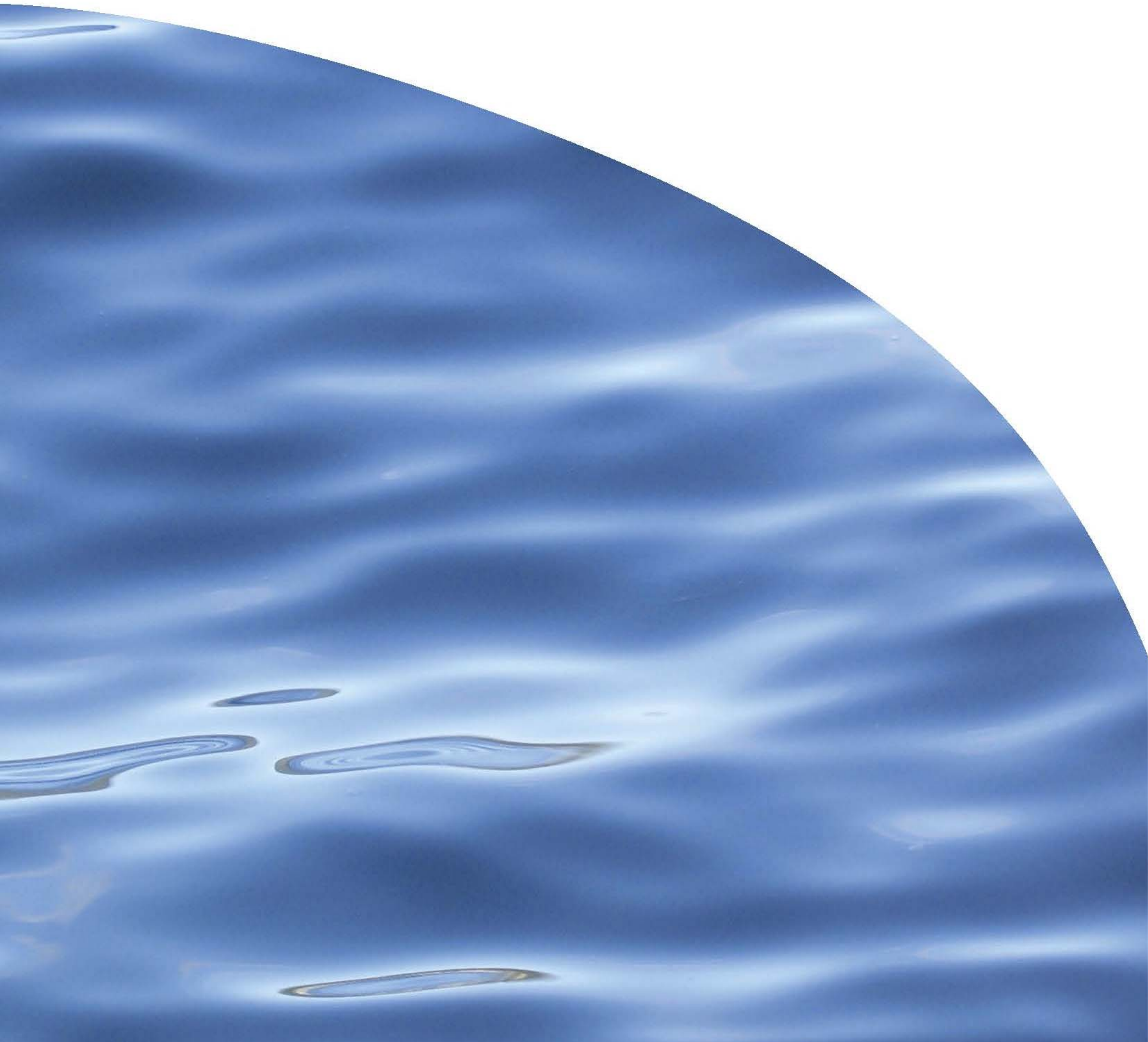
REPORT NO. 2696

SEABED REMEDIATION PILOT STUDY: FINAL REPORT



SEABED REMEDIATION PILOT STUDY: FINAL REPORT

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EXECUTIVE SUMMARY

Cawthron Institute (Cawthron) carried out research into the effectiveness of four treatments aimed at remediating areas of enriched seabed. The treatments spanned a range of methods from natural recovery to the removal of surface organic matter. Field trials took place in 2014 on the highly enriched seabed under the fallowed salmon farm site in Forsyth Bay (Marlborough Sounds).

This was a collaborative multi-agency research project, funded principally by Seafood Innovations Limited (SIL), in combination with (in reducing order of financial contributions): New Zealand King Salmon Company Limited (NZ King Salmon), Cawthron Institute (Cawthron), Ministry for Primary Industries (MPI), Sanfords Limited (Sanfords), United Fisheries Limited (UFL) and Marlborough District Council (MDC).

The primary research objective was to field-test potential treatments to accelerate seabed recovery from a highly-enriched state. However, an important secondary objective was to evaluate the potential for adverse water-column effects as a result of sediment re-suspension during treatments.

The four remediation treatments used were:

- natural recovery ('Natural')
- irrigation with oxygenated water ('Irrigation')
- harrowing (heavy raking of the seabed; 'Harrow')
- surface sediment removal ('Removal')

Sediment removal was a one-off treatment; the Harrow and Irrigation treatments were repeated three times within the first two months. The overall effects of the treatments were compared after four months. The effects were based on detailed biological and physico-chemical analysis of the seabed. Re-suspension was also monitored during each initial treatment by intensively sampling the water column using *in-situ* sensors (D-Opto™ loggers and CTD [conductivity-temperature-depth] casts and by taking physical water samples for nutrients.

Significant sediment plumes with associated reduced dissolved oxygen levels were observed during the Removal and Irrigation treatments, but the magnitude of the changes were negligible in terms of potential for significant ecological effects. However, there may be scale-related effects if the treatments are done over a larger area. For this reason, this result should be treated with some caution.

The Harrow and Irrigation treatments did not accelerate seabed remediation. However, the Removal (of surface sediment) treatment, significantly altered the physical, chemical and biological properties of the sediment, and facilitated recolonisation by infauna. Notably, copper and zinc concentrations were significantly lower inside the Removal treatments,

compared with those in adjacent sediments. This suggests that the careful removal of the top 10–15 cm of seabed sediment, may also remove a significant amount of the trace metals that have accumulated over several years. This activity also reinstated compliance with the relevant sediment quality guidelines (ANZECC 2000a).

These findings suggest that shallow sediment removal has the potential to accelerate seabed recovery and reduce copper and zinc concentrations in sediments. Effective commercial implementation of the Removal treatment remains contingent upon the development of responsible sediment disposal mechanisms. There are also potential scale-related effects that should be further evaluated with larger-scale trials. In addition, the issue of rate of re-impact is important and should also be considered.

A commercial perspective on the results of the project was provided by Mr Mark Gillard (NZ King Salmon) (Appendix 1).

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1. PROJECT OVERVIEW

In 2014, Cawthron Institute (Cawthron) led research to test the effectiveness of four potential treatments to remediation the effects of excessive organic enrichment on the seabed. Experimental treatments ranged from natural recovery to the removal of surface organic matter. Field trials took place on the highly enriched seabed under a fallowed salmon farm site in Forsyth Bay (Marlborough Sounds).

This was a collaborative multi-agency project, funded principally by Seafood Innovations Limited (SIL) in combination with (in reducing order of financial contributions): New Zealand King Salmon Company Limited (NZ King Salmon), Cawthron, Ministry for Primary Industries (MPI), Sanfords Limited (Sanfords), United Fisheries Limited (UFL) and Marlborough District Council (MDC). The results of the research were presented by Dr Nigel Keeley at the 2014 New Zealand Aquaculture Conference, Nelson and have also been prepared as a scientific manuscript that has been submitted to the scientific journal, *Aquaculture*.

The primary objective of this research was to field-test potential treatments to accelerate seabed recovery from a highly-enriched state. However, an important secondary objective was to evaluate the potential for adverse water-column effects as a result of sediment re-suspension during each treatment.

The three stages of the research project were:

Stage 1:
Objective: Obtain necessary resource consent to disturb the seabed in the prescribed manner prior to conducting the experiment.

Stage 2:
Objective: Test the effectiveness of selected seabed remediation treatments in field trials.

Stage 3:
Objective: Recommend a commercial-scale seabed remediation method.

The following sections provide a technical report of the research findings.

2. INTRODUCTION

Salmon farming in non-dispersive environments can result in excessive seabed enrichment. This comes from elevated deposition of organic-rich particles in the form of feed pellets and fish faeces (Gowen & Bradbury 1987; Buschmann *et al.* 2006). In extreme cases, sediments immediately beneath a farm can become anoxic and devoid of macrofauna, release toxic gases (*e.g.* hydrogen sulphide), and reduce overlying water quality (Brooks & Mahnken 2003a). During such events, farmed fish can become stressed, health and growth may be impaired and mortalities can result (Kiemer *et al.* 1995; Black *et al.* 1996). This of course adversely impacts production and industry reputations. Degraded seabed conditions can also breach regulatory standards (Wilson *et al.* 2009) and may result in requiring the farm to be removed or destocked for a prolonged period (commonly termed 'fallowing'). In many regions (*e.g.* New Zealand), salmon farming is space-limited and hence the need to fallow a site can equate to de-stocking, and therefore have a major effect on production.

If a fallowed site is to be restocked then the focus inevitably turns to how soon, and therefore, to anything that can be done to reduce the 'down-time'. Hence, there has been much speculation and informal trials concerning possible ways to enhance the natural recovery process. The basic premise behind accelerating recovery revolves around increasing oxygen penetration into the sediments and carbon assimilation rates by reinstating nitrification and denitrification processes, which are shut down when the sediment becomes anoxic (Bianchi 2007). Several potential treatments exist, including physical (*e.g.* harrowing, re-suspension and removal), biological (*e.g.* the addition of detritivores) and chemical approaches (*e.g.* addition of nitrate to alter redox conditions, favouring decomposition of organic matter by denitrifying bacteria). In a review of their potential effectiveness, Eriksen *et al.* (2011) concluded that these were the most promising treatments.

1. Increasing oxygen penetration by drawing surface water through the sediments.
2. 'Harrowing' the sediments.
3. Carbon capture by intercepting the particles with a subsurface structure before they reach the seabed.

Although the possible solutions are seemingly obvious, there are only a few dedicated studies that have trialled and compared their effectiveness. One widely-cited study tested harrowing *in situ* and concluded that it had the potential to significantly improve sediment quality and with that, farm production (O'Connor *et al.* 1993). However, that study failed to demonstrate that the result was directly attributable to the treatments method. Furthermore, there are other concerns and uncertainties with the approach, which mainly revolve around the potential effects of re-suspension on water quality, including the release of any associated potential contaminants (*e.g.* antifoulants or chemical therapeutants; Brooks & Mahnken 2003b; Burridge *et al.* 2010) and the implications for the wider ecology as well as the caged fish (Eriksen *et al.* 2011). Other less physically disturbing methods have been trialled, such as *in situ* bioremediation using micro-organisms (Vezzulli *et al.* 2004), opportunistic polychaetes (Kunihiro *et al.* 2008) and detritivorous fish (Katz *et al.* 2002). Some of these approaches show clear remediation potential, however, there are costs and practical issues associated with up-scaling these for commercial application.

Further experimental trials were conducted into the effectiveness of harrowing and sediment oxygenation using surface waters (Eriksen *et al.* 2012). That study initially aimed to combine *in situ* pilot-scale field trials with laboratory-based mesocosm experiments. However, the technical difficulties associated with conducting field-trials at depth were demonstrated through a failure to establish and maintain delineated sampling blocks on the seabed. Results from the mesocosm trials, however, demonstrated the potential to increase oxygen penetration to the sediments, reduce the variability in sediment quality, and provided evidence to suggest that harrowing may improve the recovery process over a longer timeframe (Eriksen *et al.* 2012). It was also concluded that the outcome of the study may have been enhanced if conducted in the field, where oxygen exchange would be less constrained.

Irrespective of the implications for farm management, if and when a site is retired, it may be socially and environmentally beneficial to address potential legacy issues by accelerating what is otherwise a relatively long recovery. While the benthic recovery from organic enrichment is substantial within one to two years, reverting to a natural functional state may take five or more years (Brooks *et al.* 2004; Pereira *et al.* 2004; Keeley *et al.* 2013). There are also other farm-derived contaminants associated with the sediments that may have longer recovery times. Copper and zinc are commonly found in elevated concentrations beneath fish farms due to their historical use in antifouling and as a feed supplement (respectively), and their propensity to bind with organically enriched sediments (Batley & Simpson 2009). Concentrations of both can exceed national sediment quality guidelines (*e.g.* ANZECC 2000a), and as they are considered reasonably conservative once bound within the sediments and buried (and therefore unable to be transported), the ability to remediate those constituents is a subject of particular interest.

The primary objective of this research was to evaluate the effectiveness of each of the chosen treatments in terms of the potential to:

- impact the water column through re-suspension (and wider environment).
- accelerate benthic recovery from a highly enriched state.

The research therefore involved intensive water-column sampling during the initial treatments, coupled with comprehensive ‘before’ and ‘after’ physico-chemical and biological sampling of the seabed environment to evaluate the effects.

Recommendations are made as to if and how any of these treatments may be advanced for commercial implementation.

3. METHODS

3.1. Study site

The study was conducted at a low-flow salmon farm situated in the Marlborough Sounds, New Zealand. Water depth across the 1.2 ha site ranged from 32 m to 34 m and the substrate comprised mud/sand overlaid with patches of biofouling drop-off (predominantly mussels) from the farm.

Prior to the field trials taking place the farm site had been fallowed (unoccupied) for four months after a 13-month period of relatively intensive farming. Prior to that occupation, the site had been fallowed for 12 months following a 2-year period of intensive use.

At the point of commencing the field trials, the seabed at the farm site was still in a highly-enriched state with blackened, flocculent organic sediments, a severely impoverished macrofaunal community and an extensive white bacterial mat (*Beggiatoa* sp.) covering the sediment surface.

3.2. Remediation treatments and experimental design

Four treatments to seabed remediation were trialled in duplicate ~15 m^2 experimental field plots haphazardly arranged within the most enriched area (*i.e.* the area previously beneath the farm, Figure 1). The treatments (Table 1) were:

1. Untreated natural recovery (‘Untreated’ treatments ‘UT1, UT2’)
2. Repeated irrigation and oxygenation (‘Irrigated’ treatments ‘IR1, IR2’)
3. Repeated harrowing (‘Harrow’ treatments ‘HA1, HA2’)
4. Simulated removal of enriched sediment layer by a custom built blade (‘Removal’ treatments ‘RE1, RE2’)

The Removal was a one-off treatment, whereas the Harrowed and Irrigated plots were treated three times; Day 0, 42 and 68. Un-impacted control sites (treatments ‘Ctl1, Ctl2’) were also established outside of the farmed area for a natural reference. After initial treatment, a small mooring block with subsurface and surface floats was placed alongside each plot, GPS coordinates were recorded and the treated area was marked out on the seabed by divers using ropes and stakes. This allowed the exact same area of seabed to be relocated for sampling and for the Harrow and Irrigated plots to be re-treated.

