INCORPORATION OF SPATIAL GRADIENTS INTO BENTHIC IMPACT ASSESSMENT

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Incorporation of Spatial Gradients into Benthic Impact Assessment

By

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A thesis submitted to the School of Graduate Studies

in partial fulfillment of the requirements for

the degree of Doctor of Philosophy

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Abstract

Before-After Control Impact (BACI) sampling designs are commonly used in environmental impact assessment and are considered the most effective for detecting changes due to anthropogenic disturbances. These designs handle local spatial variability through randomized placement of samples into a treatment stratum and one or more control strata. While BACI designs based on agronomic block procedures are appropriate for disturbances that have defined boundaries, these designs suffer from serious limitations when applied to point source disturbances where the scale of disturbance is unknown. When a contaminant disperses with distance from a point source it is suggested that a 'gradient' design will be more sensitive to change than randomized placement of samples. This requires allocating samples according to distance, rather than by random placement within randomly placed strata (blocks). This thesis develops the use of gradient designs for environmental impact assessment.

Gradient versus random sampling designs were compared using data from an oil field in the North Sea. The gradient sampling designs were more powerful than a randomized block design for point source disturbances, and enabled the scale of the disturbance to be readily identified. A gradient layout also avoids the problem of selecting adequate control sites, and gradient designs lend themselves to constructing mechanistic models that generate testable predictions of attenuating effects with increasing distance. The assumptions of gradient designs are that no natural gradients exist or that natural gradients do not effect community response to the anthropogenic disturbance. This assumption was tested using data where an increasing physical gradient due to increased wind/wave and tidal disturbance existed with distance from a sewage outfall at Manukau Harbour, New Zealand.

Statistical methods to analyze data from gradient designs are reviewed. A method using regression to estimate the magnitude of spatial gradients successfully separated sensitive and tolerant species. Ordination and gradient techniques differed in their ability to generate and test causal models and to identify the extent of the disturbance. Ordination techniques were more sensitive at detecting community change, while gradient methods lead to quantitative predictions than can be readily tested.

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To my parents, Warren and Margaret

for your love, support, and for teaching me to climb

more mountains and pick more daisies

And to Andrew, for all your love

from head to toe

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Chapter 1. Introduction

'In reality, most progress in ecological and environmental research has been made by combining field observations with controlled experimentation. However, the observational stages of this process are not designed in any rigorous way. Many ecologists now appreciate the need for a carefully laid out, formal design for experimentation, but few carry that concept to the logical conclusion, which is that concurrent field investigations should also be conducted in the framework of a sampling design. We believe that this may be one of the most important areas for future research'.

Eberhardt & Thomas (1991)

Assessments of environmental impacts are subjected to greater scientific and legal scrutiny as the risk of environmental degradation is balanced against the cost of impeding development and economic growth (Smith *et al.*, 1993; Mapstone, 1995). Decisions about environmental impacts have important consequences, whether they are correct or in error (Bernstein & Zalinksi, 1983; Andrew & Mapstone, 1987; Peterman, 1990; Fairweather, 1991). For example, an incorrect conclusion that an 'impact' has occurred (Type I error, α) may result in the unwarranted curtailment of development. On the other hand, concluding that no deleterious impact has occurred usually provides tacit support for continued development. An incorrect conclusion of 'no impact' (a Type II error, β) might mean that serious environmental degradation occurred before the real impact was noticed, possibly resulting in severe pollution, the destruction of a fishery, or the extinction of local species. The ability of an environmental assessment to correctly detect an impact is determined by the sampling design, such as the sample size, placement of samples, and the number of replicate samples. The requirement for carefully planned designs is therefore of particular concern for environmental assessment given the environmental, social and economic consequences of decisions based on these assessments.

The development of sampling methods in applied science initially focused on obtaining reliable population estimates for resource surveys. Methods for estimating a population or resource include simple random, stratified, systematic, cluster, line transects, and capture-recapture sampling methods (Cochran, 1966; Deming, 1966; Hansen *et. al.*, 1966; Kish, 1967; Stuart, 1984; Thompson, 1992; Thompson & Seber, 1996;) and more recently a proposed line transect placed along a changing resource gradient (Gillison & Brewer, 1985). Stratified sampling and the use of gradient line transects incorporate the idea of proportionally sampling a population or resource to provide a better estimate of the population abundance. For example, in stratified sampling designs, the entire population is divided into distinct sub-populations (strata). The number of samples within each stratum is then accomplished by simple random sampling.

Sampling designs were further developed for use in impact assessment following the implementation in 1969 of the U.S. National Environmental Policy Act (Turner & Gardner, 1991) which required ecological assessment of anthropogenic impacts on surrounding ecosystems. Sampling to detect anthropogenic disturbance rather than to estimate a population required special considerations of estimating spatial and temporal variability, but is still based on the principle of randomly sampling within strata. The development of stratified sampling designs to detect environmental impacts has been as follows.

1) Before/After contrasts at a single site.

Impact assessment programs initially sought to measure, at the site to be disturbed, as many biological, physical and chemical variables as the investigator could list or afford to measure (Morrisey, 1993). This design focused on describing the environment before any impact occurred and was often seen as the main part of an environmental impact assessment (Hilborn & Walters, 1981). Changes in biological variables after a disturbance could then be attributed to the activity. These monitoring programs typically sampled one site, the impacted site, and in some cases (for example oil spills), only the 'after' conditions were recorded.

2) Before/After and Control/Impact sampling of sites.

Green (1979) first recommended sampling an impact and a control site, before and after a disturbance, as an optimal design for environmental impact assessment. The design is based on the principle that if two locations (control and impact) are monitored before an anthropogenic disturbance, the impact location will show a different pattern after the disturbance than will the control location. Green's 'optimal impact design' emphasised the necessity of a control site. Green's design is based on sampling two spatial strata, a treatment and a control site. Sampling within each stratum is randomized to handle the problem of local variability. Interestingly Eberhardt (1976) had also suggested a control and an impact area should be observed before and after an event in a "station-pairs" or "ratio-method" approach. However Green (1979) is generally credited for developing stratified designs for impact assessment.

3) Repeated BACI, or BACIP.

Hurlbert's (1984) monograph raised several problems concerning the appropriate design of a sampling program where either treatments are not replicated or replicates are not statistically independent. Hurlbert termed this problem 'pseudoreplication'. For BACI designs problems arise as replicates are independent only if the contamination is randomly assigned to experimental units. The experimental units in this case are the control/impact strata. Random variation among a small number of strata can be mistaken for environmental impacts if the wrong statistical model is used. Therefore, another term for this problem could be 'strata-effects'. This led to the modification of Green's method to the before after control impact paired (BACIP) sampling design. Stewart-Oaten *et al.* (1986) and Bernstein and Zalinski (1983) developed BACI and paired BACI sampling designs. Stewart-Oaten's BACI(P) model differed from Green's model by explicitly including estimates of temporal variability, as well as spatial variability of replicates. These repeated BACI or BACIP designs sample each site several times, at random, prior to and then after the start of the potential disturbance. These authors pointed out that several periods of sampling before and after a disturbance were necessary to provide a representative picture of the mean abundance, and showed the statistical advantages of sampling at randomly chosen times. However, this design does not resolve problems of "strata effects" as only one impact and one control strata are sampled, hence contamination is not randomly assigned to experimental units and replication of units remains small.

4) Beyond BACI.

Underwood (1991) suggests that BACI designs are insufficient because any location-specific temporal difference that occurs between the treatment and control strata will be interpreted as an impact even if it has nothing to do with human disturbance. To reduce the chance of confounding between treatments and chance variation among units, Underwood (1992) suggested an asymmetrical design that compares the temporal change in a potentially impacted location with those in several randomly selected control locations. These asymmetrical designs assess natural variability of an ecosystem over several control sites and provide a more representative picture of natural variability in the affected area. For further discussion on asymmetrical designs and analyses of results see Underwood (1992, 1993, 1994).

5) Beyond block designs.

The development of BACI designs first proposed by Eberhardt (1976) and Green (1979) has greatly improved our ability to detect anthropogenic disturbances against a background of natural variability (Morrisey, 1993). However, these designs deal only with assessing an impacted area or strata versus one or more control strata. Possibly because of the historical link with designs from agricultural research, most surveys of living resources employ random and systematic block procedures or their derivatives. While these designs are appropriate for disturbances that have defined boundaries such as trawled versus un-trawled, and fertilised versus unfertilised areas, they are less appropriate for disturbances where the extent of the impact is unknown. Knowledge of the scale of an anthropogenic disturbance is important if the risks to the environment are to be correctly determined. Recently there has been considerable attention to the effects of scale in ecological studies (O'Neill, 1989, Wiens 1989, Rahel 1990, Thrush 1991, Levin 1992, Schneider 1994a 1994b). However, in environmental assessment the stratified BACI sampling designs currently used have not been developed to be able to readily determine the scale of an impact. An alternative to stratified designs for disturbances where the scale of the impact is unknown is proposed and developed in this thesis. It is here suggested that when environmental disturbances attenuate with distance, such as sewage and industrial outlets, underwater blasting events, smelter outputs etc., the use of gradient designs which sample as a function of distance from the discharge are more efficient than currently used block designs. Gradient designs are evaluated with respect to their power to detect an impact, and their ability to identify the spatial scale of a disturbance. Assumptions of the design are tested and statistical models are developed to analyse data from gradient designs.

This thesis is divided into six Chapters. Chapter 2 develops guidelines on the appropriate application and assumptions of Before After Gradient (BAG) designs and stratified Before After Control Impact (BACI) designs. Chapter 3 evaluates the power of a gradient design to detect environmental impacts where the scale of the disturbance is unknown. Chapter 4 tests methods of analysis for data from a gradient layout. Data of benthic macrofaunal density from around oil platforms in the North Sea are used to test these ideas. Chapter 5 evaluates the power of a gradient design to detect an anthropogenic impact in an environment with multiple natural physical gradients. The presence of environmental stresses such as salinity, sediment particles size, or depth gradients associated with an anthropogenic disturbance can make separation of the relative disturbances difficult. The rigor of gradient designs for detecting an anthropogenic disturbance in such environments was investigated using data of benthic macrofaunal density from Manukau Harbour, New Zealand. The final Chapter summarizes the contributions of this thesis.

Throughout this thesis, examples of species tolerant and sensitive to environmental disturbance are provided. Analysis of Covariance for species macrofaunal abundance as a function of distance was used to test whether gradients were heterogeneous among species. Regression coefficients were calculated for all species, and are provided in Appendices 1, 2, and 3. Tolerant species were identified by a negative coefficient, where species numbers were high next to the platform and decreased with distance form the pollution source. Sensitive species were identified by a positive coefficient, where species number increased with distance from the platform. The most strongly negative and positive regression coefficients are plotted to provide examples of tolerant and sensitive species.

The data sets analysed in this thesis are from surveys that recorded macrofaunal benthic abundance around oil fields in the North Sea. In the Norwegian sector, oil companies are legally required to submit results of environmental surveys around their installations to the State Pollution Board every second year. Quality control measures have been in place since 1987 (Reiersen *et al.*, 1989), and Norway has an open policy so that data is available to the public. The data sets obtained were from a 1990 survey at the Ekofisk platform, and surveys at the Gyda platform in 1987, 1990 and 1993. The Ekofisk samples were collected and identified by Akvaplan and Unilab Ltd. The Gyda samples were collected and identified by the Oil Pollution Research Unit Field Studies Council, U.K. Survey regulations in the Norwegian sector stipulate that all field studies be conducted in May or early June. The Ekofisk and Gyda field surveys were conducted in early June. Chapters 3 and 4 analyse data from the Ekofisk and Gyda oil fields. Chapter 5 analyses data of macrofaunal benthic abundance from Manukau Harbour that the author collected, sorted and identified while at the National Institute of Water and Atmospheric Sciences, New Zealand. The Manukau Harbour data was collected on the 22 November, 1996. Summary tables of the data used in this thesis are provided in Appendices 4, 5, 6, 7, and 8.

Chapter 2. Applications and assumptions of sampling designs to detect environmental impacts

2.1 Introduction

Stratified or block sampling designs are currently used to analyse most environmental impacts regardless of the nature of the disturbance. Distinction between a gradient impact, layout, statistical model and a stratified impact, layout, statistical model has not previously been considered. Attention to these distinctions will clarify the appropriate application of a sampling layout and statistical model given the environmental impact.

Block impacts are disturbances that have defined boundaries. Examples of block disturbances are trawled versus un-trawled, and logged versus un-logged areas. The appropriate sampling layout to detect a block environmental impact is to compare the impacted strata with a control area. For example, the benthic population characteristics and community composition in a trawled area (impact strata) would be compared to an un-trawled, control strata. Most environmental assessments are required to sample a minimum of once before the anthropogenic disturbance begins, and then to sample throughout the life of the activity of the project to ensure compliance monitoring. An environmental impact is detected by a significant interaction between the location (control/impact) and the time (before/after) variables. The appropriate statistical model is to test for a location or strata effect with time. An example of a block impact and the appropriate layout and statistical model is provided in Figure 2.1.

Gradient impacts are environmental disturbances that attenuate or diffuse with distance from point source disturbances. Some examples of gradient impacts are sewage outlets, underwater blasting events, and smelter outputs. The appropriate sampling layout is a gradient layout which samples as a function of distance from the discharge source. For example, changes in the benthic community composition as a function of an organic enrichment gradient with distance from a sewage outfall would be studied. The appropriate statistical model is to therefore test for an interaction in the distance (contamination gradient) and time (before/after) variables. An example of a gradient impact and the appropriate layout and statistical model is provided in Figure 2.2.

Hybrid designs occur when a block layout, or block statistical model, are applied to analyse a gradient impact and vice versa. Hybrid designs may result in a loss of sensitivity or power to detect an environmental impact. A stratified layout and statistical model should be used to detect a block impact, and are generally correctly applied in the literature. A gradient layout and statistical model should be used to detect a gradient impact. However, many examples occur in the literature where a block layout and statistical model are applied to detect gradient impacts. Examples include Holland *et al.* (1993), Reitzel *et al.* (1994), Chapman (1995), Faith *et al.* (1995), Hogg & Williams

(1996) and Otway *et al.* (1996a, 1996b). The sensitivity to detect a gradient impact is reduced if the distance information is blocked. Also, the ability to monitor changes in the spatial scale of the impact with time is reduced by sampling an impact and a control strata rather than quantifying the gradient of contamination. If a stratified statistical model is applied when effects are graded, then the variation within a block due to the graded effects will appear in, and hence inflate the error term, also reducing sensitivity (cf. Chapter 3).

The sensitivity of gradient and block designs also varies with respect to several other factors. This chapter will now evaluate both sampling designs with respect to their ability to detect the spatial scale of an impact, develop causal models, select adequate control sites, and evaluates their ability to detect impacts caused by environmental accidents where no "before" data sets exist. Table 2.1 provides a summary of these comparisons. The terminology of a gradient or block impact, sampling layout and statistical model will be used consistently throughout this thesis. The use of the term 'design' refers to all aspects of the study including the sampling layout and the statistical model used.

2.2 Use of control sites

The detection of human influences on population abundance in a particular environment is made difficult by natural variability of the population. A lack of spatial or temporal controls makes detection of an environmental impact difficult against a background of variation in numbers of organisms from site to site. The use of one or more control sites (Underwood, 1992) and several, preferably non-regular, times of sampling (Stewart-Oaten *et al.*, 1986) are, therefore, needed to provide a representative estimate of the mean abundance.

The requirements for control areas are that the site should be sufficiently distant so that it will not be affected by the potential disturbance, and yet close enough so that the control and impact areas are comparable. Exactly how far pollution will disperse in the marine environment, and whether two or more areas are comparable with respect to physical processes are both often unknown (Ellis & Schneider, 1997).

The temporally replicated BACI design assumes that the spatial scale of a putative impact is known before it occurs. For example (Underwood, 1992), a proposed outfall of warm water into an estuary may have only a local impact of a few hundred square meters. To detect this, appropriate control locations would be areas of a few hundred square meters elsewhere in the estuary. If, however, the estimated scale of impact is wrong and the outfall causes a change in abundance of some population over the whole estuary, such a sampling design will not detect it, because all the controls will be affected. Underwood

(1992) suggests that to cover this possibility, two spatial scales need to be sampled (bays, sites within bays). Therefore control locations must also be examined in other estuaries along the coast, independent of the warm water from the outfall. However, difficulties can arise as another estuary that can serve as a control may not exist, or may not be comparable with respect to physical processes.

Difficulties of satisfying basic design requirements of proper controls for block designs in lotic systems are discussed by Faith *et al.* (1991), Dostine *et al.* (1993), Humphrey & Dostine (1994) and Humphrey *et al.* (1995). A disadvantage in the use of multiple controls in a particular monitoring design is the possibility of introducing increased variability among the responses of different controls such that an impact passes undetected. For example, some of the controls may be relatively remote from the main sites of interest and so the natural variability between controls and the impacted site increases (Humphrey *et al.*, 1995). They also state that because of a lack of independence among sites in a single stream, discrete locations upstream of the disturbance still represent no more than 'pseudo-controls' or mere replicates of a single control. Selection of true controls in separate streams again could raise the possibility of one site behaving differently from others, so inflating the baseline variation in control-impact differences. While selection of valid controls for BACI designs is difficult, Underwood (1994) suggests that replication of control locations is no more difficult than choosing one control and that "clearheadness" be used to choose relevant control sites.

The assumption of block designs are that sampling procedures are consistent within the pre- and post impact periods. The assessor must also assume that the sites have in fact been correctly categorized. i.e. that the controls sites are adequate. Control sites need to be sufficiently distant so that they are not affected by the potential disturbance, and yet be close enough to be comparable with respect to physical processes to the impacted sites.

Gradient designs reduce the problem of arbitrary selection of a control site. Samples are taken as a function of distance from the contamination source. Development of contamination gradients can be monitored through time and can be used to ensure that the scale of the design is still adequate, i.e. still monitoring relative to background levels. The design does assume that no other natural environmental gradients occur, or that differences in natural gradients do not affect the pattern of community change in response to the anthropogenic disturbance. Wiens & Parker (1995) suggest that covariance analysis using measures of other environmental variables can be used to control for these effects.

2.3 Scale

The scale of the disturbance and the selection of control sites in BACI designs are interrelated problems. This is because knowledge of the scale of the disturbance is required in order to select adequate controls (cf. 2.2). However the spatial or temporal scale of the possible effects of a disturbance are often unknown before the disturbance happens. The choice of scales for sampling to detect the disturbance is then difficult if not impossible (Underwood, 1994). Green & Hobson (1970) and Underwood (1992) recognized this problem and suggested the use of hierarchical sampling of different spatial scales (bays, and sites within bays). The analytic framework and the appropriate F-ratio to detect impacts at different spatial and temporal scales were developed by Underwood (Table VI, 1992). The analysis of variance has 32 sources of variation in the model and hence requires at least 88 degrees of freedom. This is calculated using a minimum of 3 locations (2 controls and 1 impacted site), and a minimum number of sampling times (2 times of sampling, once before and once after the disturbance). A citation search of Underwood (1992) indicates that this analysis has not been adopted in any published impact assessment studies.

BACI designs are currently used to analyze point source disturbances that attenuate with distance. Smith *et al.* (1993) apply a BACI design to assess the effects of discharged cooling water from a nuclear power plant on the aquatic environment. They highlight problems such as trends in the measurements, failure to meet the assumptions of the model, confounding factors, and inadequacy of controls. BACI designs were also applied to point source disturbances such as sewage outfalls, cooling water impacts from nuclear power plants, and fuel leakage, with subsequent problems in the selection of adequate control sites (Underwood 1992, 1993). An alternative is to use a gradient design which is preferable for impacts that attenuate with distance. A gradient design enables the scale of the impact to be readily identified and avoids the use of hierarchical sampling of different spatial scales which can be costly and difficult to analyze.

Gradient designs enable spatial and temporal trends as a function of distance to be readily monitored. Development of contamination gradients relative to background levels can be monitored through time. Gradient methods also enable multiple physical disturbances that operate at different scales to be readily identified. Chapter 3 provides an example where a gradient model enabled the area affected by taint and smothering to be easily visualized. Stewart-Oaten (*in* Humphrey *et al.*, 1995) suggest that 'near' and 'far' impact sites should be included in block designs to observe whether or not purported effects wane, or show a time-lag in effect, with increasing distance from source. However the use of near and far sites again suffers from a problem that knowledge of the scale of the disturbance is required to correctly place strata.

2.4 Pseudo-replication

Eberhardt & Thomas (1991) state that the basic problem in impact studies is that evaluation of the environmental impact of a single installation of, say a nuclear power plant on a river, cannot very well be formulated in the context of the classical agricultural experimental design, since there is only one "treatment" - the particular power-generating station. The problem of only one treatment unit is therefore a real and important problem for nearly all environmental field studies, as true standard experimental designs are not feasible with only one impacted or "treatment" site. Replicates are independent only if the contaminant is randomly assigned to multiple experimental units.

Block designs

The problem of strata effects (pseudo-replication) can be overcome by having several replicated disturbed and control locations. While there are rarely replicated planned developments, there is no reason not to have replicated control locations (asymmetrical BACI, Underwood, 1994). The controls should be a representative sample of places of the same general habitat as that in which the impact is expected. There may, nevertheless, be cases where multiple controls are not available and BACI procedures are all that are available (Underwood, 1994). In such cases BACI designs can suffer from strata effects (be "pseudo-replicated") if analyzed incorrectly.

Gradient designs

Gradient designs test for changes along a contamination gradient. The assumption is, therefore, that exposures are sampled to background levels. Stratum effects could occur if most samples for gradient analysis came from the impact area only. Stratum effects would then occur because the error about the regression will be unique to those sites. Wiens & Parker (1995) note that this is a special case of pseudo-replication and it is not as great a transgression as that caused by the effect of spatial correlation on error for impact-reference and matched-pairs (BACI) designs.

