Available online at www.sciencedirect.com

ScienceDirect

Marine Pollution Bulletin 52 (2006) 1458–1466

MARINE POLLUTION BULLETIN

www.elsevier.com/locate/marpolbul

Evaluation of short-term fallowing as a strategy for the management of recurring organic enrichment under salmon cages

Catriona K. Macleod [a,b,*], Natalie A. Moltschaniwskyj [b], Christine M. Crawford [a]

[a] *Marine Research Laboratories, Tasmanian Aquaculture and Fisheries Institute, and Aquafin CRC, University of Tasmania, Hobart, Tasmania 7001, Australia*

[b] *School of Aquaculture, Tasmanian Aquaculture and Fisheries Institute, University of Tasmania, Launceston, Tasmania 7250, Australia*

Abstract

Rotation of cages within fish farm leases and the subsequent fallowing of areas of seabed is commonly used to allow recovery of infaunal communities following periods of organic enrichment. To investigate the effect of different background environmental conditions on recovery response, two Atlantic salmon (*Salmo salar*) fish farm sites in southeast Tasmania were sampled over two commercial fallowing cycles. Despite similar stocking levels and feed input there were significant differences in the way in which sediment at each farm responded to the cessation of fish stocking. Sediments at both farms showed some improvement in the community structure over a three month fallow period, but the community structure only recovered to that present before stocking not to that at the reference sites. The similarity of the impact sites to the reference sites increased from ca. 25% to 31% at one site and 11% to 27% at the other after fallowing. Rate and extent of recovery were affected by farm location, initial impact of the sediments, and length of fallow period. Initial recovery was faster at the more sheltered site than at the more exposed site, possibly reflecting differences in environmental resilience with the more sheltered location better able to assimilate organic inputs. Accordingly general fallowing management protocols may need to be adapted to reflect differences between sites. The findings of this study suggest that the recovery response of benthic communities can be predicted once baseline conditions are understood.

Keywords: Tasmania; Organic enrichment; Recovery; Aquaculture; Benthic infauna; Fallowing

1. Introduction

Faecal waste and uneaten feed from commercial finfish cage aquaculture results in organic enrichment of the underlying sediments (Black, 2001). To alleviate impacts on the sediments and to give the sediment an opportunity to recover, fish-holding cages are often removed or are left fallow for a period of time. Although environmental monitoring of the sediments is mandatory, there is no legislative requirement for fallowing in Tasmania, environmental regulations only require that there be no "unacceptable impact" associated with farming practices (Woods et al., 2004). Therefore the duration of any fallow period is largely at the discretion of the farm manager. The amount of time actually required for sediment remediation is poorly understood at present. Three months of fallowing is generally considered to be a reasonable timeframe and is regularly used.

Many studies have examined the temporal and spatial effects of organic enrichment from cage aquaculture on the benthic community structure. It takes a relatively long time for the biota to recover fully (Karakassis et al., 1999; Pohle et al., 2001; Brooks et al., 2004; Pereira et al., 2004). Nonetheless, it is evident that, in all but the very worst

* Corresponding author. Address: Marine Research Laboratories, Tasmanian Aquaculture and Fisheries Institute, and Aquafin CRC, University of Tasmania, Hobart, Tasmania 7001, Australia. Tel.: +61 3 62277237; fax: +61 3 62278035.

E-mail address: Catriona.Macleod@utas.edu.au (C.K. Macleod).

0025-326X/$ - see front matter © 2006 Elsevier Ltd. All rights reserved.
doi:10.1016/j.marpolbul.2006.05.007

cases, recovery of the sediments commences fairly quickly once farming has ceased (i.e. within a number of weeks). In one of the few studies undertaken in Australia on sediment remediation after cage fish farming, the benthic ecology of sediments at a relatively exposed Tasmanian site recovered to reference conditions after only seven weeks (Ritz et al., 1989). However, in contrast, and consistent with the majority of the literature, the benthic faunal community structure under cages at a relatively sheltered location (North West Bay, Tasmania) continued to differ from reference conditions 36 months after the cessation of farming (Macleod et al., 2004). The difference between the rates of recovery observed in the two Tasmanian studies was attributed partly to differences in background environmental conditions and partly to the expansion and intensification of the salmon farming industry since the initial study (Macleod et al., 2004).

In both Tasmanian studies, and in the majority of prior investigations, recovery of the sediments was gauged against a return to reference conditions. However, for fish farming operations to be sustainable it is not necessary for sediment condition to return to a reference state. Recovery to the extent that it does not result in progressive chemical or biological deterioration of sediments may be sufficient to support long-term farming operations. In this regard there have been no studies that have investigated appropriate fallowing regimes for such practical management of environmental condition.

Most studies of sediment recovery associated with fish farming have been restricted to single sites/leases within similar geographic areas (Ritz et al., 1989; Karakassis et al., 1999; Pereira et al., 2004), and there are very few aquaculture-based studies that specifically compare large-scale spatial variability in the sediment recovery response. One exception compared several farm sites and found significant differences in their biological recovery rate (Brooks et al., 2003, 2004). Recovery at the site in the later study (Brooks et al., 2004) was markedly slower, with the suggestion that it could take more than six years for biological recovery. Although the authors did not specifically examine why this was the case, they suggested that it may be linked to environmental differences.

Local environmental conditions can have a major influence on the rate at which sediments recover from organic enrichment (Black, 2001). Both physical and biological conditions will affect the rate of recovery response (Boesch and Rosenberg, 1981; Beveridge, 1987). Communities of highly stressed and physically variable environments may be less complex, but can recover more quickly from a disturbance than those of more benign and less variable areas (Bolam and Rees, 2003). Several studies have shown a direct relationship between the chemical condition of the sediment and the biological response (e.g. Holmer and Kristensen, 1992; Hargrave et al., 1997; Wildish et al., 2001). Measurement of redox potential and sediment sulfide concentration have been recommended as potentially useful, cost-effective approaches for assessing sediment degradation (Hargrave et al., 1997; Wildish et al., 2001; Crawford et al., 2002; Edgar et al., 2005). However, the value of these approaches in relation to sediment recovery has not yet been clearly established.