The Harrowing and Irrigation treatments were selected based on recommendations by Eriksen *et al.* (2012). The Removal treatment was added to simulate removal of the top layer of highly enriched sediments. The initial plan was to remove (by dredge or similar) the surface sediments. However, the costs associated with having a commercial dredge on site for such a small-scale trial, combined with the resource management and consent issues associated with the subsequent disposal of the waste, made this option impractical and cost-prohibitive. Hence, 'simulating' the sediment removal seemed a logical step to evaluate the potential of this treatment.

The 'pilot-scale' of the plots was determined so that the remediation treatments could be evaluated *in situ* (*i.e.* using semi-industrial sized equipment and large enough plots to collect three sets of replicate cores). This was in terms of both their effectiveness and their potential to induce re-suspension, but also at a scale that was considered a low risk in terms of potential to induce bay-wide water column effects during treatment. Resource consent was obtained from the local regulatory body to carry out the field-trials.

Table 1. Summary of treatment types, plot names, and descriptions.

	Treatment (Plot names)	Description of treatment
1.	Untreated natural recovery (UT1, UT2)	Duplicate 16 m^2 (4 m × 4 m) areas within farm site. Untreated and left to recovery naturally.
2.	Harrow/ raking (HA1, HA2)	Duplicate ~15 m^2 areas (5 m × 3 m; dictated by the width of the harrows) within farm site. Physical disturbance by 'harrowing' and 'raking'. Initial treatment was conducted by repeatedly dragging purpose-built harrow (Figure 2a) across defined area in a controlled manner using a winch from a barge anchored at four points. Re-treatment of the HA plots involved divers vigorously and repeatedly raking over the marked out area by hand with a large, long-pronged metal rake (as opposed to the initial dragging of 'harrows'). Target penetration depth was ~10 cm.
3.	Irrigated (IR1, IR2)	Duplicate ~15 m^2 areas (2 × 3 m × 2.4 m) within farm site. Surface sediments were irrigated for 3–6 hours with surface water using a boat mounted pump. The sediments were irrigated using PVC hoses perforated with an array of 200 low-pressure sprinkler heads (Figure 2). Retreatment of the IR plots involved divers repositioning the irrigation array and then repeating the pumping exercise. The estimated penetration depth was 3–5 cm.
4.	Removal (RE1, RE2)	Duplicate ~15 m^2 areas (5 m × 3 m) within farm site. Simulating removal of enriched layer on sediments to a depth of ~10 cm using custom built blade (Figure 2) pulled through the sediments slowly using a winch from a barge anchored at four points with the tow rope passing through a block attached to a 500 kg 'deadman' weight on the seabed to ensure a horizontal pull. The depth of the cut was set by adjusting the height of the guide skids on the blade.
5.	Control: un-impacted (Ctl1, Ctl2)	Duplicate 16 m^2 (4 m × 4 m) areas situated in un-impacted reference sediments approximately 170 m and 320 m from the farm in comparable depth and substrate.

The initial baseline survey involved collecting 13 near-bottom water column samples from across the site and triplicate sediment samples (Section 4.3) from within each of the plots immediately prior to the initial treatment (Day 0). Additional seabed core samples were collected from the Removal plots immediately following treatment, to examine the immediate effectiveness of removing the top layer of sediments. Extensive water column monitoring was conducted throughout the initial treatments (on Day 0, Section 4.4). The Harrow and Irrigated treatments were repeated on Days 42 and 68. Triplicate sediment samples were collected from within all of the plots again at Day 68. At the conclusion of the study, on Day 124 (after 4 months), triplicate sediment samples were collected from within and immediately adjacent to the plots to provide further spatial and statistical comparison. Samples were only taken from inside the Control plots, due to the un-impacted and temporally stable nature of those sediments. Table 2 provides a summary of the sampling regime.

Table 2. Summary of the sampling regime for the seabed remediation study.

<table>
<tr><th>Pre-treatment</th><th colspan="4">Post-treatment</th></tr>
<tr><td colspan="2">Day 0</td><td>Day 42</td><td>Day 68</td><td>Day 124</td></tr>
<tr><td>13 near-bottom water samples collected from across the site</td><td>Initial Removal, Irrigated and Harrow treatments</td><td>Irrigated and Harrow treatments repeated</td><td>Irrigated and Harrow treatments repeated</td><td></td></tr>
<tr><td>Triplicate sediment samples collected from each plot</td><td>At the Removal, Irrigation and Harrow sites CTD casts were taken and near-bottom water samples collected at 15 min intervals

Additional triplicate sediment and macrofauna samples collected from Removal plots post-treatment</td><td></td><td>Triplicate sediment samples collected from each plot</td><td>Triplicate sediment samples collected from within and immediately adjacent to each plot*</td></tr>
<tr><td colspan="2">D-Opto™ loggers (temperature, dissolved oxygen saturation and content)</td><td></td><td></td><td></td></tr>
<tr><td colspan="5">ADCP current profiler</td></tr>
</table>

* Control plots were only sampled within the plot due to the un-impacted and temporally stable nature of those sediments.

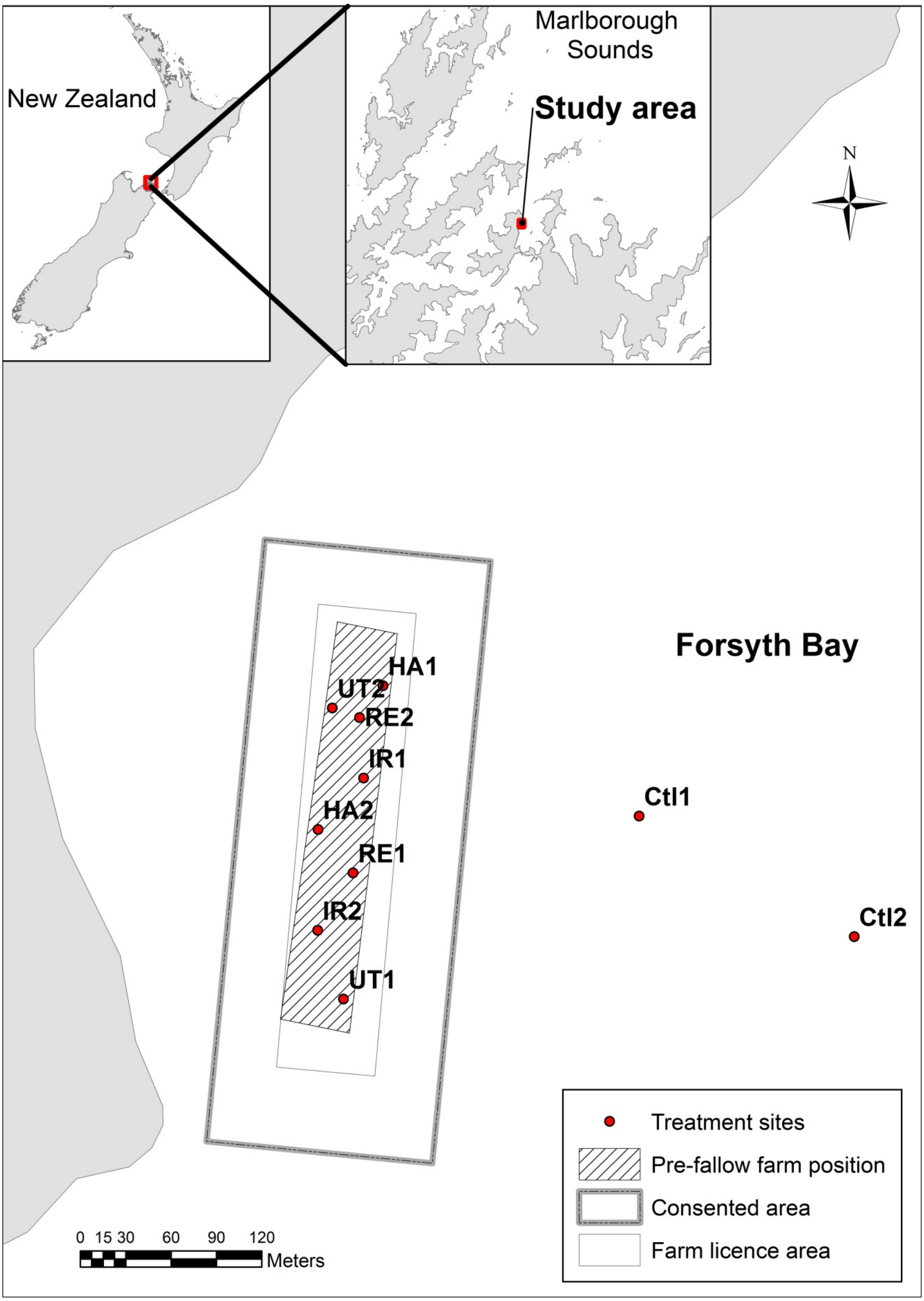
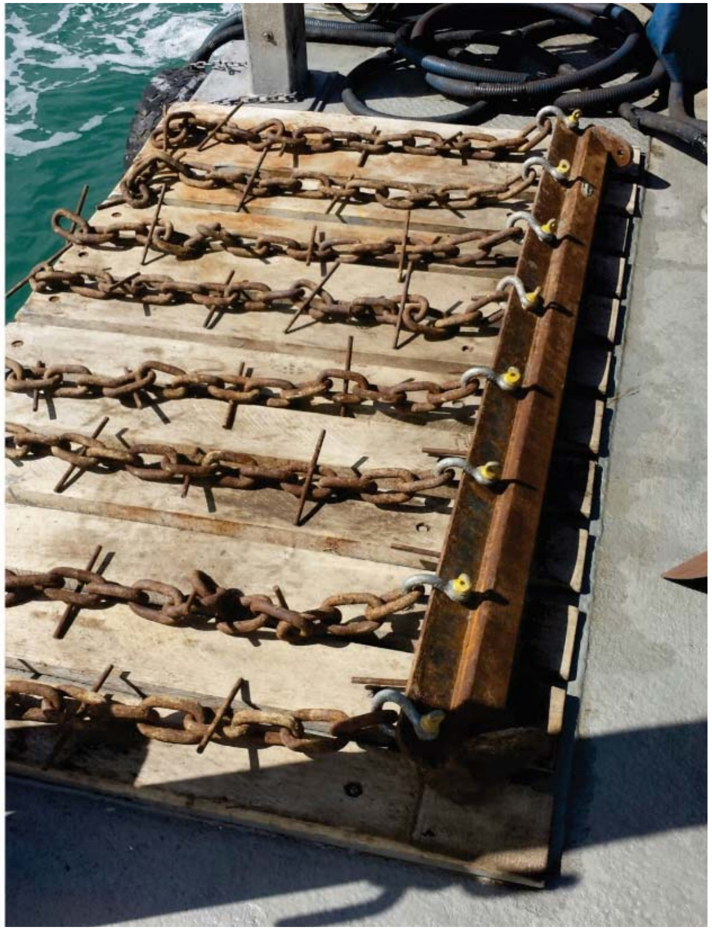


Figure 1. Site map showing arrangement of treatment plots in relation to the fallowed farm area within Forsyth Bay, Marlborough Sounds, New Zealand. UT = Untreated; IR = Irrigated; HA = Harrow; RE = Removal; Ctl = Control.



a: Harrow

b: Plow

c: Irrigator

Figure 2. Photos of the a) 'harrows' that were towed on initial treatment of the harrow plots, b) 'blade' used on simulated sediment removal plots, and c) the 'irrigator' array used in the irrigation plots.

3.1. Water column sampling

Intensive water column sampling was conducted around the site prior to, and then during, the initial treatments on Day 0. Six D-Opto (D-Opto™ Logger, Zebra-Tech Ltd., Nelson, New Zealand) remote sensors programmed to log temperature, dissolved oxygen (DO) and turbidity at 10-min intervals for the duration of the initial treatments (over two days) were deployed around site as follows (Figure 3):

- down-current 163 m to the north of the site.
- down-current 88 m to the south of the site.
- attached to the mooring lines alongside plots IR2, HA2, Clt1 and RE2.

A bottom-mounted current meter (RDI Workhorse Sentinel 600 acoustic Doppler current profiler) was also deployed central to the site to monitor currents speeds and direction throughout the study (Figure 3). The 13 baseline water column samples were collected during the two hours prior to treatment from approximately 1 m above the seabed using a van Dorn sampler. Samples were kept cool and in the dark until analysed using standardised nutrient analysis methods at Hills Laboratories. Nutrients analysed were: total nitrogen (TN), ammonia (TA), nitrite-N (nitrite), nitrate-N (nitrate) and total phosphorus (TP).

During the treatment of each of the Irrigated, Harrow and Removal plots, conductivity-temperature-depth (CTD, Seabird SBE plus v2) casts were taken through the full water column. Near-bottom (~1 m from seabed) water sampling for nutrients (as for baseline samples) was conducted at 15-minute intervals for 90 to 120 minutes from the time the treatment/disturbance began. Two CTDs were operated synchronously, whereby one stayed at the site of the treatment and the second followed the plume. Both were set up to record temperature, dissolved oxygen, turbidity, photosynthetically active radiation and salinity. The plume was tracked using a drogue suspended at 25 m depth (*i.e.* targeting near-bottom water) from a surface float with GPS logger attached (see tracks in Figure 3).

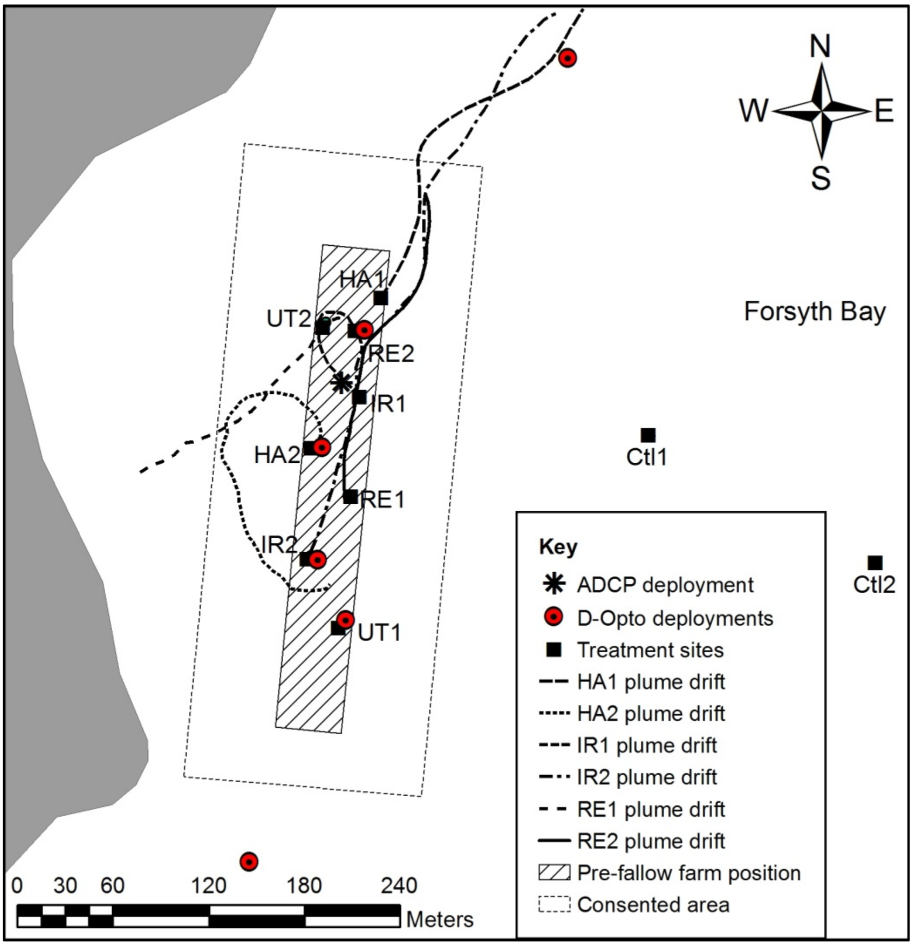


Figure 3. Site map showing the position of the acoustic Doppler current profiler (ADCP) deployment, the D-Opto logger deployments and the drogue drifts paths (25 m depth) where conductivity-temperature-depth (CTD) casts and water sampling was conducted at 15-minute intervals. UT = Untreated; IR = Irrigated; HA = Harrow; RE = Removal; Ctl = Control.