2.5 Developing causal models

Predictive monitoring is now recommended over open-ended monitoring (Paris Commission, 1989). The ICES advisory committee on the marine environment in their 1995 report emphasized that monitoring objectives should be formulated as testable, quantified hypotheses. The predictions developed in an environmental impact assessment are then tested during the monitoring program. Feedback monitoring (Gray & Jensen, 1993) results in change in management practice, if predictions are not upheld. Predictive monitoring has several advantages. It focuses directly on impacts and the underlying mechanisms, rather than on indirect indicators. It provides information needed to mitigate impacts, either via change in operating procedures, or via change in regulatory requirements. It is more efficient and cost-effective than open-ended monitoring where multiple variables must be measured.

Both block and gradient designs enable predictions to be readily developed. Gradient models test for change associated with a contamination gradient. A quantitative prediction would be to identify the change in some variable as a function of the change in distance (contamination gradient). Block designs test the null hypothesis of no change in the impact location. A quantitative prediction would be to identify the change in the response variable between the control and impact sites.

2.6 Inadequate sampling design

For most environmental accidents, there is usually no "before" data and there are no replicate treatment units, so sampling cannot be entirely randomized. Wiens & Parker (1995) consider appropriate sampling designs and assumptions for analyzing the effects of accidental environmental impacts. They conclude that in evaluating the effects of unplanned environmental impacts, post-facto study designs that document both initial effects and subsequent recovery, or that treat effects as continuous rather than categorical variables (gradient or trend designs) may be more useful than before-after comparisons. However, rather than documenting initial effects and subsequent recovery, before-after analyses are still being applied to detect environmental accidents.

Block

To overcome the problem of lack of time and resources for powerful sampling before an impact, Underwood (1994) suggests research that determines the magnitude of spatial differences and the rates of temporal change of various 'representative' populations is needed. A number of randomly chosen replicated sites could be monitored. The variance among times, locations, and their interactions provide estimates of the natural variability from a population of such locations and times. Thus, population variance estimated in such a sampling program would serve as 'before' data for any other set of locations at some subsequent time. Such a research program needs to be implemented for those habitats and populations where there is reasonable expectation that future impacts may occur. However, Wiens & Parker (1995) suggest that for environmental accidents an inclination to think first about conducting a before-after analysis is often misguided.

Gradient

Because they have the potential to document both initial impacts and subsequent recovery and are relatively robust to the effects of pseudo-replication, the impact level-by-time and impact trend-by-time designs seem particularly well suited to the analysis of unplanned impacts. Designs that treat contamination as a continuous variable (gradient, impact trend-by-time) have the potential to document contamination effects with greater precision and to detect non-linearity's in responses of the community (Wiens & Parker, 1995).

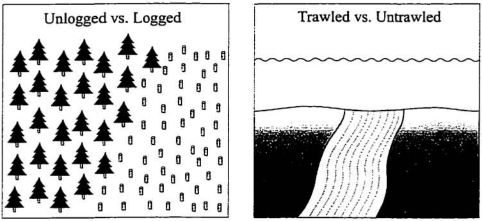
2.7 Summary

Block designs have been widely used in impact assessment regardless of the nature of the disturbance. Block sampling layout and statistical models are currently being applied to gradient impacts with a subsequent loss in the sensitivity to detect an impact. Chapter 1 and 2 suggest that a gradient layout and model may be more sensitive to detect gradient impacts than currently used block designs.

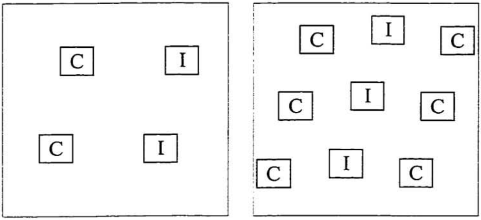
Evaluation of the power of gradient designs is required before such designs can be recommended for use in environmental impact assessment. Chapter 3 therefore evaluates the power of a gradient and a stratified layout to detect environmental impacts when the scale of the disturbance is unknown. In this chapter several problems and assumptions of gradient designs have been identified. One problem is that stratified statistical models are currently used to analyze data from a gradient layout with subsequent loss in sensitivity. Chapter 4 therefore investigates methods of analysis for data from gradient designs. The assumptions of a gradient layout are that no natural physical gradients occur in the environment. This assumption is often violated in ecological studies due to changes in environmental variables such as salinity, depth, sediment grain size associated with the anthropogenic gradient. Therefore, Chapter 5 investigates how rigorous a gradient layout is for detecting an anthropogenic disturbance in an environmental with multiple natural gradients.

Table 2.1 Comparison of gradient and block sampling designs with respect to their ability to select control sites, detect the scale of an impact, develop causal models, detect environmental accidents, and with respect to pseudoreplication

	Gradient Designs	Block Designs
Control Sites	Not a requirement of gradient designs	Difficult to select adequate control sites if applied to point source disturbances where the scale of the disturbance is unknown
Scale	Readily identified	More difficult to identify the scale of a disturbance. Green and Hobson (1970) and Underwood (1992) suggest using hierarchical sampling
Pseudoreplication	Not as great a problem for gradient designs, unless exposures are not sampled to back-ground levels	Problem of strata effects occurs because normally there is only one treatment. Multiple controls are recommended (Underwood, 1992)
Developing Causal Models	Readily identified. Causal models are of attenuating effects with increasing distance from the disturbance	Readily identified. Causal models are of changes in the impact location over time
No "Before" Data (Environmental Accidents)	Gradient designs are particularly useful in documenting effects and in monitoring subsequent recovery of environmental accidents (Wiens & Parker, 1995)	More difficult to detect using BACI designs, unless investigate impact-level-by time changes (Wiens & Parker, 1995)



1. Stratified Impact

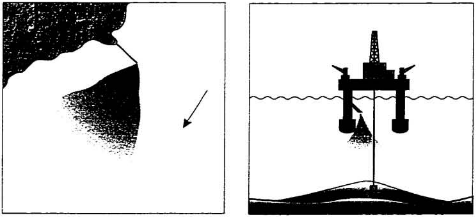


2. Stratified Layout

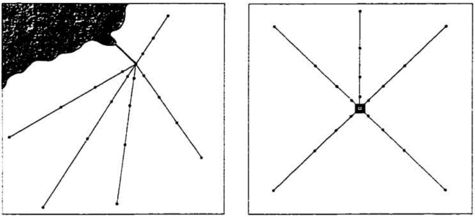
SOURCE OF VARIATION	
Before vs. After	= B
Times (Before vs. After)	= T(B)
Location (Impact vs. Control)	= L
B x L	
Time (B x L)	

3. Stratified Statistical Model

Fig 2.1 Stratified sampling design and model for block impacts



1. Gradient Impact



2. Gradient Layout

SOURCE OF VARIATION	
Before vs. After	= B
Times (Before vs. After)	= T(B)
Distance	= D
B x D	
Time (B x D)	

3. Gradient Statistical Model

Fig 2.2 Gradient sampling design and model for gradient impacts

Chapter 3. Evaluation of the power of gradient sampling designs

3.1 Introduction

Detection of anthropogenic disturbances can be made complex by variability in the abundances or rates of change of local populations (Underwood, 1993). More than any other single factor, our inability to explain natural variability places a limit on our ability to detect anthropogenic changes (Thomas, 1992). Various sampling designs and statistical analyses have been developed to handle this considerable natural variability. These designs are discussed in Chapter 1.

This evolution of sampling designs has greatly improved our ability to detect anthropogenic environmental disturbances against a background of natural variability. Since the proposal of BACI sampling designs, monitoring programs have become considerably more sophisticated, powerful and sensitive (Morrisey, 1993). 'These sophisticated designs of sampling and analysis are necessary to remove the confounding effects of existing variation from the effects of the disturbance' (Morrisey, 1993). However, these designs deal only with assessing an impacted area or strata versus one or more control strata. While the design is appropriate for detecting disturbances that have defined boundaries of impacted areas, there are many circumstances in which a disturbance attenuates with distance from a point source. In these circumstances it may be more appropriate to sample along a gradient from the disturbance.

The objective of this chapter is to compare a gradient design with the two strata, Control-Impact design used in BACI monitoring. The power of a gradient design is evaluated against a stratified Control Impact (CI) design. If a gradient design is more sensitive than a stratified design, gradient designs should be adopted to detect point source disturbances or disturbances where the scale of the impact is unknown. The power of the two sampling designs to detect differences is assessed using benthic macrofaunal data from the Ekofisk oil platform in the North Sea. To evaluate the sensitivity of the two designs we compare them using a data set with known gradient impact. A gradient layout and gradient model is compared with using a block layout, and block statistical model.

A number of questions relevant to impact assessment are also evaluated using the Ekofisk data. These include, 1) whether diagnostic characteristics of tolerant and sensitive species can be identified, 2) whether analysis based on higher taxa generates similar conclusions to analysis based on species data, and 3) whether effects of physical disturbance and chemical toxicity can be distinguished.

3.2 Methods

Exploration of the Ekofisk oil field in the North Sea began in the late 1960's (Gray *et al.*, 1990). Since 1973 several environmental investigations have been carried out around the main platform complex. The data set analysed in this chapter is from a 1990 survey that recorded benthic macrofaunal density. Sampling sites were arranged as transects radiating from the platform centre and are approximately logarithmically spaced with 38 sites located at the Ekofisk oil field (see Fig 3.1). At each site 5 grab samples were taken with a Van Veen grab (0.1 m^2). Samples were sieved on a 1 mm sieve. Samples were preserved in formalin and sorted after staining with eosin. Individuals were identified to species where possible. An additional grab sample was taken at each site for analysis of sediment particle size, organic matter, and trace metals. See Gray *et al.* (1990) for details of methods used. Note, Gray *et al.* (1990) analysed data from a 1987 survey, however the methods used to collect the data are the same as the 1990 survey.

Changes in macrofaunal abundance as a function of distance, transect arm, replicate (within site variation) sediment size and water depth were analysed using Analysis of Variance (ANOVA). Power was calculated using methods of Sokal & Rohlf (1995). To compare the gradient and CI (block) sampling designs, two situations were analysed: all replicates taken at a control and an impacted site for the stratified (CI) design versus sampling fewer times at more sites or distances (Gradient model).

The Ekofisk monitoring design sampled sites from 100 metres to 6500 metres from the platform. Twenty three distances were sampled in total. This data set enables the two models to be compared. All of the distances sampled can be analysed by regression (gradient model) for which effects are analysed as a function of distance. The data can be analysed again by grouping sites next to the platform as impacted sites, and then comparing these sites to a control site using ANOVA. Rather than sampling with distance from the disturbance only far sites are selected as controls, as in standard CI and BACI designs. Control samples are then compared against impacted sites. Sites less than 460 metres from the platform are grouped or blocked together as impacted. The analysis was also repeated by grouping sites less than 800 metres, and 1000 metres as impacted. By comparing only the impacted sites against a control, a block sampling design was tested. For both block and gradient analyses, the tolerance for making a Type I error (α) was set at 0.05.

The number of observations (n) must be equal to compare the two models. There are two reasons for requiring equal sample sizes. Firstly, the power of a test increases with sample size (Peterman 1990, Sokal & Rohlf 1995). Secondly, equal sample sizes represent similar sampling effort in the field. A gradient model samples as a function of distance, therefore samples from every site (distance) were selected to test the model. The CI model only compares the sites next to the platform with control sites. Not all sites were selected for analysis of a CI model, only the sites adjacent to the platform (impact), and sites at a control location were selected. Therefore the number of observations would be less than for the gradient design. In order to have an equal number of observations for each model only two of the five replicates for each site were randomly selected for the gradient model.

One problem that arises with a randomized design is that selection of a control site involves choosing a site that is a) close enough to the disturbance so that it is comparable to the natural variability of the disturbed environment and yet b) far enough away that it is not affected by the disturbance (cf. Chapter 2). To test how selection of a control site affects the power of a test to detect significant differences, we repeated the analysis by comparing the impacted site with several control sites. Instead of only selecting sites at 6000 metres to serve as a control, the analysis was also run using sites at 1000 meters and 2000 metres as the control.

The statistical package SAS was used to calculate p-values under the assumption of independent residuals with equal variance. Residuals were plotted against expected values. No bows or arcs were evident and so the linear statistical model was assumed to be an acceptable description of the data (Draper & Smith, 1981). The p values obtained from a theoretical and a randomized distribution were compared for different sample sizes. The results determined whether to use a theoretical or randomized distribution for the analysis of changes in benthic macrofaunal abundance.

3.3 Results

The gradient sampling design was more powerful at detecting changes in total macrofaunal abundance than the block design (Table 3.1). Regression analysis for near Ekofisk stations found a significant difference in total macrofaunal abundance relative to distance from the platform ($F_{1,6509}$=5.675, p=0.0172, β=0.86). The block design for the same data set using ANOVA of impacted and control sites was slightly less sensitive at detecting a significant difference ($F_{1,6509}$=5.05, p=0.0247, β=0.81).

Similar results were obtained when all sites were analysed. The gradient model was more sensitive ($F_{1,16490}$=4.493, p=0.0340, β=0.76). The block design was less sensitive. The variance ratio for the impacted site versus a control site at 2000 metres was not significant ($F_{1,16490}$=1.82, p=0.1771, β=0.3). With a control site at 6000 metres the variance ratio for the block design again failed to detect a significant difference ($F_{1,16490}$=1.99, p=0.1582, β=0.3). When the impacted sites were extended out to 800 metres ($F_{1,16490}$=0.91, p=0.3414, β=0.15) and 1000 metres ($F_{1,16490}$=0.76, p=0.3826, β=0.15) the block design again failed to detect a significant difference.

Theoretical and calculated (randomized) p values were compared for several data sets of differing sample sizes. The subsets of data were randomly selected from the Ekofisk data set. For each data set a theoretical p value was obtained for a simple linear model $N = \beta_o + \beta x$, where N = organism abundance, and x = distance from the platform. The response variable, 'total abundance of organisms' was then randomized and a calculated F-ratio was obtained. Randomizations were performed 500 times. Table 3.2 indicates that for small sample sizes ($\leq$ 100 observations) there can be large discrepancies between the theoretical and randomized values. At large sample sizes the values obtained from a theoretical versus a calculated distribution were similar. Examination of residuals from analysis of changes in benthic macrofaunal numbers for the gradient and block model revealed that the residuals were not normal. For non-normal residuals an F-distribution should be generated by randomization, which can be computationally laborious for large data sets. Because the p values obtained using a theoretical or a randomized distribution are similar at large sample sizes a theoretical distribution was used.

Several major trace metals were recorded along with mud content and sediment particle size in order to determine discharges due to oil related activities. Sediment sizes ranged from 1.49 to 3.47 ϕ with mud content (silt and clay fraction) elevated near the platform (Fig. 3.2). Plots of recorded trace metals (ppm) from the platform show a clear gradient with distance. Of the trace metals recorded, barium is considered the most suitable indicator of oil related activities. Barium is used in the drilling process and discharged in the drilling muds. Barium is not regarded as a toxic chemical for marine benthos and is not metabolised, as far as is known (Gray *et al.*, 1990). Therefore, the barium content of the sediments can be used as a tracer for drilling muds (Gray *et al.* 1990). Figure 3.3 shows barium concentrations (Ba=ppm) with distance (m) from the oil rig. Regression analysis of log barium against distance provides a linear model for barium;

$\ln(Ba) = 8.5024 + -0.0004 * \text{Distance}$, where $r^2 = 0.7903$.

The resulting model for the level of barium as a function of distance from Ekofisk platform is;

$$Ba = e^{8.5024} e^{-0.0004 * \text{Distance}}$$

Analysis of residuals versus expected values showed no pattern, thereby indicating that the model is acceptable. Residuals were normal.

Biological responses as a function of mud and trace metal impacts can be assessed. Analysis of variance (ANOVA) of near Ekofisk sites and far Ekofisk sites indicate that distance, sediment size, and barium are the only three, out of six explanatory variables tested, that were related to density of benthic macrofaunal organisms (Table 3.3). Numbers of benthic organisms change as a function of distance from the platform. Concentrations of trace metals and fine mud content also changed with distance from the platform. The range in depth was small: 70 to 76 m. Analysis of variance for near sites showed no significant difference ($F_{1, 16187}$=0.02, p=0.8778) for total macrofaunal abundance as a function of depth. Depth ranged from 72 to 74 metres at the near sites. There was no significant difference ($F_{2, 24822}$=2.26, p=0.1047) for the far sites, for which depth ranged from 70 to 76 metres. Changes in benthic abundance for transect, replicate, and interaction terms were not significant (Table 3.3).

Total abundance of benthic organisms decreased as a function of distance from the platform (Fig. 3.4), while species richness increased (Fig. 3.5). The relationship between species richness and distance was not a simple linear function. There appeared to be a peak of high diversity at approximately 1000 metres. Tolerant species occurred within 1000 metres of the platform (Fig. 3.6). The most abundant of the tolerant species was the polychaete *Capitella sp.* Capitellids were generally found within 150m of the platform, where they were recorded at densities as great as 1669 individuals per 0.1 m^2 (Fig. 3.6). Figure 3.7 shows examples of three species that had local extinctions or depressions next to the platform. At approximately 1000 metres the abundances of these species were similar to that found at 6000 metres. These sensitive species occurred at relatively low abundances and so changes in numbers of these species next to the platform are small. While the polychaete *Goniada maculata* did show a noticeable increase in abundance with distance from the platform, the changes in the other species were small. Changes in abundance for the other sensitive species (such as *Nephtys longoseta* and *Montacuta substriata*) are only from 0 or 1 individual per 0.1 m^2 in impacted areas, compared to 2 or 3 individuals per 0.1 m^2 at distances of 6500 metres.

Analysis of Covariance (AnCova) extends the gradient statistical model to several species or taxonomic groups. AnCova of total abundance of 30 taxonomic groups as a function of distance found no significant differences among the slopes except for copepods. There was no significant interaction term and no overall gradient within an ANCOVA of phyla abundance and distance for the Ekofisk survey (Table 3.4). AnCova was no longer able to detect gradients when species were grouped to the higher taxonomic level of phyla.

From Figure 3.6 and 3.7 it can be seen that different species of polychaetes decline or increase with distance. Numbers of *Capitella sp.* were elevated next to the platform and decreased with distance, while the abundance of *Goniada maculata* were suppressed next to the platform with numbers increasing with distance. By combining all polychaete species into one class the ability to detect changes in abundance was lost.

3.4 Discussion

Comparison of the two sampling designs indicates that the gradient model analysed by regression analysis was more powerful than randomized layout within a block (CI) design. The power of a test is the probability of rejecting the null hypothesis when it is false and the alternative hypothesis is correct (Sokal & Rohlf, 1995), hence, it is the probability of correctly identifying an impact. Besides being less powerful, the CI design has requirements in the selection of the control site that are difficult to fulfil. The requirements for control areas are that the site should be far enough away that it will not be affected by the potential disturbance, and yet close enough that the areas are comparable. Such a selection is often arbitrary as the area affected by a putative impact is not definitively known, we do not know exactly how far pollution will disperse in the marine environment, or whether two or more areas are comparable with respect to physical processes. A gradient sampling design removes the problem of selecting a control site, while being more powerful at detecting changes (biological and physical) due to disturbances.

Five CI models were tested. Of these, the design that contrasted the impacted area with a control at only 1000 metres was the most sensitive at detecting differences. Selection of a valid control must ensure that it is not affected by the disturbance. Olsgard & Gray (1995) determined the spatial area affected by contamination, defined as raised concentrations of chemicals, for three oil fields in the North Sea. They reported effects that extend out to distances of 10 km at Valhall, >1.5 km at Gyda and > 1 km at Veslefrikk. A control site at 1 km cannot, therefore, be selected as a true control as it is subject to contamination. A control at 10 km or greater may, however, no longer be controlled by the same physical and biological processes as the disturbed site.

Another advantage of the gradient model is that the results are easy to interpret and present to the public. The US National Research Council's (1990) review of marine environmental monitoring highlighted three factors that have contributed to the difficulty of obtaining useful monitoring programs. One of these is that 'information is rarely presented in a form useful in developing broad public policy or evaluating specific control strategies'. The spatial scale of an impact is an important question in policy formulation. Another important question is whether valued ecosystem components, such as shellfish stocks, are being adversely affected either by direct mortality from smothering or due to heavy metal accumulation resulting in taint. If so what is the scale of this impact, how far do the contaminants (e.g. hydrocarbons, fine mud cuttings) disperse? Such questions concerning the spatial scale of the impact are more directly communicated with a graph of gradients than with an ANOVA table. Simple plots of physical variables as a function of distance can be used to determine the spatial scale of impact. At the Ekofisk platform elevated levels of barium were recorded as far out as 6 km. Mud content of the sediment was found to be elevated within 150 metres of the platform. A gradient model enabled the area affected by taint (trace metals) and smothering (mud content) to be easily visualized.

In the case of block designs, an impacted site is compared with one or more control sites. Analysis of the data from such a sampling design leads at the simplest level to tests of variance ratio (F-tests), or to more elaborate ordination techniques. Simple univariate analysis such as an ANOVA provides information on whether or not there is a statistically significant difference between the two sites. While this design determines whether the two sites are different, it does not provide information on the spatial scale of the impact. Unless more than one control is sampled it is difficult to link the spatial scale of biotic changes to the spatial scale of the disturbance.

