Environment Waikato Technical Report 2007/50

Bayesian Network Analysis Exploring the Benthic Carrying Capacity for Finfish Farming Within the Firth of Thames

www.ew.govt.nz
ISSN 1172-4005 (Print)
ISSN 1172-9284 (Online)

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December 2007

Document #: 1264879

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Reviewed by:

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Executive Summary

Environment Waikato is currently scoping a plan change to allow for the diversification of aquaculture within existing aquaculture management areas in the region that will potentially allow for the cultivation of species other than mussels, including finfish. Currently no information about environmental effects of finfish aquaculture is available for this region. This report presents an exploration of the benthic carrying capacity of the Firth of Thames for finfish farming to provide some background information for the aquaculture diversification plan change.

An estimate of the scale of expected benthic effects associated with fish farming in the Firth of Thames Wilson Bay Marine Farming Zone was obtained by carrying out a Bayesian network analysis supported by literature studies. In addition, estimates of the likely spatial extent of benthic effects were provided based on previous published studies as a first step to predicting the footprint of finfish cages.

The Bayesian network used in this study is a modified and re-parameterised version of an existing Bayesian network that was developed to quantitatively assess the relationships between benthic fish farm impact parameters and site and farm characteristics based on data published in peer-reviewed international journals from finfish farms located in temperate zones. The scale of benthic effects expected from establishing finfish farms in the Wilson Bay Marine Farming Zone was examined using selected case studies that represent realistic farming scenarios for this area. Case studies were created by examining different combinations of fish stocking density and water depth, which are expected to influence the scale of benthic impact. The focus of this report was to highlight the differences in benthic impacts predicted from the case studies and the trends generated from varying the input parameters, rather than making predictions of absolute variable values.

The Bayesian network analysis suggested that of the examined input parameters the free water depth below fish cages has the largest effect on the severity of benthic impacts and it is recommended that the minimum free water depth below cages should be 10 m. It is general practice in finfish aquaculture to use cages between 10 and 15 m depth and allow for a similar depth of water below cages to promote the dispersal of faeces and uneaten food, consequently the most suitable areas for the installation of fish cages are those with water depths of 20 m or more. Changes in stocking density only resulted in small changes in the probability distributions of most variables.

Selected literature studies were reviewed to examine the usefulness of monitoring parameters for the assessment of benthic impacts in the Firth of Thames that were not included in the Bayesian network. Video surveys and sediment trap deployments are not recommended following problems experienced during earlier work caused by unfavourable environmental conditions. The examination of opportunistic macrofauna species was generally accepted as a good indicator of benthic impact; however, it was suggested that measurements of biogeochemical parameters may reveal earlier signs of impact and allow remedial measures to be taken if necessary to prevent severe impacts.

It is recommended that pre-impact studies should be carried out in locations chosen for fish farming to gain an understanding of these processes prior to the additional organic enrichment. This would allow changes in biogeochemical processes to be identified and limits of acceptable sediment modification to be chosen based on sound data. Additional parameters considered potentially useful for the detection of severity and spatial extent of benthic impacts once fish farms are operating are trace metals and stable isotopes.

Estimates of the spatial extent of expected benthic impacts were derived from a review of peer-reviewed literature and monitoring data from New Zealand fish farms and it was concluded that 100 m was a cautious estimate. Since the largest change of most examined parameters took place within about 50 m of the farm and the gap between farm blocks in Area A is 75 m a 50 m buffer zone between the outermost cages inside a farm block and the perimeter of the block was considered an adequate estimate of the buffer zone for initial applications before measurements are available to make realistic assessments of spatial effects. Especially if cages larger than those examined in this study (>15 m diameter) are to be installed in the Firth of Thames, it is strongly recommended that benthic impacts should be measured at high spatial and temporal resolution until sufficient information on their severity and spatial extent has been gathered to make sound recommendations on minimum buffer zones for farm blocks.

To enable a reliable detection of farm footprints, it is also recommended that the natural variability of parameters used for future monitoring is measured prior to any farming activity. This will enable the identification of changes caused by the farms and minimise the problem of separating natural from farm induced changes observed in the farm area.

1. Introduction

1.1 Background

Environment Waikato is currently scoping a plan change to allow for the diversification of aquaculture within existing aquaculture management areas in the Region. This plan change will potentially allow for the cultivation of species other than mussels, including finfish. The biggest aquaculture management area in the Region is the Wilson Bay Marine Farming Zone located in the Firth of Thames. Currently, Area A of the Wilson Bay Marine Farming Zone is consented for 470 ha of mussel longlines, and Area B of the Zone, once developed, will comprise an additional 520 ha. In addition to this, 220 ha of older farms exist within Wilson Bay.

Due to the current exclusive cultivation of mussels in the Firth of Thames, no information about environmental effects of finfish aquaculture is available for this region. To provide some background information for the aquaculture diversification plan change, this report presents an exploration of the benthic carrying capacity of the Firth of Thames for finfish farming. An estimate of the scale of expected benthic effects associated with fish farming in the Firth of Thames Wilson Bay Marine Farming Zone was obtained by carrying out a Bayesian network analysis supported by literature studies.

Scientific studies examining environmental impacts of fish farms in New Zealand and overseas have produced different conclusions concerning the severity of benthic impacts (e.g., Brown et al. 1987; Kaspar et al. 1988; Karakassis et al. 2000). These differences illustrate the complexity of processes and interactions playing a part in benthic fish farm effects but they also illustrate the uncertainty in the data originating from discrepancies in sampling and analytical protocols as well as the different temporal and spatial scales covered in different studies (Kalantzi and Karakassis 2006).

This report aims to estimate of the scale of expected benthic effects associated with fish farming in the Firth of Thames Wilson Bay Marine Farming Zone based on benthic effects observed at finfish farms in similar temperate environments. It is well known that the severity and spatial extent of benthic effects is a function of hydrographic site conditions as well as farm management and intensity (Beveridge 2004). Thus, to derive information on potential effects in the Firth of Thames from published data it is necessary to examine relationships among benthic impacts and farm and site characterising parameters. Previous attempts to examine such relationships had only limited success due to the inadequacy of using the traditional regression approach for analysing uncertain data.

An alternative method for the examination of such data is a Bayesian network analysis. Bayesian networks have been successfully used for environmental impact assessments where variables and processes may emerge at a variety of spatial, temporal and functional scales (Borsuk et al. 2004; Varis et al. 1994) and are useful for representing uncertain, ambiguous or incomplete knowledge (Sadoddin et al. 2005). NIWA has previously carried out a Bayesian network analysis to quantitatively assess the relationships between benthic fish farm impact parameters and site and farm characteristics (Giles in press). One application of the Bayesian network proposed in Giles (in press) was the examination of how specific sites might respond to fish farm impacts, which can provide decision-making support for the selection of new fish farm sites. The existing Bayesian network is based on data published in peer-reviewed international journals from a number of finfish farms located in temperate zones. Therefore, it cannot predict exact environmental impacts but it can demonstrate likely impacts and trends based on historical data.

To optimise the results of the Bayesian network application to the Firth of Thames, this report presents a re-parameterised version of the Bayesian network including New Zealand data obtained from monitoring reports of New Zealand fish farms and other grey literature. In addition, the structure of the Bayesian network was slightly modified to better fulfil the purpose of this report. For the development of the existing Bayesian network only frequently measured parameters were used. To allow an assessment of parameters that are not commonly used but which may be relevant in detecting benthic impacts in the Firth of Thames, this report also examines individual published studies of fish farm impacts that have been carried out in environments with similar hydrographic conditions.

1.2 Brief overview of benthic effects of finfish farming

Fish are typically cultivated in cages or pens in coastal areas and impact the marine environment primarily through the accumulation of waste products, such as faeces and uneaten food, on the sediment. As a consequence, fish farming can cause changes in sediment chemistry and community structure (e.g., Karakassis et al. 2000). The extent and severity of benthic impacts depends on site characteristics such as current speed and water depth. High-energy environments with strong water flow and deep sites generally have a higher capacity for fish farm activities, since waste products are dispersed and their impacts subsequently diluted (e.g., Panchang et al. 1997; Sarà et al. 2006). In addition, resuspension periodically re-exposes sediments and waste products to oxygen, which may enhance decomposition and promote oxic or suboxic, rather than anoxic conditions, in the surface sediments (Burdige 2006). A detailed review of environmental effects of marine finfish aquaculture has been carried out recently by the Cawthron Institute (Forrest et al. 2007) and further information can also be found in Pearson and Black (2001) and Hargrave (2005).

The existing Bayesian network was developed with the main aim being to quantitatively assess the relationships between benthic finfish farm impact parameters and site and farm characteristics. It identified porewater sulphide, acid volatile sulphide (AVS-S), water content, redox potential, sediment oxygen consumption, sediment ammonium (NH_4^+) release and macrofauna diversity as the parameters most sensitive to changes in site and farm characteristics. These parameters were consequently proposed as potentially useful parameters in examining how specific sites might respond to fish farm impacts and in supporting decision-making for the selection of new fish farm sites. However, it was stressed that methodological limitations have to be taken into consideration before characterising them as reliable monitoring parameters for specific applications.

2. Scope and stipulations of this report

The application of the Bayesian network in this report and the examination of individual published studies from similar hydrographic environments aim to yield information on the scale of benthic impacts that can be expected from the introduction of finfish farming in the Wilson Bay Marine Farming Zone (Figure 1). This report explores the benthic carrying capacity for finfish farming within the Wilson Bay Marine Farming Zone using selected case studies that are intended to represent realistic farming scenarios for this area. Case studies were created by examining different combinations of parameters. The selected parameters (fish stocking density and water depth) are expected to influence the scale of benthic impact and values were chosen to represent the range of probable values for fish farming operations in the Wilson Bay Marine Farming Zone. In addition to these parameters, the Bayesian network requires the site specific input parameters sediment mud content and current speed. Values of sediment mud content were available for Area A from previous work commissioned by Environment Waikato (Vopel et al. 2007) and current data were extracted from the Firth of Thames hydrodynamic model (Black et al. 2000).

Figure 1. Location of Wilson Bay Marine Farming Zone Areas A and B in the Firth of Thames.

2.1 Finfish species

The New Zealand marine finfish industry is dominated by sea-cage farming of King salmon (*Oncorhynchus tshawytscha*) in the Marlborough Sounds, Akaroa Harbour and Stewart Island (Forrest et al. 2007). Salmon prefer colder temperatures since high water temperatures can cause stress and make individuals more vulnerable to disease (Jeffs 2003). In an assessment of the feasibility of salmon farming in Northland Jeffs (2003) concluded that summer water temperatures in this region are too high for this species. A feasibility study of salmon farming in the Firth of Thames is beyond the scope of this report but because of the potentially unfavourable summer temperatures salmon is not considered the most likely candidate for cultivation in the Firth of Thames.

Over the last few years the main focus of NIWA's aquaculture research has been to assess the aquaculture potential of New Zealand species and to identify key species that were deemed to have particularly strong aquaculture and economic potential (Poortenaar et al. 2003). Through this research, as well as through analyses carried out by the New Zealand aquaculture industry, the yellowtail kingfish (*Seriola lalandi lalandi*) has been identified as being biologically suitable to New Zealand aquaculture conditions and as having strong economic and marketing prospects (Poortenaar et al. 2003, New Zealand Aquaculture Council 2006). Yellowtail kingfish have been commercially cultured in South Australia since 2001 but the largest aquaculture is located in Japan. In New Zealand a trial yellowtail kingfish farm has been established in the Marlborough Sounds, and Moana Pacific Fisheries Ltd have successfully reared small numbers of hatchery-reared yellowtail kingfish in seacages located in the entrance to Bon Accord Harbour, Kawau Island, since 2000 (Poortenaar et al. 2003). Yellowtail kingfish are typically found in temperatures between 15-24 °C (Penney 2000). NIWA has successfully cultured kingfish in temperatures between 12-22 °C, albeit with reduced appetite and growth below 14 °C (Poortenaar et al. 2003). Water temperature in the Firth of Thames ranges from about 13 to 22 °C (Broekhuizen et al. 2002). While it is beyond the scope of this report to provide a detailed assessment of the viability of yellowtail kingfish farming in the Firth of Thames, this species appears to be a likely candidate for the cultivation of finfish in the Firth of Thames. Consequently, the case studies examined in this report (see 2.3) reflect farming practices and requirements of yellowtail kingfish. Despite this focus on yellowtail kingfish, it is important to note that benthic effects of finfish farming are likely to be similar for most of the species that may be farmed in the future (Forrest et al. 2007). Therefore, the findings of this report are not restricted to the cultivation of yellowtail kingfish.