3.2. Seabed sampling

3.2.1. Physico-chemical sampling

Divers collected 55 mm diameter Perspex sediment cores at all plots. The top 40 mm of each sediment core was analysed for grain size distribution, total organic matter (TOM) content, total free sulphide (TFS), total copper (Cu) and zinc (Zn). Samples for TFS were collected from the surface sediments (0-4.5 cm depth interval) using a cut-off 5-cc plastic syringe. Redox potential (Eh_{NEH}, mV) was measured in duplicate (10 mm depth) *in situ* in each replicate core using a Thermo Scientific combination Redox/ORP electrode and the average of the two values was used to improve the estimate. During the final survey (Day 124), the triplicate sediment cores samples were also analysed for total nitrogen (TN) and total phosphorus (TP). Details of the analysis methods for each of these parameters is outlined in

Table 3. Analysis methods for physico-chemical parameters measured during the seabed remediation study.

Parameter	Unit	Analysis method
Grain-size distribution	%	Dried and analysed gravimetrically for size class fractions from silt-clay through to gravel.
Organic matter (OM)	%OM	Measured as ash-free dry weight; (Luczak *et al.* 1997).
Total free sulphide (TFS)	µM	Analysed following the methods of Wildish *et al.* (1999).
Copper (Cu)	mg kg^{-1}	Inductively coupled plasma mass spectrometry (ICP-MS) following nitric/hydrochloric acid digestion of dried sample. USEPA 200.2.
Zinc (Zn)	mg kg^{-1}	ICP-MS following nitric/hydrochloric acid digestion of dried sample. USEPA 200.2.
Redox potential	Eh_{NEH}, mV	Measured in duplicate (10 mm depth) in each replicate core using a Thermo Scientific combination Redox/ORP electrode and the average of the two values was used to improve the estimate.
Total nitrogen (TN)	mg kg^{-1}	ICP-MS, screen level, following nitric/hydrochloric acid digestion of dried sample. USEPA 200.2.
Total phosphorus (TP)	mg kg^{-1}	Catalytic combustion separation, Thermal conductivity detector.

3.2.2. Biological sampling

Divers collected triplicate 130 mm diameter (0.0132 m^2) macrofauna cores to a depth of 100 mm in the sediment. Macrofauna samples were sorted and identified to the lowest practicable level, and their abundances recorded. Macrofauna count data were used to calculate total abundance (N), number of taxa (S), Pielou's evenness index (*J'*), Shannon-Wiener diversity index (*H'*) and the AZTI's Marine Biotic Index (AMBI) and the benthic quality index (BQI) and enrichment stage (ES) (Table 4). Table 5 describes the different environmental characteristics of each enrichment stage.

Table 4. Description of biological parameters measured during the seabed remediation study.

Descriptor	Equation	Description	Reference
No. taxa (*S*)	Count (taxa)	Total number of taxa in a sample.	
No. individuals (*N*)	Sum (n)	Total number of individual organisms in a staple.	
Evenness (J')	$J' = H'/\log_e(S)$	Pielou's evenness index. A measure of equitability, or how evenly the individuals are distributed amongst the different species/taxa. Values can range from 0 to 1, where a high value indicates an uneven distribution or dominance by a few taxa.	Pielou (1966)
Diversity (*H'*)	$H' = -\sum_i(P_i\log(P_i)$ where P is the proportion of the total count arising from the *i*th species	Shannon-Wiener diversity index ($\log_e$ base). A diversity index that describes, in a single number, the different types and amounts of animals present in a collection. Varies with both the number of species and the relative distribution of individual organisms amongst the species. This index ranges from 0 for communities containing a single species to high values (> 5) for communities containing many species.	Shannon (1948)

Descriptor	Equation	Description	Reference
AZTI Marine Biotic Index (AMBI)	AMBI = [(0 × %GI + 1.5 × %GII = 3 × %GIII + 4.5 × %GIV + 6 × %GV)]/100	Pollution or disturbance classification of a site, representing the benthic community health. Values range from 0 (unpolluted) to 7 (extremely polluted). Based on single expert classification of species into five ecological groups corresponding to different sensitivity levels.	Borja *et al*. (2000)
Benthic quality index (BQI)	$BQI = (\sum_{I} (N_i/N_{class} \times ES50_{0.05i})) \times {}^{10}\log(S+1) \times (N_{total}/(N_{total}+5))$	Ecological status of a site. Values range from 0 (bad quality) to 16 (high quality). Accounts for the relative abundances of tolerant and sensitive species and for species richness. Attribution of tolerance/sensitivity is based on mathematical analysis of quantitative datasets originating from the studied area.	(Rosenberg *et al.* 2004); Labrune *et al.* (2012)
Enrichment stage (ES)		Ranges from 1 (pristine/natural) to 7 (azoic/anoxic), which is quantitatively determined from previously derived relationships with multiple biotic and physico-chemical environmental indicators (see Table 5).	(Keeley *et al.* 2012a; Keeley *et al.* 2012b)

Table 5. General descriptions of the primary environmental characteristics of the seven enrichment stages (ES) modified from Keeley *et al.* 2012b. HF = high-flow sites (mean mid-water current speeds ≥ 10 $cm.s^{-1}$), LF = low-flow sites (< 10 $cm.s^{-1}$).

ES	General description		Environmental characteristics
1.0	***Pristine end of spectrum.*** Clean unenriched sediments. Natural state, but uncommon in many modified environments	LF	Environmental variables comparable to an unpolluted / un-enriched pristine reference station.
		HF	As for LF, but infauna richness and abundances naturally higher (~2 × LF) and %organic matter (OM) slightly lower.
2.0	***Minor enrichment.*** Low-level enrichment. Can occur naturally or from other diffuse anthropogenic sources. 'Enhanced zone.'	LF	Richness usually greater than for reference conditions. Zone of 'enhancement' – minor increases in abundance possible. Mainly a compositional change. Sediment chemistry unaffected or with only very minor effects.
		HF	As for LF
3.0	***Moderate enrichment.*** Clearly enriched and impacted. Significant community change evident.	LF	Notable abundance increase; richness and diversity usually lower than reference station. Opportunistic species (*i.e.* Capitellid worms) begin to dominate.
		HF	As for LF
4.0	***High enrichment.*** Transitional stage between moderate effects and peak macrofauna abundance. Major community change.	LF	Diversity further reduced; abundances usually quite high, but clearly sub-peak. Opportunistic species dominate, but other taxa may still persist. Major sediment chemistry changes (approaching hypoxia).
		HF	As above, but abundance can be very high while richness and diversity are not necessarily reduced.
5.0	**Very high enrichment.** State of peak macrofauna abundance.	LF	Very high numbers of one or two opportunistic species (*i.e.* Capitellid worms, nematodes). Richness very low. Major sediment chemistry changes (hypoxia, moderate oxygen stress). Bacterial mat usually evident. Out-gassing occurs on disturbance of sediments.
		HF	Abundances of opportunistic species can be extreme (10 × LF ES 5.0 densities). Diversity usually significantly reduced, but moderate richness can be maintained. Sediment organic content usually slightly elevated. Bacterial mat formation and out-gassing possible.
6.0	***Excessive enrichment.*** Transitional stage between peak abundance and azoic (devoid of any organisms).	LF	Richness and diversity very low. Abundances of opportunistic species severely reduced from peak, but not azoic. Total abundance low but can be comparable to reference stations. %OM can be very high (3–6 × reference).
		HF	Opportunistic species strongly dominate, with taxa richness and diversity substantially reduced. Total infauna abundance less than at stations further away from the farm. Elevated %OM and sulphide levels. Formation of bacterial mats and out-gassing likely.
7.0	***Severe enrichment.*** Anoxic and azoic; sediments no longer capable of supporting macrofauna with organics accumulating.	LF	None, or only trace numbers of infauna remain; some samples with no taxa. Spontaneous out-gassing; bacterial mats usually present but can be suppressed. %OM can be very high (3–6 × reference).
		HF	Not previously observed—but assumed similar to LF sites.

3.3. Data analyses

Data from the CTD casts were interpolated using the Kriging method in Surfer v9 with survey time (minutes since initial treatment) on the x-axis and depth on the y-axis. Two-dimensional plots were created for DO and turbidity using standardised data range and scale divisions for comparability. Bar plots for water column, benthic and nutrient flux variables were created in R team (R Development Core Team 2011) using the barplot function. Statistical significance of differences between pairs ‘inside plot’ and ‘outside plot’ (= factor ‘Position’) at the conclusion of the study were determined using the model *lm (Variable ~ factor (Treatment) + factor (Treatment:Position)),* and the significance displayed on the plots above each pair using an asterisk code. Prior to analysis the data were analysed for normality and heterogeneity of variances, and were log transformed as required.

Macrofaunal data were also analysed multivariately using PRIMER 6 + PERMANOVA software (Clarke & Gorley 2006; Anderson *et al.* 2008). Macrofaunal count data were fourth root transformed to reduce the influence of the highly abundant taxa and then two dimensional multidimensional scaling (MDS) configuration were generated on the basis of Bray-Curtis similarities. Distance between centroids was used to display similarities between groups of samples and cluster slices were overlaid to show groupings in multivariate space. The similarity percentages (SIMPER) function was used to determine the main taxa contributing to differences/similarities between and within select groups.

4. RESULTS

4.1. Water-column effects during disturbance

Across the site, background turbidity and DO levels in the water column were typically 1.6 NTU (nephelometric turbidity unit) and 7.6 mg l^{-1} (Table 6). Incidences of elevated turbidity during treatment were usually correlated with reduced DO, which indicates high oxygen demand in the associated sediments. Turbidity plumes were evident in both Harrow and Removal treatments, with the latter having the sharpest increases in turbidity and corresponding decreases in DO (Table 6, Figure 4). The increases in turbidity were commonly on the order of 1–3 NTU (representing a 100–200% increase). The corresponding reductions in DO were approximately 0.4–0.6 mg l^{-1}, representing a ~6% reduction (to 94% of saturation). The highest turbidity of 4.2 NTU was recorded when following the plume from plot RE2. This event was also associated with the lowest DO of ~6.6 mg l^{-1}. The rate of dispersal, dissipation and return to natural background conditions was mixed between treatments with some seemingly natural again after the 90-min monitoring period, but others showing a persistent plume (Appendix 2–3). Minimal plumes were observed during both of the Irrigated treatments.

Table 6. Summary of spatial and temporal patterns evident in two-dimensional, interpolated plots of turbidity (nephelometric turbidity unit; NTU) and dissolved oxygen (DO). See examples in Figure 4 and complete set of plots in Appendices 2–3.

	Stationary	**Plume**
HA1	Increased turbidity (+1 NTU) in bottom 5 m after 45 min (no turbidity data collected prior to 45 min mark). A 0.3 mg l^{-1} reduction in DO in bottom 5–10 m commencing after 30 min, and still present at 90 min.	Sharp turbidity increase (+2 NTU) at 45 min in bottom 10 m of water column. Corresponding reduction in DO to 7 mg l^{-1} (-0.5 mg l^{-1}), strongest 5 m from seabed.
HA2	Increased near-bottom turbidity (+3 NTU) expanding to 6 m off bottom at 60 min. Less apparent after 90 min. Reduced DO with increasing proximity to seabed (-0.6 mg l^{-1} from mid-water).	Strong plume evident (+2.6 NTU, maximum ~4 NTU), particularly at 10–20 min and 50–60 min. Corresponding DO reductions, with minimum concentrations ~7 mg l^{-1}.
IR1	Minor increase in turbidity toward seabed (probably natural) with no temporal pattern. Minor DO reduction toward seabed (minimum ~7.1 mg l^{-1}).	No obvious patterns in time or space. Minor DO reduction near seabed at 0 min (before leaving farm site).
IR2	No patterns evident until 75 min mark, when near-bottom increase (+2 NTU) in turbidity occurred. Corresponding, but very minor reduction in DO.	No turbidity plume evident except for at 60 min mark when near-bottom turbidity increased by ~0.6 NTU. DO largely unaffected (~0.2 mg l^{-1} reduction, Figure 4A).
RE1	Near-bottom increase in turbidity beginning at 30 min (+2.8 NTU). Extends up to 10 m above seabed after 45–75 min. Corresponding reduction in DO (to 7 mg l^{-1}), restricted to bottom 2 m until 45 min mark. Maximum reduction 0.2–0.5 mg l^{-1}. Plume largely gone at 90 min.	Near-bottom turbidity plume (+2 NTU) evident at 30–45 min. Does not extend further than 6 m from seabed. Near-bottom DO reduced (minimum 6.9 mg l^{-1}). Possibly extending 10–15 m from seabed. Reduction on order of 0.4–0.6 mg l^{-1}.
RE2	Strong near-bottom turbidity layer, but limited to bottom 3 m. Peaked at 5.8 NTU at 70 min mark (+5 NTU from background). Consistent DO reduction with increased proximity to seabed throughout. Near-bottom DO 6.8 mg l^{-1} cf. 7.8 mg l^{-1} in mid-water.	Strong near-bottom turbidity layer (plume), limited to bottom 6–8 m. Plume not so evident after 75 min mark. Corresponding reduction in DO also dissipating after 75 min. Minimum DO was 6.6 mg l^{-1} (approx. 0.5 mg l^{-1} reduction) after 30 min in bottom 1–2 m (Figure 4B).