Identification of diagnostic characteristics of tolerant and sensitive species is an important prerequisite of impact monitoring, based on 'indicator species'. Both morphological and behavioural adaptations may render a species tolerant or sensitive to disturbances generated by drilling activity. The results from the Ekofisk analysis suggest that certain characteristics such as ability to survive in anoxic conditions, motility, and feeding ecology (Gagnon & Haedrich, 1991) may render a species tolerant or sensitive to disturbances generated by oil related activities. Two polychaetes identified as tolerant to disturbance by gradient analysis were *Capitella sp.* and *Chaetozone setosa* (Appendix 3). Both are deposit feeders capable of surviving in anoxic conditions (Fauchald & Jumars, 1979). Grassle & Grassle (1974) state that one of these species *Capitella sp.* is a pollution indicator, in that it is capable of invading areas where natural or human-made defaunation has taken place. Capitellids build temporary tubes near or at the surface of the sediment. These tubes allow the worm to feed in anoxic mud while remaining in contact with the surface to obtain necessary oxygen by irrigating the burrow (Fauchald & Jumars, 1979). The cirratulid polychaete *Chaetozone setosa* was another species with high densities close to the platform. *C. setosa* live in mud-covered tubes, and are deposit feeders. Some species of cirratulids may be extremely abundant in polluted areas (Fauchald & Jumars, 1979). The amphipod *Jassa marmorata* was identified as a tolerant species, occurring at high abundances near the platform. This is an unexpected finding as this amphipod is thought to respond negatively to oil pollution (Gray *et al.* 1990). The gradient design makes it clear that this finding was not due to chance placement of a small number of control sites i.e. not due to strata effects (pseudo-replication). Examples of three intolerant species identified by gradient analysis are *Nephtys longosetosa*, *Goniada maculata*, and *Montacuta substriata* (Appendix 3). Two were carnivorous polychaetes. One is known to be a motile predator (*Nephtys longosetosa*) and the other species (*Goniada maculata*) may move freely.

Montacuta substriata is a non-motile bivalve species, and their gills become clogged with very fine silt or mud particles. *Montacuta substriata* is found in commensal relationships with echinoids (Popham, 1940). *Goniada maculata* showed the greatest decrease in numbers closest to the platform. Further research is required to determine whether predictor characteristics of tolerant and sensitive species can be found in other disturbed areas. Identification of such characteristics may provide a predictive tool to assess likely impacts of proposed development.

Another question relevant to impact assessment is whether analysis based on higher taxa generates similar conclusions as analysis based on species data. Such a question is of importance for marine monitoring work, as the time-consuming process of species identification can be replaced by more rapid determination of class or phyla. The GESAMP workshop (Gray *et al.* 1988) on biological effects of pollutants suggested that analysis based on higher taxa may more closely reflect gradients of contamination or stress than those based on species data. Results from the analysis of gradients indicate that class is not suitable as a predictor of response to pollution. The ability to detect a gradient was lost when species were grouped into taxonomic classes. Gagnon and Haedrich (1991) compared a functional approach based on feeding type, microhabitat preference, motility pattern, and body size of benthic organisms to a taxonomic (family) based method. They found that the functional approach provided a more direct way of interpreting the effects of environmental factors on the community structure (Gagnon & Haedrich, 1991). It is therefore possible that functional groupings may prove more useful than taxonomic groupings.

Another finding from the Ekofisk data of interest to marine monitoring is the reduction in species number recorded adjacent to the platform. Rapport *et al.* (1985) and Gray (1989) stated that effects of environmental stress on assemblages leads to a reduction in species number, and an increase in dominance by opportunists. High numbers of a few tolerant species dominated the sites closest to the Ekofisk oil platform. The gradient model permits a more accurate description of change in opportunists and diversity with changes in distance from the platform than CI designs.

Olsgard & Gray (1995) identify another key question in impact assessment: Is it possible to distinguish between effects of physical disturbance and chemical toxicity? It is known that effects of heavy metals tend to reduce the number of individuals as well as the total number of species (Rygg, 1986). Severe physical disturbances tend to reduce diversity alone. These findings suggest that it may be possible to separate impacts resulting from physical and chemical disturbances. Analysis of the Ekofisk data indicated that a change in community structure to dominance by tolerant species occurred at the same spatial scale as the physical disturbance due to high levels of drilling muds adjacent to the platform. The scale of siltation was within 150 metres of the platform. *Capitella sp.* was also recorded within 150 metres of the platform. To determine potential impacts due to toxicity, total numbers of benthic organisms would need to be monitored over time.

In conclusion, a gradient design was more powerful than a control impact design in detecting changes due to anthropogenic disturbances. CI and BACI designs have been important steps in environmental impact assessment. They provided a framework for design of assessment programs. BACIP and asymmetrical BACI designs have provided even more powerful methods of detecting human induced disturbances. Our analysis indicated that further increases in power can be attained by adopting a gradient layout if the impact attenuates from a point source. One advantage of a gradient layout is that it avoids the problem of arbitrarily selecting a control site. A second advantage is that results from a gradient layout enable chemicai, physical and biological changes to be assessed as a function of distance. Gradient designs lend themselves to constructing mechanistic models that generate testable predictions of attenuating effects with increasing distance. The results are therefore easy to interpret and can be presented in a form that is useful for developing broad public policy decisions and control strategies.

Table 3.1 Comparison of results from ANOVA and regression analysis of species abundance for block and gradient sampling models

Model	Distances (m)	Replicates	n	DF	F value	Pr > F	Power
Near Ekofisk Sites							
Gradient Design	100, 150, 250, 330, 460, 500, 1000	2	6,510	1	5.675	0.0172	0.86
Block Design	Impact = 100 Control = 1000	5	6,510	1	5.05	0.0247	0.81
All Ekofisk Sites							
Gradient Design	100, 150, 250, 330, 450, 460, 500, 750, 800, 850, 1000, 1200, 1300, 1800, 1900, 2500, 3300, 3900, 4000, 4400, 5700, 5800, 6500	2	16,491	1	4.493	0.034	0.76
Block Design	Impact = 100 to 460 Control = 1000 to 2500	5	16,491	1	1.82	0.1771	0.3
Block Design	Impact = 100 to 460 Control = 4000 to 6500	5	16,491	1	1.99	0.1582	0.3
Block Design	Impact = 100 to 800 Control = 5700 to 6500	5	16,491	1	0.91	0.3414	0.15
Block Design	Impact = 100 to 1000 Control = 5700 to 6500	5	16,491	1	0.76	0.3826	0.15

Table 3.2 Comparison of theoretical (F distribution) and calculated (randomized) p values. Analysis of species abundance with distance from Ekofisk

Sample Size	Theoretical p value	Calculated p value
10	0.55	0.166
20	0.675	0.446
30	0.434	0.556
50	0.93	0.454
100	0.212	0.926
500	0.0001	<0.001
1000	0.0001	<0.001
2000	0.006	0.02
4545	0.0001	<0.001

Table 3.3 Data for ANOVA between environmental variables and fauna at near and far sites

Near Ekofisk

Source	DF	F value	Pr > F
Distance	6	4.44	0.0002
Transect	3	0.36	0.7844
Replicate	4	0.35	0.8438
Depth	1	0.02	0.8778
Sediment	13	2.44	0.0027
Barium	13	2.43	0.0028
Dis*Trans	5	0.43	0.8266
Trans*Rep	12	0.38	0.9711
Dis*Rep	24	0.34	0.999
Dis*Rep*Trans	20	0.41	0.9898

n = 16, 275
SSTot = 8,268,616.1

Far Ekofisk

Source	DF	F value	Pr > F
Distance	17	3.36	0.0001
Transect	4	0.1	0.983
Replicate	4	0.54	0.7041
Depth	2	2.26	0.1047
Sediment	18	3.05	0.0001
Barium	18	2.98	0.0001
Dis*Trans	1	0.1	0.7531
Trans*Rep	16	0.05	1
Dis*Rep	68	0.28	1
Dis*Rep*Trans	4	0.08	0.9873

n = 24, 955
SSTot = 652,907.5

Table 3.4 Analysis of covariance, abundance of phyla with distance from Ekofisk platform

Source	DF	F value	Pr > F
Phyla	32	4.99	0.0001
Distance	1	1.15	0.6985
Phyla * Distance	32	0.75	0.8443

n = 41,230
SSTot = 8,925,877.9

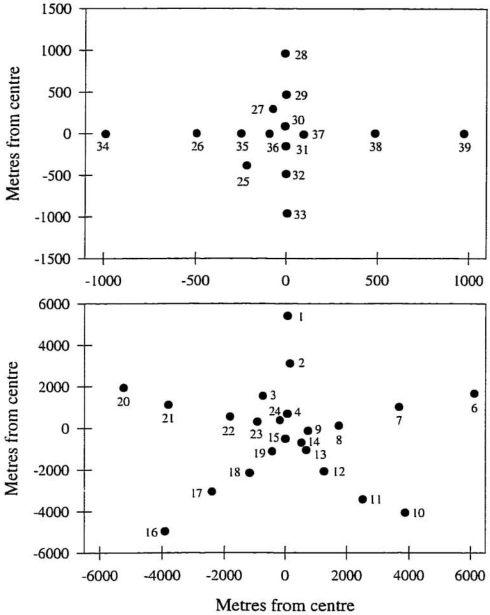


Fig 3.1 Near Ekofisk sampling sites and far Ekofisk sampling sites

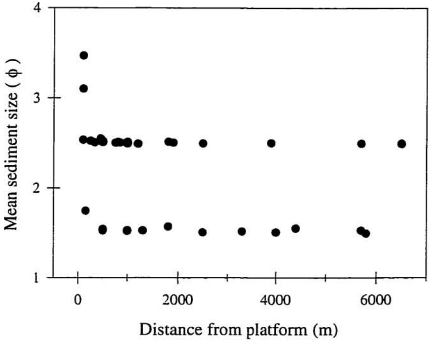


Fig 3.2 Change in mud content from oil drilling operations as a function of distance from Ekofisk platform

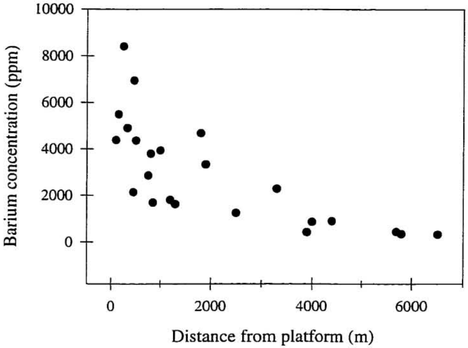


Fig 3.3 Barium concentration in sediment samples taken with distance from Ekofisk platform

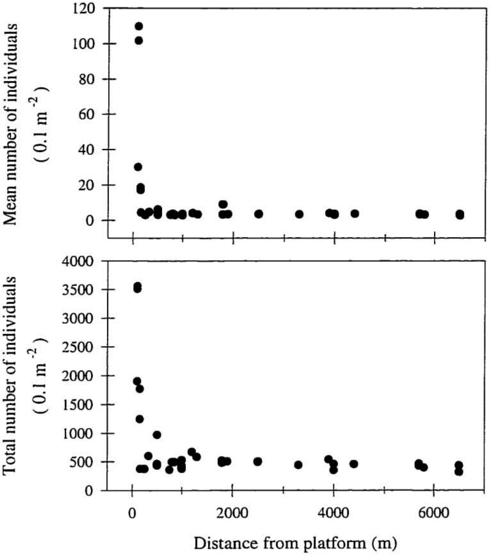


Fig 3.4 Change in mean and total number of individuals with distance from Ekofisk platform

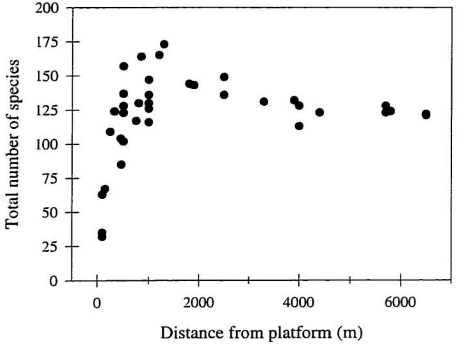


Fig 3.5 Change in total number of species with distance from Ekofisk platform

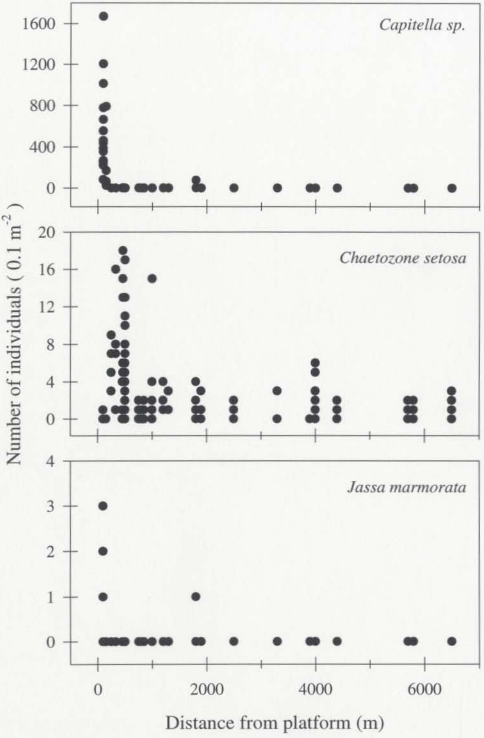


Fig 3.6 Change in abundance of three tolerant species with distance from Ekofisk platform

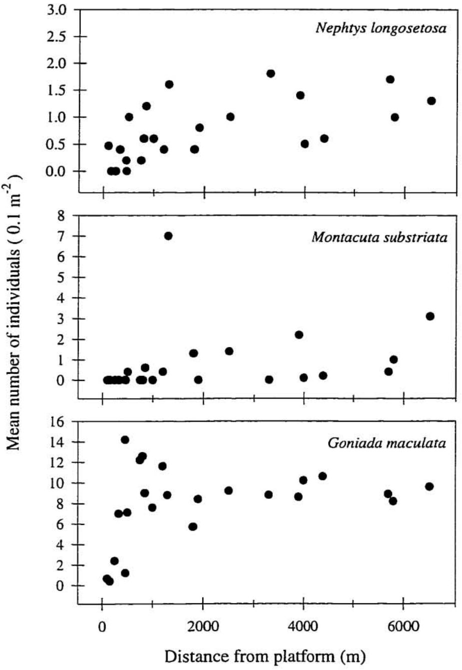


Fig 3.7 Change in abundance of sensitive species with distance from Ekofisk platform

Chapter 4. Gradient analysis

4.1 Introduction

A gradient statistical layout provides information on attenuating effects with increasing distance. Changes in the response variable as a function of the contamination gradient can then be tested. However, stratified statistical models are generally used to analyse data from a gradient layout with subsequent loss in sensitivity to detect an impact. If a stratified statistical model is applied when effects are graded, then the variation due to the graded effect will appear in, and hence inflate the error term, reducing sensitivity. Therefore, development of methods of analyzing data from gradient designs are required, and forms the basis of this chapter.

Methods to analyse change in benthic community structure include univariate (ANOVA), graphical distributional (biomass and relative abundance), and multivariate methods of classification and ordination. For a stratified sampling layout, which sample an impact and a control site preferably multiple times before and after a disturbance, univariate and multivariate statistical models have been used to analyse alteration of resident biological communities. Both Bernstein & Zalinski (1983) and Stewart-Oaten *et al.* (1986) considered that the best statistical model for BACI/P designs was to calculate the differences between the mean abundance in the two sites (control/impact) before and after the disturbance. Underwood (1992, 1993, 1994) considered how to analyze the data for asymmetrical designs using analysis of variance (ANOVA). The proposed statistical analysis using a *t*-test is, algebraically, identical to the *F*-ratio identified in the analysis of variance (Sokal & Rohlf, 1995). An environmental impact is detected by a significant interaction between the location (control/impact) and the time (before/after) variables. This means that an environmental impact affecting the abundance in that location will be detected if the temporal pattern of abundance in that location differs from the range of patterns in the set of control locations.

An equivalent statistical model for gradient designs has not been developed and hence may explain why the statistical model developed by Bernstein & Zalinski (1983) is used to analyse data from a gradient layout. Block models test for a location (control/impact) difference. The appropriate model for gradient data would be to test for a change in the response variables as a function of distance (Table 4.1).

The above analysis tests for an effect of an anthropogenic disturbance on the environment. However, statistical methods to infer species-environment relationships are needed. Exploratory methods of statistical analysis such as ordination techniques can be used to relate benthic change to environmental measurements. The statistical model would be $Y = \beta x + \varepsilon$, where Y is the response variable (a matrix of benthic densities), x is a series of principal components, β are loadings on components, and ε = residuals. Another method of analysis for point source impacts is a direct comparison of spatial gradients of response and pollution measurements before and after a disturbance. The statistical model would be $Y = \beta_o + \beta_{Dis} + Dis + \varepsilon$, where Y = response variable (benthic density, barium etc.), Dis = distance, and ε = residuals. This Before After Gradient (BAG) design utilises the same species covariance information as multivariate methods, so it was therefore of interest to determine if gradient methods are as sensitive and consistent as established multivariate methods. This chapter compares the sensitivity of multivariate methods of analysis with gradient methods.

4.2 Methods

Biological and chemical data were obtained from Ekofisk and Gyda oil fields in the North Sea. Exploration of the Ekofisk oil field began in the late 1960's. Since 1973 several environmental surveys have been carried out around the main platform complex. The data set analysed in this paper is from a 1990 survey that recorded benthic macrofaunal density. Production has occurred at Gyda since 1987. Data analysed in this paper are from three surveys conducted in 1987 (baseline survey), 1990 and 1993. Sampling sites are arranged along radii and log spaced with 38 sites located at the Ekofisk oil field. For the Gyda oil field, 18 sites were surveyed in 1987, 17 sites were surveyed in 1990, and 20 sites were surveyed in 1993. At each site 5 samples for Ekofisk and 1 sample for Gyda sites were taken with a Van Veen grab (0.1 m^2). Samples were sieved on a 1 mm sieve. Samples were preserved in formalin, stained with eosin, and sorted. Individuals were identified to species where possible. Additional grab samples were taken, one at Ekofisk and three at Gyda, for analysis of sediment particle size, organic matter, and trace metals. See Gray *et al.* (1990) and Olsgard & Gray (1995) for further details of methods used.

Statistical Analysis

Both the environmental variables and the faunal data were analysed with Principal Components Analysis (PCA) and Multidimensional Scaling (MDS). These ordination techniques operate on an n samples by p species matrix to produce a new set of variables that optimally describes the relationships among the original p variables. PCA maximises the amount of variation accounted for by the new axes; it proceeds by way of an eigen analysis on the p-by-p correlation (or covariance) matrix (Clarke & Green, 1988). MDS finds a specified number of new axes that attempt to preserve some relationship among the between-sample distances (Kruskal & Wish, 1978). MDS is further classified by whether the dissimilarities data are measured on an ordinal scale, called non-metric MDS, or an interval or ratio scale i.e. metric MDS. Non-metric MDS, PCA and metric MDS were all performed on the species abundance matrices for Gyda 1990, Gyda 1993, and Ekofisk data sets. This enabled a thorough comparison of ordination techniques with gradient methods. Also, congruent configurations obtained by the various ordination techniques indicate that a realistic ordination has been achieved. Analyses were performed on the similarity matrices using the SPSS package. Goodness-of-fit between the data and the distances in the ordination plots was measured as stress with Kruskal's stress formula 1 (Kruskal & Wish, 1978) and by RSQ, the squared correlation between the data and the distances (Norusis, 1994). These measures were used to determine whether a two-dimensional model was adequate. Non-metric MDS resulted in the lowest stress coefficients, therefore only non-metric MDS plots are presented here. Plots of the classification groupings on the original sampling stations are obtained from the MDS ordination plots. Both plots are presented for the faunal data.

Observed changes in the faunistic ordination from Gyda 1990, Gyda 1993 and for Ekofisk need to be related to measured environmental change. The method of choice for multivariate representation of community structure is often non-metric multi-dimensional scaling (MDS). This has great flexibility in accommodating biologically relevant (i.e. non-correlation based) definition of similarity in species composition of two samples, and in preserving the rank-order relations amongst these similarities in the placing of samples in an ordination. Correlation-based techniques (such as canonical correlation) are then inappropriate in linking observed biotic structure to measured environmental variables (Clarke & Ainsworth, 1993). Warwick & Clarke (1991) and Clarke & Ainsworth (1993) suggest a more natural approach is to simply compare separate sample ordinations from biotic and abiotic variables and choose that subset of environmental variables which provides a good match between the two configurations. An ordination of relevant environmental variables should closely resemble the faunistic ordination. The omission of a key environmental variable will lead to an inferior match, as will the inclusion of variables with a substantially different pattern across sites but having no effect on species composition. The omission and inclusion of environmental variables is repeated until the closest match between faunistic and environmental ordination patterns is found. Non-metric MDS was performed on all the measured environmental variables for both Ekofisk and Gyda. This included mud content, water depth, hydrocarbon content, and sediment trace metals such as barium, lead, zinc and so on.

The same faunal and environmental variables were then analysed by gradient methods for the Ekofisk oil field. Changes in abundance of all organisms as a function of distance, transect arm, replicate (within site variation), sediment size, and depth were analysed using the general linear model. Type III sums of squares were estimated. The SAS package was used to estimate sums of squares to calculate p-values under the assumption of independent residuals with equal variance. Analyses were checked by plotting residuals against expected values. If no association between the residuals and expected values was evident the model was assumed to be an acceptable description of the data (Draper & Smith, 1981). All linear models were acceptable. P values were calculated from an F-distribution, which was shown by Monte Carlo methods to produce acceptable estimates for sample sizes above 100 with these data, despite non-normal and non-homogeneous residuals (Ellis & Schneider, 1997).

To identify species responsible for site groupings from an ordination a similarity percentages program was used (Warwick *et al.*, 1990). Gray *et al.* (1990) used this program to identify species responsible for site groupings for Ekofisk. For gradient methods the magnitude and sign (+ or -) of spatial gradients was used to determine key species responsible for changes in community structure. Analysis of Covariance (ANCOVA) for species abundance as a function of distance was used to test whether gradients were heterogeneous among species. A significant interaction term indicates whether species differ in pattern of change in abundance with distance from the pollution source. Regression coefficients were calculated for all species, then ranked. A negative coefficient indicated a tolerant species, a positive coefficient indicated highly sensitive species. Mid-ranking species showed little or no gradient relative to the contamination source. The statistical model to calculate species coefficients was $Y = \beta_o + \beta_{Dis} + Dis + \varepsilon$, where Y = benthic densities, Dis = distance, and ε = residuals. The units are organism counts per grab (# m^{-2}) by distance (m^{-1}). The units therefore equal # / m^3.