The aims of this study were to assess the extent to which sediments recover under the normal production protocols employed on a commercial salmon farm in Tasmania, and to determine the variation in recovery rates between sites with differing environmental characteristics. This was done by evaluating the environmental and farm management factors which may influence recovery response. In addition, the value of established chemical condition measures of degradation (redox potential and sediment sulfide concentration) was examined to ascertain the applicability of these measures to the assessment of sediment recovery.

2. Methods

2.1. Selection of sampling locations

Two farm locations in south eastern Tasmania, Creeses Mistake (Tasman Peninsula) and Stringers Cove (Port Esperance) were sampled in this study (Fig. 1). These farms are broadly representative of the differing environments in which Atlantic salmon culture is undertaken in southwest Tasmania. Creeses Mistake is a relatively exposed, shallow (20 m) and fully marine site whereas Stringers Cove is in deeper (40 m) more sheltered waters that are occasionally subject to the freshwater influence of the nearby Esperance River.

Sediment recovery associated with standard farm production protocols was studied over two annual production cycles. Both farms employed an annual stocking regime where cages were stocked for nine months and then fallowed for three months. At each farm the study cages were circular with a circumference of 120 m. Sediment samples were collected from cage positions and references prior to the cages being stocked (TX), at the end of nine months of stocking (i.e. at the end of the stocked phase/start of fallow period—T0), and at the end of a three month fallow period (T3). In addition, during the second year samples

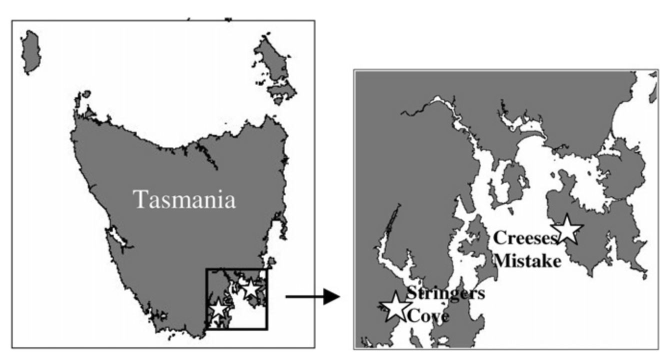

Fig. 1. Location of study farms in south eastern Tasmania. Creeses Mistake is located in Wedge Bay on the Tasman Peninsula and Stringers Cove is within Port Esperance.

were collected from both cage and reference positions at monthly intervals during the fallow period.

During the second annual production cycle, cages were restocked at Creeses Mistake in exactly the same position as in the previous cycle (positions 5 and 8, Fig. 2) however at Stringers Cove the cages were restocked adjacent to the positions used in the first cycle (i.e. cages were at positions 1 and 2 in first production cycle and at positions 1A and 2A in second cycle, Fig. 2). The Creeses Mistake cage positions had previously been farmed whereas cage positions 1, 2, 1A and 2A at Stringers Cove had not. Consequently, previously farmed cage positions 3A and 4A at Stringers Cove (Fig. 2) were analysed as replicates in the second annual farming cycle but not the first annual cycle.

Prior to sampling, each farm area was mapped using a Garmin 135 GPS Map unit coupled with a Racal differential unit. Depth and positional information were collected for all cages present on the lease at the time. In addition, reference locations, within the same depth range, but 150 m distant from the edge of selected study cages, were located using the depth contours and GPS.

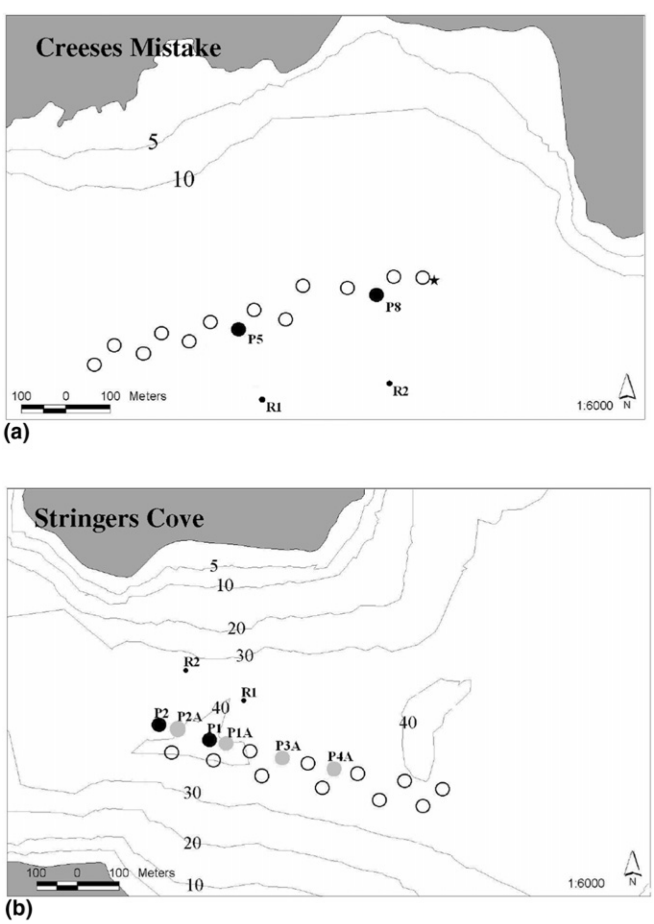

Fig. 2. Cage positions and reference sites for sediment samples for (a) Creeses Mistake and (b) Stringers Cove. Stringers Cove sample sites shown as ● were stocked in the 1st production cycle whilst those shown as ● (grey) were stocked in the 2nd cycle. Sites shown as ○ indicate the positions of other cages in the first production cycle.

2.1.1. Faunal sampling

Five replicate samples were collected from each cage position and reference using a Van Veen Grab (surface area—0.0675 m^2). Grab contents were transferred to mesh bags (mesh size 0.875 mm) and rinsed. Samples were then wet sieved to 1 mm and the retained material preserved in a solution of 10% formalin:seawater (4% formaldehyde). Samples were transferred to the laboratory for sorting and the infauna identified to the lowest possible taxonomic level and enumerated.