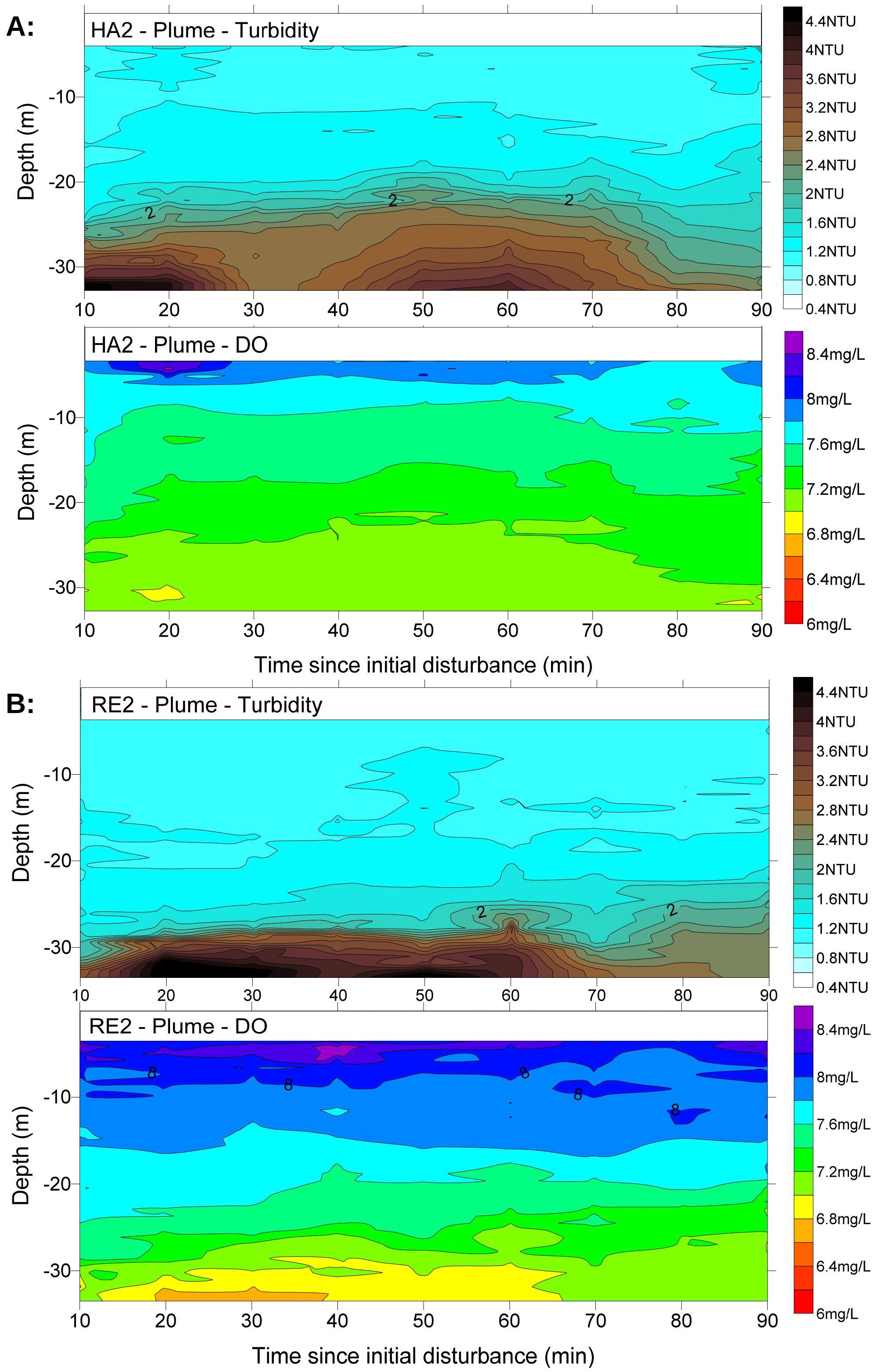


Figure 4. Temporal and spatial two-dimensional interpolation of dissolved oxygen (DO, mg l^{-1}) and turbidity (nephelometric turbidity unit; NTU) profiles from CTD (conductivity-temperature-depth) casts through water column following the plume (via near-bottom drogue) for the 90 minutes after: **A:** Harrow treatment at HA2 (top 2 plots) and **B:** Removal treatment at RE2 (bottom 2 plots). For all other treatment plots see Appendices 2–3.

The array of D-Opto™ oxygen loggers that were deployed around the site (
Figure 3) showed no substantive drops in DO in response to any of the treatments. However, a consistent minor (5–10%) reduction was observed in all instruments in response to the Removal treatment in RE2.

Near-bottom total nitrogen (TN) concentrations were within the pre-disturbance baseline range throughout most of the trials (0.17-0.28 mg l^{-1}). The only exceptions were the first sample collected following the plume in the Harrow trials, one sample taken at 90 min when following the plume from IR2, and two samples in the plume from RE1 (Figure 5). The maximum TN recorded was 0.36 mg l^{-1} at RE1 in the plume after 45 min.

Total ammonia (TA) concentrations remained low (< 0.07 mg l^{-1}) during treatments. However, were often slightly above the pre-disturbance baseline range, especially following HA2 and in the plumes from the Removal treatments (Figure 5). Nitrite (NO_2 and nitrate NO_3 concentrations also remained low (< 0.015 mg l^{-1}) and within the range of baseline conditions during treatments. These concentrations increased within the plumes of the Harrowed and Removal treatments (Figure 5).

Similarly for total phosphorus (TP), concentrations were often above the baseline range at the Harrow and Removal treatments. The Irrigated treatments remained at or near background. When following the plume from HA1 and RE2, TP concentrations increased with time to a maximum of 0.25 mg l^{-1}, whereas at HA2 the opposite trend was observed. Salinity or PAR (practical alinity unit) were also measured by the CTD casts. No effect on these parameters was detected.

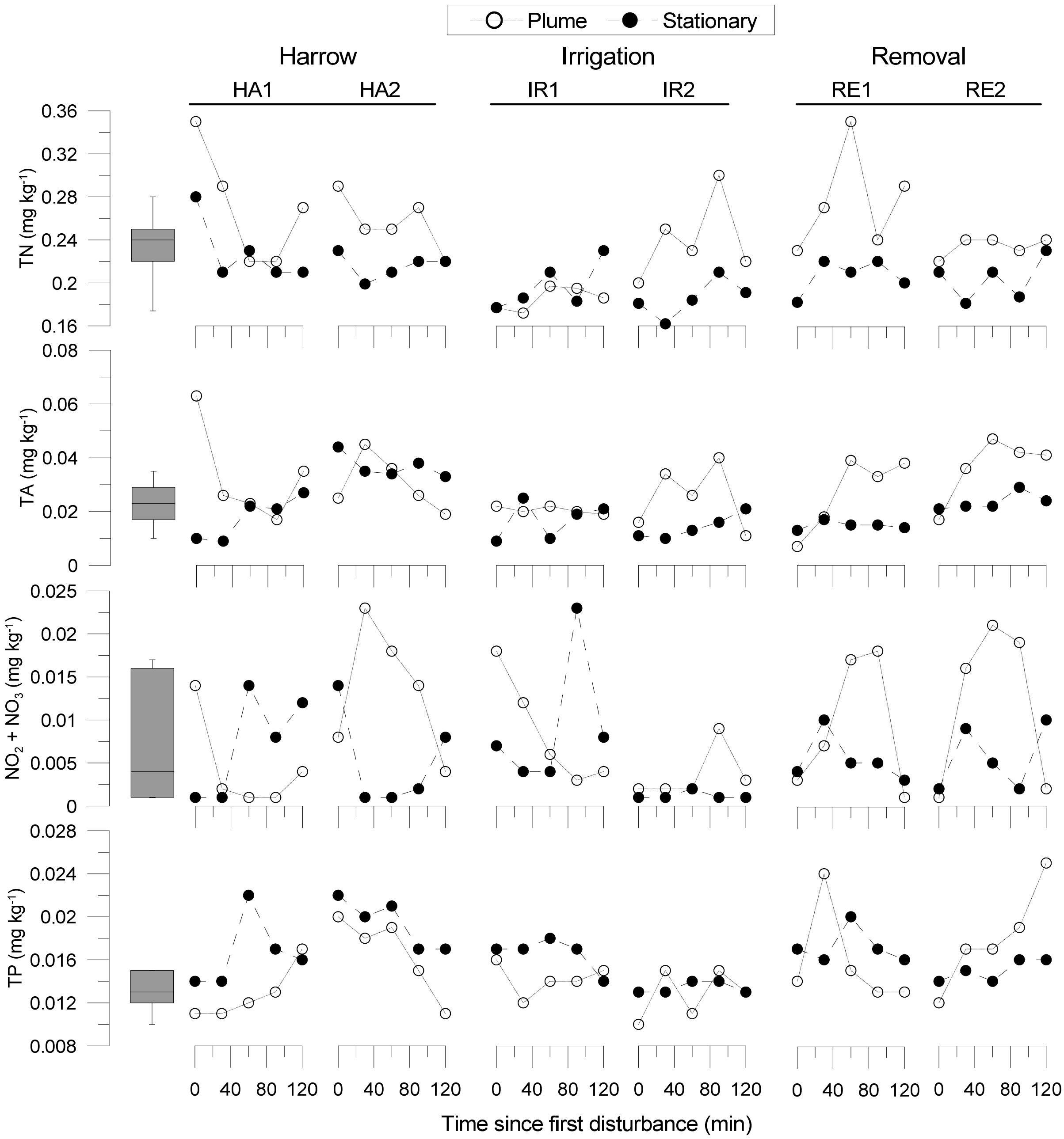


Figure 5. Near-bottom nutrient concentrations during Harrow (HA), Irrigated (IR) and Removal (RE) treatments from water sampled collected during the two hours following the treatment at the site ('Stationary') and following the plume ('Plume') in relation to the pre-disturbance baseline conditions, represented by box and whisker plots. Nutrients measured were total nitrogen (TN), total ammonia (TA), nitrite/nitrate ($NO_2^- + NO_3^-$) and total phosphorus (TP).

4.2. Biological and physico-chemical properties of sediments

4.2.1. Pre-treatment seabed conditions

Prior to treatment, the recently farmed area of seabed was highly enriched, evidenced by sediment with very high organic matter content (16–21%OM), strongly negative redox potential (-93 to -223 Eh_{NHE}, mV), very high TFS (940 to 2,153 µM) and a generally impoverished macrofauna dominated by a few enrichment tolerant taxa (0 to 5 taxa, Figure 6). Total abundance was generally low (post-peak) at all plots except RE2 and UT2, which had elevated abundances (735 and 600 individuals' $core^{-1}$, respectively). The polychaete worm *Capitella capitata* was clearly the dominant species (95% contribution according to SIMPER analysis) at all of the impacted treatment plots, except HA1. At this site it was noticeably less enriched and dominated by *Prionospio* sp. polychaetes (22% contribution), Dorvilleidae polychaetes (19%), and amphipods (17%). By comparison, the macrofaunal community at the Control stations was diverse and the seabed relatively natural. These differences produced clear clustering of samples into three groups; Controls, HA1 and the other within farm plots (Figure 6). The overall enrichment stage (ES) for these same three groups was ES 2.2–2.3 (minor enrichment), 3.7 (moderate enrichment), and between ES 5.5 and 6.0 (major enrichment), respectively (Figure 6).

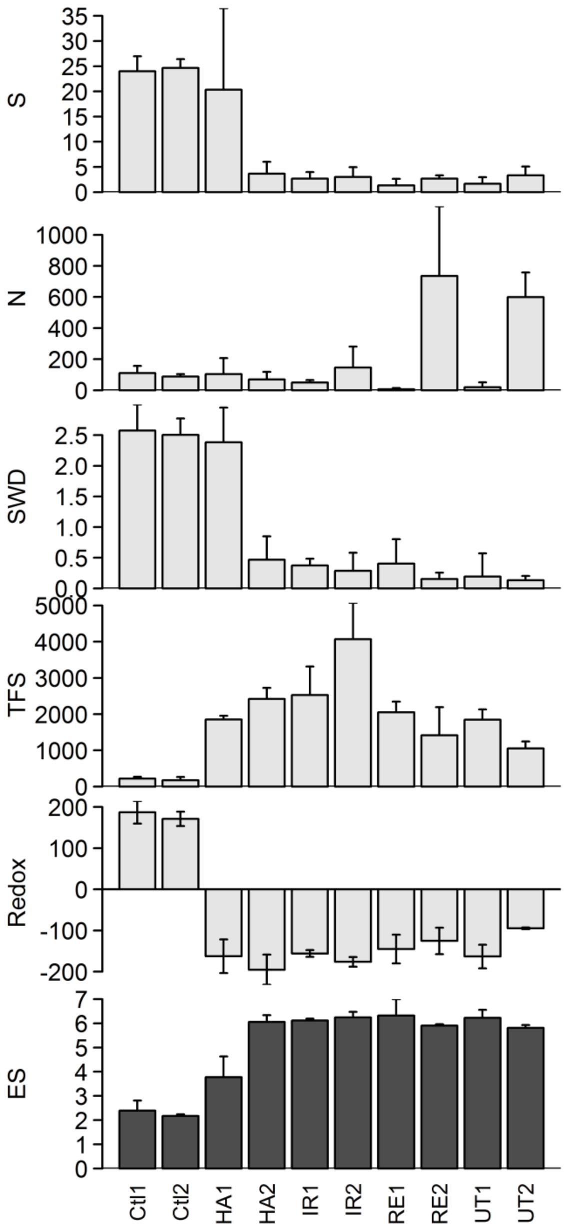


Figure 6. Summary of seabed characteristics within the treatment plots during the initial pre-treatment survey. S = number of taxa, N = number of individuals, SWD = Shannon-Wiener diversity, TFS = total free sulphides (µM), Redox = Redox potential (Eh_{NHE}, mV) and ES = enrichment stage.