2.2 Wilson Bay hydrographic conditions

2.2.1 Water depth

The Firth of Thames is a shallow estuary (<30 m) and water depth measurements at various sites within Area A (unpublished data) at different occasions have revealed a depth range from about 10 to 27 m (Figure 2). No recorded water depth measurements are available for Area B but charts and observations (K. Vopel pers. comm.) indicate that the depth range of Area B is similar to that of Area A.

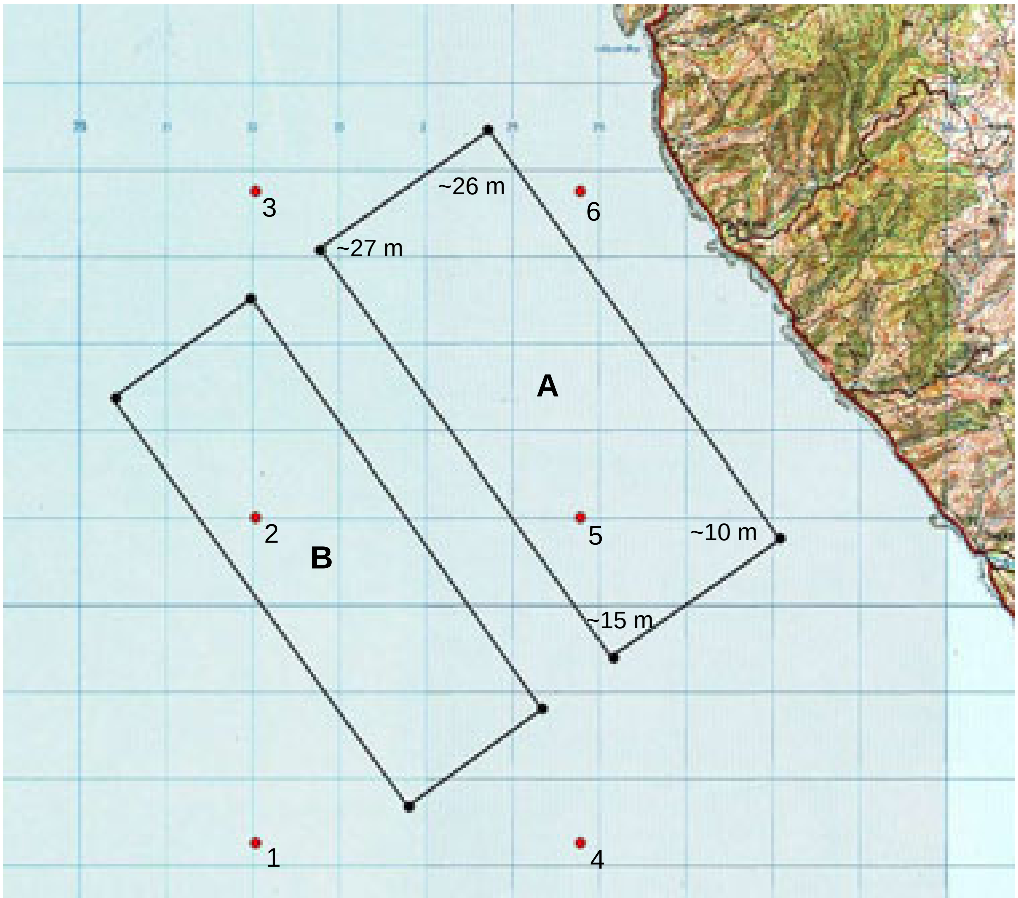


Figure 2. Approximate water depth at selected sites in Wilson Bay Marine Farming Area A. Red circles show locations where current speeds have been extracted from the hydrodynamic model.

2.2.2 Current Speed

Depth averaged current speeds were extracted for selected sites in and around the Wilson Bay Marine Farming Area (Figure 2) from the hydrodynamic model by Black et al. (2000). Simulations were made for calm conditions (no wind) and wind speeds of 15 m s^{-1} from four directions (NE, SE, SW and NW).

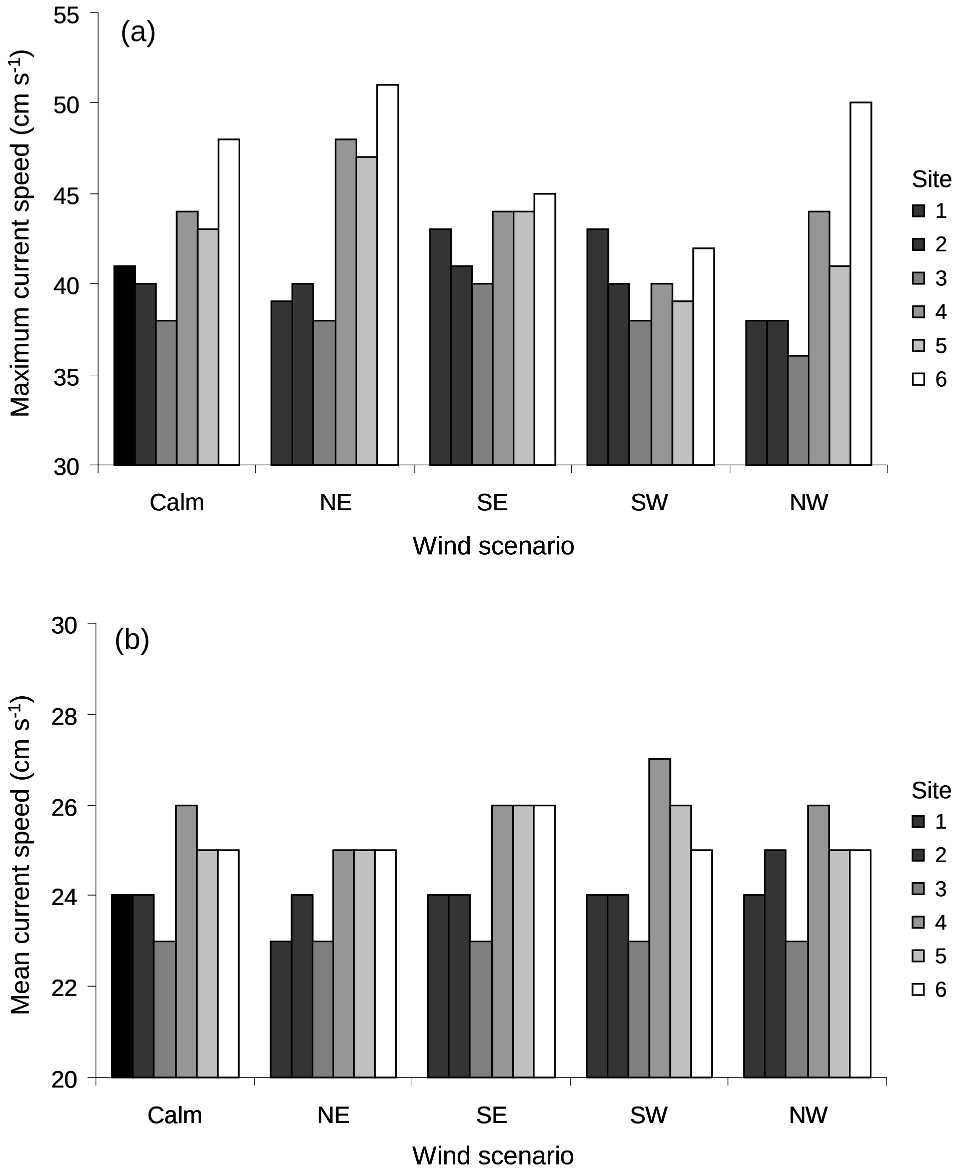


Figure 3. Predicted depth averaged (a) maximum and (b) mean current velocities at selected sites in and around the Wilson Bay Marine Farming Zone. Site locations are shown in Figure. 2.

Under most conditions current velocities were slightly higher at the sites closer to the shoreline (Figure 3). The consistently higher velocities under calm conditions indicate stronger tidal currents near the shore and the headland north of the farming area further restricts the water flow. At the three Western sites (1 to 3) maximum current velocity ranged from 36 to 43 cm s^{-1} and mean current velocity from 23 to 25 cm s^{-1}. At the sites closer to the coast (4 to 6) maximum current velocity ranged from 39 to 51 cm s^{-1} and predicted mean current velocity from 25 to 27 cm s^{-1}. The overall means for all sites and scenarios were 42.1 ± 3.8 cm s^{-1} and 24.6 ± 1.1 cm s^{-1} for maximum and mean current velocity, respectively.

2.3 Case studies

This report explores the benthic carrying capacity for finfish farming within the Wilson Bay Marine Farming Zone using a Bayesian network analysis of selected case studies. The case studies are intended to represent realistic scenarios spanning the range of feasible yellowtail kingfish culture options in the Wilson Bay Marine Farming Zone. Cases are defined based on environmental parameters and yellowtail kingfish farming practices employed in Australia as well as those recommended in kingfish research studies. In Japan, the yellowtail kingfish aquaculture industry has experienced significant problems with diseases and parasites and it has been suggested that these problems can be attributed to farming practices, such as high stocking densities (Poortenaar et al. 2003). Consequently, farming practices employed in Japan are not used for the selection of case studies in this report.

It is general practice in finfish aquaculture to use cages between 10 and 15 m depth and allow for a similar depth of water below cages to promote the dispersal of faeces and uneaten food (Poortenaar et al. 2003). This indicates that in most parts of the Wilson Bay Marine Farming Zone the depth of cages will be restricted by water depth. Established aquaculture sites in South Australia use cages as little as 4 m deep (PIRSA 2002) but it has to be taken into consideration that these cages may be too shallow for meaningful commercial development (Poortenaar et al. 2003). Three case studies were defined based on probable combinations of water and cage depth for the investigated area:

1. Water depth = 10 m, cage depth = 5 m, free water depth = 5 m
2. Water depth = 20 m, cage depth = 10 m, free water depth = 10 m
3. Water depth = 30 m, cage depth = 15 m free water depth = 15 m

Poortenaar et al. (2003) and PIRSA (2002) recommend stocking densities of yellowtail kingfish <10 kg m^{-3}. This seems low compared to published stocking densities of other finfish species and a maximum density of 20 kg m^{-3} was suggested by S. Pether (NIWA Bream Bay Aquaculture Park, pers. comm.). Consequently three case studies were selected to represent the recommended stocking densities for yellowtail kingfish but also those reported in the literature for other finfish species:

1. Stocking density = 10 kg m^{-3}
2. Stocking density = 15 kg m^{-3}
3. Stocking density = 25 kg m^{-3}

3. Bayesian network analysis

3.1 Bayesian networks

The Bayesian network used in this report is a slightly modified version of the existing Bayesian network and detailed descriptions of the construction and parameterisation processes are reported in Giles (in press).

A Bayesian network is a graphical structure in which each variable is a node and edges between the nodes represent conditional dependencies (e.g., Pearl 2000). Bayesian networks are often called probabilistic causal models, because the direction of their arrows can be used to indicate the direction of a causal dependence (Borgelt and Kruse 2002). However, the structure of the Bayesian network in this study was primarily based on the relationships between variables revealed in correlation analyses and scatterplots to best represent the literature data and therefore does not represent the complete causal structure of the underlying processes. Nodes that have no incoming arrows represent input parameters and correspond to variables that can be controlled. Variables in a Bayesian network have a range of mutually exclusive states and therefore all continuous variables were discretised into sub-ranges. These ranges were chosen to cover the range of reported values and represent their distribution as evenly as possible.

To describe relationships between each pair of connected variables Bayesian networks utilise probabilistic, rather than deterministic, expressions. Each dependence indicated by an arrow represents a conditional probability distribution that describes the relative likelihood of each value of the variable with an incoming arrow (also called nature node), conditional on every possible combination of values of the variable the arrow is pointing away from (e.g., Borsuk et al. 2004). The probabilities representing the relationships in a Bayesian network are usually estimated using data or expert judgement (Reckhow and Chapra 1999). In this study, most conditional probabilities were derived by fitting discrete probability distributions for each connected variable pair but in some instances gaps in the available data required an estimate of probabilities based on expert judgement, mainly derived from data and relationships originating from other studies of benthic enrichment. The Bayesian network was implemented using the software package Netica 1.12.