4.3 Results

The ordination plots clearly grouped sites as a function of contamination. Matches of site groupings with contamination groupings were consistent for all three platforms analysed (Figs 4.1 & 4.2). Unpolluted sites were grouped together (low barium, low THC, and % mud content <4%), intermediately polluted sites were grouped, and grossly polluted sites (high barium, high THC, and high % mud content) were grouped. Faunistic (Figs 4.1 & 4.2) and environmental (Figs 4.3 & 4.4) ordination plots are shown for Gyda 1990 and Gyda 1993. The plots obtained by our analyses were similar to faunistic and environmental ordination plots of the Ekofisk field (Gray *et al.* 1990, Warwick & Clarke 1991). To identify the underlying environmental correlates of species differences between sites, environmental ordination plots were compared to faunistic ordinations. For all three data sets MDS of barium, total hydrocarbon content (THC), and mud content showed the closest match to the faunistic MDS (Figs 4.3 & 4.4).

Biological responses to the oil drilling activities were then assessed by gradient analysis. Like MDS, gradient analysis indicated that species changes were associated with contamination gradients generated by oil drilling activities. Analysis of variance applied to near and far Ekofisk sites indicate that distance, barium (reflecting changes in trace metals), and sediment size are the only three, out of six explanatory variables tested, that were related to density of benthic macrofaunal organisms (see Table 3.3). These key environmental variables were the same as those detected using MDS. The spatial pattern of this impact was further investigated using gradient analysis. Plots of sediment trace metal concentrations from the platforms showed a clear contamination gradient with distance, with barium and THC demonstrating the largest gradient relative to background levels (Fig. 4.5). At the Ekofisk platform elevated levels of barium were recorded as far out as 6 km.

Elevated levels of silt and clay fractions due to the release of drilling cuttings, and elevated levels of organic content, were recorded within 150 m of the platforms (Fig. 4.6). These plots identify potential disturbances (trace metals, hydrocarbons, sediment accumulation, organic enrichment) that have different spatial scales of impact, with trace metals extending over the greatest spatial scale. Plots of benthic organisms identified as sensitive or tolerant from the regression coefficients were plotted to investigate biological change associated with contamination gradients. Figure 3.6 shows the total abundance of the tolerant species *Capitella sp.* at the Ekofisk platform. *Capitella sp.* were localised to within 150 m of the platform. Figure 3.7 shows three examples of sensitive species, *Nephtys longosetosa*, *Montacuta substriata*, and *Goniada maculata* at Ekofisk.

In order to identify which species accounted for the change in community structure, density gradients were ranked from negative to positive for all three data sets. Appendices 1, 2 and 3 provide lists of all recorded species at each oil field ranked by their regression coefficient. The negative coefficients indicate the species most tolerant to the contamination gradients. In all three data sets *Capitella sp.*, *Jassa marmorata*, *Chaetozone setosa*, *Philine scabra*, *Glycera alba*, *Lunatia montagui*, *Edwardsia sp.*, and *Nemertini sp.* were consistently identified as tolerant species with high abundance adjacent to the platform in polluted regions. *Amphiura filiformis*, *Goniada maculata*, *Eudorellopsis deformis*, *Scoloplos armiger*, *Abra prismatica*, *Montacuta substriata*, *Sthenelais limicola*, *Spiophanes bombyx*, and *Nephtys longosetosa* were consistently identified as sensitive species in all three data sets. Species that are intermediately ranked or neutral (regression coefficients near zero) showed the least change in abundance relative to drilling impact.

Species consistently identified as sensitive or tolerant using the strength of the gradient were nearly identical to those identified by Gray *et al.* (1990) when they used a similarities percentages (SIMPER) program applied to ordination results (Appendix 1,2, and 3). Six species (*Capitella sp., Jassa marmorata, Lunatia montagui, Glycera alba, Philine scabra, Chaetozone setosa*) and one additional grouping (Copepoda indet) were identified as tolerant by ordination based SIMPER analysis and gradient estimates. Nine species (*Montacuta substriata, Abra prismatica, Amphiura filiformis, Scoloplos armiger, Goniada maculata, Ophiura affinis, Sthenelais limicola, Ampelisca macrocephala*, and *Nephtys longosetosa*) were identified as sensitive by both methods. Gray *et al.* (1990) identified 10 species sensitive to contamination. Of the top 15 sensitive species identified using the gradient method, 7 matched the species identified by Gray *et al.* Of the top 20 sensitive species identified using the gradient method 9 matched the species identified by Gray *et al* (1990). Using SIMPER, Gray *et al.* (1990) identified 7 species tolerant to contamination. Of the top 15 tolerant species identified using the gradient method, 5 matched the species identified by Gray *et al* (1990). Of the top 20 tolerant species identified using the gradient method, all 7 species identified using SIMPER were identified.

The results of the ANCOVA indicated that an increase in the species*distance interaction term was associated with contamination gradients created by oil drilling activity.

There was no significant interaction term and no overall gradient within an ANCOVA of species abundance and distance for the Gyda 1987 baseline survey. This shows that gradients relative to the production site were absent prior to platform placement, as expected. In 1990 the interaction term was still not significant, however it was closer to the 0.05 significance level. In 1993 after 6 years of drilling operations the interaction term between species abundance and distance became highly significant (Table 4.2). The gradients in species abundance took several years to become detectable at the 95% certainty level.

Non-metric MDS provided the best fit for the data, therefore only non-metric MDS plots are presented. Non-metric MDS (level of measurement on an ordinal scale) resulted in the lowest stress coefficients indicating the best fit for the data. Stress is a measure of fit ranging from 1 (worst possible fit) to 0 (perfect fit). The stress measure for Ekofisk was 0.004 using an ordinal level of measurement, 0.072 using an interval measurement, and 0.102 using a ratio measurement level. The stress measure for Gyda 1990 was 0.010 using an ordinal level of measurement, 0.031 using an interval measurement, and 0.036 using a ratio measurement level. The stress measure for Gyda 1993 was 0.014 using an ordinal level of measurement, 0.028 using an interval measurement, and 0.035 using a ratio measurement level. Comparison between PCA, MDS and non-metric MDS revealed similar contamination groupings. This indicates that the results are representative (Green, 1979). MDS appeared slightly more sensitive at distinguishing site contamination groupings, as it identified four groupings while PCA identified three site groupings. Warwick and Clarke (1991) also found MDS to be more sensitive than PCA at identifying site groupings.

4.4 Discussion

Ordination techniques are currently used to infer species-environment relationships. For gradient designs an alternative is to directly compare response and pollution gradients. The sensitivity of gradient and ordination analyses varied with respect to several factors.

I) Sensitivity to detect an impact

Both MDS and gradient analysis identified that site differences based on species similarity matrices matched contamination gradients. MDS identified change over a greater spatial scale than gradient techniques. Multivariate techniques were therefore more sensitive at discriminating between sites or times. Warwick & Clarke (1991) and the GESAMP workshop (Gray *et al.* 1988) found multivariate methods to be more sensitive than univariate methods. Warwick & Clarke (1991) compared univariate (ANOVA), graphical distributional (biomass and relative abundance), and multivariate methods of classification and ordination. The GESAMP workshop in 1988 investigated the utility of multivariate methods such as classification (clustering), ordination (Principal Components Analysis (PCA), and Multidimensional Scaling (MDS)) and discrimination analysis with univariate analyses of a series of composite variables; such as total abundance, total biomass, dominance distributions and diversity and evenness indices. Multivariate techniques were powerful at summarising between site differences (Gray *et al.* 1988) and more sensitive than univariate analyses of composite variables (Warwick & Clarke, 1991), or gradient methods in discriminating between sites or times.

II) Developing causal models

Warwick & Clarke (1991) note that multivariate methods have the advantages of great sensitivity and generality of response, but in themselves are no more than indicators of community change. Multivariate methods produce highly abstract "factors" that are difficult to interpret and do not lend themselves to identifying the possible cause of community change or to making value judgements by policy makers or the public.

Warwick & Clarke (1991) and Clarke & Ainsworth (1993) suggest methods to link faunal MDS with environmental MDS. If the relevant environmental variables that determine community composition are correctly identified then an ordination of these variables should closely resemble the faunistic ordination. They suggest experimenting with these variables in different combinations until the closest match with the faunistic ordination is found. Warwick and Clarke (1991) provide examples where faunistic ordination patterns can be matched with ordination patterns produced from key environmental variables. However, such a process is time consuming as all environmental variables need to be tested in all possible combinations to find the optimal linking. This becomes a problem if many environmental variables were measured. Secondly while such a technique is powerful at summarising which variables 'best explain' the biotic structure and how well the community structure is explained by the full set of environmental variables measured, this technique does not enable environmental variables that affect the fauna at different scales to be readily identified.

For point source disturbances direct comparison of response and pollution gradients enable underlying causal mechanisms to be identified. Gradient analysis of the Ekofisk platform indicated a change in community structure toward dominance by a few tolerant species. This occurred at the same spatial scale as the physical disturbance associated with high levels of drilling mud adjacent to the platform. These sites were organically enriched and anaerobic due to high levels of silt and clay. This community of tolerant species was localised to within the immediate vicinity of the platform (≤150 m). Response of sensitive species occurred over larger spatial scales where trace metals but not silts were recorded. An attempt to determine the scale of the impact and what caused some ecological change is needed in order to provide relevant information to decision makers and managers (Underwood, 1996). While ordination identified that benthic community change occurred to distances of 6 km at Ekofisk, and 4 km at Gyda, it could not be used to investigate the scale of siltation and heavy metal contamination. Ordination results in a plot of points relative to two or more axes synthesised from the data. The analyst must then resort to subjective groupings of points according to proximity on axes that require interpretation (Van de Geer, 1971). Factors causing the site or species groupings must then be interpreted from unknown explanatory factors synthesised from the data. Investigating the scale of effects of different environmental variables is therefore difficult.

III) Hypothesis testing

Gradient methods enable quantitative predictions to be generated and tested. In order for marine ecology to contribute to environmental issues we need to turn statements of hypothesis into quantitative predictions rather than qualitative ones (Underwood, 1996). A quantitative prediction would be to identify the change in some variable as a function of the change in distance (contamination gradient). For example a model of exponential attenuation with distance from the platform for barium is (cf. Chapter 3);

$$Ba = c\, e^{b.r}$$

$$\ln(Ba) = \ln(c) + b\,.\,r$$

where r = radial distance from the platform, b = slope, and c = intercept

Another example is to predict whether the gradient of attenuation is anisotropic or isotropic. If a predominant current exists in the environment, elevated levels of mud and barium would be predicted to be recorded at greater distances downstream of the platform (anisotropic gradient). The use of predictive monitoring is recommended over open-ended monitoring. Predictive monitoring is advantageous as it focuses directly on impacts and the underlying mechanisms, rather than on indirect indicators (cf. Chapter 2). Predictions developed in an environmental impact statement are tested during the monitoring program. Feedback monitoring (Gray & Jensen, 1993) results in a change in regulations or mitigation if the predictions are exceeded. Suter (1996) also recommends a similar approach of presenting estimates of expected or observed effects, and also suggests presenting associated uncertainties with these estimates. Suter recommends that at contaminated sites gradients or levels of exposure must be identified, and care must be taken to match in space and time the measures of effects to these gradients or levels. The use of a gradient sampling layout (Ellis & Schneider, 1997), and testing for pollution gradients enables quantitative predictions of levels of exposure and measures of effects to be readily generated and tested.

IV) Identification of sensitive and tolerant species

A similarity percentages (SIMPER) program described by Warwick *et al.* (1990) is used to identify species responsible for site groupings from an ordination. SIMPER identifies the species responsible for the discrimination observed among sites or years, by dissection of the Bray-Curtis dissimilarity matrix. An average of contributions (d_i) to the dissimilarities between all possible cross-year cross-site pairs of replicates, for each pair of years or sites, and separately for each species (i) is computed. The average (d_i) can then be ranked across species, to give an ordering from most to least important species in the determination of that year to year or site to site difference. Another means of identifying the relationship between species and the principal components (Mante *et al.* 1995) is to compute linear correlation coefficients between species and the principal components as an aid to interpreting the ordination analysis. An alternative is to use regression to estimate species abundance gradient as a function of distance. Direct estimation of spatial gradients distinguished sensitive from tolerant species that were identified using the SIMPER program. Estimation of spatial gradients utilise the same species covariance information as multivariate methods and so was expected to be no less sensitive.

Conclusion

Both ordination and gradient methods identified a clear contamination gradient due to oil drilling activities. Once analysis indicates that there is an environmental disturbance in an area, it is necessary to identify its importance. This includes how large or extensive the impact is. Another issue that also influences managerial decisions is to attempt to determine what caused the ecological change (Underwood, 1996). Gradient techniques sample and test for change as a function of contamination gradients from the source of impact. Causal models can therefore be formulated, and the results can be readily interpreted and presented. This approach has the advantage of focusing directly on impacts and the underlying causal mechanisms, rather than on indirect indicators of change. Gradient techniques were therefore effective at determining what caused an ecological change. Ordination identified community change at greater scales than simple univariate techniques. Ordination techniques were therefore more sensitive at identifying the scale of the impact, for data collected from a gradient layout.

Table 4.1 Statistical analyses for the detection of environmental impacts using a gradient sampling design; Samples are taken at t random times before and t random times after the putative impact (distances sampled = d)

Where distance is a continuous variable

Source of Variation		Degrees of Freedom
Before versus After	= B	1
Times (in Before versus After)	= T (B)	$2 (t-1)$
Distance	= D	1
B * D		1
Time (B * D)		$2 (t-1) * 1$

Where distance is a categorical variable

Source of Variation		Degrees of Freedom
Before versus After	= B	1
Times (in Before versus After)	= T (B)	$2 (t-1)$
Distance	= D	$(d-1)$
B * D		$(d-1)$
Time (B * D)		$2 (t-1) (d-1)$

Table 4.2 Analysis of covariance, species abundance with distance

a) GYDA 1987 (baseline suvery)

Source	DF	F value	Pr > F
Distance	1	0.33	0.5645
Species	1	0.55	0.4578
Distance * Species	1	0.02	0.8926

n = 1,908
SSTot = 27,132

b) GYDA 1990

Source	DF	F value	Pr > F
Distance	1	2.2	0.1379
Species	1	4.65	0.0311
Distance * Species	1	2.82	0.0934

n = 2,363
SSTot = 3,068,774.9

c) GYDA 1993

Source	DF	F value	Pr > F
Distance	1	6.36	0.0117
Species	1	11.37	0.0008
Distance * Species	1	7.08	0.0078

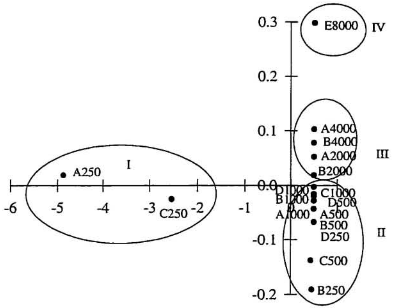
n = 3,120
SSTot = 4,944,468.4

d) Ekofisk 1990

Source	DF	F value	Pr > F
Distance	1	37.02	0.0001
Species	1	56.77	0.0001
Distance * Species	1	26.32	0.0001

n = 41,230
SSTot = 8,925,877.9

Fig 4.1 Non-metric MDS ordination by station of the macrofaunal species abundance data for Gyda 1990 oil field. Groupings are represented by the envelopes around the samples. Plots of the classification groupings on the original sample stations are provided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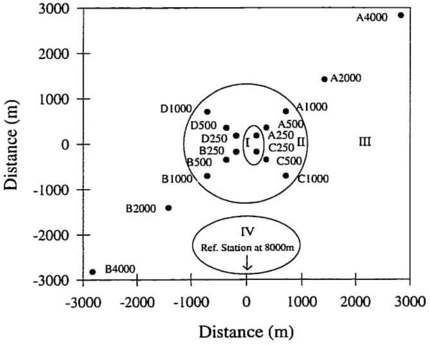
IV
A4000
B4000
A2000
III
A250
I
B2000
D1000
C1000
B1000
D500
A1000
A500
B500
D250
II
C250
C500
B250
0.3
0.2
0.1
0.0
-0.1
-0.2
-6
-5
-4
-3
-2
-1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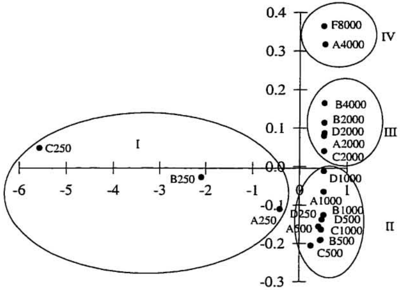
A1000
D500
A500
D250
A250
I
II
III
B250
C250
B500
C500
B1000
C1000
B2000
IV
Ref. Station at 8000m
B4000
Distance (m)
Distance (m)
3000
2000
1000
0
-1000
-2000
-3000

Fig 4.2 Non-metric MDS ordination by station of the macrofaunal species abundance data for Gyda 1993 oil field. Groupings are represented by the envelopes around the samples. Plots of the classification groupings on the original sample stations are provided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IV
B4000
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D2000
A2000
C2000
III
C250
I
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D1000
A1000
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B1000
D500
A250
A500
C1000
II
B500
C500
0.4
0.3
0.2
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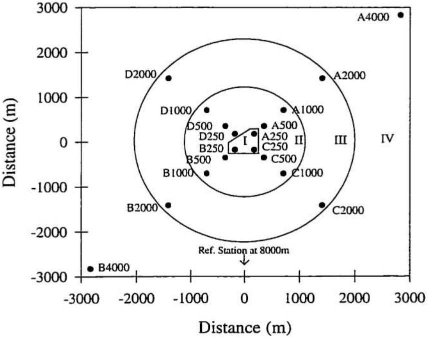
-2
-1
0
1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II
III
IV
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C500
B1000
C1000
B2000
C2000
Ref. Station at 8000m
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Distance (m)
Distance (m)
3000
2000
1000
0
-1000
-2000
-3000

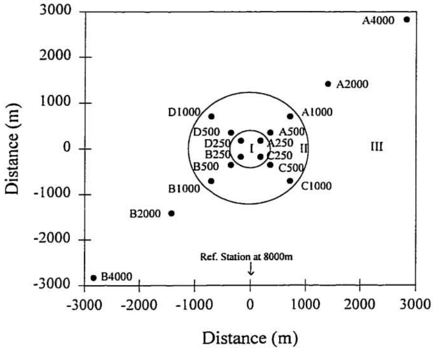


Fig 4.3 Non-metric MDS ordination by station of total hydrocarbon content, mud content, and barium for Gyda 1990 oil field. Groupings are represented by the envelopes around the original sample stations

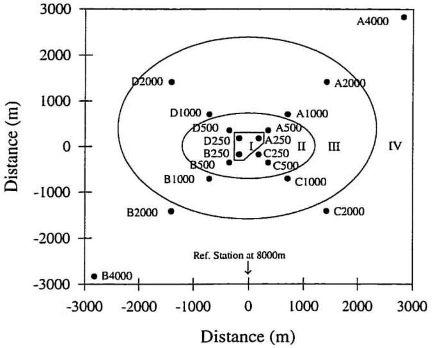


Fig 4.4 Non-metric MDS ordination by station of total hydrocarbon content, mud content, and barium for Gyda 1993 oil field. Groupings are represented by the envelopes around the original samples stations

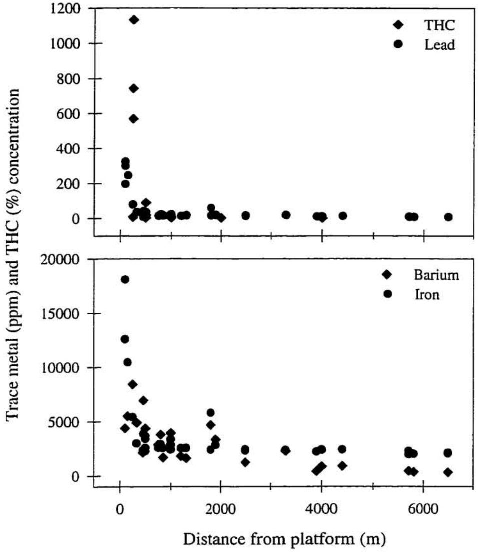


Fig 4.5 Sediment concentrations of trace metals and total hydrocarbon content (THC) with distance from Ekofisk platform

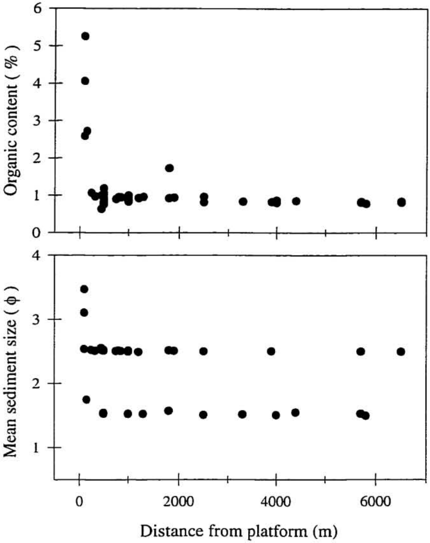


Fig 4.6 Change in sediment grain size and organic content as a function of distance from Ekofisk platform

Chapter 5. Evaluation of gradient designs to detect an anthropogenic disturbance in an environment with natural physical gradients

5.1 Introduction

Chapter 2 identified a number of assumptions and problems associated with gradient designs that required investigation. One area identified was to determine the power of gradient designs to detect a point source environmental impact. This was evaluated in Chapter 3. Another problem identified was that block statistical models are currently used to analyse data from a gradient layout. Chapter 4 investigated the use of gradient statistical models. The last area identified in Chapter 2 was the need to test the assumption of gradient designs and this will be the focus of this chapter.

Gradient designs assume that there are no natural gradients or that differences in such factors over the contamination gradients do not affect the relationship between injury and exposure (cf. Chapter 2). An assumption of no natural gradients is often violated in ecological studies. The presence of large salinity gradients, sedimentary particle size gradients, or other environmental gradients associated with an anthropogenic disturbance can distort the pattern of ecological change observed. Pearson & Rosenberg

(1978) also note that in estuaries or other areas where there are salinity gradients organisms may be exposed to the combined stress of organic enrichment and reduced and/or fluctuating salinities. Separation of the relative effects of these two factors is often difficult, even more so when other environmental factors (e.g., variation in tidal flow, sedimentary structure, temperature, etc.,) also influence the species distribution.