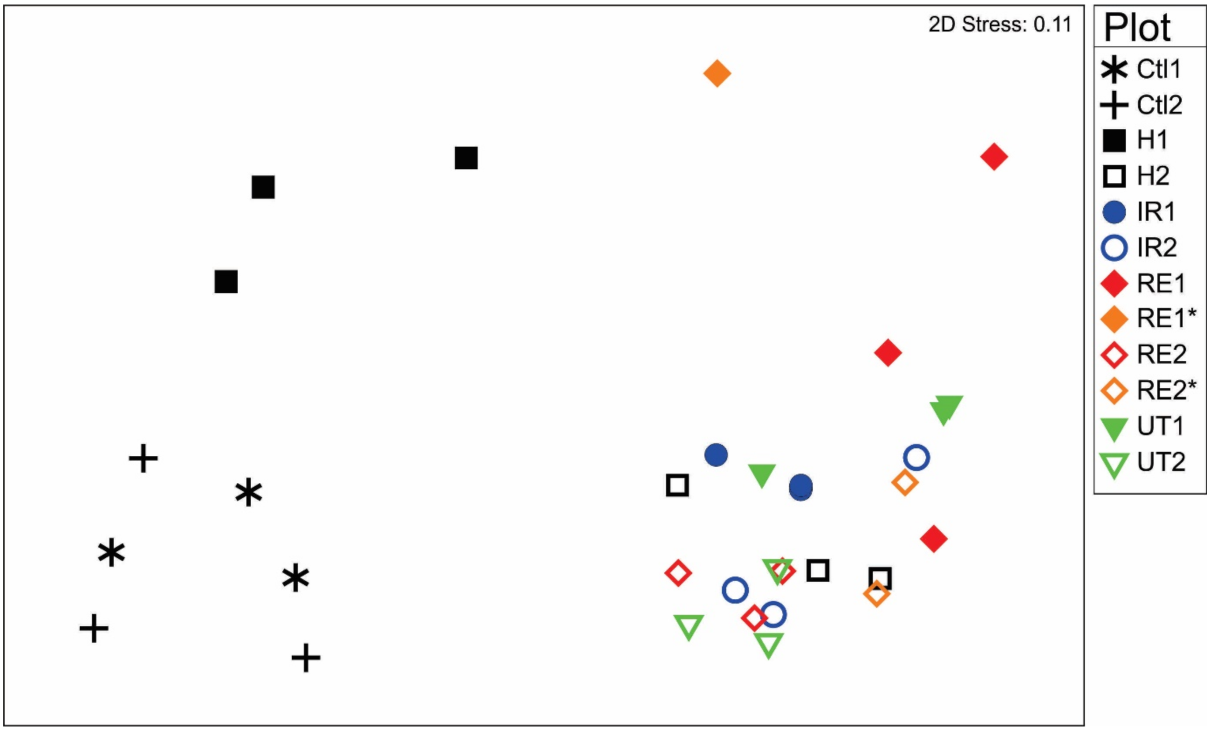


Figure 7. Two-dimensional multidimensional scaling (MDS) configuration on the basis of Bray-Curtis similarities of fourth-root transformed macrofaunal count data from initial pre-treatment survey. * (next to RE1 and RE2) indicate samples taken inside plots immediately after removal.

4.2.2. Post-treatment seabed conditions

At the conclusion of the study, four months after the initial treatment, the seabed within the general farmed area had undergone some recovery. This was indicated by reduced TFS concentrations, increased S and other macrofauna diversity measures in all treatments including the untreated natural recovery plots. Howe3ver, it was still highly enriched. The main 'Plot' factor in the statically models was highly significant ($P < 0.01$) in all cases, indicating heterogeneity between plots. This was mainly driven by the differences at the control sites, but also the HA1 plot, which was initially less enriched.

Paired contrasts between samples collected inside and outside of the treatments indicated that the Harrow, Irrigation and Untreated treatments had had no significant ($P < 0.05$) effect on the majority of the benthic characteristics (Figures 8–10). The exceptions being TFS, which was significantly higher inside the IR1 ($P = 0.039$) treated area, and mud content, which was higher inside both IR1 ($P = 0.037$) and HA2 ($P = 0.028$). Total phosphorus was also significantly higher inside UT1 plot at the conclusion of the experiment.

However, the physico-chemical characteristics inside and outside of the Removal treatment were significantly different in many cases (Figures 8–10). Total organic matter and TFS were both significantly reduced inside the treated areas ($P < 0.01$), and the medium and fine sand sediment grain size components were significantly reduced—being offset by increases in the mud component (Figure 8). Number of taxa (S) and BQI index was approximately twice as high inside the Removal treatment. However, the AMBI was not significantly different (Figure 9). The Removal treatment also had the net effect of significantly reducing the TN, TP and zinc content of the sediments (Figure 10). Overall, ES was significantly reduced by approximately 1.5 following the Removal treatments, representing the difference between being classified very highly enriched and moderate-highly enriched (Figure 11).

The relative differences between the macrofaunal assemblages that were found inside and outside the treatment plots at the conclusion of the study are displayed in Figure 12. Substantial differences between the 'In' and 'Out' pairs (represented by greater distances) were evident in the two RE plots and at IR1. The seabed outside of the treated IR1 plot had higher abundances of several enrichment tolerant species (early colonisers), most notably nematodes, *C. capitata* and *Prionospio* sp. polychaete worms. By contrast, seabed inside both of the treated Removal plots had greater abundances of most species, especially the polychaetes Dorvilleidae, *Prionospio multicristata*, *Capitella capitata* and *Armandia maculata*, and amphipods. Macrofaunal communities at the Control sites were different to those in the treated plots, suggesting none of the plots had completely recovered from farm impacts.

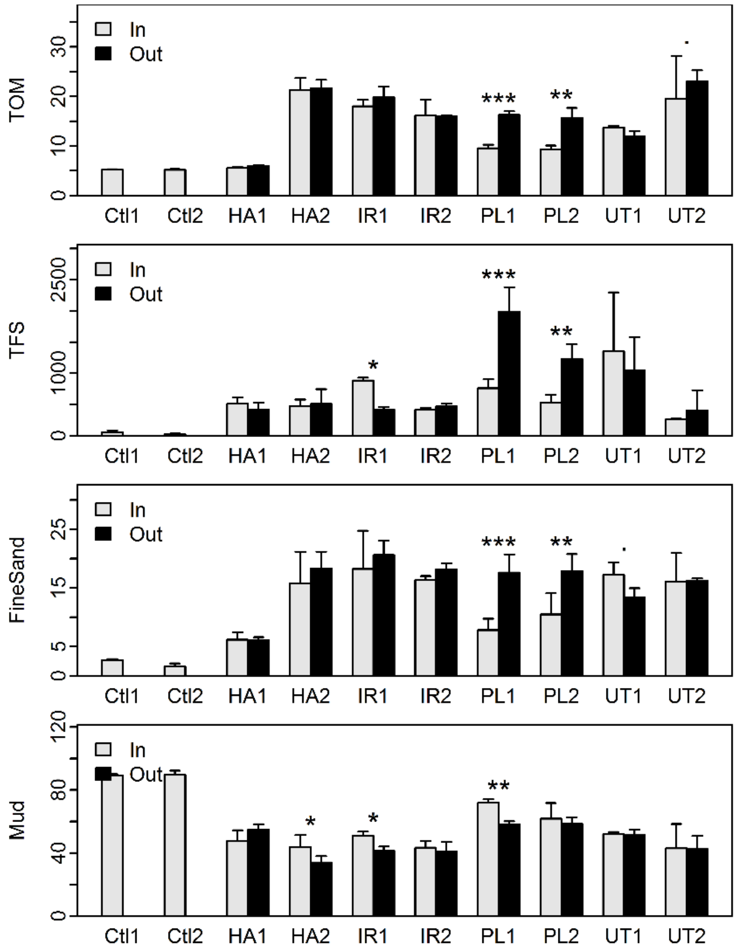


Figure 8. Comparison of physico-chemical attributes (total organic matter [TOM], total free sulphides [TFS], fine sand and mud) of sediments inside ('In') and outside ('Out') of the treatments plots at the conclusion of the study, four months after initial treatment. Error bars represent standard error. The significance of the difference between each pair (In/Out) represented by: '·' = $P < 0.1$; '*' = $P < 0.05$; '**' = $P < 0.01$; '***' = $P < 0.001$. No outside samples were collected at the control sites. Ctl = Control, HA = Harrow, IR = Irrigation, RE = Removal, UT = Untreated.

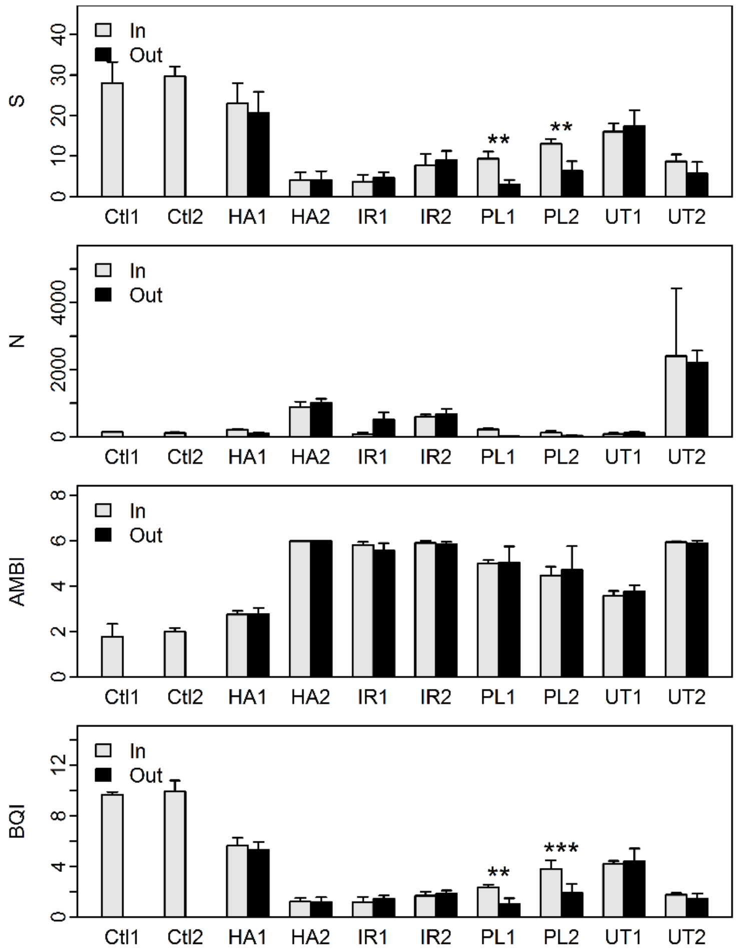


Figure 9. Comparison of macrofaunal parameters (number of taxa [N], number of individuals [S], AZTI Marine Biotic Index [AMBI], benthic quality index [BQI]) from samples taken within ('In') and outside ('Out') the treatments plots at the conclusion of the study, four months after initial treatment. Error bars represent standard error. The significance of the difference between each pair (In/Out) represented by: '·' = P < 0.1, '*' = P < 0.05, '**' = P < 0.01, '***' = P < 0.001. No outside samples were collected at the control sites. Ctl = Control, HA = Harrow, IR = Irrigation, RE = Removal, UT = Untreated.

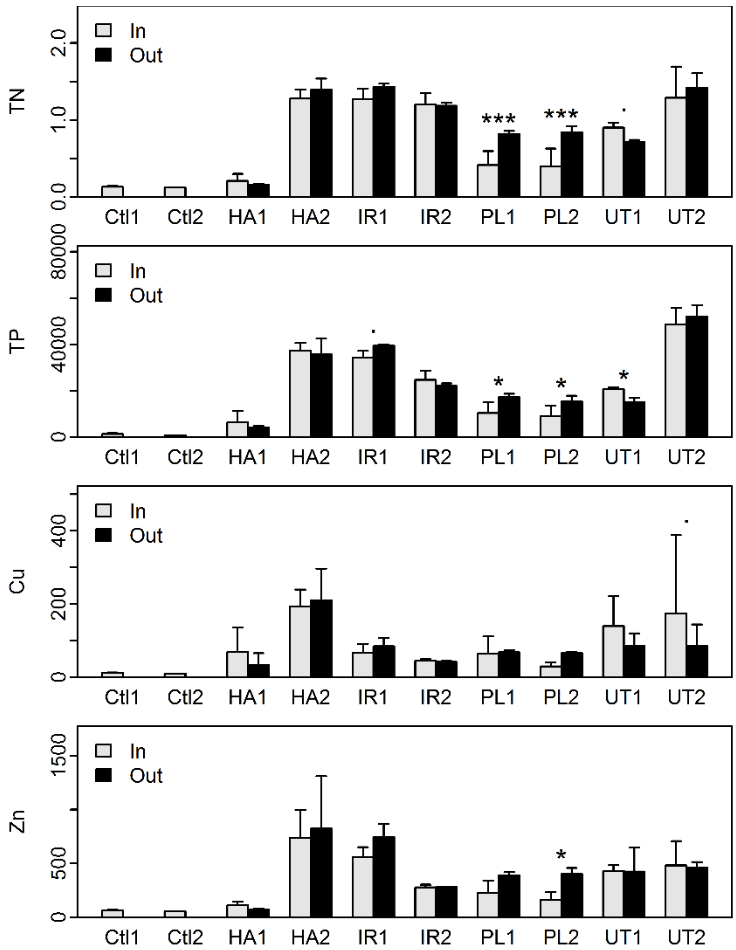


Figure 10. Comparison of sediment nutrients (total nitrogen [TN] and total phosphorus [TP]) and metals (copper [Cu] and zinc [Zn]) from samples taken within ('In') and outside ('Out') the treatments plots at the conclusion of the study, four months after initial treatment. Error bars represent standard error. The significance of the difference between each pair (In/Out) represented by: '·' = P < 0.1, '*' = P < 0.05, '**' = P < 0.01, '***' = P < 0.001. No outside samples were collected at the control sites. Ctl = Control, HA = Harrow, IR = Irrigation, RE = Removal, UT = Untreated.

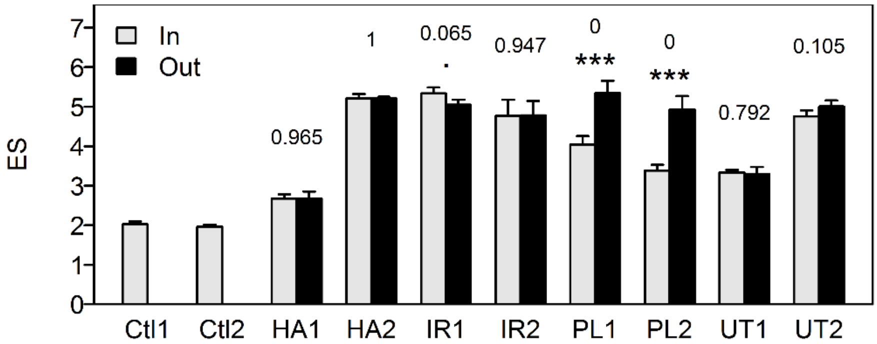


Figure 11. Comparison of overall enrichment stage in sediments sampled from within ('In') and outside ('Out') the treatment plots at the conclusion of the study, four months after initial treatment. Error bars represent standard error. The significance of the difference between each pair of bars represented by '·' = P < 0.1, '*' = P < 0.05, '**' = P < 0.01, '***' = P < 0.001. No outside samples were collected at the control sites. Ctl = Control, HA = Harrow, IR = Irrigation, RE = Removal, UT = Untreated.