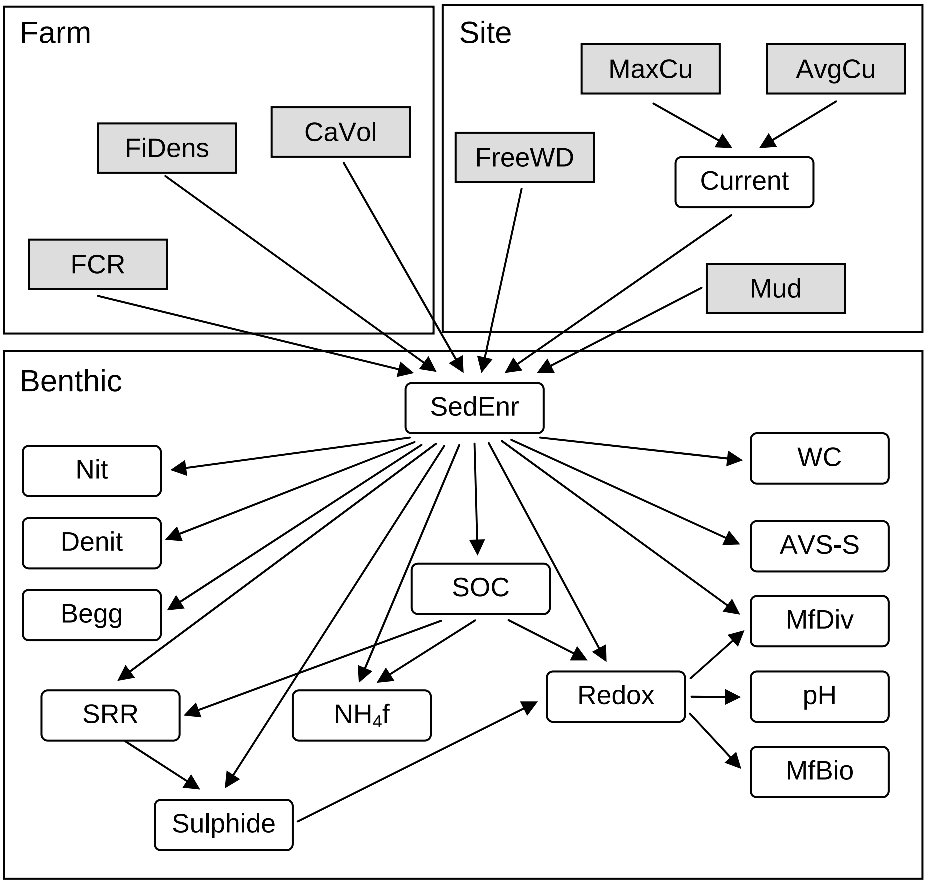


Figure 4. Bayesian network of benthic impacts of fish farming. Shaded rectangular nodes represent input variables and rounded nodes represent variables that are calculated from the states of the variables influencing them according to their conditional probability tables. Variables are grouped into farm characterising (Farm), site characterising (Site) and benthic impact (Benthic) variables. Variable abbreviations are explained in Table 1.

3.2 Bayesian network for assessment of potential benthic effects of finfish farming

The Bayesian network developed for the assessment of potential benthic effects of finfish farming comprises three farm, five site and 14 benthic variables (Figure 4, Table 1). Benthic variables were linked to farm and site characterising variables through a variable called sediment enrichment (SedEnr) that represented the three parameters organic matter (OM), organic carbon (OC) and nitrogen (N; Table 2; Giles in press). Similarly, the variable Current regime (Current) was introduced to combine reported average and maximum current speeds into one variable, thereby classifying the general current regime (Giles in press).

Table 1. Variables used in the Bayesian network. The abbreviations used in this report and parameter units are shown.

Variable	Abbreviation	Unit
Farm characteristics		
Cage volume	CaVol	m^3
Fish density	FiDens	$kg\ m^{-3}$
Food conversion ratio	FCR	–
Site characteristics		
Free water depth below cage	FreeWD	m
Sediment mud fraction	Mud	%
Average current speed	AvgCu	$cm\ s^{-1}$
Maximum current speed	MaxCu	$cm\ s^{-1}$
Benthic biogeochemical parameters		
surficial (0–2 cm)		
Porewater sulphides	Sulphide	$mmol\ L^{-1}$
Acid volatile sulphide S	AVS-S	$mg\ g^{-1}$
Water content	WC	%
Redox potential (at 2 cm)	Redox	mV
pH	pH	–
Sediment–water fluxes and redox reactions		
Sediment oxygen consumption	SOC	$mmol\ m^{-2}\ d^{-1}$
Sediment–water ammonium flux	NH_4f	$mmol\ m^{-2}\ d^{-1}$
Sulphate reduction rate	SRR	$mmol\ m^{-2}\ d^{-1}$
Nitrification	Nit	$mmol\ m^{-2}\ d^{-1}$
Denitrification	Denit	$mmol\ m^{-2}\ d^{-1}$
Benthic biological		
Shannon–Wiener diversity index	MfDiv	–
Macrofauna biomass	MfBio	$g\ m^{-2}$
Beggiatoa spp. mats	Begg	–
Derived variables		
Current regime	Current	–
Sediment Enrichment	SenEnr	–

The variables fish density (FiDens) and free water depth below cage (FreeWD) were varied according to the case studies described in section 2.3. Values for the remaining input variables Current, Mud, cage volume (CaVol) and FCR were chosen based on data available from previous work conducted by the NIWA. As presented in section 2.2, currents in the Wilson Bay Marine Farming Zone are strong and average 42.1 cm s^{-1} and 24.6 cm s^{-1} for maximum and mean current velocity, respectively. While there are some variations predicted among the six examined sites in the area most values were represented by the same states of the variables MaxCu (current velocity range

Table 2. Organic matter (OM), organic carbon (OC) and nitrogen (N) contents (% dry weight) representing the four states of the Bayesian network variable Sediment Enrichment (SedEnr).

SedEnr	OM (%)	OC (%)	N (%)
low	<5	<1.4	<0.25
moderate	5–10	1.4–2.8	0.25–0.50
high	10–20	2.8–5.7	0.50–1.00
very High	>20	>5.7	>1.00

30–45 cm s^{-1}) and AvgCu (current velocities >15 cm s^{-1}). At some sites the predicted maximum current speed was slightly above 45 cm s^{-1}. However, higher current speeds reduce benthic impact and thus using the indicated variable states is acceptable for an assessment of impact severity. Measurements of sediment mud content of selected sites in Area A indicate that these sediments are very muddy (Vopel et al. 2007). No measurements were available for Area B. However, based on previous work in the area it was assumed that sediments contain similarly high mud fractions. As a result, the variable Mud was set to its highest state (mud content >70%). Poortenaar et al. (2003) recommend 15 m × 15 m × 15 m cages during development of yellowtail kingfish farms. Most cages examined in the literature studies were circular with diameters ranging from 2.5 to 24 m. In 42 studies cages were <10 m, in 7 cages were 10–20 m and only 2 studies examined cages >20 m diameter. For the analysis in this report a cage diameter of 15 m was chosen. The food conversion ratio (the number of kg food used to produce 1 kg of fish, FCR) for yellowtail kingfish is largely unknown but likely to be reasonably good (PIRSA 2003). The variable FCR was set to its second lowest state, defined by FCRs of 1.2–1.6, based on research conducted at the NIWA Bream Bay Aquaculture Park (S. Pether, pers. comm.).

3.3 Predicted benthic effects of finfish farming in the Firth of Thames

The results of the Bayesian network analysis represent probability distributions of the variables, that is, the probabilities of each variable state being attained in the particular case study. For some variables, several states had similar probabilities, reflecting uncertainty in the prediction of variable values. In some instances the calculated probabilities for different case studies were similar. This could be an indication that the variable is insensitive to changes in the varied input parameter(s). Alternatively, it could indicate that the data used to parameterise the Bayesian network did not contain sufficient detail to enable an analysis of the variable's response to these variations. It is important to stress that the Bayesian network cannot predict exact environmental impacts. It was not parameterised or calibrated with site specific data, thus its purpose is to demonstrate the scale of expected benthic impacts and trends based on literature data. The focus of this report is to highlight the differences in benthic impacts predicted from the case studies and the trends generated from varying the input parameters, rather than making predictions of absolute variable values. The probabilities of all variable states for a specific case study sum to one, e.g., if increasing stocking density induces a reduction in the probability of seeing 'low enrichment', 'moderate enrichment' and 'high enrichment' it induces a corresponding increase in the probability of seeing 'very high enrichment'.

The calculated probability distributions of the benthic impact parameters are shown in Figures 5 and 6. In addition, values are presented in Appendix 1. Sediment enrichment was predicted to be most likely moderate for free water depths below cages of 10 or 15 m but when the free water depth was reduced to 5 m the probability of very high impact was considerably enhanced (Figure 5). In these shallow environments the probability of very high impact was predicted to be up to 36 %, which was 1.6 times higher than the highest probability calculated for the 10 m (23 %) and 15 m (21 %) free water depth cases.

Changes in the free water depth below cages, particularly the reduction from 10 to 5 m, also generated the most apparent response in most other variables (Figure 6). For example, the likelihood of *Beggiatoa* spp. mats occurring under cages increased from 28 to 32 % when free water depth was reduced from 15 to 10 m (stocking density 15 kg m^{-3}) but to 43 % when the water depth was further reduced to 5 m. When comparing the same cases the probability of porewater sulphide levels >6 mmol L^{-1} increased from 3.9 to 4.5 % following a reduction of free water depth from 15 to 10 m but reached 8.0 % after a reduction to 5 m. Similarly, the probability of Redox potentials <–100 mV increased considerably more for a reduction from 10 to 5 m free water depth than for a reduction from 15 to 10 m. Wildish et al. (2004b) categorised sediments at a salmon farm into normal, oxic, hypoxic and anoxic conditions based on sulphide concentration and Redox potential together with sedimentary observations for macrofauna of Pearson and Rosenberg (1978) and sediment profile images (SPI) and the benthic habitat quality (BHQ) of Nilsson and Rosenberg (2000). Sulphide concentrations above 1.3 mmol L^{-1} in combination with Redox potentials below 0 mV represented hypoxic conditions and sediments with sulphide concentrations >6 mmol L^{-1} and Redox potentials <–100 mV were classified as anoxic. These threshold values give an indication of the severity of benthic impacts and illustrate that the probabilities of these variable states should be minimised. However, since the classification was based on a combination of more factors than porewater sulphide concentration and Redox potential alone, it cannot be used to assess the severity of benthic impacts predicted from the Bayesian network.

Increasing stocking generally increased benthic impacts but these effects were not large. This was partly due to the problem that stocking densities vary over the growing season, but typically only one stocking density was reported. The reported value usually represented the maximum stocking density or the stocking density at the time of measurements. This inconsistency created uncertainty that is reflected in the results of the Bayesian network analysis.

For some variables only sparse data were available so, while it was possible to calculate probability distributions, it is important to note that these are only very vague estimates of the expected values. These variables are denitrification, nitrification and sulphate reduction.

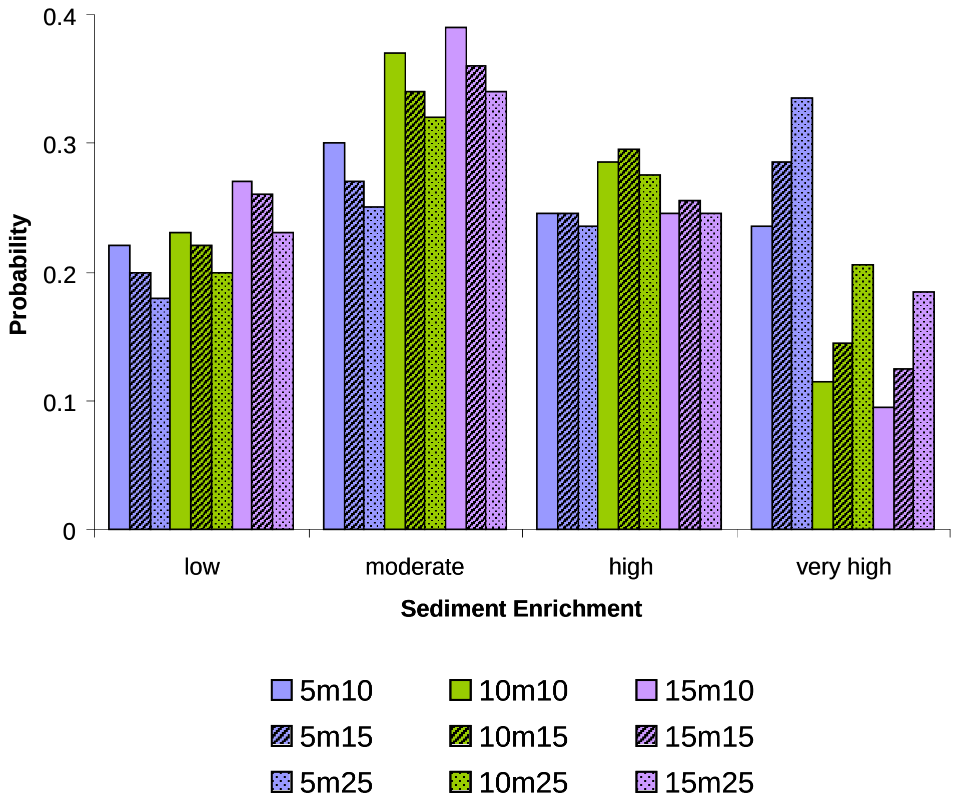
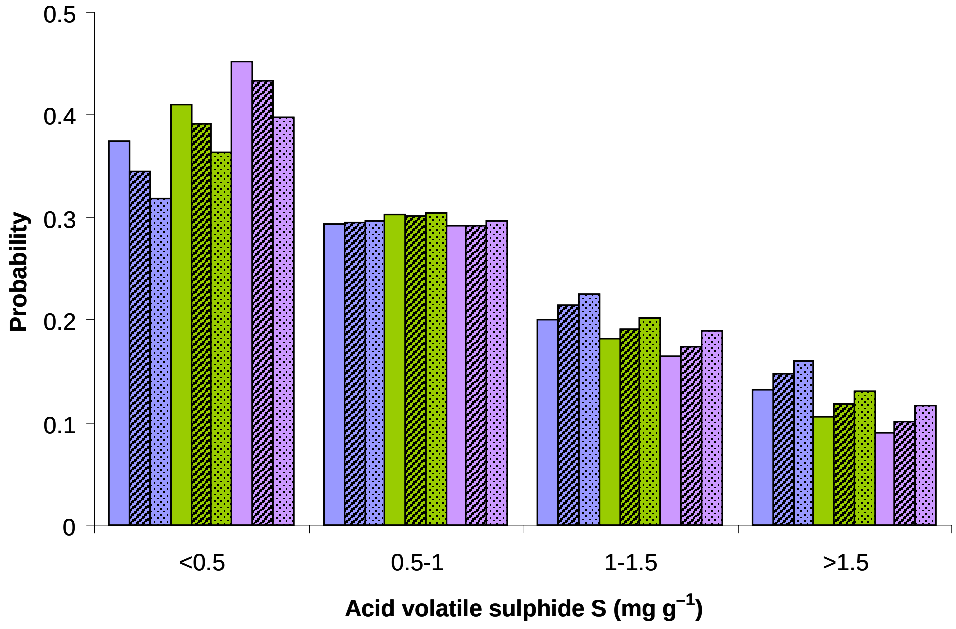