2.1.2. Redox & sulfide assessment

Three replicate cores (perspex tubes 250 mm length × 45 mm internal diameter) were taken at each cage position and reference site for measurement of redox potential and sulfide concentration. Redox and sulfide were measured at 3 cm depth using a WTW Redox Probe and a Cole-Parmer 27502-40 silver/sulfide electrode respectively. Sulfide was sampled according to the method described by Wildish et al. (1999), with 2 ml of anti-oxidant buffer added to a 2 ml sediment sample prior to measurement.

2.1.3. Statistical analysis

As the aim of the present study was to evaluate the rate and extent of recovery associated with farm fallowing three levels of recovery were considered:

Level 1 recovery—improvement in sediment condition (i.e. biologically and chemically).
Level 2 recovery—return to pre-stocking sediment condition.
Level 3 recovery—return to reference sediment condition.

Abundance data were square root transformed to reduce the influence of abundant taxa and the Bray–Curtis similarity index was used because of the robustness of this statistic to zero-inflated data sets (Clarke, 1993). Replicate samples were used to generate a mean value for each cage and reference site.

An ordination plot using non-metric multidimensional scaling (nMDS) was used to identify differences in the community structure between reference sites and cages at each farm over the fallow period. The significance of differences between conditions prior to stocking, at the reference positions, and at the end of the fallow period was assessed using the ANOSIM randomisation test provided by the Plymouth Routines in Multivariate Ecological Research (PRIMER) software package (PRIMER, 2006).

Initial impact levels were assessed using the Bray–Curtis similarity of the full species abundance (square root transformed) data set between cage positions and associated references immediately prior to fallowing, i.e. cages that had a community structure with a high similarity to their respective reference site were considered to be less impacted that cages that had a low similarity to their respective reference site. The Bray–Curtis similarity matrix was also used to determine the relationships between farm location, farming practices and the period of time that the site had been fallowed. This analysis was used to examine absolute changes over time and the full species dataset was used to allow detection of subtle compositional changes in community structure during the relatively short fallow period. The Bray–Curtis similarities between the start (T0) and end (T3) of fallowing, between the end of fallowing (T3) and pre-stocking (TX), and between recovered condition (T3) and reference condition at the end of fallow period (R3) were used as relative measures of level 1, 2 and 3 recovery, respectively.

The rate of change in the community structure at each cage during the fallow period was determined from the gradient of the regression line generated from changes in the monthly Bray–Curtis similarity over the fallowing period. Univariate statistical analysis of similarity measures was used to examine differences in recovery over time. Student *t*-tests were used to compare the average rate and magnitude of change in the community structure between the farm locations. Regression analysis was used to examine the rate of change at the references and cages within farms over the fallow period as a function of initial impact. The association between initial impact, farming factors (feed input and number of adjacent cages), and rate and level of change were examined using Pearson correlation coefficients.

Comparisons of the average redox potential and sulfide concentration between cage and reference sites (across both farms) were made using Student *t*-tests.

3. Results

At both farm locations the community structure of the cage positions changed during the fallow period in a manner that reflected recovery from organic enrichment. Over the three month fallow period (i.e. between T0 and T3) the community structure at all cage positions changed significantly (ANOSIM, Rho = 0.25, $p = 0.016$), with cage positions at both farms recovering to pre-stocking condition (ANOSIM, Stringers—Rho = 0.100, $p = 0.185$, Creeses—Rho = 0.41, $p = 0.114$) (Fig. 3). However, neither farm location recovered to reference conditions (ANOSIM, Stringers—Rho = 0.92, $p < 0.00$, Creeses—Rho = 0.80, $p < 0.001$) (Fig. 3).

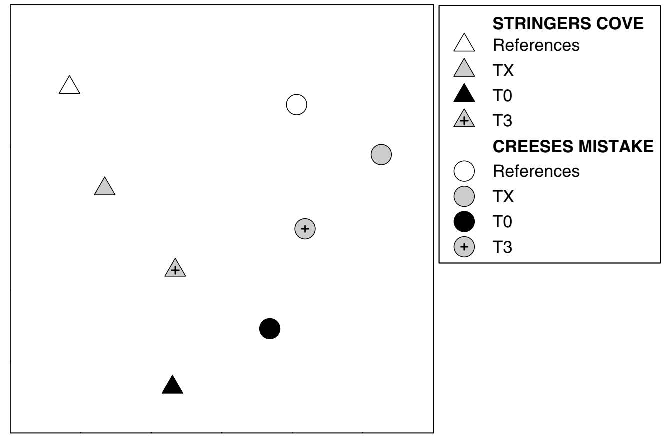

Fig. 3. Non-metric multidimensional scaling plot of community similarities at the two farms. Stress = 0.03. (TX—pre-stocking, T0—start of fallow period, T3—end of three month fallow period.)

The reference communities were very different between farm locations (ANOSIM, Rho = 1.000, $p < 0.001$), but within farms the reference communities were similar both spatially and temporally (Table 1). Regression analysis of the monthly Bray–Curtis similarities for the reference positions over the fallowing period shows no evidence of any significant variation in the reference community structure at either farm location over the fallow period (Creeses Mistake: $r^2 = 0.64$, $F = 1.81$, df = 1,2, $p = 0.407$; Stingers Cove: $r^2 = 0.78$, $F = 3.59$, df = 1,2, $p = 0.309$). At Stringers Cove the changes in the cage communities over the fallow period (T0–T3) were greater than in the reference communities ($t = -4.56$, df = 6, $p = 0.004$), i.e. similarity

Table 1
Average similarity (±SE) (using Bray–Curtis similarity index) (a) between cage positions and equivalent references at the start of the fallow period (T0–R0), (b) between the cage positions at the start and end of the fallow period (T0–T3), (c) rate of change as the gradient of change in similarity determined from the change in similarity between the cages each month over the fallow period, (d) average daily feed input (±SE) and (e) average number of adjacent stocked cages (±SE) over period that the cage was stocked