Data of benthic macrofaunal abundance from the Manukau Harbour sewage outlet, New Zealand, were collected to test this assumption. Manukau Harbour is a dynamic and rigorous environment in terms of movements of water and sediments (Turner *et al.*, 1995). This harbour is subject to physical wind/wave and tidal disturbance (Commito *et al.*, 1995; Thrush *et al.*,1996) as well as the anthropogenic disturbance created by the sewage outlet. The organic enrichment gradient decreases with distance from the outfall while the wind wave disturbance increases with distance (Bell *et al.*, *in press*). The aim of this study is to therefore determine how effective gradient designs are for detecting an anthropogenic disturbance in an environment with natural physical gradients.

Wiens & Parker (1995) suggest that covariance analysis using measures of other environmental variables can be used to separate effects of multiple gradients. Another method is to make *a priori* predictions (Hewitt *et al.*, 1996) of the expected community pattern as a response to the various gradients. The use of analysis of covariance and the testing of *a priori* predictions were used to try and separate the relative effects of organic and physical disturbance on the benthic communities.

Predictions

Organic Enrichment

Pearson & Rosenberg (1978) reviewed 47 publications of effects of organic enrichment and pollution on marine benthic communities. A consistent pattern to the faunal changes observed along a gradient of increasing organic input to marine sediments was observed (Fig. 5.1). In the enriched (sewage sludge) area, biomass and faunal density of a few tolerant species are very high. Further away from the organically enriched area, the number of species increased, at first slowly, but after having passed the ecotone point the number increased more rapidly towards an asymptotic value. The 'ecotone' point on this environmental gradient is a transition zone. On the heavily polluted side of this point, the community is composed of a few pollution-tolerant opportunistic species. On the less polluted side of the ecotone point the different transitory assemblages gradually approach the composition of the community in the unpolluted environment. The community at the ecotone point consists of species from both adjacent communities (Pearson, 1975; Pearson & Rosenberg, 1978; Pearson *et al.*, 1986). Recent studies of effects of organic enrichment on marine communities support the community change described by Pearson and Rosenberg (Mirza & Gray, 1981; Essink, 1984; Whitlatch & Zajac, 1985; Weston, 1990). Therefore, we predicted that;

(1) Discharges of organic wastes change the physiochemical properties of sediments and create low oxygen concentrations in the bottom water. The effects are most pronounced in the vicinity of the outfall and decrease progressively with distance from the discharge source due to mixing and dilution.

(2) Consistent patterns to the faunal changes associated with an organic enrichment gradient occur. High numbers of a few small opportunistic (tolerant) species occur adjacent to the discharge source. A maximum in the number of species is reached with distance from the outfall, before a decline to the lower species numbers usual in the unenriched environment. In the unenriched environment a complicated faunal structure involving burrow complexes of large species intermingled with smaller tube dwelling and burrowing species occurs. This complex physical and faunal structure is made up of a mosaic of interrelated micro-organisms which encompass a wide range of physical sizes, with large individuals often forming the dominant component in the structure of the whole community.

(3) A decline in suspension-feeding organisms and an increase in deposit feeding organisms occurs as organic input increases. Suspension feeders reach a maximum in the centre of the gradient, and carnivores in the areas of both high and low inputs.

Physical wind/wave disturbance

The hydrodynamic regime (mainly tidal currents) largely determines the sedimentary characteristics of an area. Tidal currents determine to a large degree the nature of the bottom substrate but, in addition, they influence the stability of the sediment, and the nature of the food supply for benthic organisms (Sanders, 1958; Warwick & Uncles, 1980). A number of studies exist that examine a relationship between grain size (reflective of hydrodynamic conditions) and trophic structure (Sanders, 1958; Grange, 1977). Grange (1977) was able to relate grain size and animal abundance, and found that the percentage of deposit feeders was inversely proportional to grain size. Deposit feeders were found in fine sand whereas suspension feeding organisms were most abundant in sand of medium grain size. Warwick & Uncles (1980) also found that species in the sand wave zone tended to be more robust filter feeders while those in sedimentary areas include more delicate deposit feeders. Therefore we predicted that;

(4) A decline in deposit feeders and an increase in filter feeders occurs as physical disturbance due to wind/wave and tidal velocity increases.

5.2 Methods

Manukau Harbour is located on the west coast of the North Island, New Zealand. The Auckland city sewage treatment facility has an outfall into the Manukau Harbour.

Sampling sites were arranged along two transects taken with distance from the outfall (Fig 5.2). Six sites were taken on transect two, but only five sites were sampled on transect one because of time limitations due to rising tides. At each site 5 core samples (10 cm diameter by 15 cm depth) were taken. Samples were sieved (500 μm mesh) and fixed in 5 % formalin and 0.1 % Rose Bengal. In the laboratory, macrofauna were sorted, identified to the lowest possible taxonomic level, counted and preserved in 70 % alcohol. Surficial sediment (0-2 cm) was collected at each core sampling site. Replicate samples from a particular sampling location were pooled to assess grain size and organic matter. Grain size was determined by size fractionation through sieving. Samples were reacted with 6% hydrogen peroxide and left for 24 hours. Samples were then wet sieved (2mm, 500μm, 125μm, and 63μm sieves) and each fraction was dried in a 60 °C oven and weighed. Organic matter was determined from ash free dry weight. Water velocity values for mean ebb and flood tides were generated from an oceanographic model of the harbour's tidal regime (Bell *et al., in press*).

Data analysis

Changes in macrofaunal abundance as a function of distance from the outfall, transect, and replicate (within site variation) were analyzed using Analysis of Variance (ANOVA). The statistical package SAS was used to calculate p values under the assumption of independent residuals with equal variance. Residuals were plotted against expected values. If no association between the residuals and expected values were evident the model was assumed to be an acceptable description of the data (Draper & Smith, 1981).

The variability in macrofaunal community structure in relation to the changing wind/wave disturbance and organic loading gradients was examined using multivariate analyses. Data were analyzed using non-metric multidimensional scaling (MDS), detrended canonical correspondence analysis (DECORANA) and canonical correspondence analysis (CANOCO). The ordination configurations obtained from DECORANA and CANOCO were similar, therefore only results from CANOCO are presented. Congruent configurations obtained indicates that a realistic ordination has been achieved. Non-metric MDS was performed on the species abundance matrices using the SPSS package (Norusis, 1994). CANOCO was used to identify relationships between the community structure and the environmental variables using PC-ORD (McCune & Mefford, 1995). Environmental variables included the organic content, sediment grain size (% <63μm, 63μm, 125μm, 500μm, 2mm), and the mean velocity of ebb and flood tides (cm/s).

Another method to identify species responsible for changes in community structure is to calculate the magnitude and sign of spatial gradients (See Chapter 4). Analysis of Covariance (ANCOVA) for species abundance as a function of distance was used to test whether gradients were heterogeneous among species. Regression coefficients were calculated for all species and then ranked.

5.3 Results

Analysis of variance indicates that distance is the only explanatory variable tested that was related to density of benthic macrofaunal organisms (Table 5.1). Changes in benthic abundance for transect, replicate and interaction terms were not significant. Change in the numbers of benthic organisms as a function of distance is associated with changes in organic enrichment and in physical disturbance gradients.

Organic content of sediment samples was recorded in order to determine discharges due to the sewage outfall. Organic enrichment occurred within 1,400m (T1/S2, T2/S1, T2/S2) of the outfall (Fig. 5.3). Sediment grain size was recorded to determine the physical disturbance due to wind/wave disturbance. Sediment grain size indicated an increase in wind/wave disturbance with distance from the outfall (Fig. 5.4). Mud content (silt/clay fraction) was elevated within 1,400m of the outfall. These sites are sheltered from the predominant wind/wave disturbance by Puketutu Island. The percentage of sand (125μm to 2mm) and tidal velocity (Table 5.2) increases for sites beyond Puketutu Island.

Total abundance of benthic organisms decreased as a function of distance while species richness increased (Fig. 5.5). The relationship between species richness and distance was not a simple linear function. Species richness was reduced at sites 1 and 2 within 1,400 meters of the outfall. These sites were characterized by high numbers of a few tolerant species. There appeared to be a peak of high diversity at site 3, transect 2 (distance=2,300m). Diversity dropped to levels similar to sites adjacent to the outfall at the far sites (4,200m and 4,600m).

MDS grouped sites as a function of the organic and physical disturbance (Fig. 5.6). Sites 1 and 2 (T1/S2, T2/S1, T2/S2) on both transects were grouped. These sites are polluted (high organic content), with low wind/wave disturbance (high mud content), and low tidal velocity. Unpolluted sites with high physical disturbance (low organic content, high % sand) were grouped together. Results of CANOCO (Fig. 5.7) did not group sites in relation to organic enrichment or physical disturbance as expected. Canonical correspondence analysis of the macrofaunal abundance data indicating site positions (Fig 5.8) and species positions (Fig. 5.9) are graphed separately. CANOCO grouped sites at 1700 and 2300 meters (T2/S3, T1/S3) from the outfall as being in environments of high mud and organic content. However, sites at 800 meters (T2/S1 & T1/S2) were found in habitats with a higher sand content (Fig. 5.8). These groupings do not correspond to simple distance plots which indicate that the sites within 1,400m of the outfall are subject to high organic enrichment and low physical disturbance. The species groupings were also hard to interpret. For example, species found on the left hand side of the diagram are species identified as being associated with sandy habitats, while species on the lower right hand side of the diagram are those found in muddy environments (Fig. 5.9). Therefore, *Aquilaspio*, *Heteromastus*, *Anthopleura*, *Comine*, *Nucula*, *Methalimendon* and *Austrovenus* are associated with sandy habitats, while *Cirolana*, *Cossura*, *Boccardia*, *Helice*, *Phoronid*, *Nemertean*, *Ophelidae* and *Ostracod sp.* are associated with muddy, enriched habitats. These groupings are not consistent with the species groupings identified using covariance analysis or the plots of sensitive and tolerant species as discussed below.

Another method of determining which species account for the change in community structure, is to rank density gradients from negative to positive (Table 5.3). The negative coefficients indicate the species most tolerant to the contamination gradient. Species identified as tolerant with high abundance adjacent to the outfall in polluted regions were *Heteromastus filiformis*, *Aquilaspio aucklandica*, *Arthritica bifurca*, *Boccardia sp.*, *Torridaharpinia hurleyi*, *Owenia fusiformis*, *Nereid*, *Nemertean* etc. Species identified as sensitive were *Magelona dakini*, *Macomona liliana*, *Cossura sp.*, *Trochodota dendyi*, *Orbinia papillosa*, *Colurostylis lemurum* etc. Benthic organisms identified as sensitive or tolerant from the regression coefficients were plotted to investigate biological change associated with the pollution and physical disturbance gradient. Figure 5.10 provides examples of three tolerant species identified using regression analysis. *Heteromastus filiformis*, *Boccardia sp.*, and *Aquilaspio aucklandica* occur at high densities within 1,700 m of the outfall. These sites are those identified as being polluted with low physical disturbance. Figure 5.11 provides examples of two sensitive species, *Magelona dakini* and *Cossura sp.* These species occur at sites identified as having low organic content and high physical disturbance. The species identified as sensitive or tolerant to organic enrichment using density gradients are different than those identified using CANOCO. Where possible, taxa were allocated to feeding guilds principally following Grange (1977) with further information provided by Fauchald & Jumars (1979). A change from suspension feeders to deposit feeders along a gradient of increasing organic enrichment was not observed (Table 5.3).

5.4 Discussion

Elevated levels of organic content were most pronounced in the vicinity of the outfall as predicted. However, a progressive decrease in the level of organic enrichment with distance did not occur (Prediction 1). Wind/wave disturbance created a sharp boundary of enriched versus un-enriched areas, rather than the predicted gradient of attenuation to background levels. The plots of organic content and sediment grain size for Manukau Harbour indicate that elevated levels of organic enrichment occur within 1,400m of the outfall. These sites were sheltered from the predominant wind/wave disturbance by Puketutu Island. Beyond Puketutu Island the wind/wave disturbance and tidal currents increase. The hydrodynamic regime largely determines the sedimentary characteristics of an area (Sanders, 1958; Warwick & Uncles, 1980). Water movement is one of the critical factors controlling the deposition of organic material at the sediment-water interface. In areas of strong currents little deposition will take place. Only the heaviest particles settle out and such coarse sediments retain little organic material (Warwick & Uncles, 1980). Most organically enriched sediments are in areas where low current speeds are prevalent for at least some of the time (Warwick & Uncles, 1980). In such areas finer particles form a greater proportion of the sediments and the organic content is correspondingly higher (Sanders, 1958). The predicted gradient of impact was therefore modified by sediment transport due to increased hydrodynamic factors for sites beyond Puketutu Island.

A gradient of community change was observed for Manukau Harbour, however, the unenriched sites showed a different pattern of community structure than expected (Prediction 2). The review by Pearson and Rosenberg (1978) found high numbers of a few tolerant species within the vicinity of a discharge source. This occurred for Manukau where high numbers of a few small tolerant species such as *Heteromastus*, *Aquilaspio* and *Boccardia*, were recorded within 1,400m of the outfall. The Pearson and Rosenberg model (1978) suggests that a maximum in species richness occurs after an 'ecotone' point. At Manukau Harbour the number of species rose to 46 at 2,300 m as compared to

20 species in the enriched zone. Pearson and Rosenberg's (1978) model then suggests that after this transition zone the species number will decline to the lower species number usually found under normal conditions. However, their model predicts that diversity in the unperturbed communities is still higher than the enriched zone where only a few tolerant species can survive in the anoxic conditions. The unperturbed or normal community consists of a complicated faunal structure dominated by large individuals (Pearson and Rosenberg, 1978). At Manukau Harbour, this pattern was not observed. The far, unenriched sites had low species diversity similar to the diversity recorded at the organically enriched sites. These sites were dominated by small sized polychaetes, contrary to the prediction. The effects of physical wind/wave disturbance on macrobenthic communities may result in the different community composition found at the far unenriched sites than predicted.

Shallow water coastal and estuarine habitats, such as Manukau Harbour, are considered to be dynamic and physically rigorous environments, characterized by large predictable and unpredictable fluctuations in environmental variables and subject to continual disturbance. The most obvious impacts of hydrodynamic conditions on the macrobenthic communities are likely to arise from wind-generated wave activity leading to increased turbulence and currents, which in turn, result in increased physical disturbance of the surface sediment (Turner *et al.*, 1995). Wind will also contribute to desiccation stress at times of low tide, with potential effects on the survival of some species. Less obvious impacts are the effects of sedimentary and hydrodynamic conditions on the relative success of larval, juvenile and adult transport and settlement (Turner *et al.*, 1995). Pearson and Rosenberg (1978) note that the presence of large salinity gradients, sedimentary particle size gradients, or other environmental stresses associated with organic enrichment may distort the pattern of ecological change observed and would give rise to considerable differences in the resulting species distributions. At Manukau Harbour the physical disturbance may result in the different gradient of community structure than predicted.

Prediction 3, that a decline in suspension-feeding organisms and an increase in deposit feeding organisms occurs as organic enrichment increases was not supported. Prediction 4, that a decline in deposit feeders and an increase in filter feeders occurs as a response to physical disturbance was also not supported. The ratio of filter feeders to deposit feeders did not change noticeably with distance from the outfall. Pearson & Rosenberg (1978) consider the changing relationship between the various trophic groups composing the resident communities along a gradient of organic enrichment, and show the decline of the suspension-feeding element and the increase in deposit-feeders as organic input to the sediment increases. A decrease in suspension feeders in the enriched areas may be due to physical clogging of ciliary and siphonal mechanisms, an inability to withstand lower oxygen tensions, and the increasing sedimentary instability brought about by deposit feeders (Pearson & Rosenberg, 1978). At Manukau the enrichment tolerant species identified using spatial gradients were mainly deposit feeders. However, a noticeable increase in suspension feeders did not occur with distance from the outfall. Pridmore *et al.* (1990) highlight problems with inferring relationships between trophic structure and grain size. Firstly, allocating animals to trophic groups can be difficult. Studies of some taxa have indicated that feeding behaviour is too variable to allow allocation to distinct groups. Some organisms are capable of switching their mode of feeding, examples of plasticity in feeding behaviour are provided in Fauchald & Jumars (1979) and by Pridmore *et al.* (1990). Snelgrove & Butman (1994) also provide evidence for plasticity in feeding mode as a function of flow and sediment transport. Many species of surface deposit feeders, for example, are now known to be facultative suspension-feeders (Hughes, 1969; Buhr & Winter, 1977; Dauer *et at.*, 1981) evidently in response to flow and elevated fluxes of suspended particles (Levinton, 1991, Taghon & Greene, 1992). Finally, in shallow and turbid environments, where surface sediments and associated diatom mats are frequently resuspended, differentiation between deposit feeders, suspension feeders and grazers etc., is further confounded.

There is little evidence that sedimentary grain size alone is the primary determinant of infaunal species distributions (Snelgrove & Butman, 1994). Snelgrove & Butman (1994) suggest a shift in focus towards understanding relationships between organism distributions and the dynamic sedimentary and hydrodynamic environment. Grain size covaries with sedimentary organic matter content, pore-water chemistry, and microbial abundance and composition, all of which are influenced by the near-bed flow regime. Hence, until there is further research evaluating the many aspects of hydrodynamic and sediment transport regimes, it is unlikely any meaningful predictive relationships will emerge (Snelgrove & Butman, 1994).

Conclusions

A priori predictions are one method to relate patterns to process. Another method is to use exploratory statistical analyses to relate community composition to known variation in the environment. CANOCO was developed to analyse and visualise the relationship between many species and many environmental variables (Ter Braak, 1986, 1987, 1988). However, in this research CANOCO was not effective at identifying species-environment relationships. The sites recorded with the highest organic and mud content were grouped by CANOCO as occurring in sandy habitats relative to other sites. Species identified by CANOCO as being associated with muddy habitats included *Cossura*. This species was identified by estimation of spatial gradients as sensitive to organic enrichment, with abundance increasing with distance from the outfall. *Heteromastus* and *Aquilaspio* were grouped as occurring in environments with a higher sand content, again estimation of spatial gradients and simple distance plots suggest that these species are found in the muddy enriched sites adjacent to the sewage outfall.

Reliance on *post-hoc* statistical methods to separate effects of multiple gradients is therefore to be cautioned.

An alternative to determine species responsible for community change is to use direct estimation of spatial gradients. Estimation of the magnitude of spatial gradients successfully separated sensitive and tolerant species. The use of *a priori* predictions was also effective at inferring relevant processes. *A priori* predictions about the distribution of benthic communities were supported for the organically enriched sites. However, the *a priori* predictions of the scale of disturbance from enrichment, and the predictions of community structure change to suspension feeding communities dominated by large individuals at the far un-enriched sites were not supported. It is suggested that the physical wind/wave disturbance modified the scale of enrichment and the community structure observed. The use of *a priori* predictions and spatial gradients were therefore more effective at describing the species-environment relationship than the exploratory technique of choice (CANOCO).

While the occurrence of other environmental disturbances along a gradient of anthropogenic disturbance makes interpretation of community pattern more difficult, knowledge of the differences in the expected and observed patterns greatly increases ecological understanding of how to link patterns and processes. The use of a gradient sampling layout together with testing *a priori* predictions enables impacts of the anthropogenic and natural environmental disturbances to be interpreted. However this is an area requiring future research. Greater efforts to include analysis of spatial patterns (density variations and habitat gradients) in both field experiments and survey programmes should be a priority for benthic ecologists. Time series data and spatial pattern analysis of larger scale phenomena may also enable general mechanistic hypothesis to be generated and tested.