Figure 5. Predicted probability distributions of variable sediment enrichment calculated for the case studies. Cases are defined by free water depth below cages (case 1 = 5 m, case 2 = 10 m, case 3 = 15 m) and stocking density (case 1 = 10 kg m^{-3}, case 2 = 15 kg m^{-3}, case 3 = 25 kg m^{-3}). Cases are coded by colour (blue = 5 m free water depth, green = 10 m free water depth, purple = 15 m free water depth) and shading (no shading = 10 kg m^{-3} stocking density, diagonal lines = 15 kg m^{-3} stocking density, dots = 25 kg m^{-3} stocking density).



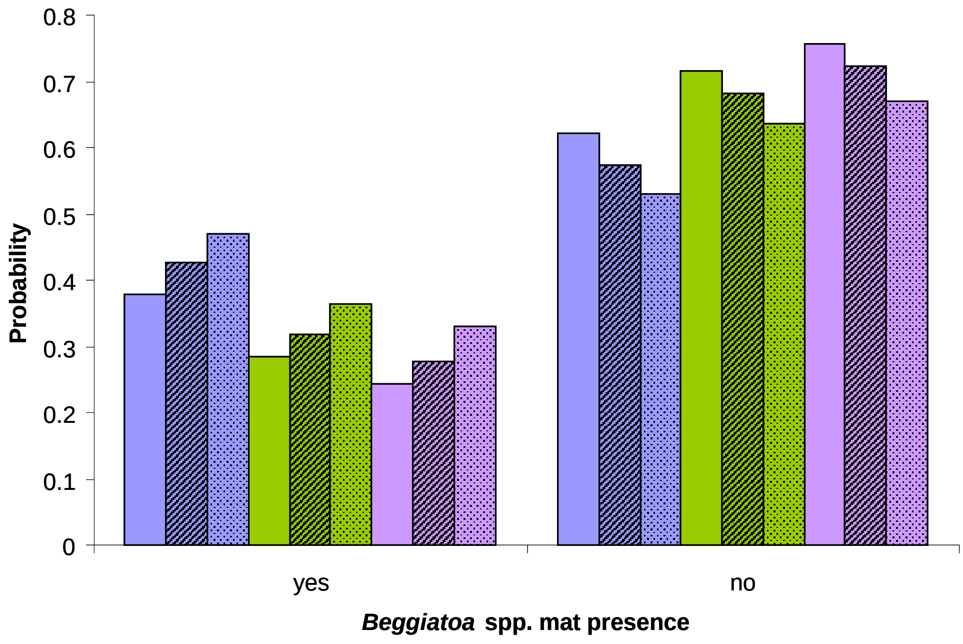
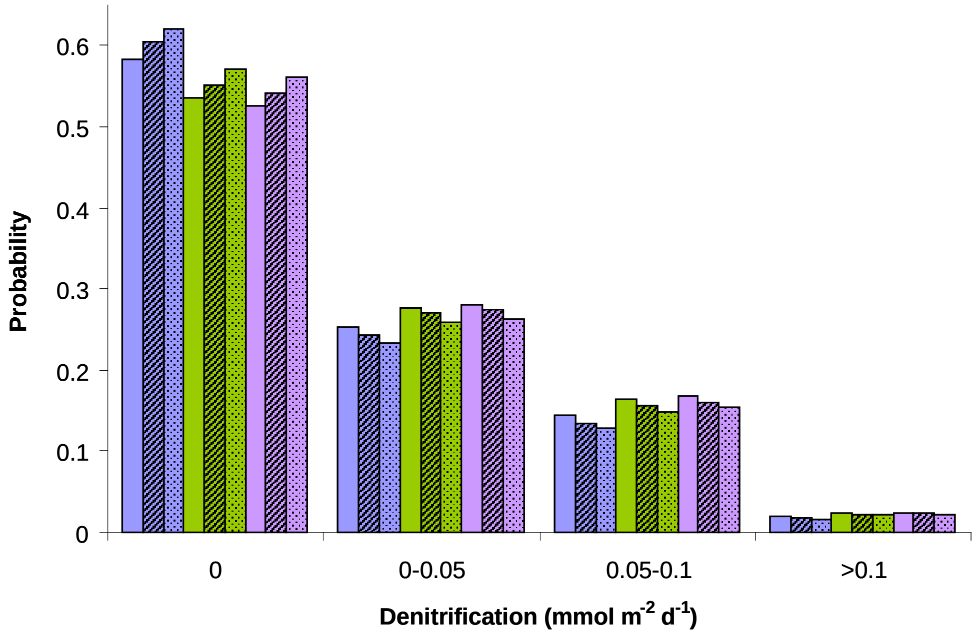


Figure 6. Predicted probability distributions of benthic impact variables (in alphabetical order). Cases are coded by colour and shade as described in Figure 5.



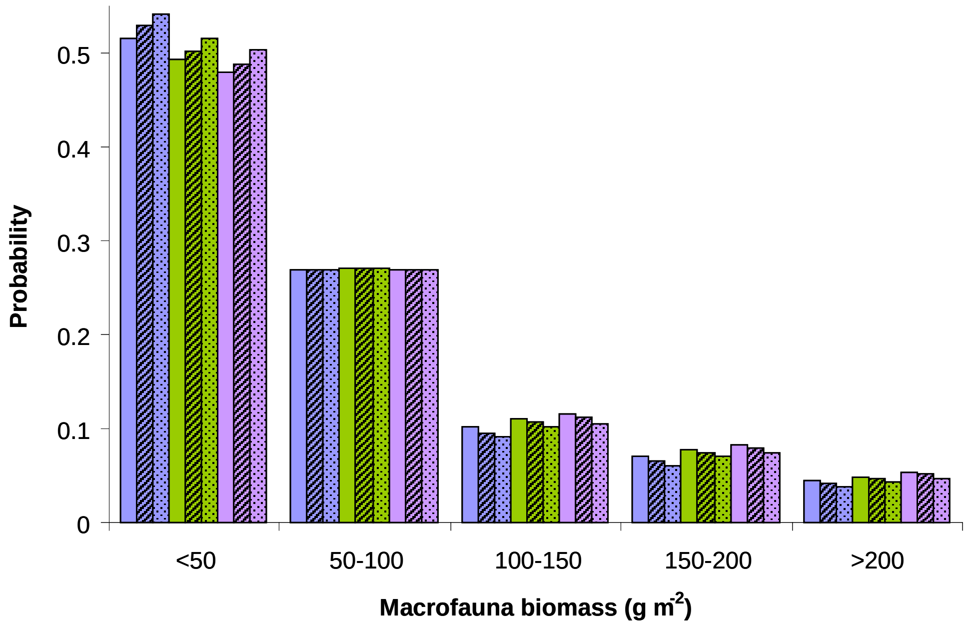
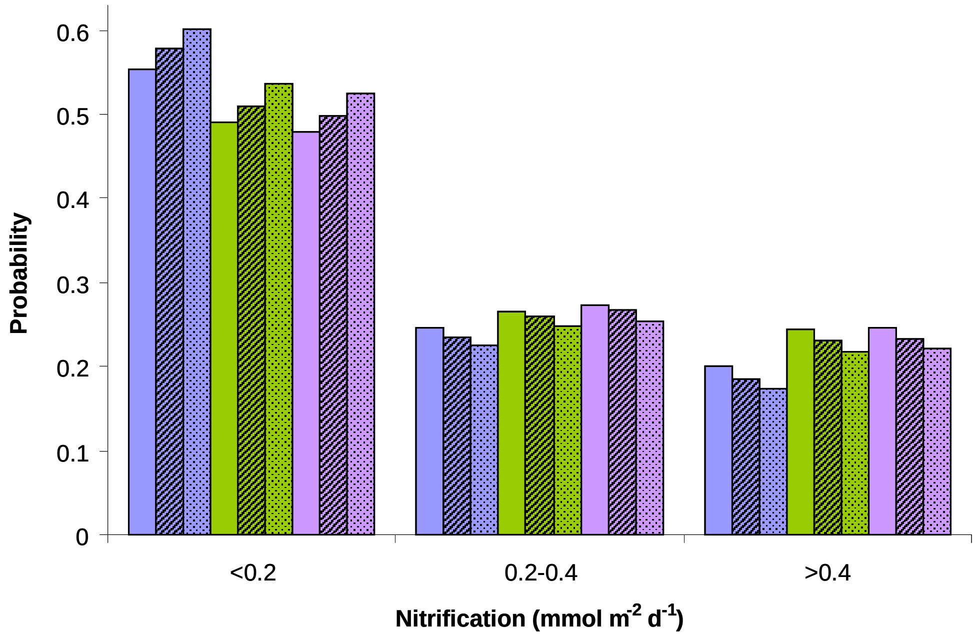
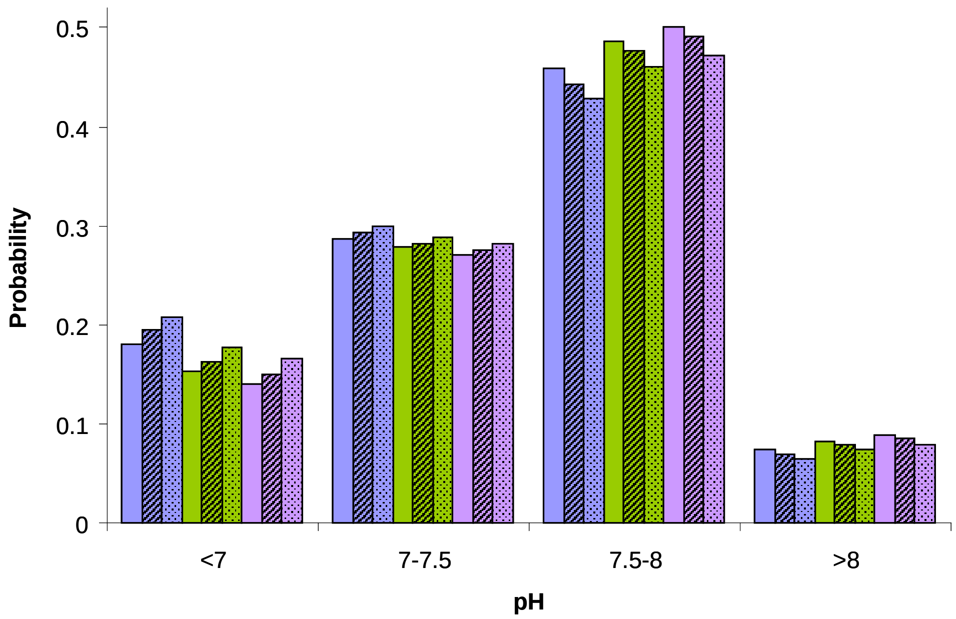