Position	Similarity between T0 and R0	Similarity between T0 and T3	Rate of change*	Average feed input (kg/day)	Average no of adjacent cages
Stringers					
PC1-Cages	8.50(±1.24)	15.45(±7.05)	11.85(±2.78)	519(±20.0)	1.41(±0.39)
PC2-Cages	13.74(±2.60)	29.80(±6.00)	5.84(±5.05)	324(±11.3)	1.49(±0.24)
PC1-Refs		63.17(±1.00)			
PC2-Refs		68.24(±0.01)	0.54(±0.97)		
Creeses					
PC1-Cages	18.31(±3.33)	49.63(±4.78)		491(±36.0)	1.75(±0.09)
PC2-Cages	31.29(±3.35)	31.16(±3.07)	−5.75(±6.01)	304(±52.5)	1.69(±0.03)
PC1-Refs		59.03(±0.35)			
PC2-Refs		50.77(±0.89)	2.62(±8.46)		

Monthly data was unavailable from Creeses Mistake or references in first production cycle. (Note larger numbers indicate greater similarity levels and therefore less change.)

levels were significantly lower. At Creeses Mistake the overall change at the cage sites was less conspicuous and was within the range of the reference communities over the same period ($t = -1.19$, df = 4, $p = 0.301$) (Table 1).

Overall the two farms were similar in the degree to which they differed from the references at the end of the fallow period (T3–R3) ($t = 0.65$, df = 3.5, $p = 0.554$) and in the extent of the change in the community structure that occurred over the fallow period (T0–T3) ($t = 1.93$, df = 8, $p = 0.089$) (Fig. 4). However, the average rate of change differed between the two sites ($t = -2.33$, df = 7.9, $p = 0.048$). Whilst at both locations the community structure showed a level of recovery, at Stringers Cove the rate of change over the three months fallow period was positive whilst at Creeses Mistake it was negative. A positive rate indicates that the community became more similar with each subsequent month, whereas a negative rate indicates that the similarity levels decreased in each subsequent month. For this to occur, and still be indicative of recovery, it is clear that there must be differences in the components of the community affected, i.e. a positive response suggests that the species representing unimpacted conditions are increasing whilst a negative rate suggests that the species indicative of the impacted community are decreasing. The greater the magnitude of the rate the faster the recovery in the community structure occurred.

The community structures and recovery response differed markedly between farm locations (Fig. 3). The community structure was most similar between farms when the impact levels were greatest (i.e. T0), but even at this stage there were still significant differences between the farms (ANOSIM, Rho = 0.83, $p = 0.006$), and these differences persisted over the fallow period (ANOSIM, Rho = 0.75, $p < 0.001$). Initial impact levels at each of the farm locations were markedly different (i.e. community structures prior to fallowing were different) (ANOSIM, Rho = 0.47, $p = 0.003$). However, the farms were similar in the extent to which the initial impact differed from the respective reference communities (T0–R0) ($t = 2.74$, df = 1, 7, $p = 0.05$) (Fig. 4). Initial impact affected the rate of change ($r^2 = 0.58$, $F = 8.25$, df = 1, 7, $p = 0.028$). When the initial impact was greater (i.e. Bray–Curtis similarity between T0 and R0 was small) the rate of change over the fallow period was faster (Fig. 5).

The magnitude of the initial impact also affected the magnitude of the change in the community structure over the fallow period (i.e. Bray–Curtis similarity between T0 and T3) (Fig. 6). In this case the data suggest that there may be difference in the response at Creeses Mistake from that at Stringers Cove. There was a strong linear relationship between initial impact and extent of recovery at Stringers Cove ($r^2 = 0.84$, $F = 27.33$, df = 1, 5, $p = 0.006$).

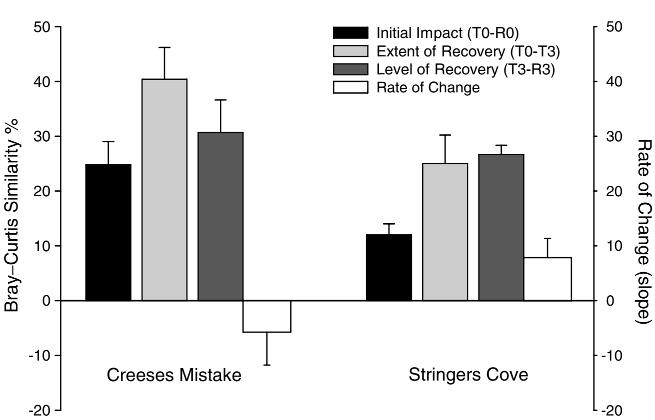

Fig. 4. Mean Bray–Curtis similarity (± SE) for initial impact (T0–R0), extent of impact (T0–T3), level of recovery (T3–R3), and the mean rate of change (±SE) at Creeses Mistake and Stringers Cove.

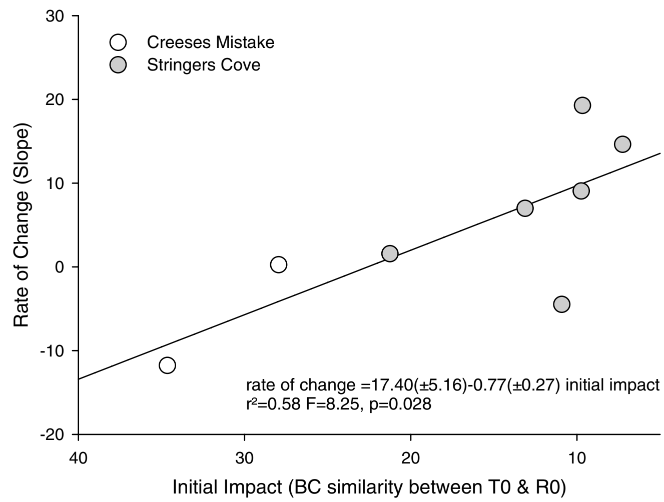

Fig. 5. Initial impact versus rate of change over fallow period (slope of regression for monthly similarity levels). Note that low initial impact levels correspond to high similarity levels (T0–R0) whilst high initial impacts are consistent with low similarity levels (T0–R0). Values in brackets are SE's of regression parameters.