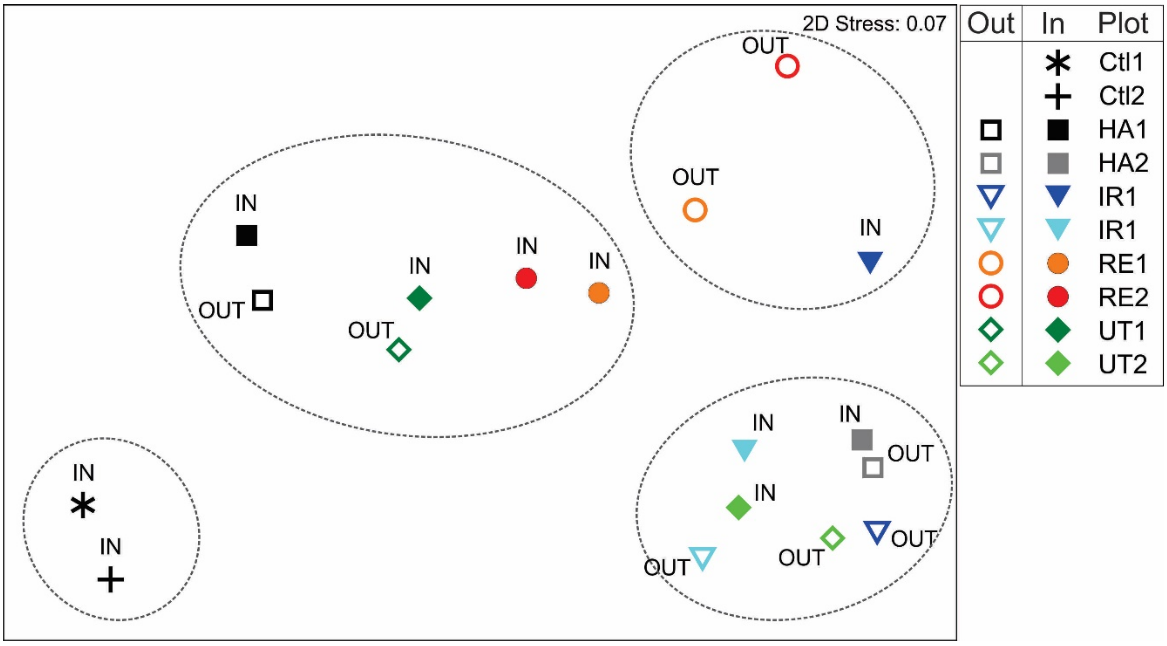


Figure 12. Two-dimensional multidimensional scaling (MDS) configuration of distances between centroids on the basis of Bray-Curtis similarities of fourth-root transformed macrofauna count data from final survey (Day 124). Clusters overlaid at 50% distance level.

5. DISCUSSION

The level of water column disturbance associated with the pilot-scale seabed remediation trials was relatively minor. The highest incident of increased turbidity occurred following the Removal trials and represented an approximate 4-fold increase from a background level of ~1 NTU to 4.2 NTU, but returned to background levels by 90 minutes post-treatment. Notable plumes also occurred in conjunction with Harrow treatment. Following the Removal treatments, DO levels decreased correspondingly, from approximately 7.8 mg l^{-1} to 6.8 mg l^{-1} (in the most affected area near the seabed), which represented a reduction to 87% saturation. To put these changes in to context, near-bottom DO levels in the outer Pelorus Sound are commonly in the range of 8–10 mg l^{-1} (Gibbs *et al.* 1991) and the Australia and New Zealand water quality guidelines specify a lower limit of 90% saturation in marine areas to avoid risk of adverse effects (ANZECC 2000b). Associated nutrient release during the disturbances was also biologically insignificant, with only very minor increases detected in near-bottom waters relative to concentrations pre-treatment and other previously recorded levels (Gibbs *et al.* 1992) and all remained well below national guidelines standards (ANZECC 2000b). Additionally, in most cases the observed plumes dissipated appreciably after 90 minutes. Therefore, the disturbances associated with these pilot scale trials did not pose any significant ecological threats in terms of reduced water quality.

Although the changes in water quality were relatively minor during this experiment, and it may therefore be inferred that the techniques are appropriate from an environmental perspective, there may be scale-related effects and the result should be treated with caution. For example, the total area that was harrowed was approximately 30 m^2, and this was conducted in controlled manner using a winch. A substantially larger plume is likely to develop if the entire 12,000 m^2 site was to be harrowed, and at speed, when being towed behind a boat. With regards to the Removal method, which was intended to simulate removal of sediments (*i.e.* dredging), the re-suspension effects are likely to be somewhat dependant on the method of dredging. A carefully operated suction dredge, for example, may have minimal disturbance on the seabed due to the extraction of the otherwise re-suspended particles. Therefore, although this study provides a valuable indication as to the likely water-column effects, if up-scaling of one of these treatments was to occur, then further investigations should be conducted to evaluate the effects at a semi-commercial scale (*e.g.* over 100–500 m^2) to ensure the risk is properly evaluated.

In terms of the effectiveness of the three proposed treatments for remediating the seabed, only the Removal treatment consistently and significantly improved benthic condition. At the conclusion of the 4-month trial period, the sediments within the Removal plots had significantly lower TOM, TFS, TN, TP and Zn (and lower but not significantly, Cu). The Removal plots also had a physically different sediment grain size and a correspondingly higher number of taxa and BQI. Total abundance was marginally increased, but not significantly, while the AMBI remained comparable. This suggests that the Eco-Group classification structure of the macrofauna concerned (and therefore enrichment tolerance, see:Borja *et al.* 2000; Keeley *et al.* 2012a) was comparable, despite the greater taxa richness. It can therefore be concluded that the 'removal' of the top 10–15 cm of sediment effectively removed much of the organically enriched layer along with some of the associated contaminants, and as a result the underlying sediments could be recolonised more readily by wider range of macrofauna. This acceleration in recolonisation and improved sediment chemistry resulted in an overall reduction in ES scores in the Removal treatments by 1.0–1.5 ES.

Although the observed reductions in copper and zinc concentrations appeared relatively minor, they had a critical net effect of reducing the average value to below levels in the relevant sediment quality guidelines (*e.g.* ANZECC Interim Sediment Quality Guidelines, ANZECC 2000b). For example at the conclusion of the experiment, the average zinc concentrations inside RE2 plot was 160 mg kg^{-1} compared to 400 mg kg^{-1} outside of the plot. The ANZECC ISQG-Low[1] trigger threshold for zinc is 200 mg kg^{-1}. Likewise, the copper concentration was 29 mg kg^{-1} inside the plot, and 65 mg kg^{-1} outside of the plot. The ISQG-Low trigger threshold for copper is 65 mg kg^{-1}. Similar results occurred at RE1, although the reductions were less pronounced, partly due to a high degree of variability in the copper result, which is probably the result of anti-fouling paint chips in one of the samples spiking the result (Sneddon *et al.* 2012). This finding is significant, because it suggests that careful removal of the top 10–15 cm of sediment can remove a significant fraction of the trace metals that have accumulated over several years and reinstate the site's compliance with ANZECC (2000b) guidelines. A logical next step in this investigation would be to profile the trace metal concentrations through the sediments to determine the optimum depth of sediment for removal.

Sediment removal would require a safe and effective waste disposal mechanism to be economically viable and environmentally sound. If this was not possible, leaving the sediments *in situ* may be the more prudent option. The two main disposal options that exist are: deep sea disposal (reliant on dilution and dispersal), and containment on land (with or without pre-treatment). There are numerous issues that arise when considering these options that would need to be carefully considered before heading down this path, but this is beyond the scope of this study. It is, however, useful to note that there are potentially useful precedents set by other industries for both of these practices (*e.g.* dredge spoil), as well as a reasonable body of information that considers their effects (*e.g.* Bolam *et al.* 2005). Should removal prove impractical for any of these reasons, it may be preferable to leave the contaminated layer to be buried by additional sedimentation. In a recent study in to the fate and toxicity of

[1] If the lower sediment quality guideline (ISQG-Low) for a particular contaminant is not exceeded, the chemical is unlikely to cause any biological impact on organisms inhabiting that sediment.

elevated copper concentrations in sediments, MacLeod *et al.* (2004) found that benthic recovery was seemingly unaffected due to limited biological availability of the copper, and therefore that continuation of farming may be advisable. This was because the greater depositional rates associated with salmon farms would theoretically accelerate burial of the contaminated layer below the 'biologically active' zone. However, the validity of such an '*in situ* capping' approach is contingent upon there being no further inputs (preventing further accumulation) and additionally, it does not entirely deal with the potential legacy issue of leaving a chemically altered sediment profile. Hence, the best approach will likely be determined by situation specifics and societal demands.

While this study has helped to elucidate the potential to accelerate benthic recovery from a highly enriched state, the related issue of rate of re-impact is also important and remains poorly described. Sediment can become highly enriched from a near-natural state in the space of three months in response to heavy biodeposition from salmon farms, and this rate of re-impact is likely adversely influenced by the presence of residual impacts (Keeley *et al.* 2015). This is because a system's ability to respond to inputs is influenced by the inherent physical and biological properties of the sediments (Macleod *et al.* 2007). In the case of the Removal treatment, the underlying sediments were exposed, which had different physico-chemical properties and were initially largely depauperate. Some recolonisation occurred, however, this was primarily by opportunistic species, which tend to inhabit the surface layers, as opposed to larger, deep-burrowing organisms indicative of the later stages of succession (Pearson & Rosenberg 1978). Exactly how this affected the ability of the macrofauna to utilise the 'new' upper layer of the sediment matrix, remains unknown. Therefore the effects on long-term recolonisation and the ability to respond to farm reintroduction are also undetermined. The latter is a particularly important consideration if sediment removal was to be considered as a maintenance strategy (*i.e.* repeated routinely to maintain a farm).

There are also some other potential issues that this study did not consider in detail, which should be addressed before prior to any larger-scale treatments. For example, dinoflagellate cysts can remain dormant in the sediments for many years and if released into the water column under the right conditions may result in proliferation or blooms (Eriksen *et al*. 2012). It would therefore be advisable to examine the sediments for the presence of these potential pathogens as part of a site remediation pre-assessment process.

Overall, shallow sediment removal (of the top 10–15 cm of sediment) appears to have the greatest potential as a method for accelerating seabed recovery and reducing copper and zinc concentrations in sediments. Effective commercial implementation of this approach, however, will be contingent upon the development of responsible sediment disposal mechanisms. There are also potential scale-related effects that should be further evaluated with larger-scale trials. The response of remediated sediments and the rate of re-impact also require further examination in future trials.

6. ACKNOWLEDGEMENTS

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8. APPENDICES

Appendix 1. Commercial perspective on Cawthron Remediation project—Mr Mark Gillard (The New Zealand King Salmon Co. Limited).

Seafood Innovations Limited

Commercial perspective on Cawthron Remediation Project –

Mark Gillard

The New Zealand King Salmon Company Ltd

In 2014 Cawthron carried out a series of trials to determine whether there were benefits from a range of processes to improve the recovery of heavily impacted seabed under a salmon farm.

The farm chosen was a NZ King salmon site situated in a low flow area, and in a state of fallow. The seabed was heavily impacted.

Recovery to its original state through natural process takes many years. It has been demonstrated for this farm that full recovery can be up to 10 years. This is a long time for a site to be unused. Shortening this period or allowing for semi continual use will be greatly beneficial.

A range of methodologies were used to assess improvement, and the best was determined to be sediment removal.

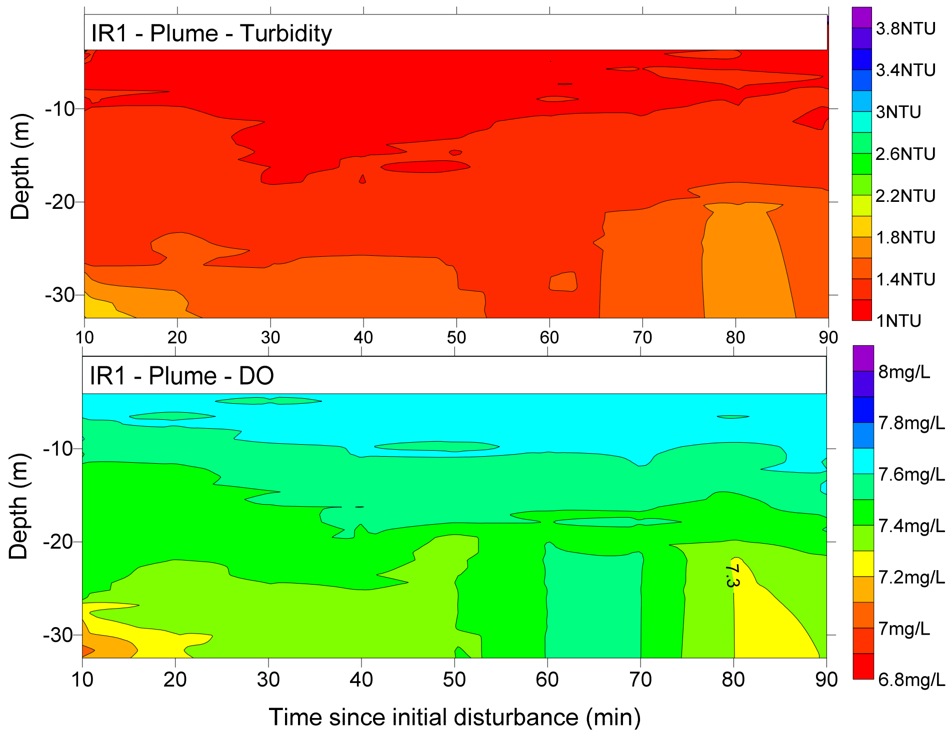
With the correct equipment we could envisage removing a thin layer from the seabed to accelerate assimilation, recolonisation and allow continued use of the site. However the problem of suspended noxious material from the process potentially affecting fish above and other sea life is a potential problem to be overcome before full scale sediment removal could be contemplated.

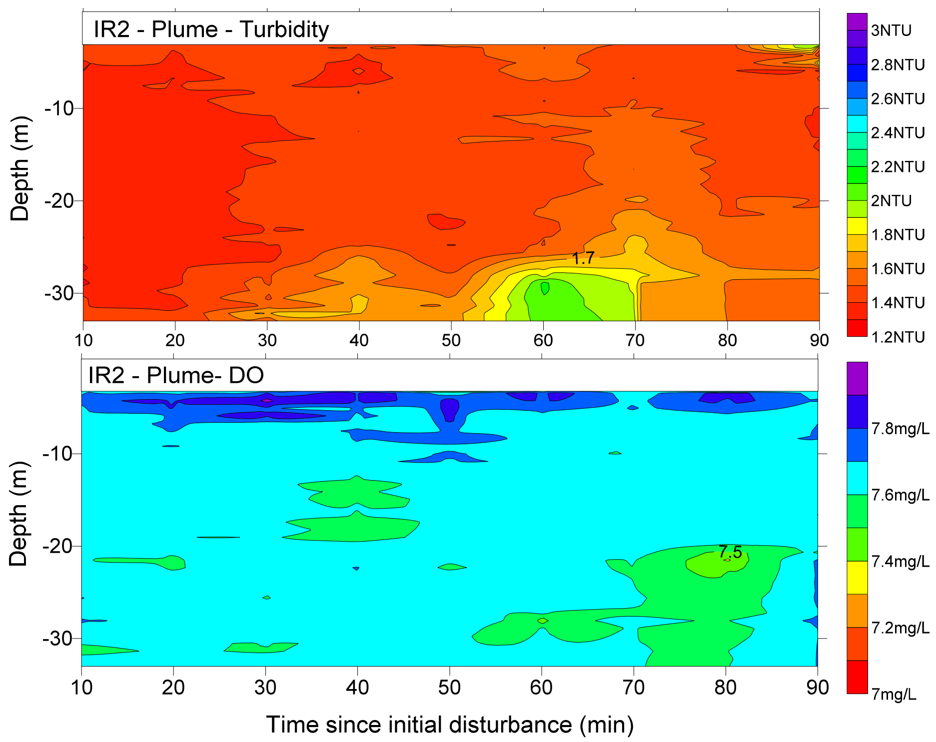
The trials were carried out over a very limited area and constraints. Therefore it is imperative that a larger commercial scale trial be carried out before consideration could be given to adopting it for commercial use. Determination of the correct equipment including collecting, handling, storage, transport and disposal requires further work. Options might include a vacuum dredge and bulk tanker fitted with a suitable treatment process/plant.

