Distance ($F_{1,138}$ = 6.5, p < 0.0002), Location ($F_{1,2}$ = 5.5, p = 0.0012), and their interaction ($F_{1,138}$ = 3.55, p = 0.0008), on infaunal composition, but no differences between Sites ($F_{2,138}$ = 1.46, p = 0.097). Pairwise tests indicated that the 20, 50 and 100 m samples differed from the 1000 m samples at the north lease (p < 0.03) but not from the 0 m samples, or at the south lease (Fig. 1). In addition, the square root of the component of variation for Distance (11.1) was low relative to that for Location (17.3) and especially the residual (57.3), indicating that Distance only accounted for a small proportion of the variation in infaunal community composition. Neither total abundance nor taxonomic richness showed any response to Distance ($F_{1,141}$ = 0.90, p = 0.34; $F_{1,141}$ = 0.28, p = 0.60, respectively).

Only 4 taxa of the 59 found in the samples occurred with a total abundance of 20 or more (n = 140 samples). Three of these were families of polychaetes: Capitellidae (63 ind.), Cirratulidae (206 ind.) and Spionidae (65 ind.). The fourth was a family of tanaid crustaceans: Apseudidae (62 ind.). The next most abundant taxon was the polychaete family Maldanidae, which was represented by only 17 individuals.

Epifauna

For the video transects radiating out from lease edges, there was a significant Site effect ($F_{2,48}$ = 11.8, p < 0.0002), but no effect of Distance ($F_{1,2}$ = 0.91, p = 0.66), or any other factor. For the video transects radiating out from individual cages, there was a complex pattern of small-scale variation in epifaunal composition, indicated by numerous higher-order interactions in the PERMANOVA. While there do appear to be effects of Distance (i.e. quarter) on epifaunal composition, these effects change depending on what Direction the transect took from the cage, and what cage it radiated out from ($F_{9,290}$ = 2.6, p < 0.0002; Fig. 2). Breaking down the analysis to the separate Locations showed that these differences were only present at the north lease. There is thus no clear indication that YTK farming influences epifaunal assemblages.

Seagrass

While the PERMANOVA indicated a highly significant effect of Site ($F_{3,54}$ = 5.58, p < 0.0002), there was no overall effect of Distance on seagrasses ($F_{2,3}$ = 1.75, p = 0.21). The number of leaves per quadrat varied from 41.5 at one of the lease sites, to 113 at one of the Fitzgerald Bay control sites, with a large amount of variation between lease sites. The number of shoots showed a very similar pattern, ranging from 13 to 43. Maximum leaf length was 600 to 650 mm for most sites, with the exception of one of the Fitzgerald Bay control sites, where it was 848 mm, and one of the external control sites, where it was only 511 mm. Leaf width was more variable, ranging between 6.9 mm and 12.6 mm, with the external control sites having the widest leaves. Total aboveground biomass ranged from 77 to 97 g dry weight m^{-2} at the external control sites, but was only 74 % of this at the within-bay control sites, and 44 % at the sites adjacent to leases. The highest epiphyte biomass (3.9 mg cm^{-2}) occurred at one of the external control sites, while the lowest was at one

Table 2. Variation in sediment chemical composition, stable isotope ratios and sediment porewater nutrient concentrations at Fitzgerald Bay, South Australia, as a function of location (north lease, south lease) and proximity (control versus lease) to finfish aquaculture cages. Data are means ± SE