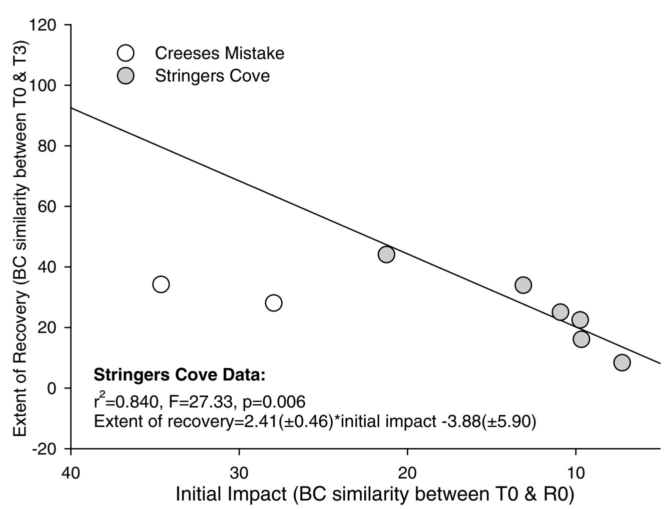

Fig. 6. Initial impact versus magnitude/extent of recovery over fallow period (BC similarity between T0 and T3). Regression line and equations shown are for Stringers Cove data only. Note that low initial impact levels correspond to high similarity levels (T0–R0) whilst high initial impacts are consistent with low similarity levels (T0–R0). Values in brackets are SE's of regression parameters.

When the initial impact was greatest the extent of change in community structure over the fallow period was also greater. The cage positions which had the highest initial impact (i.e. differed most from the references prior to fallowing (T0–R0)) recovered most over the fallow period (i.e. had the lowest similarities (T0–T3)), whilst the least change occurred at cage positions with the lowest initial impact (Fig. 6). There were insufficient data to establish such a relationship at Creeses Mistake.

The final recovered condition at the end of the three month fallow period (T3–R3) was also strongly related to the initial impact level ($r^2 = 0.70$, $F = 13.72$, df $= 1,2$, $p = 0.01$). As the initial impact level increased the similarity between the final community structure and the reference community decreased (Fig. 7). This relationship was exponential and therefore when the initial impact levels were high (i.e. <20% similarity) there was a much greater difference between the final community structure and the reference community than at low initial impact levels (Fig. 7). Overall sediments at Stringers Cove had higher impact levels than at Creeses Mistake.

Farming practices can have a major influence on impact level and recovery response. Feed input was relatively consistent among cage positions farmed in the same production cycle, but differed between cycles (Table 1). In the second production cycle there was a marked reduction in the feed input at both farms. However, there was no evidence of any relationship between feed input and rate of recovery ($r^2 = 0.33$, $F = 3.00$, df $= 1,6$, $p = 0.134$), extent of recovery ($r^2 = 0.40$, $F = 4.02$, df $= 1,6$, $p = 0.092$) or initial impact ($r^2 = 0.37$, $F = 3.556$, df $= 1,6$, $p = 0.108$). All of the cage positions were subject to the additional impact of at least one adjacent cage throughout the stocked phase with several cages having two adjacent cages operational during the stocked phase (Table 1). However, there was also no evidence to suggest that adjacent cages affected the initial impact levels ($r^2 = 0.02$, $F = 0.124$, df $= 1,6$, $p = 0.736$).

Interestingly, there was a marked difference in the level of recovery between the different production cycles, with the greatest change occurring after the second production cycle (Fig. 8). On the whole, the community structure after

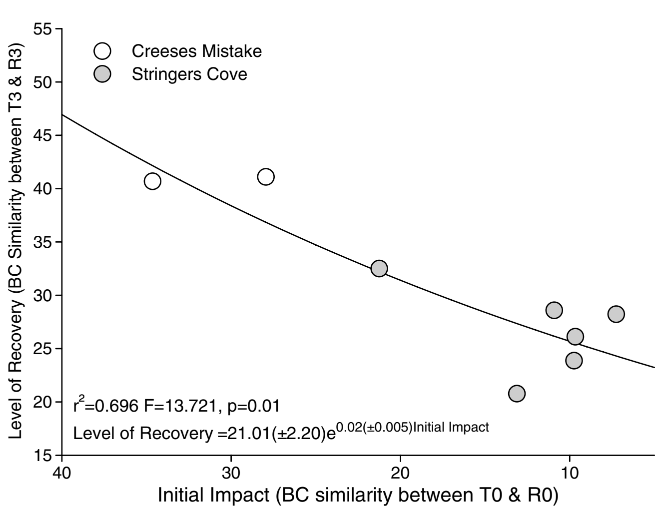

Fig. 7. Initial impact versus recovered status at end of fallow period as defined by the similarity of final community to reference conditions. Note that low initial impact levels correspond to high similarity levels (T0–R0) whilst high initial impacts are consistent with low similarity levels (T0–R0). Values in brackets are SE's of regression parameters.