Figure 6. (continued)





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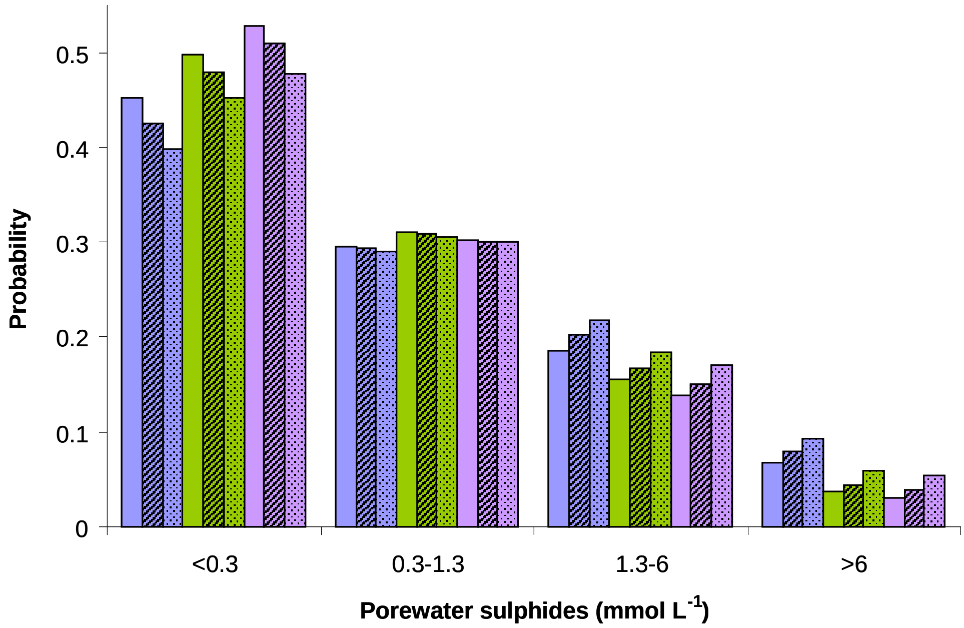

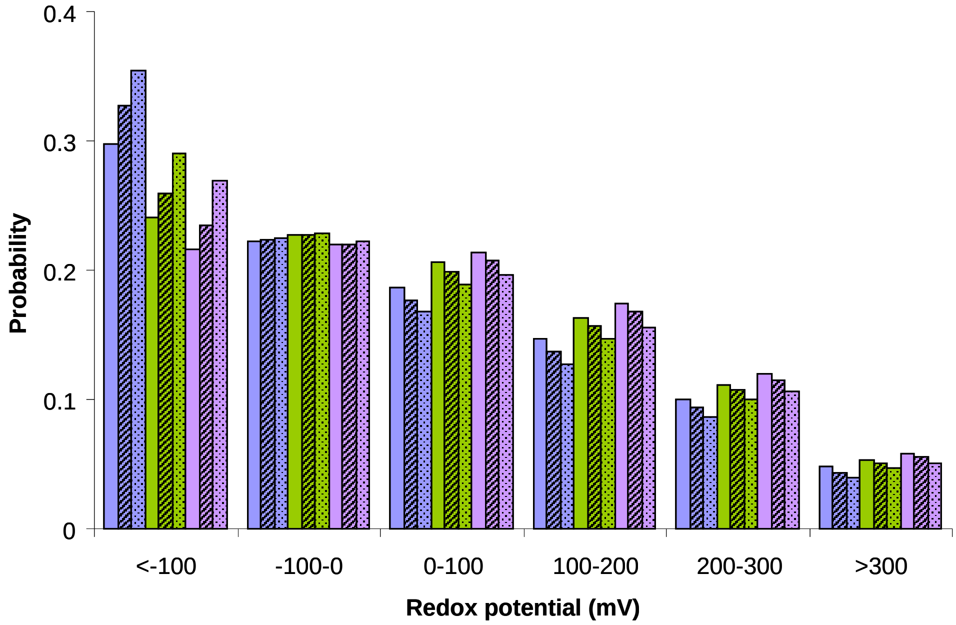

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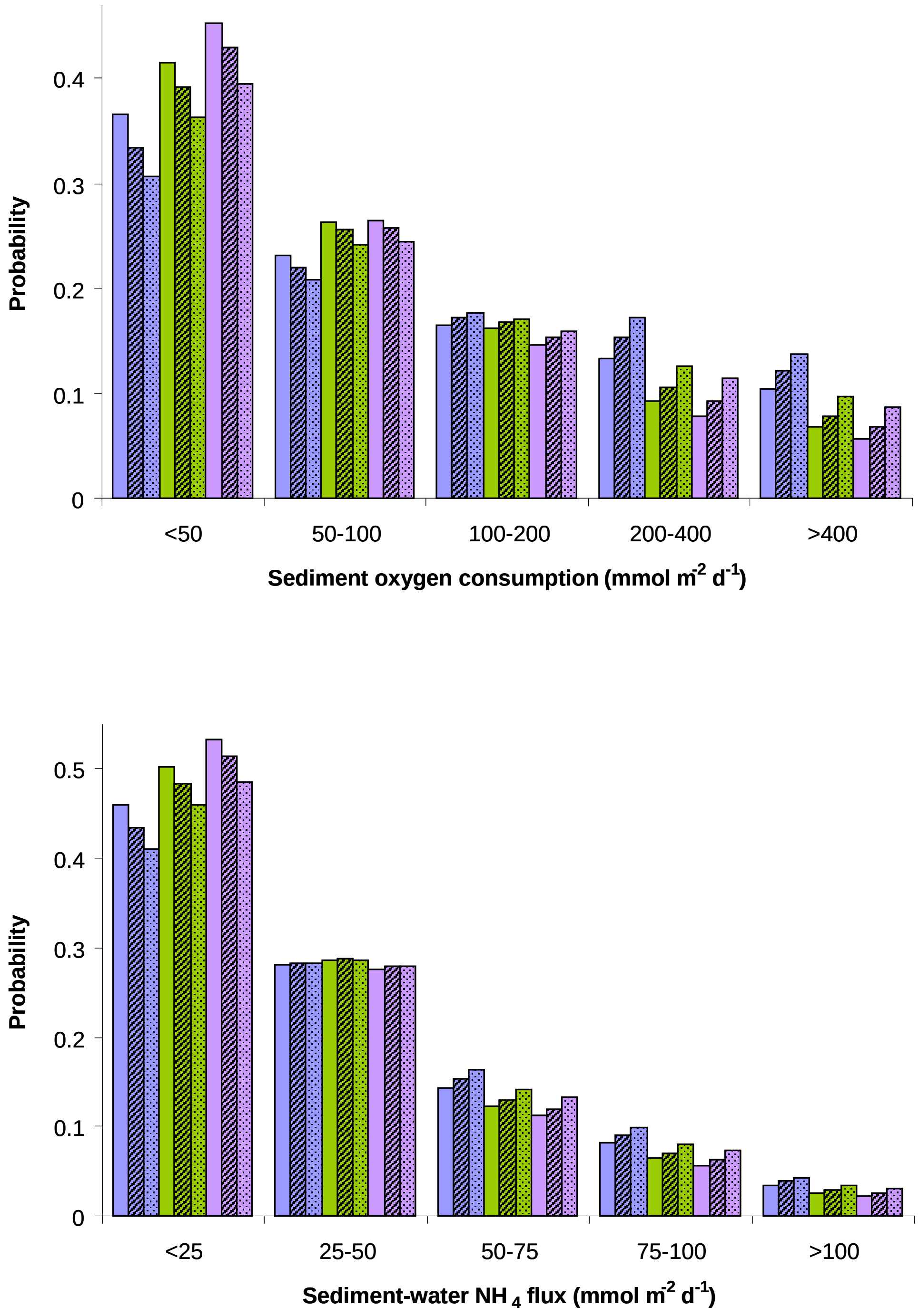

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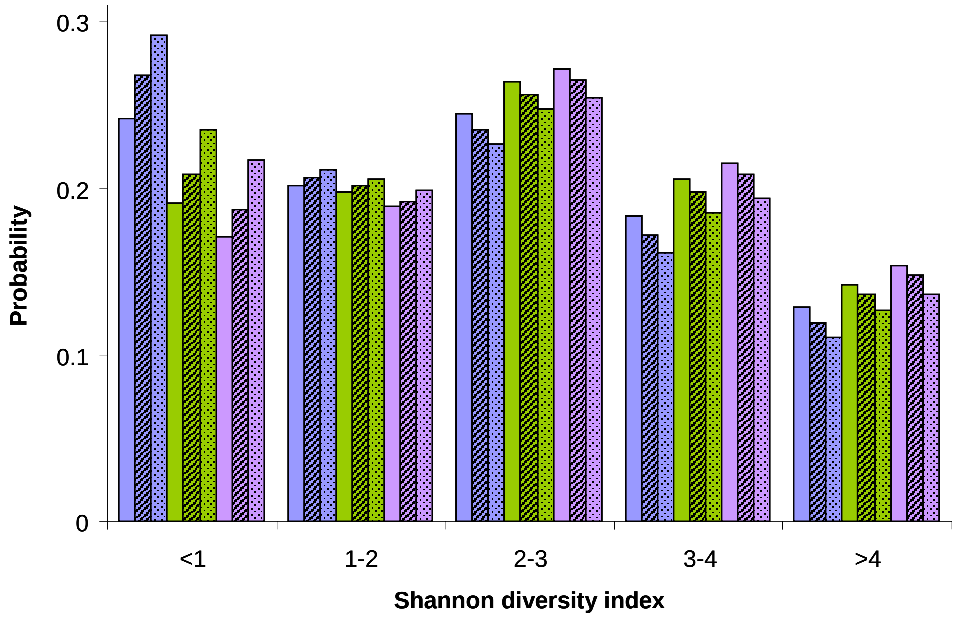

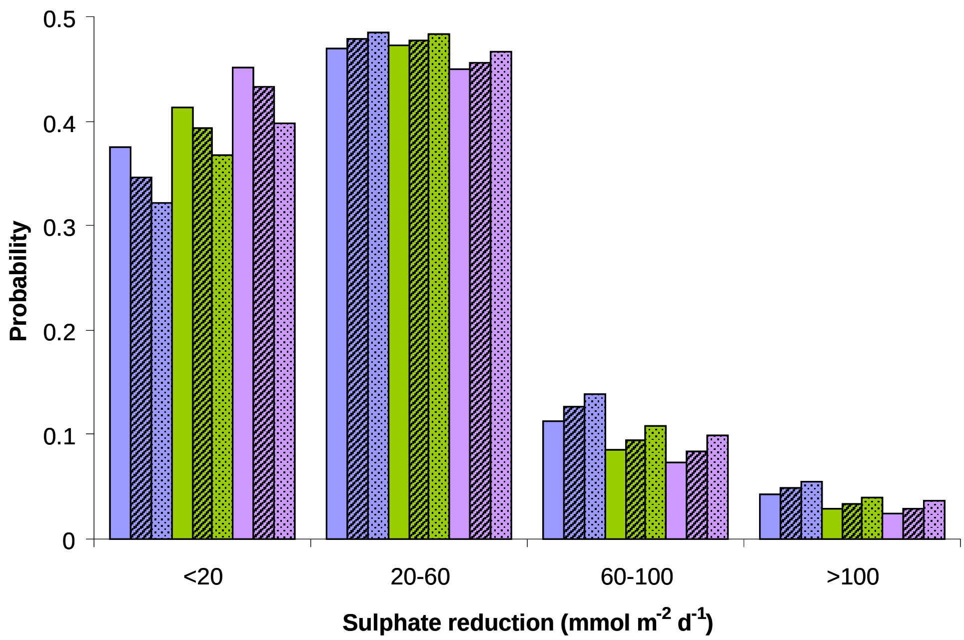

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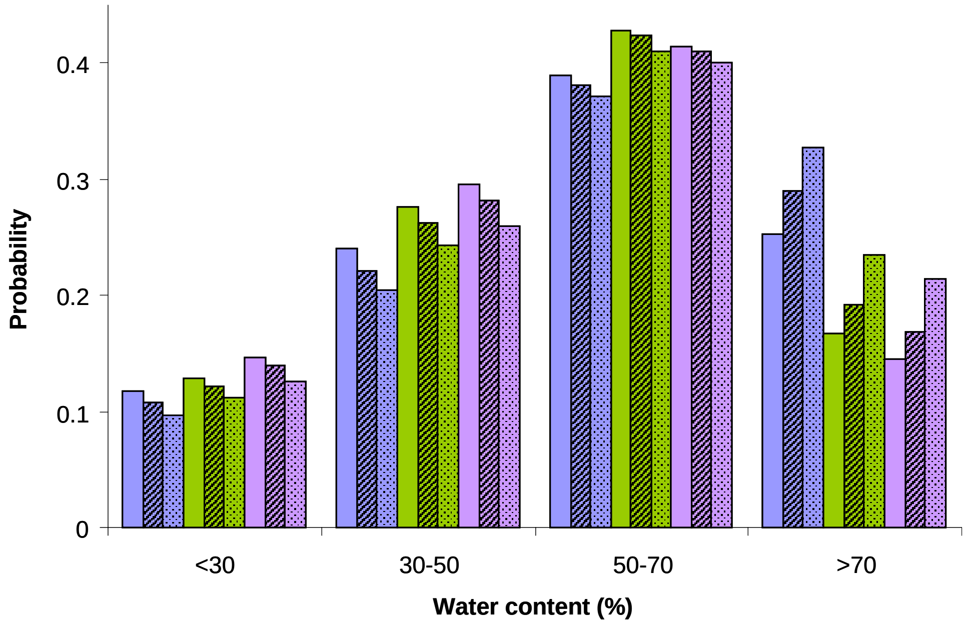

Figure 6. (continued)

4. Sensitivity to increases in FCR

The FCR of yellowtail kingfish were estimated as 1.2–1.6 (see section 3.2). Other culture species may have higher FCRs and thus the sensitivity of the Bayesian network results to increased FCRs was examined. Three case studies were selected representing low, moderate and high environmental impact (Table 3). For each selected case the variable FCR was increased to >1.6–2.0 and the calculated probabilities were compared. The benthic impact parameters showed similar trends and therefore only one variable, sediment enrichment, was presented.