Site	Organic carbon (%)	Nitrogen (%)	Phosphorus (%)	$\delta^{13}C$ (ppm)	$\delta^{15}N$ (ppm)	Porewater ammonia (µM N)	Porewater phosphate (µM P)
North lease	1.41 ± 0.12	0.2 ± 0.004	0.060 ± 0.095	−15.83 ± −0.03	3.55 ± 0.10	42 ± 12	32 ± 11
North control	1.32 ± 0.19	0.185 ± 0.005	0.035 ± 0.040	−16.53 ± −0.08	3.35 ± 0.06	19 ± 4	1.0 ± 0.6
South lease	1.04 ± 0.18	0.125 ± 0.009	0.034 ± 0.036	−17.23 ± −0.06	3.28 ± 0.05	47 ± 25	4.1 ± 3.9
South control	0.60 ± 0.16	0.105 ± 0.003	0.028 ± 0.028	−17.15 ± −0.06	3.75 ± 0.42	32 ± 17	2.3 ± 0.6

of the lease sites (1.8 mg cm^{-2}). The within-bay control sites tended to have low epiphyte biomass, while the external control sites tended to have high epiphyte biomass. The average N content was 1.02%, with the 2 lease sites having the highest and lowest values. C content averaged 33.1%, with the lease sites being intermediate between the within-bay controls and outside bay controls.

Sediments

The chemical composition of the sediments varied with both Distance ($F_{1,12}$ = 5.24, p = 0.011), and Location ($F_{1,12}$ = 22.5, p = 0.0002). OC, N and P all tended to be higher adjacent to cages than at control sites (Table 2). While at both locations the OC concentrations were higher at the farm site than the control site ($F_{1,12}$ = 9.4, p = 0.01), this difference was accentuated at the southern lease, where C levels were lower. At the southern lease, OC at the farm site was 73% higher than that at the control, versus 7% for the northern lease. N content also varied with Location ($F_{1,12}$ = 192, p < 0.001), and was 12% higher at farm sites than at control sites ($F_{1,1}$ = 9.8, p = 0.009). There were no differences in P content with either factor (p > 0.09). Although differences in carbon isotope ratios were all highly significant, they were only very small (maximum difference of 0.7 between control and cage samples at the north lease); there were no differences in the nitrogen isotope ratios.

Porewater nutrient concentrations also varied with Distance ($F_{1,4}$ = 4.34, p = 0.049), but not Location ($F_{1,4}$ = 2.21, p = 0.19; Table 2). Porewater P concentrations varied with the interaction between Distance and Location ($F_{1,4}$ = 36, p = 0.004; Table 2). Concentrations were particularly high at the northern farm site, and in both cases the farm sites had higher concentrations than the control sites. These increases were 3077 and 80% for northern and southern farms, respectively. However, for ammonia, there were no statistically significant differences due to any of the factors tested.

DISCUSSION

The clearest responses to YTK aquaculture documented here were in some of the chemical variables, both in the water column and the sediments. There were also highly significant effects of Distance from cages on infaunal and epifaunal assemblages; however, the patterns in these groups were obscured by high levels of small spatial-scale variation seemingly unrelated to aquaculture, and it is difficult to determine if these Distance effects are an aquaculture effect per se. Indeed, most of the variables studied varied between Locations (north/south), despite these being only 3 to 4 km apart and not varying greatly in environmental variables such as depth and wind/wave exposure. The interpretation of these effects is made more difficult by the lack of data from before the commencement of aquaculture, a problem which is common to many similar studies.

Water column impacts

The significant multivariate differences in water column nutrients with Distance were driven primarily by ammonia ($NH_4^+ + NH_3$), the only individual nutrient to vary, with concentrations being 81% higher next to cages than at control sites 1 km away from any cages. A preliminary model of N loads from YTK aquaculture indicates that 60% of N in feed inputs is lost as soluble excretion products (Fernandes & Tanner 2008), which in teleost fish are delivered mostly as ammonia (Forster & Goldstein 1969, Gowen & Bradbury 1987), and as such it is not surprising that concentrations of this nutrient are elevated next to cages. The samples analysed here were collected in late spring (November 2004), when it is likely that phytoplankton would be at or near their peak in productivity, and thus absorbing a maximal amount of ammonia; hence, this pattern is unlikely to be due to seasonally low phytoplankton abundance. There was no evidence of an increase in phytoplankton abundance, chl *a* levels, seagrass epi-

phyte biomass, or a change in seagrass nutrient composition in close proximity to cages, however, suggesting that while this ammonia may be rapidly utilised, it is also rapidly dispersed and is not retained in plant biomass close to the cages.