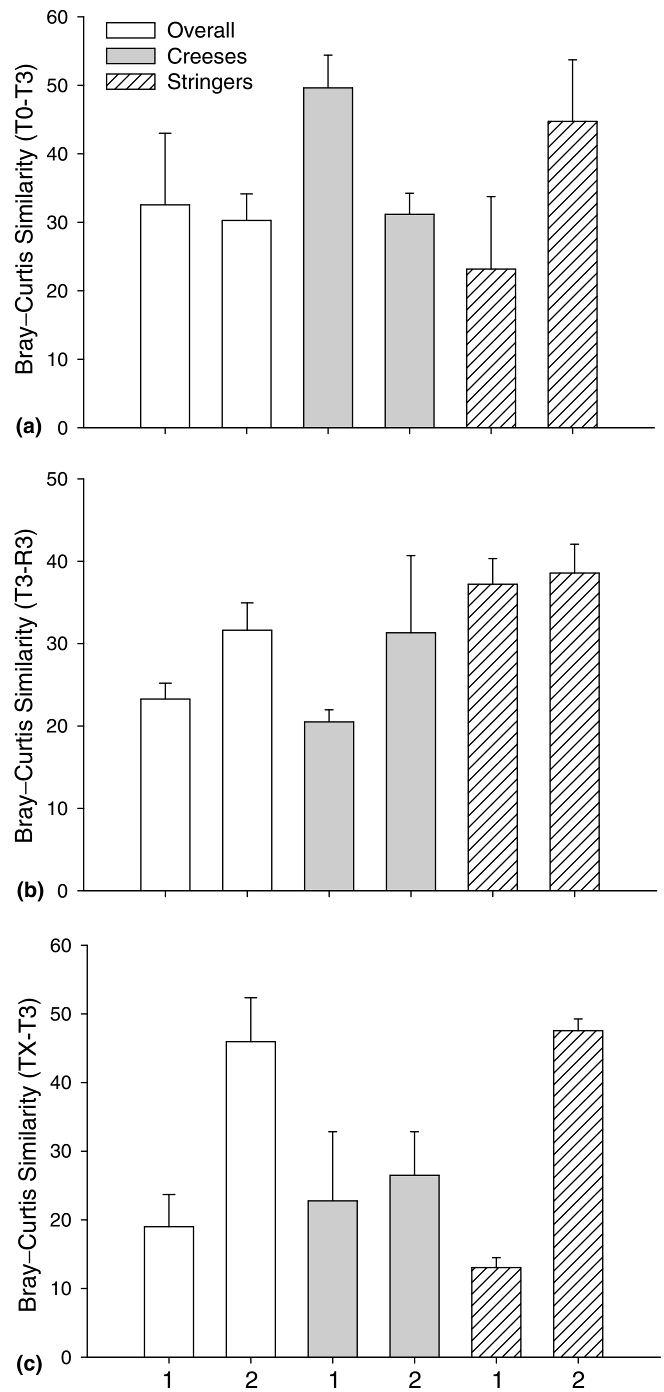

Fig. 8. Mean Bray–Curtis similarity level (±SE) in each production cycle (1 or 2) between (a) start and end of fallow period (T0–T3), (b) end of fallow period and equivalent reference (T3–R3) and (c) pre-stocking and at the end of fallow period (TX–T3) for the cage communities at Creeses Mistake and at Stringers Cove.

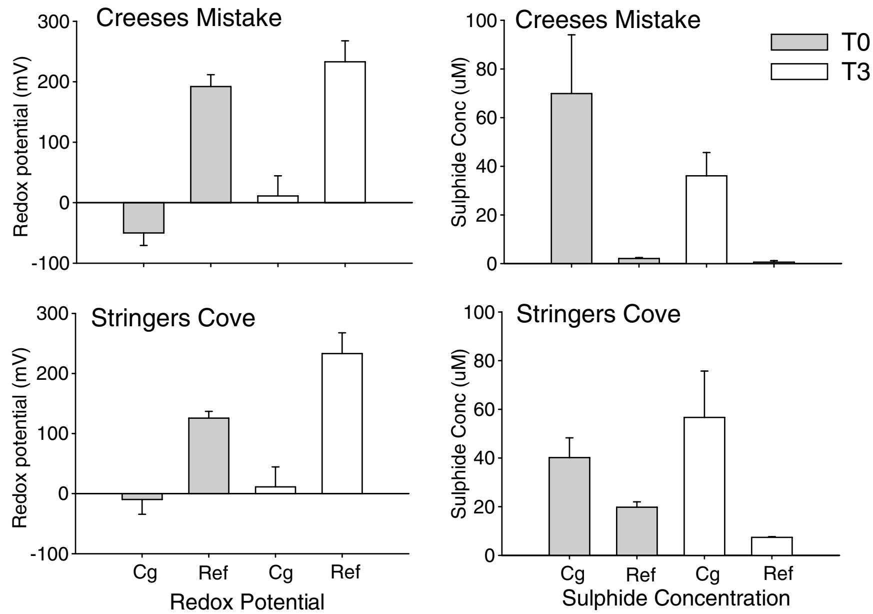

Fig. 9. Mean redox potential (±SE) and sulfide concentration (±SE) at cages and references at Creeses Mistake and at Stringers Cove at the start (T0) and end (T3) of the fallow period.

the second production cycle returned to a state more closely resembling the pre-stocking conditions (TX–T3) ($t = -3.39$, df = 8, $p = 0.009$). Although there was no significant difference in the extent of the recovery (T0–T3) between production cycles at either farm (Creeses—$t = 3.25$, df = 2, $p = 0.083$; Stringers—$t = -1.55$, df = 3, $p = 0.218$) there was a greater difference between the final community and the equivalent reference (T3–R3) at Creeses in the first production cycle than in the second cycle (i.e. less similarity) (Creeses—$t = -13.77$, df = 1, $p = 0.046$; Stringers—$t = -0.29$, df = 3, $p = 0.791$).

Redox potential and sulfide concentration clearly indicated significant differences between the cage and reference sediments both immediately prior to fallowing and at the end of the three month fallow period (Redox T0 – $t = -9.85$, df = 16, $p < 0.001$; T3 – $t = -6.09$, df = 16, $p < 0.001$; Sulfide T0–$t = 2.51$, df = 14, $p = 0.025$; T3 – $t = 3.24$, df = 14, $p = 0.006$) (Fig. 9). Redox potential was consistently lower at the cage positions than at the reference sites whilst sulfide levels were higher at the cage positions. However, over the fallow period there was no significant change in either the sulfide concentration (Creeses—$t = -0.80$, df = 4, $p = 0.470$; Stringers—$t = 1.302$, df = 7, $p = 0.234$) or the redox potential (Creeses—$t = -1.56$, df = 5, $p = 0.179$; Stringers—$t = -1.80$, df = 10, $p = 0.101$) at either farm. Physico-chemical measures of sediment condition were effective in determining impact, but were not sensitive indicators of recovery.

4. Discussion

In this study, three levels of recovery were considered: a measurable improvement in sediment condition, a return to pre-farming condition and a return to reference condition. From the perspective of commercial aquaculture operations, recovery to pre-stocking conditions may be sufficient to sustain ongoing farming. At both farm locations the sediments at the cage positions recovered to pre-stocking levels over the three month fallow period, but they did not return to reference conditions. Despite farming practices, including the stocking levels and feed input, being similar at both farms there were marked differences in the overall recovery response and in the rate of change in the benthic community structure between the farm locations. This implies that the relationship between organic load and sediment recovery is not simple and different locations may need different fallowing strategies even when production protocols are similar. Spatial variability in recovery response has been reported in other studies. Two recent studies of salmon farms in the Broughton Archipelago, Canada showed marked differences in recovery rate. Sites in the 2004 study were estimated to have recovered in <6 months, whereas the sites in a 2003 study were estimated to require >6 years for recovery (Brooks et al., 2003, 2004).