As expected, an increased FCR, that is, an increased amount of food required to obtain a certain mass of fish, enhances benthic impact through the increased amount of material lost to the environment (Figure 7). The level of enhancement is moderate (0–24% change in calculated probabilities of variable states) and differs among the examined cases. An increase in FCR has the highest effect when impact is low (7–24% change in calculated probabilities, Figure 7a) but seems to be less relevant when impacts are severe. If fish farms will be established in the Firth of Thames monitoring will be carried out to ensure that benthic impacts are minimised and therefore it is likely that the FCR of fish cultured in the Firth of Thames will play an important factor in the level of benthic impact.

Estimating the amount of material lost to the environment from fish culture can be done using two methods: direct, through sampling and analysis of the water column and of sedimenting particulate material; and indirect, using a mass balance approach. The former approach requires the deployment of sediment traps and is only practical for estimating uneaten food and faecal solids (Beveridge 2004). Based on previous work with yellowtail kingfish faeces done by NIWA it is considered unlikely that faecal material could be identified in sediment traps (Giles, unpublished work). In addition, sediment trap deployments in the Firth of Thames are problematic (see section 5.2) and, consequently, reliable direct measurements of FCRs are not

Table 3. Cases representing low, moderate and high benthic impact chosen for the assessment of Bayesian network sensitivity to increases in FCR. Case references are explained in Figure 5.

Impact	Case reference	Free water depth (m)	Stocking density (kg m^{-3})
Low	15m10	15	10
Moderate	10m15	10	15
High	5m25	5	25

considered possible in the Firth of Thames. A mass balance approach for the estimate of material lost to the environment would be the preferred method but requires a substantial amount of field and laboratory data, including the FCR (Beveridge 2004). Therefore, it is recommended to obtain estimates of FCRs for species considered for cultivation in the Firth of Thames prior to farm developments.

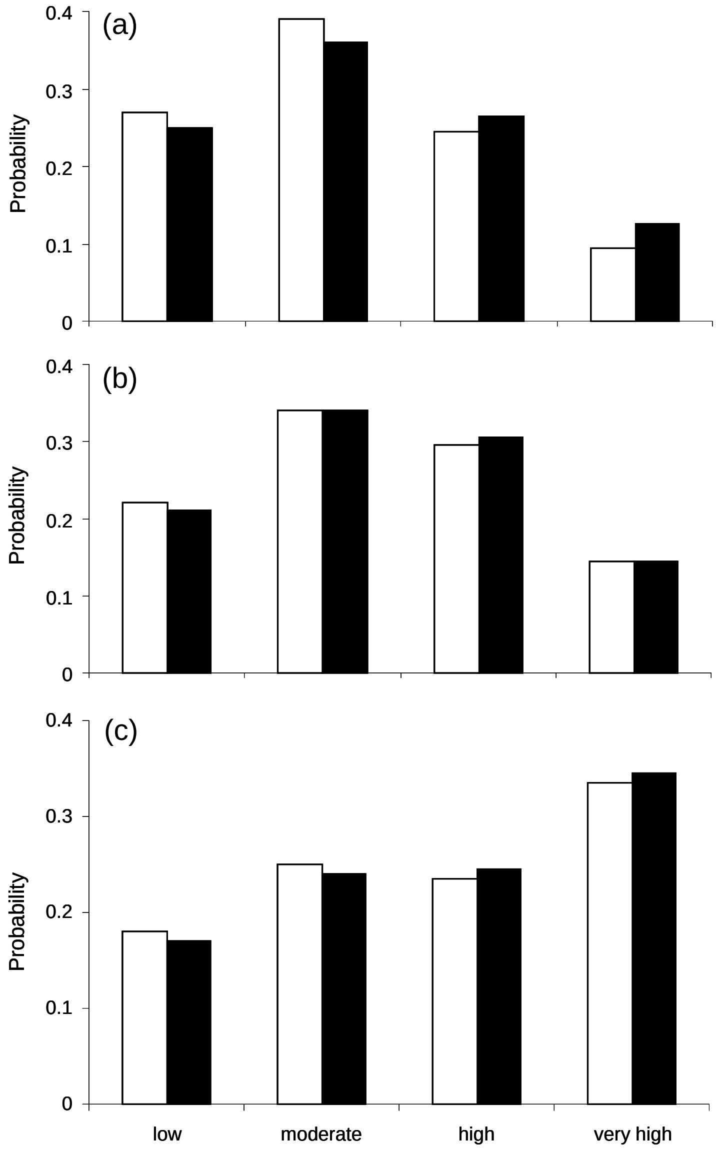

Figure 7. Predicted probability distributions of variable sediment enrichment calculated for (a) low, (b) moderate and (c) high impact with FCRs between 1.2 and 1.6 (white bars) and >1.6 and 2.0 (black bars).

5. Review of selected literature studies

The sections present a review of literature studies carried out in areas with hydrographic conditions similar to those observed in the Firth of Thames. The focus is on studies that examined parameters that have not been included in the Bayesian network and the usefulness of these additional parameters for the assessment of benthic effects in the Firth of Thames is discussed.

5.1 Trace metals and video surveys

Kempf et al. (2002) carried out an environmental monitoring study at the site of a marine trout farm located in the Bay of Cherbourgh (English Channel), France. At their study site the maximum current velocity ranged from 30 to 55 cm s^{-1} and the water depth was approximately 11 m with a large tidal range (2.5–5.3 m). Brown trout were cultured in net pens of 20 m × 15 m × 6.5 m (2000 m^3) arranged on four rafts, carrying four net pens each. The free water depth below net pens was 6–12 m depending on tide. During the study fish biomass ranged from 140–350 t with a temporary maximum of 450–500 t, corresponding to stocking densities of 4.4–15.6 kg m^{-3}. The food conversion ratio was 1.3 to 2. These farm and site characteristics are similar to the characteristics in the Firth of Thames and the parameters selected for the case studies examined in this report.

In addition to examining organic matter, carbon and nitrogen, macrofauna and sediment oxygen consumption, Kempf et al. (2002) analysed sediments for trace metals, and a video survey was carried out. Metals such as copper (Cu) and zinc (Zn) were incorporated as trace elements in the fish feed used at the farm. These elements are largely eliminated with the faeces and become trapped in the sediment's fine fraction. The amounts of both metals in the sediment were generally low but elevated below the net pens. The authors found a negative relationship between trace metal content and mud fraction and suggested that this may be a result of trace metal dilution in the increase of the mud fraction. The video survey complemented the knowledge of benthic macro-organisms and isolated spots of *Beggiatoa* spp. mats were noticed. However, the authors discussed the technical and financial difficulties of carrying out video surveys around fish farms and suggest fitting remotely operated cameras on net pens as a compromise for a quick monitoring and initial diagnosis of the sediments.

When comparing the results of Kempf et al. (2002) to the Firth of Thames it must be noted that the average mud fraction in their study area (<16–22 %) was considerably lower than the mud fraction in the Wilson Bay Marine Farming area. The authors suggested that a higher mud content may dilute the trace metal content in the sediments, which is an interesting aspect when estimating the potential consequences of trace metal input to the muddy sediments in the Firth of Thames. However, depending on the metal content of fish feed likely to be used in the area, it would be prudent to carry out trace metal analyses to ensure that the effects are low.

Fitting cameras to fish cages in the Firth of Thames is not a feasible option for monitoring the sea floor due to the high turbulence and consequently low visibility. The low water clarity reduces the quality of sediment surface video footage taken by divers (Vopel et al. 2007) and consequently is not recommended for the monitoring of fish farm impacts in the Firth of Thames.

5.2 Biogeochemical processes and sediment trap deployments

A model of benthic response to organic enrichment caused by salmon net-pen aquaculture was developed by Findlay and Watling (1997) based on the ratio between oxygen supply and oxygen demand. Oxygen supply was calculated using three key environmental variables: flow velocity, temperature and oxygen concentration of the bottom water. Comparisons of measured carbon sedimentation rates and rates of benthic metabolism revealed strong linear correlations and consequently in the model sediment oxygen demand was predicted from measured carbon sedimentation rates. The authors conclude that their model synthesises the understanding that sediment oxygen delivery and sediment oxygen consumption both play a role in determining benthic community structure and, if these two parameters are measured and compared, then the level of impact can be predicted for net-pen aquaculture.

While this modelling approach was successful for the farm locations examined in this study it is unlikely to be useful for the Firth of Thames. One study site examined by Findlay and Watling (1997) had currents (maximum = 52.3 cm s^{-1}, mean = 21.5 cm s^{-1}) and water depth (11.4 m) similar to the Firth of Thames; however, the sediment at this site was poorly-sorted gravel. The authors comment that, in shallow-water systems, sediment trap measurements of deposition rates can be confounded by resuspension followed by settling. They suggest that in these cases a comparison of the organic and inorganic content of sediments can be used to correct for the effect of resuspension on deposition rates. Previous sediment trap measurements in the Firth of Thames indicated that measured sedimentation rates are seriously biased by a combination of sediment derived from rivers draining farm land and high current speeds and wind generated waves that resuspend sediment locally as well as on nearby extensive intertidal mudflats (Giles et al. 2006). In their study, resuspension was estimated to account for 56 to 87% of measured sedimentation and consequently only relative differences between sites, rather than absolute values, were discussed. These deployments were made in the Western Firth of Thames. However, it is highly likely that resuspension would affect sediment trap measurements in the Eastern Firth of Thames to a similar extent. Consequently, sediment trap deployments and the modelling approach suggested by Findlay and Watling (1997) are not recommended for the Wilson Bay Marine Farming Zone.

Nevertheless, Findlay and Watling (1997) raised some important aspects that are relevant to the benthic impact assessment in the Firth of Thames. They reviewed previous studies that showed that if enrichment was sufficient to generate an oxygen demand greater than the supply deliverable by diffusion, decomposition of organic matter was mediated by anaerobic processes and virtually all oxygen consumption could be related to the reoxidation of reduced alternative electron acceptors. Under these conditions natural sediments will be devoid of macrofauna and *Beggiatoa* spp. mats will form at the sediment-water interface. The authors reasoned that if the ability of the benthos to oxidise organic matter aerobically could be determined and compared to the organic flux originating from the net-pens, a prediction of benthic impact would result. This demonstrates the importance of understanding sediment biogeochemical processes, particularly the pathways of organic matter decomposition, for a comprehensive assessment of benthic impacts of fish farming. While the changes in macrofauna community structure and the occurrence of sulphide-oxidising bacteria are important and commonly used measures of benthic impact, severe impacts could possibly be avoided by focussing on biogeochemical impact parameters that indicate enrichment effects before they lead to faunal changes or *Beggiatoa* spp. mat formations.

5.3 Opportunistic macrofauna species

The Bayesian network includes macrofauna diversity but no information on the macrofauna species comprising the benthic community. Tomassetti and Porrello (2005) evaluated the polychaete population changes as a result of organic enrichment produced by a marine fish farm. The marine organism considered as the best indicator of polluted marine habitat is the endobenthic polychaete species complex *Capitella capitata* (Tomassetti and Porrello 2005). It consists of a number of morphologically similar sibling species and their major adaptation strategies are a high capacity for long-term anaerobiosis and/or the ability to oxidise sulphide. The results of this study showed a rapid change of the assemblage located beneath the cages related to the fish biomass reared and a high dominance of the opportunistic *Capitella* spp. The authors also state that changes in abundance and biodiversity of benthic assemblages are a consequence of changes in sediment chemistry. Thus, while the dominance of a low number of opportunistic species with a high abundance is related to environmental conditions and can be used as a good indicator of gross impacts, measurements of biogeochemical parameters may provide earlier signs of impact and allow remedial measures to be taken if necessary to prevent severe benthic impacts.