The lack of response of most water column nutrients to the presence of YTK cages parallels the results of a number of other studies. For example, a number of studies have looked for effects of tuna farming off Port Lincoln, South Australia, on water quality, and failed to detect any (Clarke et al. 2000). It is generally considered that this lack of an effect is due to high uptake rates by phytoplankton and high water movement (Doglioli et al. 2004). However, a meta-analysis of over 50 studies showed that aquaculture generally had a large effect on water column nutrients, with ammonium being most affected (Sarà 2007a). As we found here, however, aquaculture often does not appear to lead to increases in phytoplankton abundance (Gowen & Bradbury 1987, Merceron et al. 2002), probably due to high rates of transfer up the food web (Pitta et al. 2009).

Benthic impacts

We found clear differences in organic C and N content, and porewater P, with samples collected adjacent to cages having elevated levels compared to those collected at control sites at least 1 km away from any cage. Stable isotope ratios, however, did not show any clear patterns. In contrast, previous studies have found sediment isotope ratios to be useful indicators of environmental impacts of aquaculture (Yamada et al. 2003), although in some cases only N was affected (Sarà et al. 2004). Sediment OC loadings have also been used successfully at other locations to detect benthic impacts of aquaculture (Carroll et al. 2003), although they were not useful in a study of fallowing of salmon farming sites in Tasmania (Macleod et al. 2004). In the study by Carroll et al. (2003), however, the infaunal assemblages at some sites indicated a severe disturbance at a site 50 m from the edge of a cage, while sediment chemistry indicated that the site was normal. Thus, at least in that study, sediment chemistry was a much less sensitive indicator of disturbance than was infaunal composition. Other physico-chemical parameters that have been shown to at least sometimes respond to the presence of aquaculture are sediment redox and sulphide levels (Brooks et al. 2003, Macleod et al. 2004, Edgar et al. 2005). Again, however, the study by Brooks et al. (2003) showed recovery of a suite of sediment physico-chemical parameters almost as soon as harvesting was completed, whereas infauna took a further 6 mo to recover.

While there is a statistically significant effect of distance from cages on the infaunal composition, the responses of individual taxa are not always consistent with the hypothesis that this is due to organic enrichment around cages (Tanner & Bryars 2007). The Capitellidae are well known to be indicators of organic enrichment (Grassle & Grassle 1974, Pearson & Rosenberg 1978), and while they did display increased abundance in the immediate vicinity of cages (0 to 20 m), they also showed high abundance at the southern control (1000 m) site. Even at the 0 m sites, capitellid abundance was below 365 m^{-2}, which is well below typical abundances of 10s to 100s of thousands m^{-2} seen in areas with high levels of organic enrichment (e.g. Brooks et al. 2003, Pereira et al. 2004). Cirratulidae are also known indicators of organic enrichment (Glasby 2000), and their pattern of abundance at the north lease suggests organic enrichment, although that at the south lease does not. While mean abundances reach higher peaks than do the Capitellidae, again the peak abundance of 1378 m^{-2} is relatively low by worldwide standards. Spionidae have been recorded in densities up to 5670 m^{-2} in organically enriched areas (Yokoyama 2002), but in this study did not show any clear trends with distance from cages, and only reached maximum abundances of 325 m^{-2}.

The low densities of taxa generally considered to be good indicators of organic enrichment suggest that while there is some level of organic enrichment in these sediments, it is very low. This conclusion is supported by the lack of effects on infaunal taxon richness and total abundance. In fact, total abundances are very low in comparison to what has been found in the southern Spencer Gulf using the same methods, where Loo & Drabsch (2005) found a mean abundance at control sites of 5644 m^{-2}. This compares to a mean of 1049 m^{-2} at the 1000 m sites in Fitzgerald Bay, lower than the abundances for individual taxa in many other studies (see previous paragraph). In organically enriched areas, there is typically a low richness, but high abundance, with a few opportunistic taxa dominating the assemblage (Pearson & Rosenberg 1978, Pereira et al. 2004).