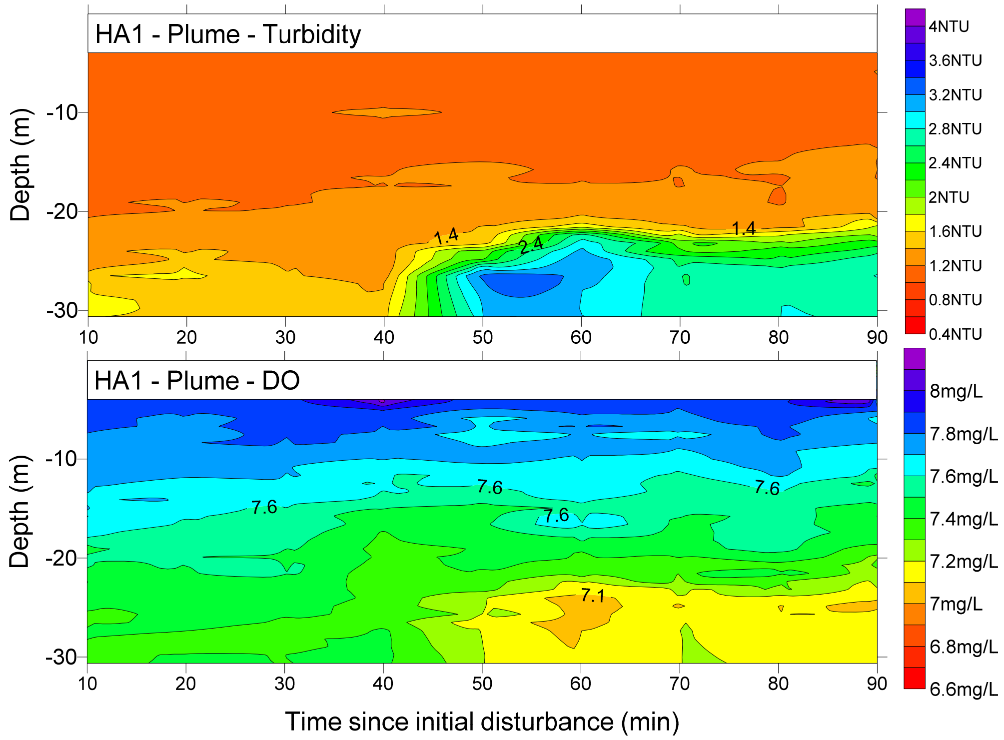
Disposal/use is another area that could be problematical given the type of material being processed; however this is doable in our opinion but requires investigation. Vacuum dredges could be adapted from existing plant and equipment currently being used in the aquaculture or other industries. Some discussion has already been had with a potential manufacturer who has shown interest.

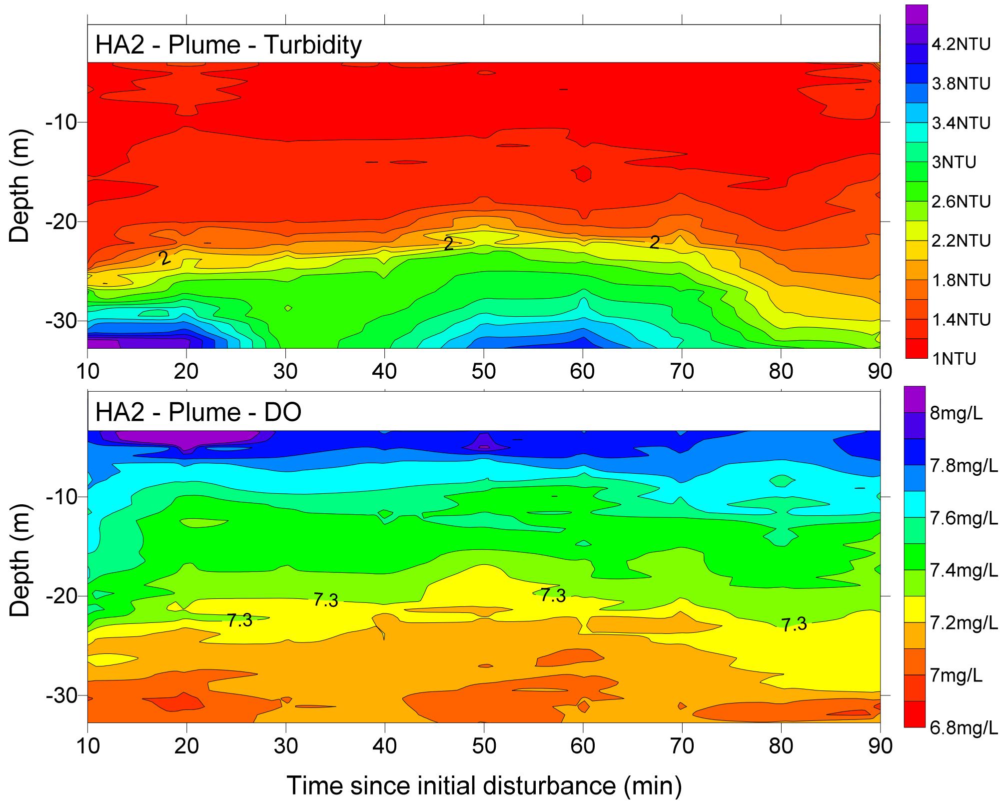
In summary the trial has given some good information regarding the merit of various options to improve the seabed, and should be investigated further to identify a commercially sensible process for maintaining the seabed under salmon farms in a state compliant with consent conditions and without compromising the environment.

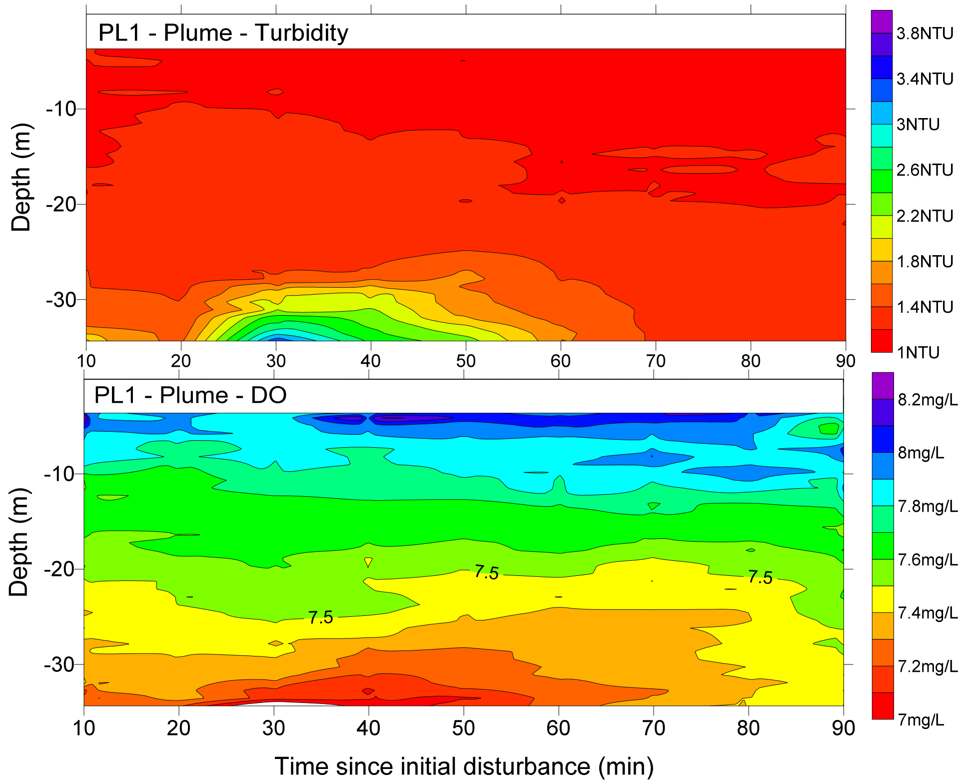
Appendix 2. Temporal and spatial two-dimensional interpolation of dissolved oxygen (DO, mg l^{-1}) and turbidity (nephelometric turbidity units; NTU) profiles from CTD (conductivity-temperature-depth) casts through the water column following the plume (via near-bottom drogue) for the 90 minutes after each treatment.

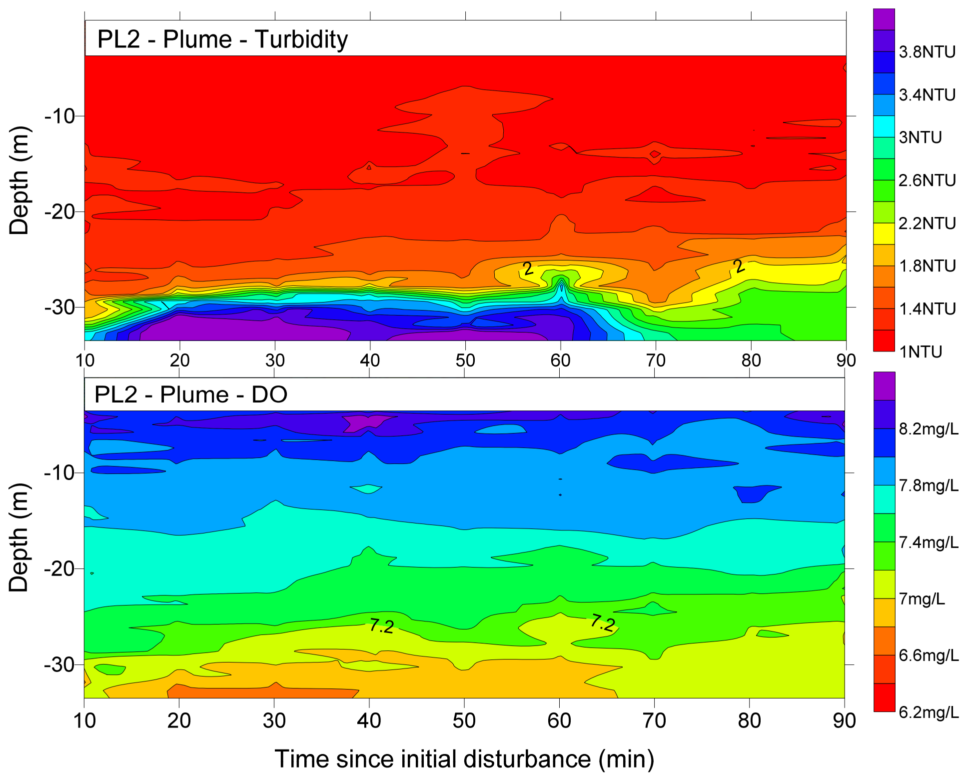




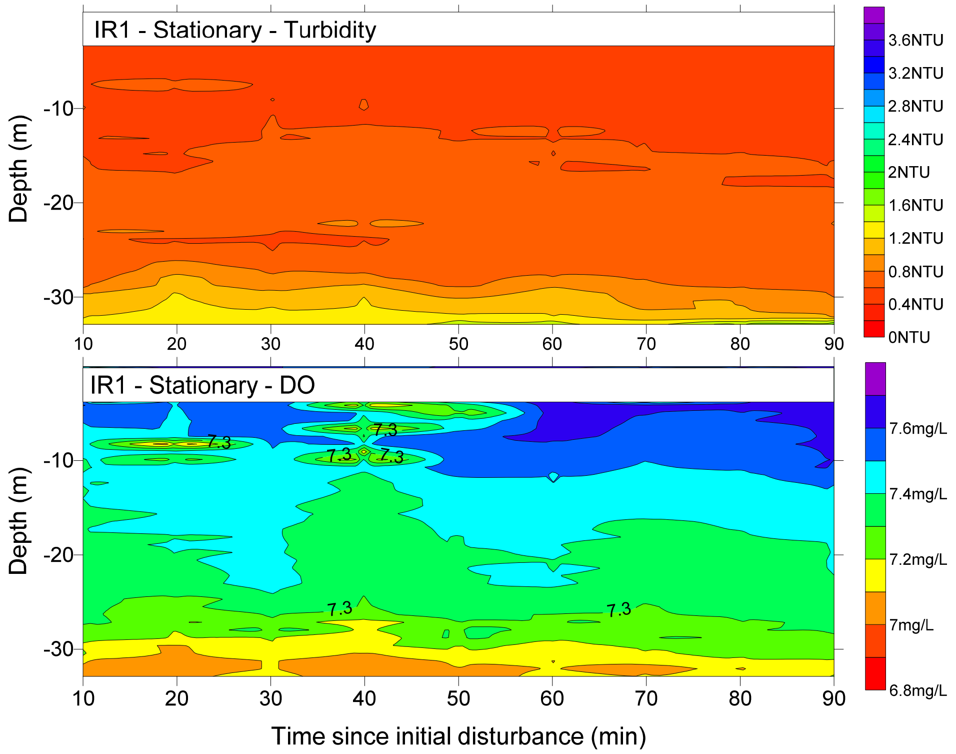

HA1 - Plume - Turbidity
HA1 - Plume - DO
Depth (m)
Time since initial disturbance (min)
4NTU
3.6NTU
3.2NTU
2.8NTU
2.4NTU
2NTU
1.6NTU
1.2NTU
0.8NTU
0.4NTU
8mg/L
7.8mg/L
7.6mg/L
7.4mg/L
7.2mg/L
7mg/L
6.8mg/L
6.6mg/L