5.4 Stable isotopes

Sarà et al. (2006) used stable isotopes to examine differential effects of fish farm waste on the water column and sediments. The isotopic signal of fish waste is different from the isotopic signal of natural material (phytoplankton, terrigenous and sand microflora) and, therefore, this method allows the identification of the spatial distribution of fish farming. Sarà et al. (2006) detected some effects of organic loading up to 1000 m from the cages but no impact that implied biological consequences and environmental cost. The key findings of this study were that: (1) the relative area of influence of the impacts of fish farms seems to proportionally increase with increasing current velocities, (2) sediments appear to mirror the real dispersion better than the water column and (3) the distribution of waste from the cages seems to be dependent on movements at the bottom of the water column, which confirms the recently identified role played by resuspension movements. Stable isotopes could provide a useful approach for assessing the spatial extent of benthic impacts in the Firth of Thames if farms are established. Because of the distinctive isotopic signal of fish feed, stable isotope measurements can easily be linked to the farm, whereas this connection is difficult to establish for commonly measured compounds such as total organic matter, carbon or nitrogen.

6. Estimates of farm footprints in the Firth of Thames

6.1 Spatial extent of benthic impacts

6.1.1 Estimates based on peer-reviewed literature

A previous review of literature information on the spatial extent of benthic fish farm impacts showed that most parameters changed rapidly with distance from the farm (Giles in press) and suggested that impacts were confined to a radius of about 40 - 70 m around farms. It was demonstrated that the footprint of a farm depends on the parameter examined and that some parameters reached reference values at closer distance to the farm than others. This review study also confirmed that impacts were less intense but further spread in deep areas, whereas impacts in shallow areas appeared more intense but confined to a smaller area around the farm. Due to a lack of available literature data only a limited number of parameters could be examined and consequently it was emphasised that the footprint of a farm may extend beyond the estimated distance. Furthermore, specific local environmental conditions may provide higher or lower capacities for fish farm activities than those of the reviewed studies and, therefore, the size of a farm footprint may be outside the 40–70 m range suggested.

6.1.2 Estimates based on New Zealand monitoring data

An investigation of the spatial extent of benthic impacts observed at fish farms in New Zealand indicated slightly larger footprints than suggested from the peer-reviewed literature. Giles (in press) estimated that the spatial extent of benthic impact was about 40 m based on changes in sediment oxygen consumption and total phosphorus, 50 m for organic matter and 70 m for macrofauna diversity (measured as Shannon diversity index). Govier and Bennet (2007a–d) measured organic matter and macrofauna diversity at salmon farms in Te Pangu Bay, Waihinau Bay, Otanerau Bay and Ruakaka Bay (Figure 8). Organic matter declined rapidly outside the farm but levels were still elevated at a distance of 50 m. Data between 50 and 150 m were sparse but when interpolating the trend it seems that reference levels were reached at about 100 m. The Shannon diversity index values were very scattered and two values were still reduced at 150 and 250 m distance. It is not known if these values (from the same farm) were low due to farm impacts of naturally low diversity. When these values were ignored the spatial extent was about 100 m. However, more data are necessary to confirm this.

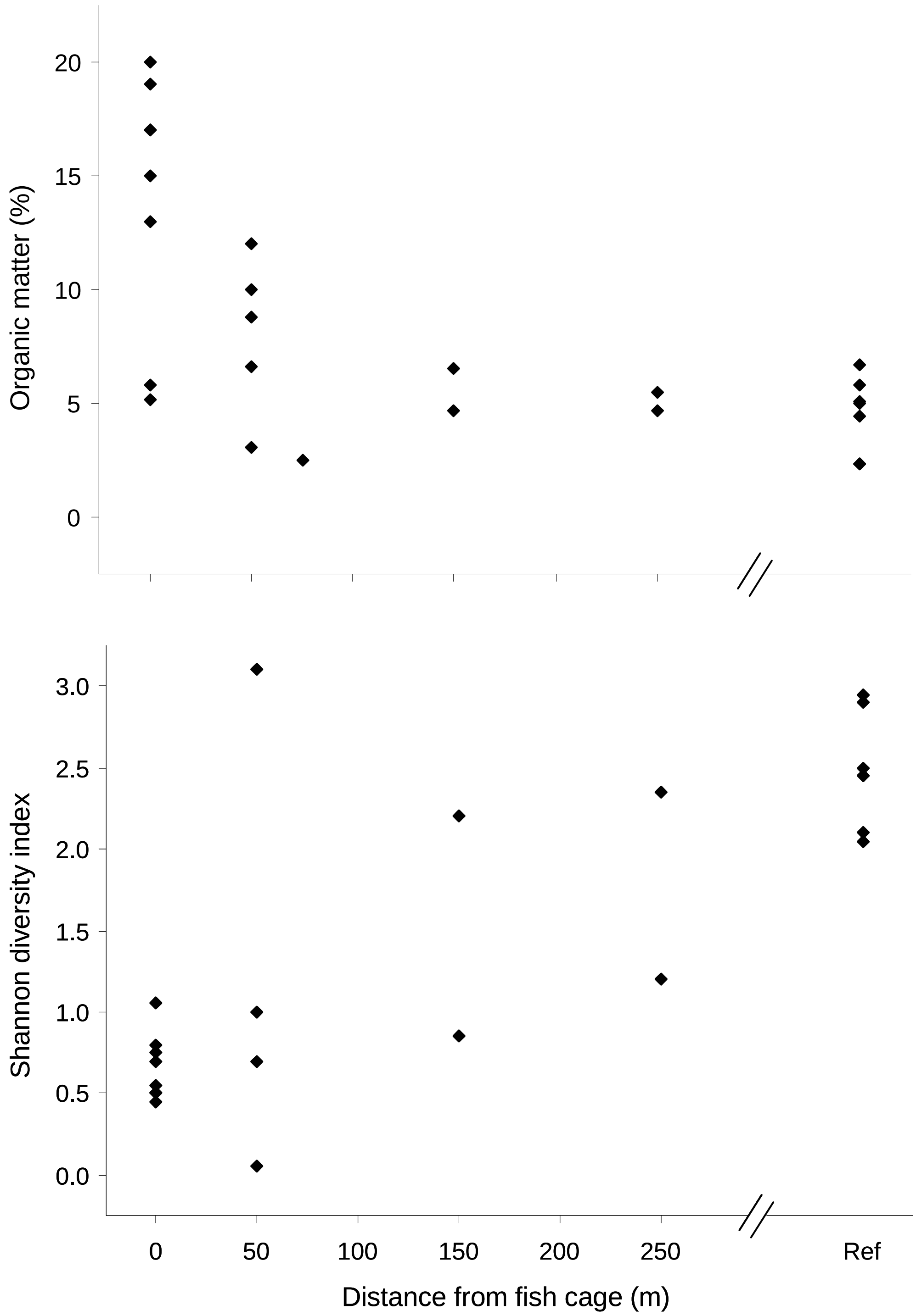

Figure 8. Organic matter and Shannon diversity index values measured at increasing distance from fish farms in Te Pangu Bay, Waihinau Bay, Otanerau Bay and Ruakaka Bay and at reference sites (Govier and Bennet 2007a–d).

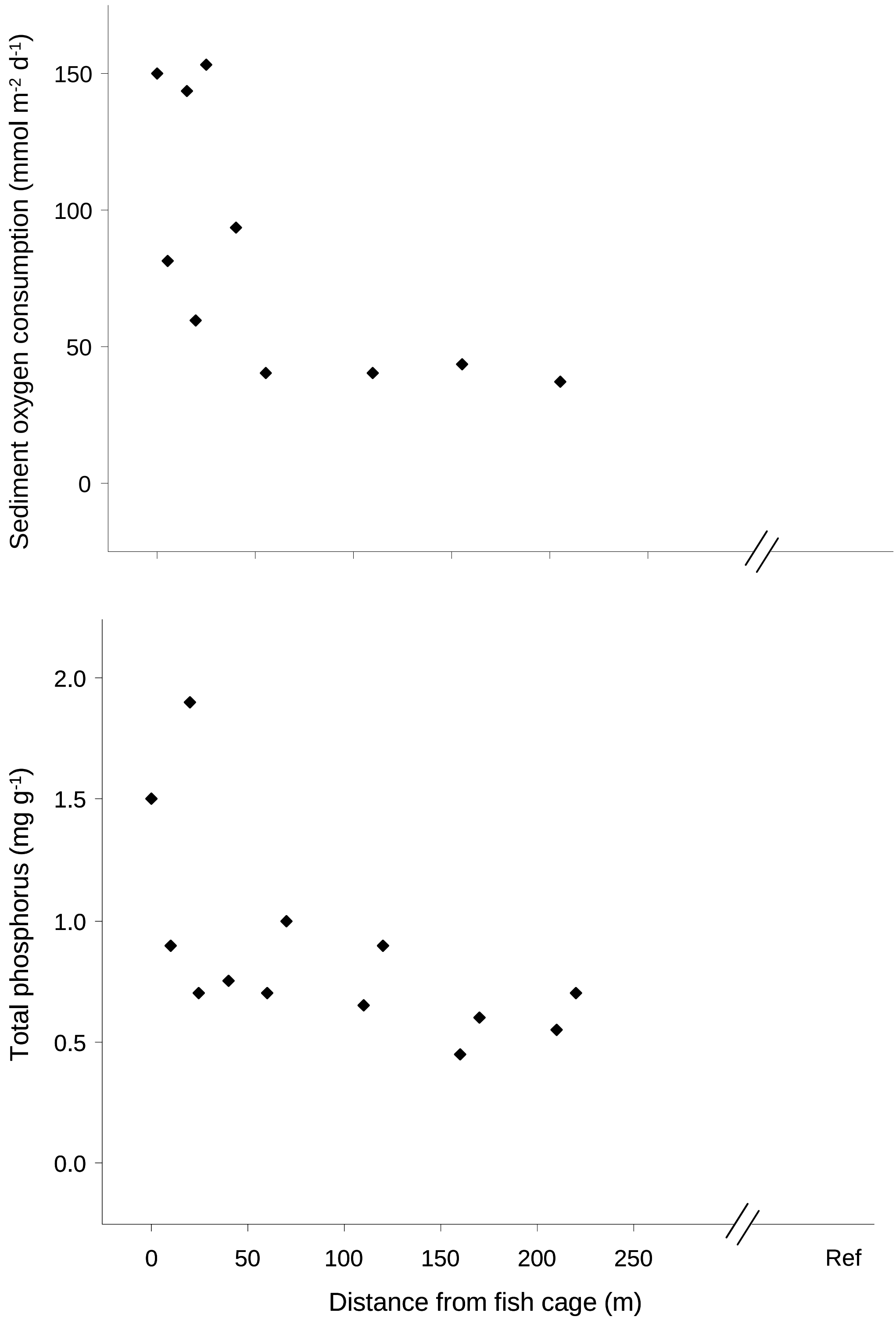

Figure 9. Sediment oxygen consumption and total phosphorus measured at increasing distance from fish farms in Big Glory Bay (Roper et al. 1988). Sediment oxygen consumption values are averaged from transects at five different farms and total phosphorus values were measured at two farms. No data were available from sites identified as reference sites.

Roper et al. (1988) measured sediment oxygen consumption and total phosphorus at salmon farms in Big Glory Bay in 1987/88 (Figure 9). No measurements were provided for reference sites but the data suggest that sediment oxygen consumption declined to background values at about 50 m distance from the farm. Total phosphorus also declined rapidly outside the farm but still showed a slight decrease up to about 150 m distance. It is important to note that these measurements of total phosphorus were made in the late 1980's and fish farm management has since made considerable advances leading to less environmental impact. For example, feed composition has been improved to ensure better feed assimilation and less waste material. Consequently, impact distances should not be based on these data.