In the current study, the results for both infauna and epifauna were equivocal. While there was an apparent effect of distance from cages on both groups, the patterns found did not clearly indicate an effect of organic enrichment. In the case of infauna, it is possible that there are patterns in the level of organic enrichment of the sediments, or other parameters such as grain size which are unrelated to aquaculture and that these could have confounded our ability to detect clear gradient effects away from cages. Similar patterns have been found in the tuna farming zone off Port Lincoln, where there are distinct regions of sediments with high and low organic loadings (Fernandes et al. 2006). In

the case of epifauna, there appears to be significant small-scale spatial variability that is not associated with aquaculture. However, it also appears that proximity to aquaculture sites increases this variability, a pattern that is fairly common in assemblages exposed to low to moderate levels of disturbance (see Dernie et al. 2003, Wear & Tanner, 2007). This higher level of variability makes it more difficult to detect temporal trends (Thrush et al. 2001) and differences in mean abundance/composition between sites. High levels of temporal variability have been shown to precede population extinction in some systems (e.g. Drake & Griffen 2010), and it is interesting to speculate as to whether high levels of small-scale spatial variability might provide a similar early warning of potential ecosystem level changes.

No effects of aquaculture on seagrass structure were detected. Previous studies across a number of locations have shown that finfish aquaculture generally causes a decrease in seagrass shoot density, biomass and primary productivity (Holmer et al. 2008, Apostolaki et al. 2009). Most studies that have investigated epiphytes have found increased loading on seagrass leaves closer to the finfish farms (Delgado et al. 1999, Cancemi et al. 2003). However, Ruiz et al. (2001) found that epiphytic cover did not decrease in close proximity to farms, and Bryars (2003) reported that seagrasses in the vicinity of 2 finfish cages in South Australia appeared 'healthy' in respect to epiphyte load, although no quantitative data were collected. In a parallel study, we (Tanner et al. 2006) showed that modelled patterns of C deposition around pens in Fitzgerald Bay are highest north and south of pens, with deposition dropping off very rapidly in an east-west direction. Given that seagrasses were collected several hundred metres west of the lease sites, it is very unlikely that they would experience increased rates of sedimentation. This location also means that they are not affected by shading from the pens, or other physical impacts.

In a concomitant study examining demersal fish assemblages in and around Fitzgerald Bay, Williams (2004) found no differences between sites adjacent to cages and control sites >1 km from cages. There were also no differences detected between Fitzgerald Bay and nearby locations (~5 to 20 km north and south). This lack of response contrasts with a number of other studies, which have shown marked aggregation of demersal fish around aquaculture cages (e.g. Tuya et al. 2006, Dempster et al. 2009), and may indicate that particulate wastes are insufficient to promote aggregation, that wastes are being consumed by pelagic fish before they reach the seafloor, or that demersal fish are highly mobile and only aggregate at cages during and shortly after feeding, which occurs once a day. Attempts to survey pelagic fish assemblages using baited remote video, gill nets and divers were unsuccessful, so no analyses of this group could be undertaken.

CONCLUSIONS

The clearest impacts of YTK aquaculture were seen in some of the physico-chemical variables, with both water and sediment chemistry being affected. Responses in the benthic fauna/flora were also evident, although somewhat ambiguous due to substantial small-scale variation seemingly unrelated to aquaculture, and the lack of data from prior to the commencement of aquaculture. Any impacts of aquaculture on the infauna are relatively small compared to those that have been documented elsewhere, with key polychaete families that normally show strong responses to organic enrichment having relatively low abundances. For infauna, the multivariate analysis detected patterns that were not observed in some of the univariate indices that are commonly employed.

The lack of unequivocal responses in the biotic data sets probably reflects the low production levels in Fitzgerald Bay, along with its relatively open nature, which exposes it to moderate levels of wave activity and currents. However, with production increasing since this study was undertaken, the physico-chemical responses provide a timely early warning of potentially more severe impacts in the future.

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