Impact level is one of the primary factors affecting recovery response (Rosenberg et al., 2002). The overall change in the community structure over the fallow period at Stringers Cove was greater than at Creeses Mistake and at the commencement of fallowing, initial impact was greater at Stringers Cove than at Creeses Mistake. These differences in impact level were not related to either feed input or the presence of adjacent cages. Local hydrographic conditions can influence impact levels and recovery rates, mitigating or exacerbating impact and recovery (Black, 2001). It has been suggested that in quiescent areas the impact may be greater and recovery may take much longer than in more hydrodynamically energetic areas (Pearson and Rosenberg, 1978; Holmer, 1991; Black, 2001). Stringers Cove was more sheltered than Creeses Mistake, suggesting that local environmental characteristics may have a major role in determining the initial impact.

At the end of the fallow period, the extent to which the farm sediments differed from the reference conditions was broadly comparable. Consequently, even though the initial impacts differed, with equivalent time periods for recovery, the two locations returned to similar levels. This suggests that contrary to previous findings (Gowen et al., 1988; Holmer, 1991; Black, 2001), recovery was faster at the more sheltered site (Stringers Cove) than at the more exposed site (Creeses Mistake). This may be due to differences in the background ecology and natural resilience of the systems (Boesch and Rosenberg, 1981; Snelgrove and Butman, 1994; Rosenberg et al., 1997). Regional differences in hydrodynamic conditions, such as wave exposure, may also affect the faunal composition (Edgar et al., 2005). Sheltered locations, such as Stringers Cove, tend to be naturally depositional with higher levels of organic material (Hall, 1994). Consequently, the stable state community structure of this system would reflect adaptation to a higher organic load (Boesch and Rosenberg, 1981; Llansó, 1992). This would facilitate recovery from aquaculture impacts in two ways. Firstly, there would be a natural reservoir of species able to colonise the improving sediments early in the recovery phase resulting in a more rapid return to stable state conditions and secondly because the natural conditions are already slightly organically enriched the community would attain the reference condition more quickly (Rosenberg et al., 2002). Furthermore, wave or tidal disturbance may influence sediment stability at the more exposed sites impairing the ability of species to recolonise sediments and slowing recovery dynamics (Thrush et al., 1992). Further investigation of the ecological relationships and function of the communities is needed to clarify the actual mechanisms involved.

At Stringers Cove the rate of change was positive signifying that similarity levels increased over time. Consequently, the greatest change in the community structure occurred in the first month, indicating that the recovery process was initially rapid but stabilized over time. In contrast, the rate of change at Creeses Mistake was negative, which indicates that each month there was a greater difference in the community structure, i.e. the initial recovery response at Creeses Mistake was slow but accelerated after the first month. Information of this kind is important for managing recovery in different systems as it indicates that changing the length of the fallow period will have a significant influence on the recovery response. Shortening the fallow period would be likely to have a greater negative effect at Creeses Mistake where recovery was slow to start with. Consequently, it is important to understand the nature of each individual farming environment in order to manage the length of time required for recovery.

Physico-chemical parameters, such as redox and sulfide, are recommended as useful approaches for monitoring the impacts of fish farming (Hargrave et al., 1997; Wildish et al., 1999; Crawford et al., 2002; Edgar et al., 2005). Regular measurement of redox potential is currently a requirement in both the baseline and ongoing environmental monitoring programs for salmonid farms in Tasmania (Woods et al., 2004). There are clearly measurable changes in sediment chemistry in the period immediately following the cessation of organic inputs (e.g. Eleftheriou et al., 1982; Brown et al., 1987; Weston, 1990; Brooks et al., 2003; Macleod et al., 2004). In a recent study evaluating the broad-scale impacts of fish farming in Tasmania, Edgar et al. (2005), specifically identified redox measured at 40 mm depth as a very sensitive indicator, able to distinguish farming effects from reference conditions. The present study agrees with the findings of Macleod et al. (2004) and suggests that although redox potential and sulfide concentration appeared to be good indicators of deteriorating sediment conditions (especially those associated with major impacts) these measures returned to reference levels very quickly and as such were poor indicators of the biological condition of recovering sediments.

Although recovery rate differed between farm locations, recovery response can be predicted once the baseline environmental characteristics are understood. In order to manage sediment recovery most effectively the differences between farm locations need to be recognized and management measures tailored accordingly. Cage positions within a lease should be managed individually, initial impact levels need to be established for each cage position in order to plan for recovery and initial impact and recovery should be evaluated based on an understanding of the ecology not just in relation to the status of the sediment chemistry or production levels. This study reinforces the findings of Macleod et al. (2004) which concluded that local benchmarks are extremely important to ensure appropriate evaluation of both impact and recovery.

Acknowledgements

This work formed part of a project of the Co-operative Research Centre for Sustainable Aquaculture of Finfish (Aquafin CRC) and received funds from the Australian Government's CRC Program, the Fisheries Research and Development Corporation (FRDC) and other CRC participants. The authors would particularly like to acknowledge the support of the Tasmanian Aquaculture Industry both collectively through the Tasmanian Salmon Growers Association, as well as the individual support of Tassal Ltd. We would like to acknowledge the field and laboratory assistance of Susan Forbes and to thank Dr. Graham Edgar for his valuable comments on the draft manuscript.

References

Beveridge, M.C.M., 1987. Cage Aquaculture. Fishing News Books. Farnham, UK.

Black, K.D., 2001. Environmental Impacts of Aquaculture. Sheffield Academic Press Ltd., UK.

Boesch, D.F., Rosenberg, R., 1981. Response to stress in marine benthic communities. In: Barent, G.W., Rosenberg, R. (Eds.), Stress Effects on Natural Ecosystems. John Wiley & Sons, New York, pp. 179–197.