6.2 Implications for buffer zones of farm blocks in the Wilson Bay Marine Farming Zone

Based on the estimates made in the previous section and by Giles (in press) a cautious estimate of the spatial extent of a fish farm appears to be approximately 100 m but the largest change of most parameters was observed within about 50 m of the farm. To minimise cumulative effects of neighbouring farm blocks the minimum distance between cages in neighbouring blocks should be 100 m but ideally 200 m should be allowed. Farm blocks in the Wilson Bay Marine Farming Zone Area A are 110 m by 250 m with a 75 m distance between them. If buffer zones (area without cages, measured as the distance between the outermost cages inside a farm block and the perimeter of the block) of 50 m were established the minimum distance between cages in neighbouring farm blocks would be 175 m. This is close to the ideal distance suggested above and therefore considered acceptable as a first estimate. However, a 50 m buffer zone is not realistic for the farm blocks in Area A that are only 110 m wide. The prevailing current flow is approximately parallel to the long side of the farm blocks. Considering that the major spatial distribution of benthic impacts occurs in the direction of the prevailing current flow it may be acceptable to allow smaller buffer zones along the long sides of farm blocks, allowing fish cages to be installed in the farm blocks of Area A. It is important to note that the spatial extent of specific fish cages will depend on a complex network of processes and environmental influences that cannot be predicted from literature data alone. The distances derived here are only ballpark figures based on measurements made at other locations and are by no means are intended to define performance standards or guidelines for farm operations.

Poortenaar et al. (2003) suggest that large cages (50–150 m diameter) could be used once the farm technology and expertise are developed. The lack of literature data available for cages of this size range precluded an assessment of the influence of cage size on the severity of benthic impacts but it is possible that large cages lead to more severe impacts. If large cages are to be installed in the Firth of Thames it is strongly recommended that benthic impacts should be measured at high spatial and temporal resolution until sufficient information on their severity and spatial extent has been gathered to make sound recommendations on minimum buffer zones for farm blocks.

7. Summary and recommendations

7.1 Bayesian network analysis

The Bayesian network analysis suggested that of the examined input parameters the free water depth below fish cages has the largest effect on the severity of benthic impacts. Probabilities of variable states indicating benthic impact increased more following a reduction from 10 to 5 m water depth than following a reduction from 15 to 10 m. Therefore it is recommended that the minimum free water depth below cages should be 10 m. In this study a 10 m free water depth was based on a total water depth of 20 m and a cage depth of 10 m. If fish cages are to be installed in shallower water it is recommended to reduce the cage depth and ensure a 10 m water column below cages. Changes in stocking density only resulted in small changes in the probability distributions of most variables. This was mainly due to limited and/or inconsistent data available for the Bayesian network parameterisation.

The aim of the Bayesian network analysis was to obtain some information about the scale of expected benthic impacts and the sensitivity of these impacts to different farm and site characteristics by examining the predicted probability distributions of selected parameters. Of these parameters, some are more difficult and/or expensive to measure than others and before selecting monitoring parameters methodological limitations must be considered. For example, Redox potential has been disregarded as a reliable parameter, particularly in reference sediments where oxic conditions prevail (Wildish et al. 2004a, Brooks and Mahnken 2003). A detailed discussion of the limitations of parameters can be found in Giles (in press).

7.2 Reviewed studies

The review of selected literature studies revealed several monitoring parameters that have not been included in the Bayesian network, either because only sparse data were available or because no consistent relationships could be detected with other parameters. The relevance of these additional parameters for detecting benthic impacts in the Firth of Thames was examined. Video surveys and sediment trap deployments were not recommended following problems experienced during earlier work caused by unfavourable environmental conditions. The examination of opportunistic macrofauna species was generally accepted as a good indicator of benthic impact. However, it was suggested that measurements of biogeochemical parameters may reveal earlier signs of impact and allow remedial measures to be taken if necessary to prevent severe impacts. Particularly relevant for the understanding of biogeochemical changes due to fish farming activity are changes in organic matter decomposition pathways. It is recommended that pre-impact studies should be carried out in locations chosen for fish farming to gain an understanding of these processes prior to the additional organic enrichment. This would allow changes in biogeochemical processes to be identified and limits of acceptable sediment modification to be chosen based on sound data.

Additional parameters used for the monitoring of fish farm impacts in the reviewed studies were trace metals and stable isotopes. The significance of trace metal input to the sediments depends on the metal content in the fish feed so more information on fish feed potentially used in the Firth of Thames is required before making an assessment on the importance of trace metal measurements. Sediment trace metal contents measured in the reviewed study were generally low and based on the high mud content in the Firth of Thames. It is possible that effects would be even less severe in this area. Stable isotope measurements could be a useful tool to detect the spatial extent of benthic impacts once fish farms are operating.

7.3 Spatial extent of benthic effects

Estimates of the spatial extent of expected benthic impacts were derived from a review of peer-reviewed literature and monitoring data from New Zealand fish farms and it was concluded that 100 m was a conservative estimate. Since the largest change of most examined parameters took place within about 50 m of the farm and the gap between farm blocks is 75 m a 50 m buffer zone between the outermost cages inside a farm block and the perimeter of the block was considered an adequate estimate of the buffer zone for initial applications before measurements are available to make realistic assessments of spatial effects. The major spatial distribution of benthic impacts occurs in the direction of the prevailing current flow and, therefore, it may be acceptable to allow smaller buffer zones along the long sides of farm blocks, allowing fish cages to be installed in Area A where blocks are only 110 m wide. It is important to note that the spatial extent of specific fish cages will depend on a complex network of processes and environmental influences that cannot be predicted from literature data alone. Especially if large (>15 m diameter) cages are to be installed in the Firth of Thames it is strongly recommended that benthic impacts should be measured at high spatial and temporal resolution until sufficient information on their severity and spatial extent has been gathered to make sound recommendations on minimum buffer zones for farm blocks. To enable a reliable detection of farm footprints, it is also recommended that the natural variability of parameters used for future monitoring is measured prior to any farming activity. This will enable the identification of changes caused by the farms and minimise the problem of separating natural from farm induced changes observed in the farm area.

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9. Appendix

Table 1. Predicted probability distributions of all benthic impact variables in Bayesian network calculated for the case studies. Cases are defined by free water depth below cages (case 1 = 5 m, case 2 = 10 m, case 3 = 15 m) and stocking density (case 1 = 10 kg m^{-3}, case 2 = 15 kg m^{-3}, case 3 = 25 kg m^{-3}).

	5m10	5m15	5m25	10m10	10m15	10m25	15m10	15m15	15m25
Sediment Enrichment									
low	0.22	0.20	0.18	0.23	0.22	0.20	0.27	0.26	0.23
moderate	0.30	0.27	0.25	0.37	0.34	0.32	0.39	0.36	0.34
high	0.25	0.25	0.24	0.29	0.30	0.28	0.25	0.26	0.25
very high	0.24	0.29	0.34	0.12	0.15	0.21	0.10	0.13	0.19
Acid volatile sulphide S (mg g^{-1})									
<0.5	0.37	0.34	0.32	0.41	0.39	0.36	0.45	0.43	0.40
0.5-1	0.29	0.29	0.30	0.30	0.30	0.30	0.29	0.29	0.30
1-1.5	0.20	0.21	0.23	0.18	0.19	0.20	0.17	0.17	0.19
>1.5	0.13	0.15	0.16	0.11	0.12	0.13	0.09	0.10	0.12
Beggiatoa spp. mats									
yes	0.38	0.43	0.47	0.28	0.32	0.36	0.24	0.28	0.33
no	0.62	0.57	0.53	0.72	0.68	0.64	0.76	0.72	0.67
Denitrification (mmol m^{-2} d^{-1})									
0	0.58	0.60	0.62	0.54	0.55	0.57	0.53	0.54	0.56
0-0.05	0.25	0.24	0.23	0.28	0.27	0.26	0.28	0.28	0.26
0.05-0.1	0.14	0.14	0.13	0.16	0.16	0.15	0.17	0.16	0.15
>0.1	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02

Table 1. (continued)

	5m10	5m15	5m25	10m10	10m15	10m25	15m10	15m15	15m25
Macrofauna biomass (g m^{-2})									
<50	0.52	0.53	0.54	0.49	0.50	0.52	0.48	0.49	0.50
50-100	0.27	0.27	0.27	0.27	0.27	0.27	0.27	0.27	0.27
100-150	0.10	0.10	0.09	0.11	0.11	0.10	0.12	0.11	0.11
150-200	0.07	0.07	0.06	0.08	0.07	0.07	0.08	0.08	0.07
>200	0.04	0.04	0.04	0.05	0.05	0.04	0.05	0.05	0.05
Nitrification (mmol m^{-2} d^{-1})									
<0.2	0.55	0.58	0.60	0.49	0.51	0.54	0.48	0.50	0.52
0.2-0.4	0.25	0.24	0.22	0.26	0.26	0.25	0.27	0.27	0.25
>0.4	0.20	0.19	0.17	0.24	0.23	0.22	0.25	0.23	0.22
pH									
<7	0.18	0.19	0.21	0.15	0.16	0.18	0.14	0.15	0.17
7-7.5	0.29	0.29	0.30	0.28	0.28	0.29	0.27	0.27	0.28
7.5-8	0.46	0.44	0.43	0.49	0.48	0.46	0.50	0.49	0.47
>8	0.07	0.07	0.06	0.08	0.08	0.07	0.09	0.09	0.08
Porewater sulphides (mmol L^{-1})									
<0.3	0.45	0.42	0.40	0.50	0.48	0.45	0.53	0.51	0.48
0.3-1.3	0.30	0.29	0.29	0.31	0.31	0.30	0.30	0.30	0.30
1.3-6	0.19	0.20	0.22	0.16	0.17	0.18	0.14	0.15	0.17
>6	0.07	0.08	0.09	0.04	0.04	0.06	0.03	0.04	0.05

Table 1. (continued)

	5mS10	5mS15	5mS25	10mS10	10mS15	10mS25	15mS10	15mS15	15mS25
Redox potential (mV)									
<-100	0.30	0.33	0.35	0.24	0.26	0.29	0.22	0.24	0.27
-100-0	0.22	0.22	0.22	0.23	0.23	0.23	0.22	0.22	0.22
0-100	0.19	0.18	0.17	0.21	0.20	0.19	0.21	0.21	0.20
100-200	0.15	0.14	0.13	0.16	0.16	0.15	0.17	0.17	0.16
200-300	0.10	0.09	0.09	0.11	0.11	0.10	0.12	0.12	0.11
>300	0.05	0.04	0.04	0.05	0.05	0.05	0.06	0.06	0.05
Sediment oxygen consumption (mmol m^{-2} d^{-1})									
<50	0.37	0.33	0.31	0.41	0.39	0.36	0.45	0.43	0.40
50-100	0.23	0.22	0.21	0.26	0.26	0.24	0.26	0.26	0.24
100-200	0.16	0.17	0.18	0.16	0.17	0.17	0.15	0.15	0.16
200-400	0.13	0.15	0.17	0.09	0.11	0.13	0.08	0.09	0.11
>400	0.10	0.12	0.14	0.07	0.08	0.10	0.06	0.07	0.09
Sediment–water ammonium flux (mmol m^{-2} d^{-1})									
<25	0.46	0.43	0.41	0.50	0.48	0.46	0.53	0.51	0.48
25-50	0.28	0.28	0.28	0.29	0.29	0.29	0.28	0.28	0.28
50-75	0.14	0.15	0.16	0.12	0.13	0.14	0.11	0.12	0.13
75-100	0.08	0.09	0.10	0.06	0.07	0.08	0.06	0.06	0.07
>100	0.03	0.04	0.04	0.03	0.03	0.03	0.02	0.03	0.03

Table 1. (continued)

	5mS10	5mS15	5mS25	10mS10	10mS15	10mS25	15mS10	15mS15	15mS25
Shannon–Wiener diversity index									
<1	0.24	0.27	0.29	0.19	0.21	0.23	0.17	0.19	0.22
1-2	0.20	0.21	0.21	0.20	0.20	0.21	0.19	0.19	0.20
2-3	0.24	0.23	0.23	0.26	0.26	0.25	0.27	0.26	0.25
3-4	0.18	0.17	0.16	0.21	0.20	0.19	0.22	0.21	0.19
>4	0.13	0.12	0.11	0.14	0.14	0.13	0.15	0.15	0.14
Sulphate reduction rate (mmol m^{-2} d^{-1})									
<20	0.37	0.35	0.32	0.41	0.39	0.37	0.45	0.43	0.40
20-60	0.47	0.48	0.49	0.47	0.48	0.48	0.45	0.46	0.47
60-100	0.11	0.13	0.14	0.09	0.09	0.11	0.07	0.08	0.10
>100	0.04	0.05	0.05	0.03	0.03	0.04	0.03	0.03	0.04
Water content (%)									
<30	0.12	0.11	0.10	0.13	0.12	0.11	0.15	0.14	0.13
30-50	0.24	0.22	0.20	0.28	0.26	0.24	0.29	0.28	0.26
50-70	0.39	0.38	0.37	0.43	0.42	0.41	0.41	0.41	0.40
>70	0.25	0.29	0.33	0.17	0.19	0.24	0.14	0.17	0.21