Table 5.1 Data for ANOVA between environmental variables and macrofauna for Manukau Harbour

Source	DF	F Value	Pr > F
Distance	1	43.46	0.0001
Transect	1	1.32	0.25
Replicate	1	0.24	0.6218
Dis*Trans	1	0.16	0.6911
Trans*Rep	1	1.21	0.2723
Dis*Rep	1	0.24	0.6222
Dis*Trans*Rep	1	0.81	0.3693

n = 3,795
SSTot = 400,070.6

Table 5.2 Mean tidal currents for Manukau Harbour survey sites

Transect	Distance (m)	Peak ebb (cm/s) mean tide	Peak flood (cm/s) mean tide
1	800	25	16
	1700	30	36
	2900	19	30
	3600	25	27
	4200	26	23
2	800	16	9
	1400	25	35
	2300	44	38
	2600	43	19
	3300	33	9
	4600	31	19

Table 5.3 Strength of density gradient, as estimated by regression for Manukau Harbour (n = 3,795)

Species	Coefficent	Feeding Guild
Heteromastus	-0.02230268	Surface deposit feeder, motile, non-jawed
Aquilaspio aucklandica	-0.01636695	Surface deposit feeder, discretely motile, tentaculate
Arthritica bifurca	-0.00976996	Suspension feeder
Boccardia sp.	-0.00934195	Surface deposit feeder, discretely motile, tentaculate
Torridaharpina hurleyi	-0.00252242	
Owenia fusiformis	-0.00215429	Filter & deposit feeder, discretely motile, tentaculate
Nucula hartivigiana	-0.00122876	Deposit feeder
Nereididae	-0.00105504	Carnivore, motile, jawed
Halicarcinus whitei	-0.00100429	
Nemertean	-0.00076341	
Waitangi brevirostris	-0.00058541	
Amphipod ii	-0.00022017	
Helice crassa	-0.00020043	Deposit feeder
Cominella glandiformis	-0.0001676	Carnivore
Anthopleura aureoradiata	-0.00015869	
Ostracod ii	-0.00013809	
Austrovenus stutchburyi	-0.00013562	Suspension feeder
Notoacmea spp.	-0.00011277	
Glycera americana	-0.00006727	Carnivore
Soletellina siliqua	-0.00003069	Deposit feeder
Nebalacea	-0.00002371	
Scolecolepides sp.	0	Surface deposit feeder

Methalimendon sp.	0.00001255	
Ophelliidae	0.00001631	Non-selective deposit feeder
Mactra ovata	0.00002146	
Gastropod i	0.00002167	
Exosphaeroma chilensis	0.00002961	
Ostracod i	0.0000309	
Cyclaspis thomsoni	0.00003273	
Cirolana sp.	0.00003809	
Phoronid	0.00003906	Surface deposit feeder, motile, non-jawed
Halicarcinus cookii	0.00003981	
Scaleworm	0.00004206	carnivore, motile, jawed
Ruditapes sp.	0.00004206	
Corophid	0.00004206	
Edwardsii	0.00004206	
Phyllodocid	0.00004517	Carnivore
Syllid	0.00004517	Carnivore
Euchone	0.00004517	Filter feeder, sessile, tentaculate
Aricidea	0.00004517	Non-selective sub-surface deposit feeder
Aminotrypan	0.00004517	
Felaniella zelandica	0.00004517	
Paraonid ii	0.00004517	Non-selective sub-surface deposit feeder
Exosphaeroma falcatum	0.00004871	
Zeacumantus lutulentus	0.00004871	Surface deposit feeder
Hermit crab	0.00004914	
Goniada emerita	0.00005011	Carnivore, dicretely motile, jawed
Macroclymenella stewartensis	0.00005697	Sub-surface deposit feeder
Lepidodontidae	0.00005697	
Eteone neo durantiaca	0.00005783	Carnivore

Chitin	0.00006567	
Maldonidae	0.00006567	
Exogonidae	0.0000676	Carnivore
Aglaphamus macroura	0.00006921	Carnivore, motile, non-jawed
Cominella fusiformis	0.00006921	
Xymene plebeius	0.00006921	Carnivore
Amphipod	0.00007232	
Polydora sp.	0.00007232	Filter & surface feeder
Sphaerosyllis semiverrocosa	0.00007232	Carnivore, motile, non-jawed
Oligochaete	0.00007436	
Microspio sp.	0.00007564	Surface deposit feeder
Diloma subrostrata	0.00008112	
Paraonid	0.00008616	
Colurostylis lemurum	0.00009506	
Orbinia papillosa	0.00009635	Non-selective sub-surface deposit feeder
Trochodota dendyi	0.00011105	
Cossura sp.	0.00029828	Sub-surface deposit feeder, motile, non-jawed
Macomona liliana	0.00053165	Surface deposit feeder
Magelona dakini	0.0077029	Surface deposit feeder, discretely motile, tentaculate

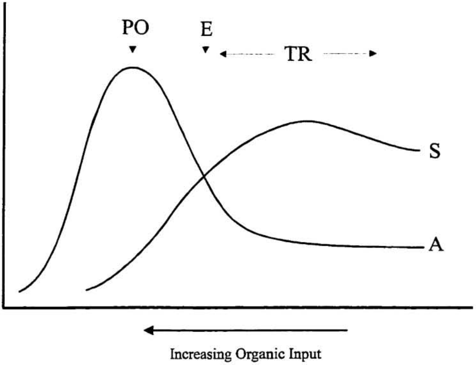
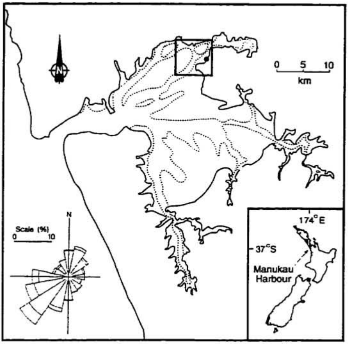


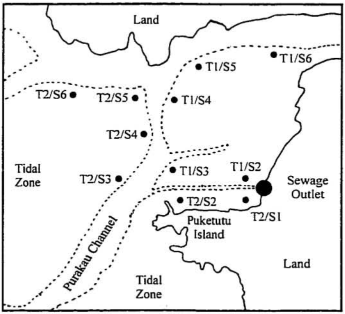
Fig 5.1 Generalised species abundance diagram along a gradient of organic enrichment: S, species number; A, total abundance; PO, peak of opportunists; E, ecotone point; TR, transition zone. (From Pearson & Rosenberg, 1978)

Fig 5.2 (a) Location of Manukau Harbour, New Zealand. Dotted line indicates area of sand-flat exposed at spring low tide. Wind rose data for Auckland International Airport. (b) Map indicating the location of the sites sampled at Manukau Harbour (T = Transect, S = Site).

(a)


0 5 10
km
Scale (%)
0 10
N
174°E
37°S
Manukau
Harbour

(b)


Land
T1/S6
T1/S5
T2/S6
T2/S5
T1/S4
T2/S4
Tidal
Zone
T2/S3
T1/S3
T1/S2
Sewage
Outlet
T2/S2
T2/S1
Puketutu
Island
Purakau Channel
Land
Tidal
Zone

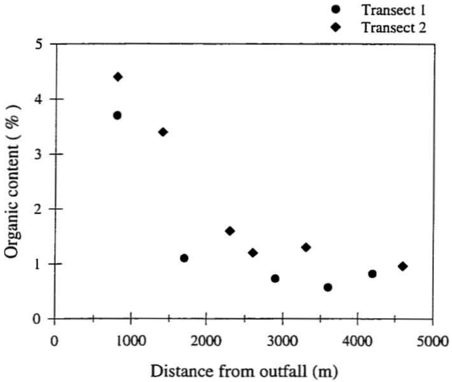


Fig 5.3 Change in sediment organic content as a function of distance from the Manukau Harbour sewage outfall

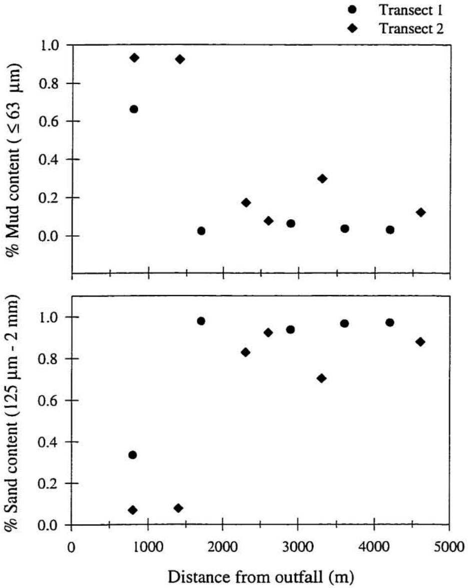


Fig 5.4 Change in sediment grain size with distance from the Manukau Harbour sewage outfall

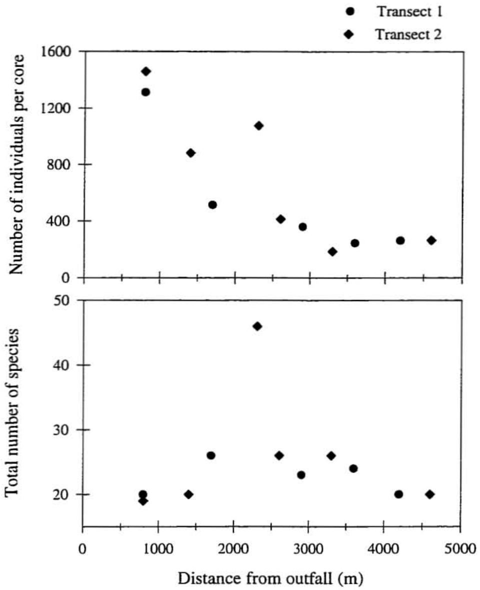


Fig 5.5 Change in macrofaunal abundance and species richness with distance from Manukau Harbour sewage outfall

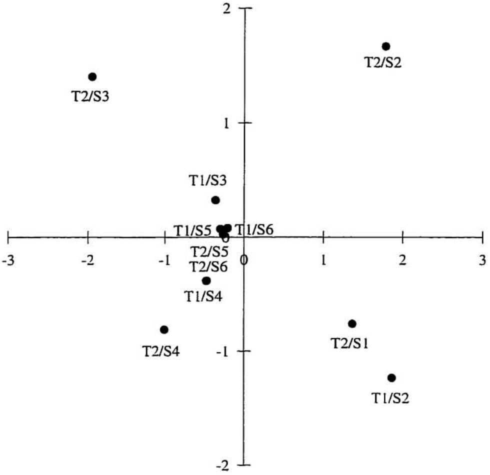


Fig 5.6 Non-metric MDS ordination by station of the macrobenthos species abundance data for Manukau Harbour sewage outfall (T = Transect, S = Site)

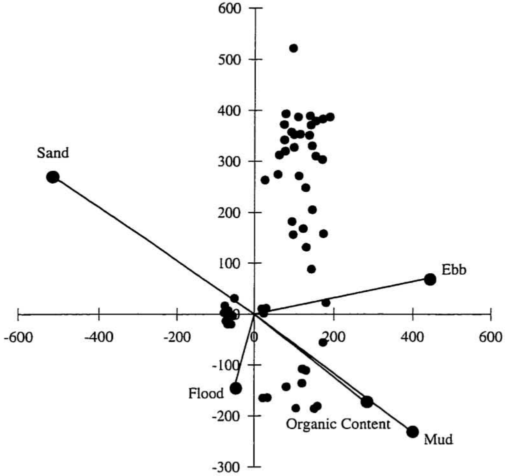


Fig 5.7 Canonical correspondence analysis of environmental variables (lines) and sampling sites (•), using macro-faunal abundance data from Manukau Harbour

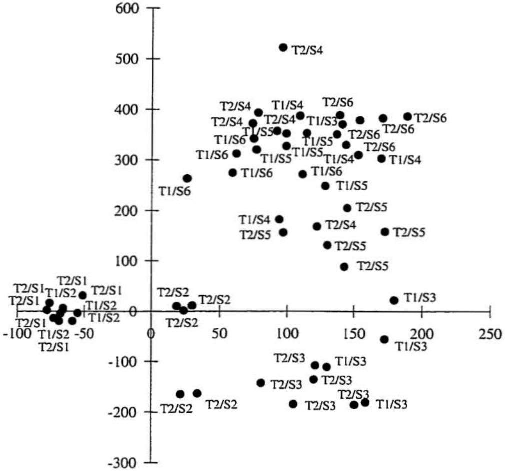


Fig 5.8 Canonical correspondence analysis of the macrofaunal abundance data, indicating site positions for Manukau Harbour (T = Transect, S = Site)

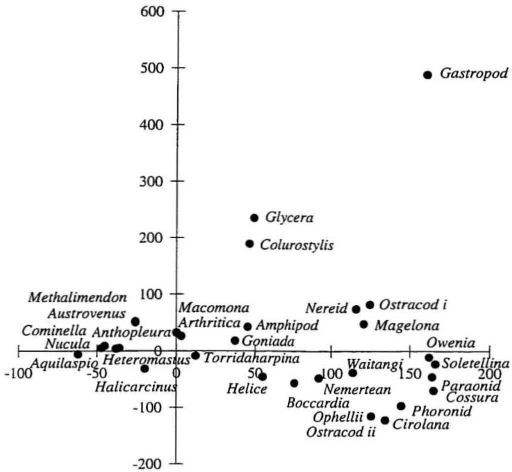


Fig 5.9 Canonical correspondence analysis of the macrofaunal abundance data, indicating species positions for Manukau Harbour

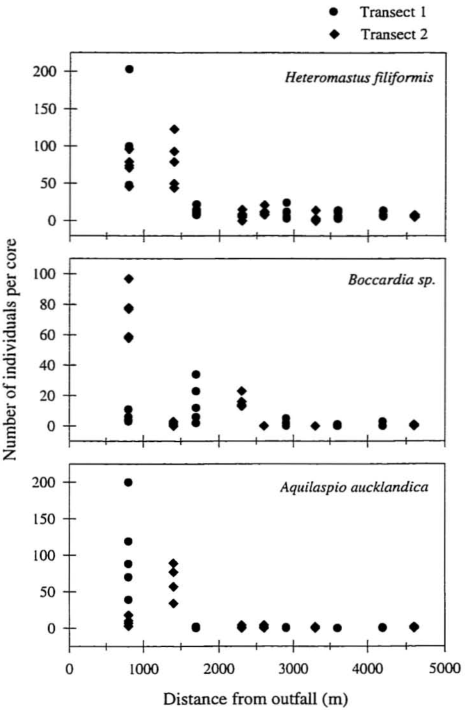


Fig 5.10 Change in abundance of tolerant species with distance from the Manukau Harbour sewage outfall

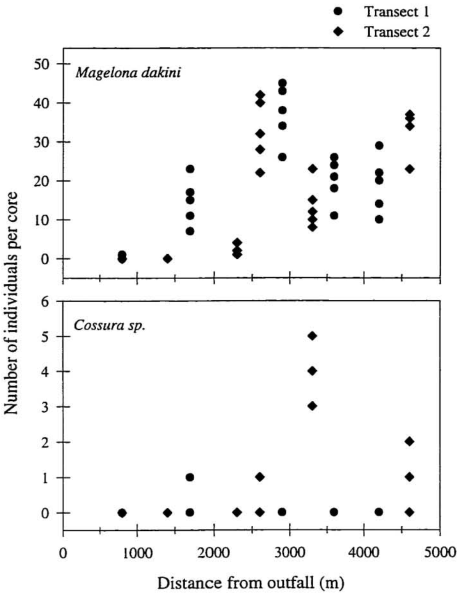


Fig 5.11 Change in abundance of sensitive species with distance from Manukau Harbour sewage outfall

Chapter 6. Summary

Experimental designs based on strata have been developed to detect and monitor environmental impacts. These designs are appropriate for disturbances that have defined boundaries. However many disturbances diffuse in the environment and the scale of the disturbance is unknown. The use of block sampling designs to detect these gradient impacts causes difficulties in selecting appropriate control sites, and in monitoring temporal and spatial changes. An alternative gradient design is proposed and developed in this thesis. Chapter 2 evaluates the appropriate application and assumptions of gradient and block designs. A stratified layout and statistical model should be used to detect a block impact, and are generally consistently used in the literature. A gradient layout and statistical model should be used to detect a gradient impact. However many examples occurred in the literature where block sampling layouts are applied to detect gradient impacts. Chapter 3 therefore evaluated the power of a block and a gradient sampling layout to detect a gradient impact. Chapter 3 concludes that a gradient layout is more powerful to detect gradient impacts and enabled spatial and temporal community changes to be readily identified. Block statistical models have been developed and are applied correctly to analyse block impacts. However, again block statistical models have been used to analyse data from a gradient layout. A lack of appropriate gradient statistical models may have resulted in the application of widely known and applied block models to analyse gradient impacts. However if a stratified statistical model is applied when effects are graded, then the variation within a block due to the graded effects will appear in, and hence inflate the error term, reducing sensitivity. Chapter 4 evaluates statistical methods of analysis for data from a gradient sampling layout. The assumptions of gradient designs are also highlighted and tested. The primary assumption is that no natural gradients exist or that natural gradients do not effect community response to the anthropogenic disturbance. This assumption was tested using data where an increasing physical gradient due to increased wind/wave and tidal disturbance existed with distance from a sewage outfall (Chapter 5). However this is an area requiring future research. Greater effort to include analysis of spatial patterns (density variations and habitat gradients) in both field experiments and survey programmes was identified as a research priority.

Areas of impact assessment requiring further research have been identified in this thesis. The GESAMP workshop (Gray *et al.* 1988) on effects of pollutants suggests that analysis based on higher taxa may more closely reflect gradients of contamination or stress than those based on species data. This has important implications for impact assessment as the time-consuming process of species identification could be replaced by more rapid determination of class or phyla. Results from Ekofisk analysis of gradients indicate that taxonomic class is not suitable as a predictor of response to pollution. It is suggested that rather than grouping species by taxonomy, functional groupings could be tested.

Both morphological and behavioral adaptations may render a species tolerant or sensitive to disturbances generated by drilling activity and other environmental disturbances. The results from Ekofisk suggest that certain characteristics such as ability to survive in anoxic conditions, motility, and feeding ecology may render a species sensitive or tolerant. Further research in this area is required to determine whether predictor characteristics can be found in other disturbed areas. It is suggested that identification of such characteristics would provide a predictive tool to assess likely impacts of proposed development.

The use of explicit descriptions of spatial patterns to infer relevant processes is another area requiring further research. Wiens (1986) notes that descriptions of spatial patterns of community structure or population heterogeneity can provide a stepping stone to inferring relevant processes. This approach was used successfully to gain further understanding of the relative importance of organic loading and physical disturbance on the benthic community at the Manukau Harbour. While the use of *a priori* predictions was powerful in inferring relevant processes, a reliance on *post hoc* statistical methods to infer species-environment relationships was not useful in inferring species-environment relationships.

The contribution of this thesis was to develop an alternative sampling design to detect point source disturbances where the scale of the impact is unknown. The assumptions of gradient designs are highlighted and areas requiring further research identified. Findings specific to environmental impact assessment that differ from findings of previous researchers are also identified. These are potential areas for future research.

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Appendix 1. Strength of density gradient, as estimated by regression for Gyda 1990 oil field (n = 2,363)

SPECIES	REGRESSION COEFFICIENT (u = # / m^3)	SPECIES IDENTIFIED BY GRAY
Capitella capitata	-0.048282	* Tolerant
Chaetozone setosa	-0.002843	* Tolerant
Cerebratulus sp.	-0.001451	
Philine scabra	-0.000804	* Tolerant
Ophiura sp.	-0.000772	
Lunatia montagui	-0.000384	* Tolerant
Nemertini	-0.00024	
Glycera cf. alba	-0.000217	* Tolerant
Prionospio cirrifera	-0.000209	
Tryphosites longipes	-0.0002	
Glycera sp.	-0.000196	
Lunatia alderi	-0.00013	
Anaitides groenlandica	-0.000116	
Pseudocuma similis	-0.000088411	
Hippomedon denticulatus	-0.000087525	
Pariambus typicus	-0.000082429	
Amphilocus spencebatei	-0.000059384	
Timoclea ovata	-0.000052958	
Cirratulus cirratus	-0.000052515	
Paraonis gracilis	-0.000049191	
Argissa hamatipes	-0.000046089	
Glycinde nordmanni	-0.000036561	
Luidia sarsi	-0.000035453	
Okenia pulchella	-0.00002659	
Tmetonyx cicada	-0.000021715	
Ampharete cf. finmarchica	-0.000020164	
Nephtys hombergii	-0.000020164	
Diastlyis cf. lucifera	-0.000020164	

Pagurus cf. bernhardus	-0.000020164
Diastylis laevis	-0.000017948
Philine cf. quadrata	-0.000017727
Brania sp.	-0.000016397
Catriona gymnota	-0.000016397
Apherusa bispinosa	-0.000016397
Stenothoe marina	-0.000016397
Perioculodes longimanus	-0.000010193
Scolelepis tridentata	-0.000008863
Petalosarsia declivis	-0.000008863
Pseudocuma longicornis	-0.000008863
Chone sp.	0.000006204
Ophelina modesta	0.000006204
Samytha sexcirrata	0.000006204
Clathrus cf. trevelyanus	0.000006204
Astarte cf. montagui	0.000006204
Phascolion strombi	0.000006204
Nereis zonata	0.000011079
Acanthocardia echinata	0.000012409
Spio filicornis	0.000016175
Gattyana cirrosa	0.000018391
Aphrodita aculeata	0.000019942
Corophium crassicorne	0.000019942
Ampelisca macrocephala	0.000025925
Autolytus sp.	0.000032351
Aonides paucibranchiata	0.000036339
Apistobranchus tullbergi	0.000036339
Aricidea minuta	0.000036339
Enipo elisabethae	0.000036339
Eteone sp.	0.000036339
Euclymene sp.	0.000036339
Hydroides norvegica	0.000036339
Lumbrineris fragilis	0.000036339
Myriochele sp.	0.000036339
Pectinaria koreni	0.000036339
Graphis albida	0.000036339
Anomia ephippium	0.000036339

Gari fervensis	0.000036339
Hiatella artica	0.000036339
Montacuta substriata	0.000036339
Musculus niger	0.000036339
Lembos longipes	0.000036339
Hyas coarctatus	0.000036339
Tharyx sp.	0.00004343
Cirratulus sp.	0.000043652
Synchelidium sp.	0.000050299
Arctica islandica	0.000053623
Diastyloides biplicatata	0.000060049
Anaitides subulifera	0.000072679
Caulleriella sp.	0.000072679
Ampelisca brevicornis	0.000072679
Parapleustes bicuspis	0.000072679
Photis longicaudata	0.000072679
Ampharete sp.	0.000078883
Cylichna cylindracea	0.000085088
Campylaspis rubicunda	0.000087747
Lebbeus polaris	0.000087747
Westwoodilla caecula	0.000095945
Diplocirrus glaucus	0.00009661
Magelona sp.	0.00009661
Nephtys cirrosa	0.00009661
Thelepus cincinnatus	0.00009661
Abra prismatica	0.00009661
Montacuta ferruginosa	0.00009661
Ampelisca tenuicornis	0.00009661
Echinocardium cordatum	0.00009661
Ophelia limacina	0.000106
Spatangus purpureus	0.000109
Owenia fusiformis	0.000111
Nephtys caeca	0.000121
Leucothoe lilljeborgi	0.000133
Abludomelita obtusata	0.000135
Campylaspis costata	0.000136
Chaetoderma nitidulum	0.000137