Bolam, S.G., Rees, H.L., 2003. Minimizing impacts of maintenance dredged material disposal in the coastal environment: a habitat approach. Environmental Management 32, 171–188.

Brooks, K.M., Stierns, A.R., Mahnken, C.V.W., Blackburn, D.B., 2003. Chemical and biological remediation of the benthos near Atlantic salmon farms. Aquaculture 219, 355–377.

Brooks, K.M., Stierns, A.R., Backman, C., 2004. Seven year remediation study at the Carrie Bay Atlantic salmon (*Salmo salar*) farm in the Broughton Archipelago, British Columbia, Canada. Aquaculture 239, 81–123.

Brown, J.R., Gowen, R.J., McLusky, D.S., 1987. The effect of salmon farming on the benthos of a Scottish sea loch. Journal of Experimental Marine Biology and Ecology 109, 39–51.

Clarke, K.R., 1993. Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology 18, 117–143.

Crawford, C.M., Macleod, C.K.A., Mitchell, I.M. 2002. Evaluation of techniques for environmental monitoring of salmon farms in Tasmania. TAFI Technical Report No. 8. January 2002. Tasmanian Aquaculture and Fisheries Institute—University of Tasmania, Hobart, Australia.

Edgar, G.J., Macleod, C.K., Mawbey, R.B., Shields, D., 2005. Broad-scale effects of marine salmonid aquaculture on macrobenthos and the sediment environment in southeastern Tasmania. Journal of Experimental Marine Biology and Ecology 327, 70–90.

Eleftheriou, A., Moore, D.C., Basford, D.J., Robertson, M.R., 1982. Underwater experiments on the effects of sewage sludge on a marine ecosystem. Netherlands Journal of Sea Research 16, 465–473.

Gowen, R.J., Brown, J.R., Bradbury, N.B., McLusky, D.S. 1988. Investigations into benthic enrichment, hyper-nutrification and eutrophication associated with mariculture in Scottish coastal waters (1984–1988). Department of Biological Sciences, University of Stirling, 289 pp.

Hall, S.J., 1994. Physical disturbance and marine benthic communities: life in unconsolidated sediments. Oceanography and Marine Biology: an Annual Review 32, 179–239.

Hargrave, B.T., Phillips, G.A., Doucette, L.I., White, M.J., Milligan, T.G., Wildish, D.J., Cranston, R.E., 1997. Assessing benthic impacts of organic enrichment from marine aquaculture. Water, Air and Soil Pollution 99, 641–650.

Holmer, M. 1991. Impacts of aquaculture on surrounding sediments: generation of organic-rich sediments. In: De Pauw, N. and Joyce, J. (Eds.) Aquaculture and the Environment. European Aquaculture Society Special Publication No. 16, Gent, Belgium, pp. 155–176.

Holmer, M., Kristensen, E., 1992. Impact of marine fish farming on metabolism and sulphate reduction of underlying sediments. Marine Ecology Progress Series 80, 190–201.

Karakassis, I., Hatziyanni, E., Tsapakis, M., Plaiti, W., 1999. Benthic recovery following cessation of fish farming: a series of successes and catastrophes. Marine Ecology Progress Series 184, 205–218.

Llansó, R.J., 1992. Effects of hypoxia on estuarine benthos: The lower Rappahannock River (Chesapeake Bay), a case study. Estuarine, Coastal and Shelf Science 35 (5), 491–515.

Macleod, C.K., Crawford, C.M., Moltschaniwskyj, N.A., 2004. Assessment of long-term change in sediment condition after organic enrichment: defining recovery. Marine Pollution Bulletin 49, 79–88.

Pearson, T.H., Rosenberg, R., 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanography and Marine Biology: an Annual Review 16, 229–311.

Pereira, P.M.F., Black, K.D., McLusky, D.S., Nickell, T.D., 2004. Recovery of sediments after cessation of marine fish farm production. Aquaculture 235, 315–330.

Pohle, G., Frost, B., Findlay, R., 2001. Assessment of regional benthic impact of salmon mariculture within the Letang Inlet, Bay of Fundy. ICES Journal of Marine Science 58, 417–426.

Ritz, D.A., Lewis, M.E., Ma Shen, 1989. Response to organic enrichment of infaunal macrobenthic communities under salmonid seacages. Marine Biology 103, 211–214.

Rosenberg, R., Nilsson, H.C., Hollertz, K., Hellman, B., 1997. Density-dependent migration in an *Amphiura filiformis* (Amphiuridae, Echinodermata) infaunal population. Marine Ecology Progress Series 159, 121–131.

Rosenberg, R., Agrenius, S., Hellman, B., Nilsson, H.C., Norling, K., 2002. Recovery of marine benthic habitats and fauna in a Swedish fjord following improved oxygen conditions. Marine Ecology Progress Series 234, 43–53.

Snelgrove, P.V.R., Butman, C.A., 1994. Animal-sediment relationships revisited: cause versus effect. Oceanography and Marine Biology: an Annual Review 32, 111–177.

Thrush, S.F., Pridmore, R.D., Hewitt, J.E., Cummings, J.V., 1992. Adult infauna as facilitators of colonisation on intertidal sandflats. Journal of Experimental Marine Biology and Ecology 159, 253–265.

Weston, D.P., 1990. Quantitative examination of macrobenthic community changes along an enrichment gradient. Marine Ecology Progress Series 61, 233–244.

Wildish, D.J., Akagi, H.M., Hamilton, N., Hargrave, B.T. 1999. A recommended method for monitoring sediments to detect organic enrichment from mariculture in the Bay of Fundy. Canadian Technical Report of Fisheries and Aquatic Sciences No. 2286. p. 31.

Wildish, D.J., Akagi, H.M., Hamilton, N., 2001. Sedimentary changes at a Bay of Fundy salmon farm associated with site fallowing. Bulletin of the Aquaculture Association of Canada 101 (1), 49–56.

Woods, G., Brain, E., Shepherd, C., Paice, T. 2004. Tasmanian Marine Farming Environmental Monitoring Report: Benthic Monitoring 1997–2002. Marine Environment Section, Department of Primary Industry, Water and Environment, Tasmania.