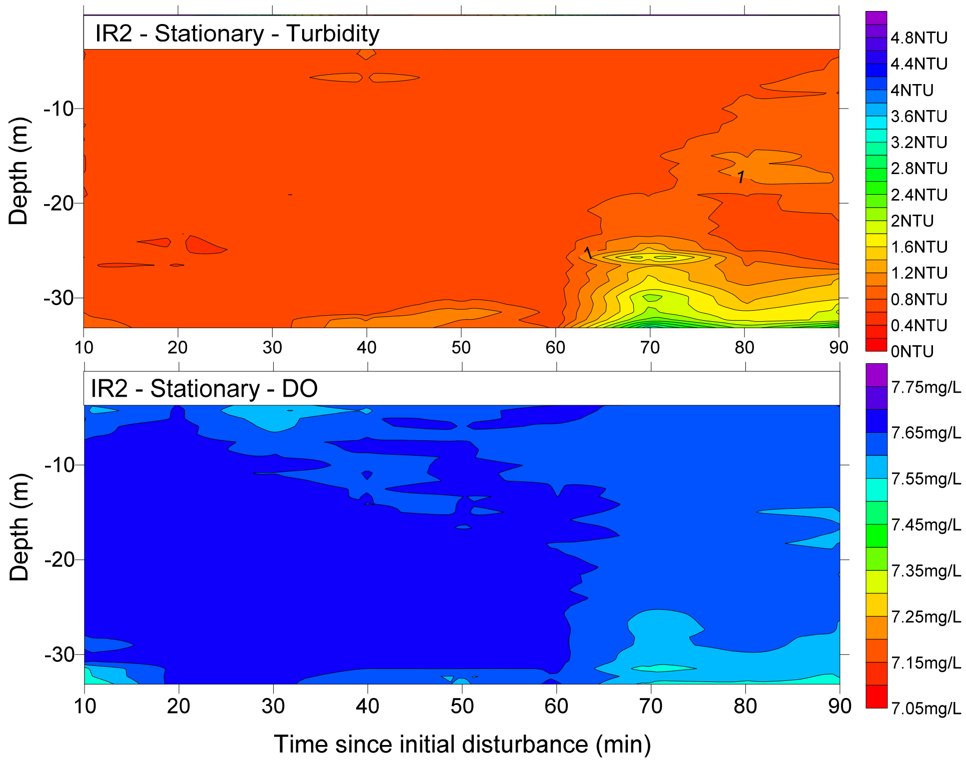

HA2 - Plume - Turbidity
HA2 - Plume - DO
Depth (m)
Time since initial disturbance (min)
4.2NTU
3.8NTU
3.4NTU
3NTU
2.6NTU
2.2NTU
1.8NTU
1.4NTU
1NTU
8mg/L
7.8mg/L
7.6mg/L
7.4mg/L
7.2mg/L
7mg/L
6.8mg/L

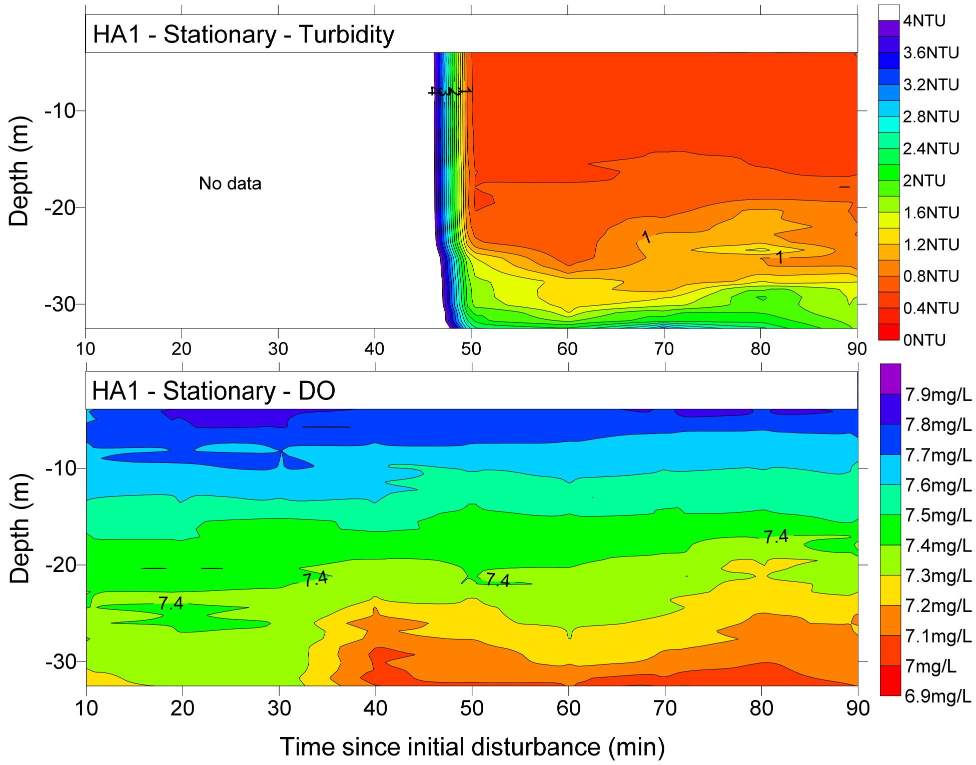

PL1 - Plume - Turbidity
Depth (m)
-10
-20
-30
10
20
30
40
50
60
70
80
90
3.8NTU
3.4NTU
3NTU
2.6NTU
2.2NTU
1.8NTU
1.4NTU
1NTU
PL1 - Plume - DO
7.5
8.2mg/L
8mg/L
7.8mg/L
7.6mg/L
7.4mg/L
7.2mg/L
7mg/L
Time since initial disturbance (min)

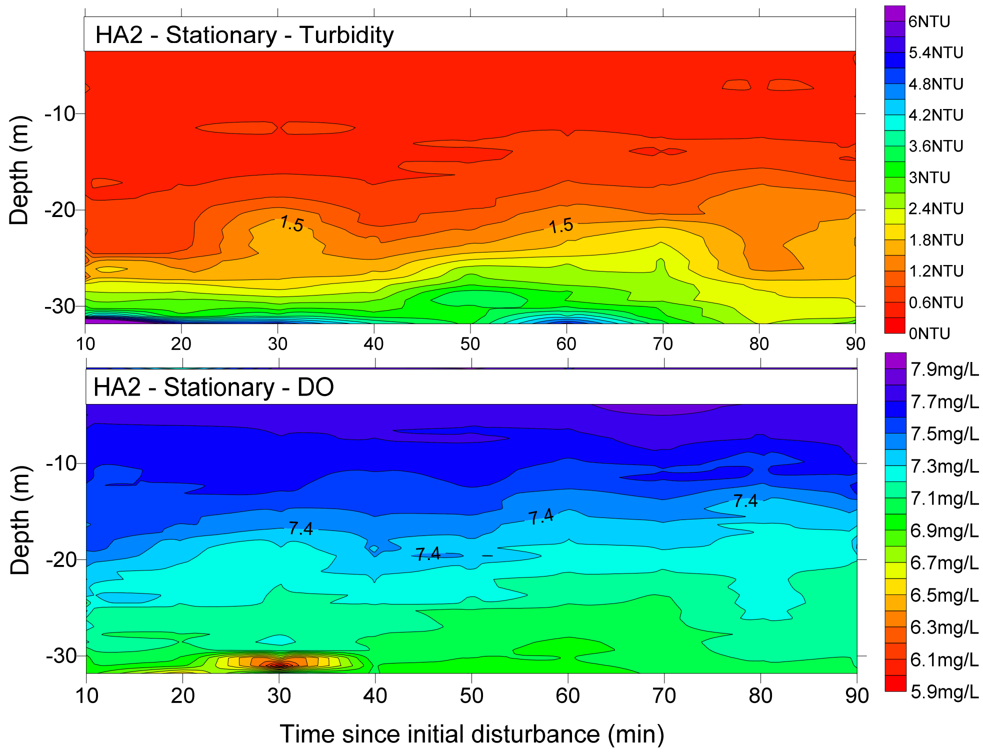

PL2 - Plume - Turbidity
Depth (m)
-10
-20
-30
10
20
30
40
50
60
70
80
90
2
3.8NTU
3.4NTU
3NTU
2.6NTU
2.2NTU
1.8NTU
1.4NTU
1NTU
PL2 - Plume - DO
7.2
8.2mg/L
7.8mg/L
7.4mg/L
7mg/L
6.6mg/L
6.2mg/L
Time since initial disturbance (min)

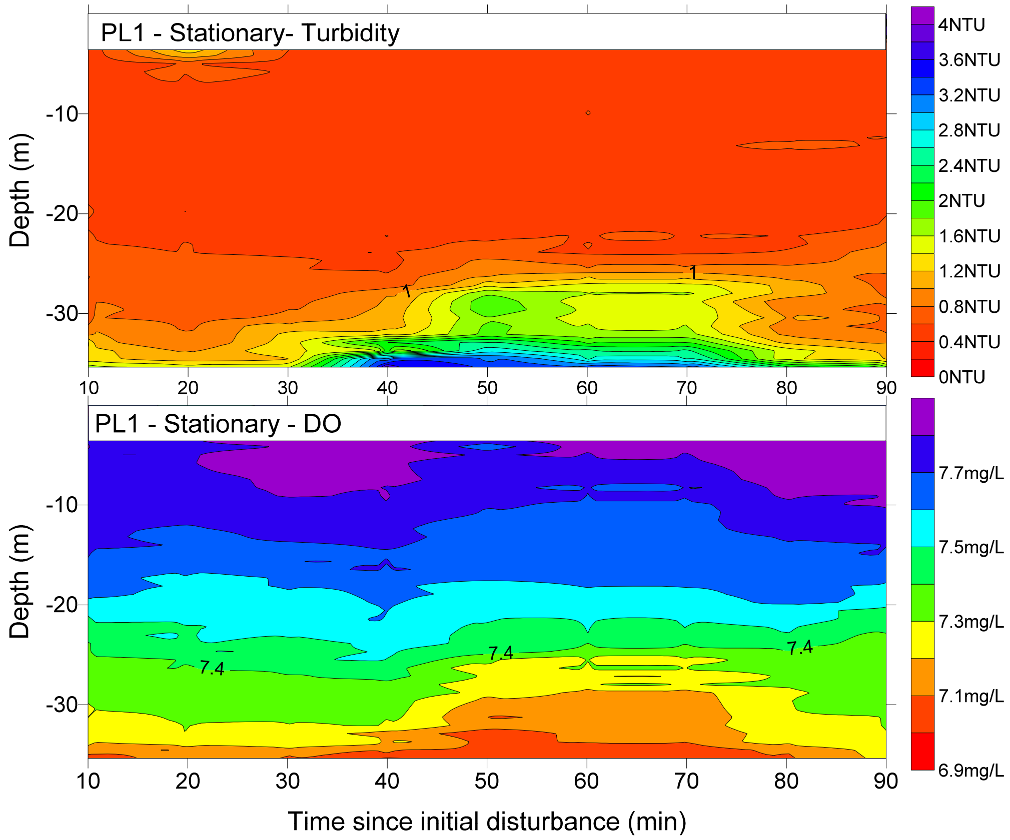
Appendix 3. Temporal and spatial two-dimensional interpolation of dissolved oxygen (DO, mg l^{-1}) and turbidity (nephelometric turbidity units; NTU) profiles from CTD (conductivity-temperature-depth) casts through water column following at each treatment site (stationary) for the 90 minutes after each treatment.

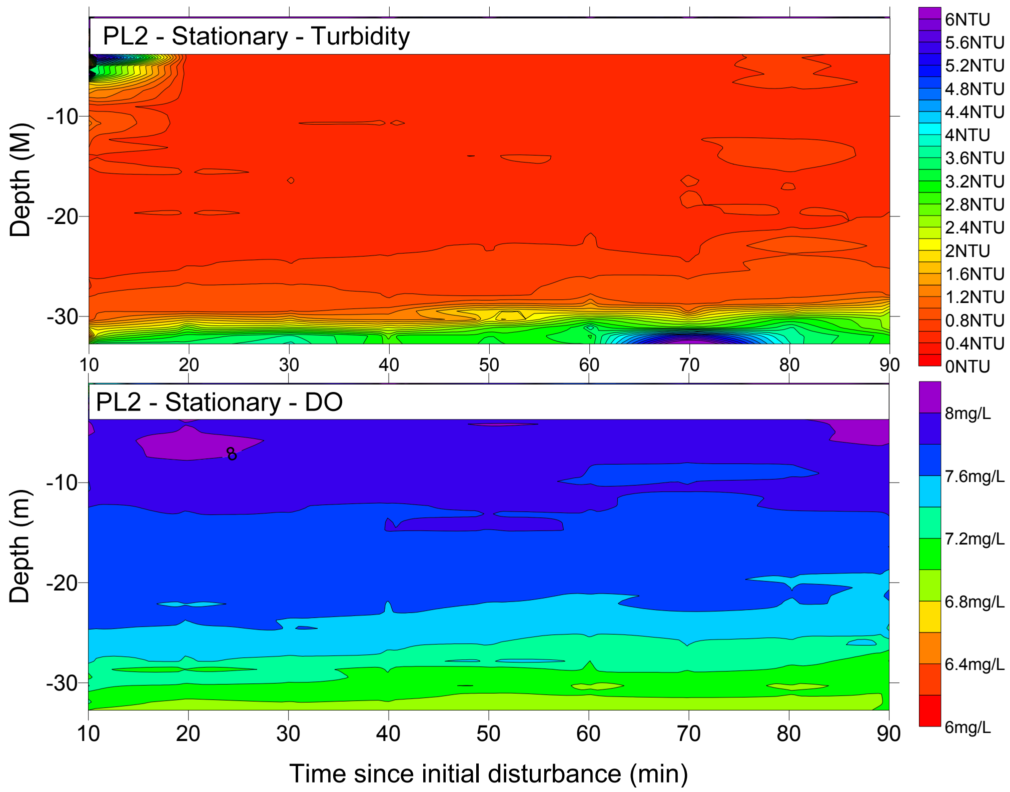





HA1 - Stationary - Turbidity
No data
Depth (m)
-10
-20
-30
10
20
30
40
50
60
70
80
90
4NTU
3.6NTU
3.2NTU
2.8NTU
2.4NTU
2NTU
1.6NTU
1.2NTU
0.8NTU
0.4NTU
0NTU
HA1 - Stationary - DO
7.4
7.9mg/L
7.8mg/L
7.7mg/L
7.6mg/L
7.5mg/L
7.4mg/L
7.3mg/L
7.2mg/L
7.1mg/L
7mg/L
6.9mg/L
Time since initial disturbance (min)


HA2 - Stationary - Turbidity
Depth (m)
-10
-20
-30
1.5
10
20
30
40
50
60
70
80
90
6NTU
5.4NTU
4.8NTU
4.2NTU
3.6NTU
3NTU
2.4NTU
1.8NTU
1.2NTU
0.6NTU
0NTU
HA2 - Stationary - DO
7.4
7.9mg/L
7.7mg/L
7.5mg/L
7.3mg/L
7.1mg/L
6.9mg/L
6.7mg/L
6.5mg/L
6.3mg/L
6.1mg/L
5.9mg/L
Time since initial disturbance (min)


PL1 - Stationary- Turbidity
PL1 - Stationary - DO
Depth (m)
-10
-20
-30
10
20
30
40
50
60
70
80
90
1
7.4
Time since initial disturbance (min)
4NTU
3.6NTU
3.2NTU
2.8NTU
2.4NTU
2NTU
1.6NTU
1.2NTU
0.8NTU
0.4NTU
0NTU
7.7mg/L
7.5mg/L
7.3mg/L
7.1mg/L
6.9mg/L


PL2 - Stationary - Turbidity
PL2 - Stationary - DO
Depth (M)
Depth (m)
-10
-20
-30
10
20
30
40
50
60
70
80
90
8
Time since initial disturbance (min)
6NTU
5.6NTU
5.2NTU
4.8NTU
4.4NTU
4NTU
3.6NTU
3.2NTU
2.8NTU
2.4NTU
2NTU
1.6NTU
1.2NTU
0.8NTU
0.4NTU
0NTU
8mg/L
7.6mg/L
7.2mg/L
6.8mg/L
6.4mg/L
6mg/L