Virgularia mirabilis	0.000139	
Pectinaria auricoma	0.000145	
Parathemisto sp.	0.00015	
Phoronis cf. muelleri	0.00015	
Polydora sp.	0.00016	
Hemilamprops rosea	0.000161	
Bathyporeia elegans	0.000163	
Sphaerodorum flavum	0.000169	
Phaxas pellucidus	0.00018	
Cerianthus lloydi	0.000182	
Synchelidium maculatum	0.00019	
Poecilochaetus serpens	0.000193	
Sosane gracilis	0.000193	
Chamelea gallina	0.000211	
Ophiodromus flexuosa	0.000212	
Harpinia antennaria	0.000266	
Phtisica marina	0.000333	
PLATYHELMINTHES	0.000431	
Aricidea simonae	0.000466	
Edwardsia sp.	0.000563	
Echinocardium flavescens	0.000616	
Ericthonius rubricornis	0.000618	
Antalis sp.	0.000764	
Nephtys assimilis	0.000844	
Mysella bidentata	0.001041	
Spiophanes kroeyeri	0.001208	
Pholoe inornata	0.001653	
Nephtys longosetosa	0.001844	* Sensitive
Paramphinome jeffreysii	0.002303	
Goniada maculata	0.002357	* Sensitive
Spiophanes bombyx	0.002357	
Astropecten irregularis	0.002577	
Sthenelais limicola	0.00398	* Sensitive
Eudorellopsis deformis	0.004486	
Amphiura sp.	0.005357	
Scoloplos armiger	0.005381	* Sensitive
Amphiura filiformis	0.00805	* Sensitive

Appendix 2. Strength of density gradient, as estimated by regression for Gyda 1993 oil field
(n = 3,120)

SPECIES	REGRESSION COEFFICIENT (u = # / m^3)	SPECIES IDENTIFIED BY GRAY
Chaetozone setosa	-0.052499	* Tolerant
Corymorpha nutans	-0.006252	
Nemertini sp.A	-0.003262	
Phoronis mulleri	-0.001078	
Philine scabra	-0.000856	* Tolerant
Glycera alba	-0.000435	* Tolerant
Dosinia cf lincta	-0.000426	
Nemertini sp.B	-0.0004	
Arctica islandica	-0.000365	
Edwardsia spp.	-0.000323	
Lunatia montagui	-0.000282	* Tolerant
Parapleustes bicuspis	-0.000267	
Lunatia alderi	-0.000238	
Ophelia limacia	-0.000231	
Bathyporeia elegans	-0.000191	
Pectinaria auricoma	-0.000171	
Chone infundibuliformis	-0.000143	
Caulleriella spp.	-0.000109	
Cirratulus cirratus	-0.000104	
Jassa marmorata	-0.000103	* Tolerant
Nephtys longosetosa	-0.000102	
Eulimella cf ventricosa	-0.000089829	
Scalibregma inflatum	-0.000089354	
Asteropecten irregularis	-0.000086027	
Myriochele oculata	-0.000071293	
Parathermisto gaudichaudi	-0.000065589	
Nepthys caeca	-0.000061312	
Retusa umbilicata	-0.000060837	

Tmetonyx cicida	-0.00005751
Platyhelminthes sp.	-0.000056084
Paramphinome jeffreysii	-0.000055608
Pontocrates altamarinus	-0.000055133
Timoclea ovata	-0.000051331
Mya cf truncata	-0.000047529
Ampharete lindstroemi	-0.000047053
Eusyllis blomstrandi	-0.000045152
Anaitides groenlandica	-0.000032319
Scolelepis tridentata	-0.000032319
Thyasira flexuosa	-0.000029943
Eteone flava	-0.000027567
Ophelina modesta	-0.00002519
Scionella lornensis	-0.00002519
Stenothoe marina	-0.00002519
Glycera lapidum	-0.000024715
Diplocirrus glaucus	-0.000022338
Cochlodesma praetenue	-0.000021863
Acteon tornatilis	-0.000018061
Dosinia lupinus	-0.000018061
Nereis zonata	-0.000017586
Diastylis laevis	-0.00001711
Pariambus typicus	-0.00001711
Capitella capitata	-0.00001616
Euchone southerni	-0.00001616
Argissa hamatipes	-0.00001616
Tryphosites longipes	-0.00001616
Macropipus pusillus	-0.00001616
Thyasira equalis	-0.00001616
Phascolion strombi	-0.00001616
Priapulus caudatus	-0.00001616
Labidoplax buskii	-0.00001616
Chaetopterus variopedatus	-0.000015684
Synchelidium sp.	-0.000015209
Thracia cf villosiuscula	-0.000015209
Hydroides norvegica	-0.000013783
Spiochaetopterus typicus	-0.000013783

Astacilla longicornis	-0.000013783
Hippomedon denticulatus	-0.000013783
Astarte montagui	-0.000013783
Ophiura texturata	-0.000013783
Schistomysis ornata	-0.000013308
Virgularia mirabilis	-0.000010456
Thelepus cincinnatus	-0.00000903
Ampelisca sp.	-0.00000903
Iphimedia obesa	-0.00000903
Philine cf quadrata	-0.00000903
Harmothoe imbricata	-0.000008555
Scaphander punctostriatus	-0.00000808
Exogone hebes	0.000000475
Nepthys ciliata	0.000000475
Paradoneis lyra	0.000000475
Praxillura longissima	0.000000475
Spio filicornis	0.000000475
Phthisica marina	0.000000475
Meganyctiphanes norvegica	0.000000475
Macropipus holsatus	0.000000475
Pagurus bernhardus	0.000000475
Spisula subtruncata	0.000000475
Echinocyamus pusillus	0.000000475
Chone sp.	0.000003327
Ophelina acuminata	0.00000808
Amphilochus spencebatei	0.000010932
Mysia undata	0.000012357
Myriochele danielsseni	0.000015684
Chone duneri	0.000019487
Ophiodromus flexuosa	0.000019487
Phyllodoce rosea	0.000019487
Paradulichia typica	0.000019487
Philine sp.	0.000019487
Echinocardium pennatifidum	0.000019487
Lysilla loveni	0.000019962
Aonides paucibranchiata	0.000020437
Cirratulus sp.	0.000020437

Pholoe inornata	0.000038498
Nicomache lumbricalis	0.000038973
Montacuta tenella	0.000039873
Lembos longipes	0.000040399
Eteone longa	0.000048479
Photis reinhardi	0.00004943
Musculus niger	0.00004943
Polycirrus spp.	0.00005751
Trichobranchus roseus	0.000057985
Synchelidium haplocheles	0.000057985
Pseudopolydora antennata	0.00005846
Pseudocuma similis	0.00005846
Chamelea striatula	0.000059886
Westwoodilla caecula	0.000060837
Antalis entale	0.000063688
Diastylis sp.	0.000086027
Polydora caeca	0.000087452
Magelona alleni	0.000094106
Ampelisca tenuicornis	0.000096483
Lagis koreni	0.000096958
Sphaerodorum gracilis	0.000098384
Anobothrus gracilis	0.000098859
Echinocardium flavescens	0.000104
Melita obtusata	0.000115
Abra nitida	0.000118
Hemilamprops rosea	0.000135
Levinsenia gracilis	0.000144
Cerianthus lloydii	0.000147
Ampelisca brevicornis	0.000173
Chaetoderma sp.	0.00019
Owenia fusiformis	0.000192
Cylichna cylindracea	0.000237
Nuculoma tenuis	0.00025
Phaxas pellucidus	0.000253
Lucinoma borealis	0.000258
Aricidea spp.	0.000259
Gari fervensis	0.000272

Nephtys hombergii	0.000279	
Euclymene praetermissa	0.000309	
Ophiura affinis	0.000359	* Sensitive
Montacuta substriata	0.000442	* Sensitive
Photis longicaudata	0.000481	
Echinocardium cordatum	0.000654	
Sthenelais limicola	0.000698	* Sensitive
Goniada maculata	0.000853	* Sensitive
Montacuta ferruginosa	0.001028	
Scoloplos armiger	0.001226	* Sensitive
Abra prismatica	0.001302	* Sensitive
Spiophanes bombyx	0.001399	
Eudorellopsis deformis	0.00168	
Spiophanes kroyeri	0.002125	
Harpinia antennaria	0.003597	
Amphiura filiformis	0.011772	* Sensitive
Mysella bidentata	0.015793	

Appendix 3. Strength of density gradient, as estimated by regression for Ekofisk oil field
(n = 41,230)

SPECIES	REGRESSION COEFFICIENT (u = # / m^3)	SPECIES IDENTIFIED BY GRAY
Capitella capitata	-0.02424233	* Tolerant
Myriochele oculata (ref. to sp.)	-0.00383279	
Chaetozone setosa	-0.000458633	* Tolerant
Copepoda indet.	-0.00042256	* Tolerant
Glycera alba	-0.00030296	* Tolerant
Acteon tornatilis	-0.000241363	
Nemertini indet.	-0.000175832	
Eudorella sp.	-0.000168974	
Edwardsia sp.	-0.000154624	
Lunatia montagui	-0.000142856	* Tolerant
Prionospio cf. cirrifera	-0.000130789	
Pholoe minuta	-0.000108394	
Chaetoderma nitidulum	-0.00009312	
Paramphinome jeffreysii	-0.0000787182	
Thyasira sarsi	-0.0000749268	
Lumbrineris fragilis	-0.0000695546	
Philine scabra	-0.0000478086	* Tolerant
Cephalaspidea indet.	-0.0000390887	
Pectinaria auricoma	-0.0000374655	
Jassa marmorata	-0.0000353702	* Tolerant
Gari sp.	-0.0000313562	
Nuculoma tenuis	-0.0000306048	
Ampharete falcata	-0.000023346	
Aphrodita aculeata	-0.0000232359	
Metopa cf. solsbergi	-0.0000200852	
Hippomedon cf. propinquus	-0.0000190765	
Aphroditidae indet.	-0.0000181515	
Priapulida indet.	-0.0000146783	

Nematoda indet.	-0.0000140687
Eudorellopsis deformis	-0.0000132585
Bivalvia indet.	-0.0000124448
Phoronis sp.	-0.000012144
Tryphosites longipes	-0.0000118097
Timoclea ovata	-0.0000115049
Lunatia alderi	-0.0000107542
Nereis zonata	-0.0000097633
Lucinoma borealis	-0.0000075972
Nudibranchia indet.	-0.0000071867
Westwoodilla caecula	-0.0000067033
Tmetonyx cicada	-0.0000066665
Caligidae indet.	-0.0000060101
Typosyllis sp.	-0.0000060023
Amphicteis gunneri	-0.0000054445
Ophiura albida	-0.0000051078
Phaxas pellucidus	-0.0000050014
Menigrates obtusifrons	-0.0000048845
Phyllodocidae indet.	-0.0000047037
Philine cf. quadrata	-0.0000046173
Ophelina sp.	-0.0000045691
Diastylis sp.	-0.0000045499
Nephtys ciliata	-0.0000044018
Hyperia cf. galba	-0.0000042806
Opheliidae indet.	-0.0000041643
Spisula subtruncata	-0.0000040977
Ampharete finmarchica	-0.000003761
Troschelia berniciensis	-0.0000030012
Phascolion strombi	-0.0000028948
Owenia fusiformis	-0.0000026049
Scopelocheirus crenatus	-0.0000025298
Protodorvillea kefersteini	-0.0000025106
Pycnogonidae indet.	-0.0000025106
Campylaspis rubicunda	-0.0000025106
Idotea pelagica	-0.0000025106
Spiochaetopterus sp.	-0.0000024433
Mytilidae indet.	-0.0000024433

Lunatia sp.	-0.0000022009
Campylaspis costata	-0.0000022009
Chone sp.	-0.0000020393
Buccinum undatum	-0.0000020393
Scolelepis tridentata	-0.0000020258
Philine quadrata	-0.0000019719
Petalosarsia declivis	-0.0000019719
Ophelina acuminata	-0.000001962
Ophiura texturata	-0.0000016353
Modiolus sp.	-0.0000015679
Musculus niger	-0.0000015679
Notomastus latericeus	-0.0000015006
Pomatoceros sp.	-0.0000015006
Sabellidae indet.	-0.0000015006
Gitana sarsi ?	-0.0000015006
Paramphilochoides intermedius	-0.0000015006
Sphaerodorum flavum	-0.0000014134
Schistomysis ornata	-0.0000012986
Epimeria cf. cornigera	-0.0000012986
Metopa cf. alderi	-0.0000012986
Orchomene sp.	-0.0000012986
Stegocephaloides christianiensis	-0.0000012986
Meganyctiphanes norvegica	-0.0000012986
Arctica islandica	-0.0000010462
Perioculodes longimanus	-0.0000008832
Philine sp.	-0.0000008704
Cylichna cylindracea	-0.0000008173
Laonice cirrata (refer to Laonice sp.)	-0.000000577
Eupagarus bernhardus	-0.0000004806
Thracia phaseolina	-0.0000003459
Ditrupa sp.	-0.0000003076
Maldanidae indet.	-0.0000002212
Pectinaria koreni	-0.0000002212
Astarte sp.	-0.000000173
Nephtys assimilis	0
Nichomache sp.	0
Caudofoveata indet.	0

Gari costulata	0
Ampelisca brevicornis	0
Iphimedia minuta	0
Thysanoessa longicaudata	0
Crangon crangon	0
Asteroidea indet.	0
Spatangoidea indet.	0
Harmothoe sp.	0.0000000099
Spintheridae indet.	0.0000005004
Gastropoda indet.	0.0000006103
Ophiodromus flexuosa	0.0000006876
Eugyra arenosa	0.0000010335
Iphimedia obesa	0.0000010398
Harmothoe fragilis	0.0000012411
Mysella bidentata	0.0000012419
Ditrupa arietina	0.0000014432
Cirripedia indet.	0.000001799
Placostegus sp.	0.000002607
Scaphander punctostriatus	0.000002607
Amphiuridae indet.	0.000002607
Turbonilla crenata	0.0000027417
Antalis vulgaris	0.0000027417
Luidia sarsi	0.0000027417
Leptosynapta inhaerens	0.0000027417
Leptasterias muelleri	0.0000028381
Dosinia exoleta	0.0000030210
Cylichna alba	0.0000030983
Scaphopoda indet.	0.0000032804
Lysianassidae indet.	0.0000032904
Scalibregma inflatum	0.0000033286
Lebbeus polaris	0.0000035306
Trichobranchus roseus	0.0000036192
Thyasiridae indet.	0.0000041367
Polycarpa fibrosa	0.0000048583
Aonides paucibranchiata	0.0000050312
Flabelligeridae indet.	0.0000050312
Neoamphitrite grayi	0.0000050312

Polychaeta indet.	0.0000050312
Montacuta sp.	0.0000050312
Hyas coarctatus	0.0000050312
Nereimyra punctata	0.0000051659
Modiolus modiolus	0.0000051659
Nichomache borealis	0.0000052339
Hippomedon cf. denticulatus	0.0000054834
Argissa hamatipes	0.0000056280
Antalis cf. vulgaris	0.0000058875
Scolelepis cf. foliosa	0.0000060222
Astacilla longicornis	0.0000061086
Leucothoe cf. lilljeborgi	0.0000061086
Spatangus purpureus	0.0000061086
Eteone sp.	0.0000061937
Echinocardium sp.	0.0000067253
Mysia undata	0.0000077729
Pariambus typicus	0.0000086774
Cirratulidae indet.	0.0000095918
Metopa alderi	0.00000964
Amphilochus manudens	0.0000100624
Abra prismatica	0.0000102552
Synchelidium haplocheles	0.0000108641
Apherusa cf. bispinosa	0.0000111398
Pseudocuma sp.	0.0000112086
Nephtys hombergii	0.000011297
Sosane gracilis	0.0000113915
Lanice conchilega	0.0000114574
Maldane glebifex	0.0000117268
Cumacea indet.	0.0000119096
Polynoidae indet.	0.0000120351
Aricidea simonae	0.0000122236
Ophiura sarsi	0.0000124583
Spatangus sp.	0.0000132968
Hemilamprops rosea	0.0000134577
Spiophanes bombyx	0.0000134875
Amphilochus spencebatei	0.0000137469
Phtisica marina	0.0000140070

Ampharetidae indet.	0.0000143437	
Acanthocardia echinata	0.0000153347	
Anaitides groenlandica	0.0000156919	
Corophium affine	0.0000173931	
Echinocardium cordatum	0.0000177780	
Spiophanes kroeyeri	0.0000180282	
Cerianthus sp.	0.0000190319	
Harpinia antennaria	0.0000199037	
Montacuta ferruginosa	0.0000202702	
Hippomedon denticulatus	0.0000215106	
Platyhelminthes indet.	0.0000216588	
Pholoe pallida	0.0000236477	
Antalis entalis	0.0000239093	
Hemichordata indet.	0.0000241474	
Ericthonius fasciatus	0.0000251561	
Pennatulacea indet.	0.0000233245	
Euclymene droebachiensis	0.0000266191	
Harmothoe cf. andreapolis	0.0000269282	
Synchelidium cf. tenuimanum	0.0000302837	
Nephtys caeca	0.0000309883	
Ampelisca macrocephala	0.0000324485	* Sensitive
Opisthobranchia indet.	0.0000333919	
Polydora sp.	0.0000433976	
Venus striatula	0.0000557304	
Astarte montagui	0.0000562947	
Cirratulus cirratus	0.0000642406	
Echinocardium flavescens	0.0000659587	
Astropecten irregularis	0.0000660523	
Porifera indet.	0.000071584	
Parathemisto gaudichaudii	0.000084967	
Chaetognatha indet.	0.00009893	
Paraonis gracilis	0.0001314801	
Ophiura affinis	0.000132698	* Sensitive
Nephtys longosetosa	0.000137398	* Sensitive
Montacuta substriata	0.00021966	* Sensitive
Scoloplos armiger	0.0004101047	* Sensitive
Anthozoa indet.	0.000443599	

Sthenelais limicola	0.000589775	* Sensitive
Goniada maculata	0.000686421	* Sensitive
Amphiura filiformis	0.000921858	* Sensitive

Appendix 4. Summary of Gyda 1987 baseline data. Species recorded, and the number of samples in which species were present

Investigated by: Oil Pollution Research Unit Field Studies Council, U.K.
Units: Antall individer pr. grabbskudd (0.1 sq.m, weighted Van Veen grab)
Stations: 18 stations x 1 replicate

SPECIES	TOTAL NUMBER	SAMPLES PRESENT
Cerianthus lloydi	15	9
Edwardsia sp.	45	17
Virgularia mirabilis	3	3
Platyhelminthes	2	2
Cerebratulus sp.	1	1
Nemertini sp1	71	18
Ampharete cf. baltica	2	2
Ampharete cf. finmarchica	1	1
Ampharete sp.	2	2
Anaitides groenlandica	8	6
Anaitides subulifera	2	2
Aonides paucibranchiata	33	13
Apistobranchus tullbergi	5	4
Aricidea minuta	3	3
Aricidea simonae	21	12
Chaetoparia nilssoni	1	1
Chaetozone setosa	25	10
Chone cf. duneri	3	2
Cirratulus cf. cirratus	5	4
Cirratulus sp.	3	3
Diplocirrus glaucus	2	2
Enipo elisabethae	1	1

Eteone sp.	8	7
Euclymene sp.	2	2
Eumida sp.	1	1
Gattyana cirrosa	2	2
Glycera cf. alba	1	1
Goniada maculata	99	18
Harmothoe castanea	2	2
Harmothoe sp.	2	2
Myriochele sp.	21	3
Nephtys caeca	13	9
Nephtys longosetosa	156	18
Ophelia limacina	112	17
Ophelina acuminata	11	1
Ophelina modesta	2	2
Owenia fusiformis	8	7
Paramphinome jeffreysii	3	3
Pectinaria auricoma	2	2
Pectinaria koreni	1	1
Pholoe inornata	43	16
Polydora sp.	2	2
Prionospio cirrifera	7	4
Scolelepis tridentata	2	2
Scoloplos armiger	213	18
Sosane gracilis	15	12
Spiophanes bombyx	305	18
Spiophanes kroeyeri	51	15
Sthenelais limicola	54	17
Tharyx sp.	37	16
Trichobranchus roseus	1	1
Lunatia alderi	3	3
Lunatia montagui	4	3
Philine cf. quadrata	3	2
Philine cf. scabra	7	5

Chaetoderma nitidulum	23	13
Abra nitida	1	1
Abra prismatica	126	18
Acanthocardia echinata	6	6
Arctica islandica	2	2
Cerastoderma scabrum	1	1
Montacuta ferruginosa	5	2
Montacuta substriata	59	8
Musculus niger	7	7
Mysella bidentata	17	10
Phaxas pellucidus	47	18
Spisula elliptica	6	5
Venus striatula	8	6
Antalis sp.	12	9
Campylaspis rubicunda	1	1
Diastylis cf. lucifera	2	2
Diastylis laevis	1	1
Eudorellopsis deformis	417	18
Hemilamprops rosea	1	1
Petalosarsia declivis	2	2
Pseudocuma similis	1	1
Ampelisca brevicornis	1	1
Ampelisca macrocephala	155	18
Amphilochus spencebatei	2	2
Argissa hamatipes	1	1
Bathyporeia cf. elegans	43	16
Byblis gaimardi	1	1
Corophium crassicorne	34	15
Gammaropsis nitida	4	2
Harpinia antennaria	1	1
Hippomedon denticulatus	9	9
Lembos longipes	21	11
Parathemisto sp.	2	2

Pariambus typicus	1	1
Perioculodes longimanus	6	5
Protomedeia fasciata	1	1
Synchelidium maculatum	10	7
Tryphosites longipes	4	4
Westwoodilla caecula	10	7
Crangon allmanni	1	1
Eupagurus cf. bernhardus	2	2
Golfingiidae indet.	3	3
Phascolion strombi	1	1
Phoronis cf. muelleri	124	17
Astropecten irregularis	25	9
Luidia sarsi	1	1
Amphiura filiformis	77	17
Ophiura affinis	3	3
Echinocardium cordatum	2	2
Echinocardium flavescens	9	9
Spatangus purpureus	1	1

Appendix 5. Summary of Gyda 1990 data. Species recorded, and the number of samples in which species were present

Investigated by: Oil Pollution Research Unit Field Studies Council, U.K.
Units: Antall individer pr. grabbskudd (0.1 sq.m, weighted Van Veen grab)
Stations: 17 stations x 1 replicate

SPECIES	TOTAL NUMBER	SAMPLES PRESENT
Cerianthus lloydi	33	14
Edwardsia sp.	44	12
Virgularia mirabilis	3	3
PLATYHELMINTHES	36	14
Cerebratulus sp.	108	14
Nemertini	47	13
Ampharete cf. finmarchica	1	1
Ampharete sp.	3	2
Anaitides groenlandica	8	6
Anaitides subulifera	2	1
Aonides paucibranchiata	1	1
Aphrodita aculeata	2	2
Apistobranchus tullbergi	1	1
Aricidea minuta	1	1
Aricidea simonae	21	11
Autolytus sp.	4	3
Brania sp.	1	1
Capitella capitata	2401	6
Caulleriella sp.	2	1
Chaetozone setosa	169	9
Chone sp.	1	1
Cirratulus cirratus	31	11

Cirratulus sp.	4	3
Diplocirrus glaucus	1	1
Enipo elisabethae	1	1
Eteone sp.	1	1
Euclymene sp.	1	1
Gattyana cirrosa	6	4
Glycera cf. alba	16	7
Glycera sp.	14	8
Glycinde nordmanni	2	2
Goniada maculata	179	17
Hydroides norvegica	1	1
Lumbrineris fragilis	1	1
Magelona sp.	1	1
Myriochele sp.	1	1
Nephtys assimilis	42	14
Nephtys caeca	5	5
Nephtys cirrosa	1	1
Nephtys hombergi	1	1
Nephtys longosetosa	191	16
Nereis zonata	3	3
Ophelia limacina	5	4
Ophelina modesta	1	1
Ophiodromus flexuosus	5	3
Owenia fusiformis	13	10
Paramphinome jeffreysii	87	15
Paraonis gracilis	3	3
Pectinaria auricoma	4	3
Pectinaria koreni	1	1
Pholoe inornata	63	16
Poecilochaetus serpens	2	1
Polydora sp.	4	4
Prionospio cirrifera	13	7
Samytha sexcirrata	1	1

Scolelepis tridentata	1	1
Scoloplos armiger	78	6
Sosane gracilis	2	1
Sphaerodorum flavum	3	2
Spio filicornis	2	2
Spiophanes bombyx	49	8
Spiophanes kroeyeri	15	3
Sthenelais limicola	208	16
Tharyx sp.	24	7
Thelepus cincinnatus	1	1
Catriona gymnota	1	1
Clathrus cf. trevelyanus	1	1
Cylichna cylindracea	4	2
Graphis albida	1	1
Lunatia alderi	13	7
Lunatia montagui	20	6
Okenia pulchella	3	2
Philine cf. quadrata	2	2
Philine scabra	109	15
Chaetoderma nitidulum	50	13
Abra prismatica	1	1
Acanthocardia echinata	2	1
Anomia ephippium	1	1
Arctica islandica	5	5
Astarte cf. montagui	1	1
Chamelea gallina	11	5
Gari fervensis	1	1
Hiatella artica	1	1
Montacuta ferruginosa	1	1
Montacuta substriata	1	1
Musculus niger	1	1
Mysella bidentata	27	4
Phaxas pellucidus	6	4

Timoclea ovata	3	3
Antalis sp.	20	7
Campylaspis costata	22	1
Campylaspis rubicunda	2	2
Diastlyis cf. lucifera	1	1
Diastylis laevis	5	5
Diastyloides biplicatata	3	3
Eudorellopsis deformis	349	16
Hemilamprops rosea	12	7
Petalosarsia declivis	1	1
Pseudocuma longicornis	1	1
Pseudocuma similis	7	6
Abludomelita obtusata	6	4
Ampelisca brevicornis	2	1
Ampelisca macrocephala	6	4
Ampelisca tenuicornis	1	1
Amphilocus spencebatei	5	5
Apherusa bispinosa	1	1
Argissa hamatipes	12	8
Bathyporeia elegans	25	14
Corophium crassicorne	2	2
Ericthonius rubricornis	17	1
Harpinia antennaria	4	2
Hippomedon denticulatus	12	7
Lembos longipes	1	1
Leucothoe lilljeborgi	2	2
Parapleustes bicuspis	2	1
Parathemisto sp.	6	3
Pariambus typicus	28	11
Perioculodes longimanus	2	2
Photis longicaudata	2	1
Phtisica marina	7	4
Stenothoe marina	1	1

Synchelidium maculatum	24	12
Synchelidium sp.	16	9
Tmetonyx cicada	5	3
Tryphosites longipes	17	8
Westwoodilla caecula	10	7
Hyas coarctatus	1	1
Lebbeus polaris	2	2
Pagurus cf. bernhardus	1	1
Phascolion strombi	1	1
Phoronis cf. muelleri	23	11
Astropecten irregularis	198	15
Luidia sarsi	4	1
Amphiura filiformis	215	14
Amphiura sp.	164	14
Ophiura sp.	269	15
Echinocardium cordatum	1	1
Echinocardium flavescens	7	2
Spatangus purpureus	3	2

Appendix 6. Summary of Gyda 1993 data. Species recorded, and the number of samples in which species were present

Investigated by: Oil Pollution Research Unit Field Studies Council, U.K.
Units: Antall individer pr. grabbskudd (0.1 sq.m, weighted Van Veen grab)
Stations: 20 stations x 1 replicate

SPECIES	TOTAL NUMBER	SAMPLES PRESENT
Corymorpha nutans	865	19
Virgularia mirabilis	8	5
Cerianthus lloydii	75	20
Edwardsia spp.	96	19
Platyhelminthes sp.	27	15
Nemertini sp.A	231	14
Nemertini sp.B	74	17
Ampharete lindstroemi	11	7
Anaitides groenlandica	2	1
Anobothrus gracilis	18	9
Aonides paucibranchiata	3	3
Aricidea spp.	85	18
Capitella capitata	1	1
Caulleriella spp.	66	19
Chaetopterus variopedatus	2	2
Chaetozone setosa	3348	19
Chone duneri	1	1
Chone infundibuliformis	9	4
Chone sp.	2	2
Cirratulus sp.	23	10
Cirratulus cirratus	52	13
Diplocirrus glaucus	33	17
Eteone flava	2	1
Eteone longa	2	2
Euchone southerni	1	1
Euclymene praetermissa	10	6

Eusyllis blomstrandi	5	1
Exogone hebes	1	1
Glycera lapidum	3	3
Glycera alba	44	15
Goniada maculata	150	18
Harmothoe imbricata	2	2
Hydroides norvegica	1	1
Lagis koreni	4	2
Levinsenia gracilis	12	9
Lysilla loveni	2	2
Magelona alleni	8	3
Myriochele danielsseni	33	1
Myriochele oculata	30	15
Nepthys caeca	11	6
Nepthys ciliata	1	1
Nephtys hombergii	67	18
Nephtys longosetosa	65	14
Nereis zonata	3	2
Nicomache lumbricalis	2	1
Ophelia limacia	84	13
Ophelina acuminata	7	4
Ophelina modesta	2	2
Ophiodromus flexuosus	1	1
Owenia fusiformis	43	15
Paradoneis lyra	1	1
Paramphinome jeffreysii	43	19
Pectinaria auricoma	30	11
Pholoe inornata	46	15
Phyllodoce rosea	1	1
Polycirrus spp.	1	1
Polydora caeca	14	11
Praxillura longissima	1	1
Pseudopolydora antennata	3	2
Scalibregma inflatum	12	8
Scionella lornensis	2	2
Scolelepis tridentata	2	2
Scoloplos armiger	60	13

Sphaerodorum gracilis	7	6
Spio filicornis	1	1
Spiochaetopterus typicus	1	1
Spiophanes bombyx	338	17
Spiophanes kroyeri	286	16
Sthenelais limicola	203	18
Thelepus cincinnatus	1	1
Trichobranchus roseus	2	2
Diastylis laevis	4	3
Diastylis sp.	11	8
Eudorellopsis deformis	394	20
Hemilamprops rosea	4	4
Pseudocuma similis	3	3
Astacilla longicornis	1	1
Ampelisca sp.	1	1
Ampelisca brevicornis	4	3
Ampelisca tenuicornis	3	3
Amphilochus spencebatei	3	3
Argissa hamatipes	1	1
Bathyporeia elegans	29	15
Harpinia antennaria	83	8
Hippomedon denticulatus	16	12
Iphimedia obesa	1	1
Jassa marmorata	8	5
Lembos longipes	5	3
Melita obtusata	3	2
Paradulichia typica	1	1
Parapleustes bicuspis	28	2
Parathermisto gaudichaudi	32	16
Pariambus typicus	4	3
Phthisica marina	1	1
Photis longicaudata	11	3
Photis reinhardi	4	2
Pontocrates altamarinus	9	8
Stenothoe marina	2	2
Synchelidium haplocheles	2	2
Synchelidium sp.	3	2

Tmetonyx cicida	4	3
Tryphosites longipes	1	1
Westwoodilla caecula	3	3
Meganyctiphanes norvegica	1	1
Schistomysis ornata	2	2
Macropipus holsatus	1	1
Macropipus pusillus	1	1
Pagurus bernhardus	1	1
Chaetoderma sp.	154	19
Acteon tornatilis	2	2
Cylichna cylindracea	9	7
Eulimella cf ventricosa	6	3
Lunatia alderi	24	10
Lunatia montagui	26	8
Philine cf quadrata	1	1
Philine scabra	119	19
Philine sp.	1	1
Retusa umbilicata	7	4
Scaphander punctostriatus	3	3
Abra nitida	24	10
Abra prismatica	60	10
Arctica islandica	97	19
Astarte montagui	1	1
Chamelea striatula	71	18
Cochlodesma praetenue	4	4
Dosinia cf lincta	43	13
Dosinia lupinus	2	1
Gari fervensis	8	5
Lucinoma borealis	23	11
Montacuta substriata	9	3
Montacuta tenella	2	2
Montacuta ferruginosa	21	2
Musculus niger	9	8
Mya cf truncata	10	9
Mysella bidentata	788	15
Mysia undata	6	4
Nuculoma tenuis	5	3

Phaxas pellucidus	12	5
Spisula subtruncata	1	1
Thracia cf villosiuscula	3	3
Thyasira equalis	1	1
Thyasira flexuosa	2	2
Timoclea ovata	97	18
Antalis entale	4	4
Phascolion strombi	1	1
Priapulus caudatus	1	1
Phoronis mulleri	362	19
Asteropecten irregularis	14	9
Amphiura filiformis	533	16
Ophiura affinis	80	15
Ophiura texturata	1	1
Echinocardium cordatum	15	4
Echinocardium flavescens	8	4
Echinocardium pennatifidum	1	1
Echinocyamus pusillus	1	1
Labidoplax buskii	1	1

Appendix 7. Summary of Ekofisk data. Species recorded, and the number of samples in which species were present

Investigated by: Akvaplan A/S, Unilab A/S
Units: Antall individer pr. grabbskudd (0.1 sq.m, weighted Van Veen grab)
Stations: 39 stations x 5 replicates (Station 5 is missing)

SPECIES	TOTAL NUMBER	SAMPLES PRESENT
PORIFERA		
Porifera indet.	38	20
ANTHOZOA		
Anthozoa indet.	112	28
Cerianthus sp.	195	102
Edwardsia sp.	191	97
Pennatulacea indet.	25	20
PLATYHELMINTHES		
Platyhelminthes indet.	30	29
NEMERTINI		
Nemertini indet.	457	133
NEMATODA		
Nematoda indet.	11	10
POLYCHAETA		
Ampharete falcata	203	67
Ampharete finmarchica	3	3
Ampharetidae indet.	7	6
Amphicteis gunneri	3	3

Anaitides groenlandica	21	19
Aonides paucibranchiata	1	1
Aphrodita aculeata	15	13
Aphroditidae indet.	9	8
Aricidea simonae	27	23
Capitella capitata	9755	26
Chaetozone setosa	413	119
Chone sp.	1	1
Cirratulidae indet.	9	7
Cirratulus cirratus	99	58
Ditrupa arietina	2	1
Ditrupa sp.	2	2
Eteone sp.	5	4
Euclymene droebachiensis	13	13
Flabelligeridae indet.	1	1
Glycera alba	210	92
Goniada maculata	1480	176
Harmothoe cf. andreapolis	28	20
Harmothoe fragilis	2	2
Harmothoe sp.	3	3
Lanice conchilega	3	3
Laonice cirrata (refer to Laonice sp.)	2	2
Lumbrineris fragilis	94	70
Maldane glebifex	3	3
Maldanidae indet.	1	1
Myriochele oculata (ref. to sp.)	2069	70
Neoamphitrite grayi	1	1
Nephtys assimilis	1	1
Nephtys caeca	31	27
Nephtys ciliata	2	1
Nephtys hombergi	383	157
Nephtys longosetosa	152	92
Nereimyra punctata	1	1

Nereis zonata	7	5
Nichomache borealis	8	6
Nichomache sp.	2	1
Notomastus latericeus	1	1
Opheliidae indet.	10	8
Ophelina acuminata	4	4
Ophelina sp.	3	3
Ophiodromus flexuosus	27	24
Owenia fusiformis	20	16
Paramphinome jeffreysii	59	31
Paraonis gracilis	230	103
Pectinaria auricoma	33	26
Pectinaria koreni	1	1
Pholoe minuta	148	91
Pholoe pallida	24	23
Phyllodocidae indet.	3	3
Placostegus sp.	1	1
Polychaeta indet.	1	1
Polydora sp.	30	19
Polynoidae indet.	8	8
Pomatoceros sp.	1	1
Prionospio cf. cirrifera	103	60
Protodorvillea kefersteini	1	1
Sabellidae indet.	1	1
Scalibregma inflatum	3	3
Scolelepis cf. foliosa	2	2
Scolelepis tridentata	1	1
Scoloplos armiger	289	95
Sosane gracilis	18	17
Sphaerodorum flavum	7	7
Spintheridae indet.	2	2
Spiochaetopterus sp.	1	1
Spiophanes bombyx	7	7

Spiophanes kroeyeri	8	8
Sthenelais limicola	589	157
Trichobranchus roseus	98	65
Typosyllis sp.	4	1
GASTROPODA		
Acteon tornatilis	263	96
Buccinum undatum	1	1
Cephalaspidea indet.	20	13
Cylichna alba	7	6
Cylichna cylindracea	4	4
Gastropoda indet.	4	4
Limacina sp.	0	0
Lunatia alderi	6	3
Lunatia montagui	85	55
Lunatia sp.	1	1
Nudibranchia indet.	11	10
Opisthobranchia indet.	18	17
Philine cf. quadrata	2	1
Philine quadrata	1	1
Philine scabra	33	15
Philine sp.	12	7
Scaphander punctostriatus	1	1
Troschelia berniciensis	2	2
Turbonilla crenata	1	1
CAUDOFOVEATA		
Caudofoveata indet.	1	1
Chaetoderma nitidulum	267	115
BIVALVIA		
Abra prismatica	6	5
Acanthocardia echinata	8	8

Arctica islandica	23	20
Astarte montagui	57	44
Astarte sp.	2	2
Bivalvia indet.	13	13
Dosinia exoleta	4	4
Gari costulata	0	0
Gari sp.	22	18
Lucinoma borealis	25	22
Modiolus modiolus	1	1
Modiolus sp.	1	1
Montacuta ferruginosa	20	17
Montacuta sp.	1	1
Montacuta substriata	135	34
Musculus niger	1	1
Mysella bidentata	10	10
Mysia undata	2	2
Mytilidae indet.	1	1
Nuculoma tenuis	32	26
Phaxas pellucidus	9	8
Spisula subtruncata	5	5
Thracia phaseolina	4	3
Thyasira sarsi	89	56
Thyasiridae indet.	2	2
Timoclea ovata	79	50
Venus striatula	112	65
SCAPHOPODA		
Antalis cf. vulgaris	2	2
Antalis entalis	23	20
Antalis vulgaris	1	1
Scaphopoda indet.	1	1

P Y C N O G O N I D A		
Pycnogonidae indet.	1	1
C O P E P O D A		
Caligidae indet.	3	3
Copepoda indet.	4877	194
C I R R I P E D I A		
Cirripedia indet.	1	1
M Y S I D A C E A		
Schistomysis ornata	1	1
C U M A C E A		
Campylaspis costata	1	1
Campylaspis rubicunda	1	1
Cumacea indet.	4	4
Diastylis sp.	2	2
Eudorella sp.	118	49
Eudorellopsis deformis	152	92
Hemilamprops rosea	35	31
Petalosarsia declivis	1	1
Pseudocuma sp.	17	14
I S O P O D A		
Astacilla longicornis	1	1
Idotea pelagica	1	1
A M P H I P O D A		
Ampelisca brevicornis	0	0
Ampelisca macrocephala	12	12
Amphilochus manudens	2	1
Amphilochus spencebatei	3	3

Apherusa cf. bispinosa	2	2
Argissa hamatipes	5	5
Corophium affine	6	6
Epimeria cf. cornigera	1	1
Ericthonius fasciatus	5	1
Gitana sarsi ?	1	1
Harpinia antennaria	5	4
Hippomedon cf. denticulatus	2	2
Hippomedon cf. propinquus	14	10
Hippomedon denticulatus	10	7
Hyperia cf. galba	2	2
Iphimedia minuta	0	0
Iphimedia obesa	9	5
Jassa marmorata	15	8
Leucothoe cf. lilljeborgi	1	1
Lysianassidae indet.	4	4
Menigrates obtusifrons	4	4
Metopa alderi	10	5
Metopa cf. alderi	1	1
Metopa cf. solsbergi	8	4
Orchomene sp.	1	1
Paramphilochoides intermedius	1	1
Parathemisto gaudichaudii	356	137
Pariambus typicus	6	5
Perioculodes longimanus	18	18
Phtisica marina	7	7
Scopelocheirus crenatus	2	2
Stegocephaloides christianiensis	1	1
Stenothoe marina	1	1
Synchelidium cf. tenuimanum	8	6
Synchelidium haplocheles	15	13
Tmetonyx cicada	133	62
Tryphosites longipes	15	15

Westwoodilla caecula	17	17
EUPHAUSIACEA		
Meganyctiphanes norvegica	1	1
Thysanoessa longicaudata	1	1
DECAPODA		
Crangon crangon	1	1
Eupagarus bernhardus	4	4
Hyas coarctatus	1	1
Lebbeus polaris	2	2
Phascolion strombi	7	7
PRIAPULIDA		
Priapulida indet.	14	11
PHORONIDA		
Phoronis sp.	686	147
ASTEROIDEA		
Asteroidea indet.	0	0
Astropecten irregularis	91	65
Leptasterias muelleri	3	3
Luidia sarsi	1	1
OPHIUROIDEA		
Amphiura filiformis	1006	154
Amphiuridae indet.	1	1
Ophiura affinis	223	98
Ophiura albida	3	3
Ophiura sarsi	7	7
Ophiura texturata	1	1

ECHINOIDEA		
Echinocardium cordatum	10	10
Echinocardium flavescens	64	38
Echinocardium sp.	11	11
Spatangoidea indet.	0	0
Spatangus purpureus	1	1
Spatangus sp.	4	3
HOLOTHUROIDEA		
Leptosynapta inhaerens	1	1
HEMICHORDATA		
Hemichordata indet.	19	17
CHAETOGNATHA		
Chaetognatha indet.	553	161
TUNICATA		
Eugyra arenosa	3	3
Polycarpa fibrosa	3	3

Appendix 8. Summary of Manukau Harbour data. Species recorded, and the number of samples in which species were present

Samples collected by: Cummings, V.J., Ellis, J.I. and Thrush, S.F.
Samples identified by: Ellis, J.I.
Units: 10 cm diameter by 15 cm depth core samples
Stations: 11 stations x 5 replicates (Transect 1 / Site 1 was not sampled)

SPECIES	TOTAL NUMBER	SAMPLES PRESENT
AMPHIPODA		
Amphipod	1	1
Amphipod ii	18	10
Corophid	2	2
Methalimendon sp.	24	18
Torridaharpina hurleyi	195	31
Waitangi brevirostris	124	30
BIVALVIA		
Arthritica bifurca	537	30
Austrovenus stutchburyi	138	26
Felaniella zelandica	1	1
Macomona liliana	326	54
Mactra ovata	9	4
Nucula hartivigiana	352	35
Ruditapes sp.	2	2
Soletellina siliqua	47	17
CRUSTACEA		
Halicarcinus cookii	5	2
Halicarcinus whitei	61	17
Helice crassa	20	11
Notoacmea spp.	84	9

Nebalacea	11	3
POLYPLACOPHORA		
Chitin	2	2
CNIDARIA		
Anthopleura aureoradiata	32	15
Edwardsii	2	2
CUMACEA		
Colurostylis lemurum	10	7
Cyclaspis thomsoni	5	2
ECHINOIDEA		
Trochodota dendyi	6	6
GASTROPODA		
Cominella fusiformis	2	2
Cominella glandiformis	14	10
Diloma subrostrata	27	10
Gastropod i	4	3
Xymene plebeius	2	2
Zeacumantus lutulentus	1	1
DECAPODA		
Hermit crab	2	2
ISOPODA		
Cirolana sp.	1	1
Exosphaeroma chilensis	6	3
Exosphaeroma falcatum	1	1
NEMERTEAN		
Nemertean	53	17
OLIGOCHAETA		
Oligochaete	3	1

OSTRACODA		
Ostracod i	9	8
Ostracod ii	91	10
PHORONIDA		
Phoronid	6	5
POLYCHAETA		
Aglaphamus macroura	2	2
Aminotrypan	1	1
Aquilaspio aucklandica	929	27
Aricidea	1	1
Boccardia	580	30
Cossura spp.	27	11
Eteone neo durantiaca	3	3
Euchone	1	1
Exogonidae	1	1
Glycera americana	14	10
Goniada emerita	40	27
Heteromastus filiformis	1671	53
Lepidodontidae	1	1
Macroclymenella stewartensis	1	1
Magelona dakini	865	42
Maldanid	2	2
Microspio	6	4
Nereid	83	26
Ophelliidae	8	3
Orbinia papillosa	2	2
Owenia fusiformis	489	21
Paraonid	3	3
Paraonid ii	1	1
Phyllodocid	1	1
Polydora	1	1
Scaleworm	2	2
Scolecolepides sp.	3	2
Sphaerosyllis semiverrocosa	1	1
Syllid	1	1

IMAGE EVALUATION
TEST TARGET (QA-3)

150mm

6"

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