RETROSPECTIVE ANALYSIS OF MARINE BIOLOGICAL DATA FROM PORT VALDEZ, ALASKA: A CASE STUDY IN LONG-TERM MONITORING

A

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ABSTRACT

Efforts to understand anthropogenic effects within the Port Valdez study area provide a simple and adaptive model for developing and refining hypotheses to measure the structure of and detect change in nearshore and benthic habitats. Drivers of change detected by this study are the 1964 Prince William Sound earthquake and the oil transportation and salmon aquaculture industries within the fjord. The study area is a glacial-outwash fjord characterized by strong seasonal and spatial environmental gradients due to glacial influences including seasonally low salinity, high suspended sediment loads, and subsequent high sedimentation rates. Direct and indirect effects from intolerance to low salinity are important in organizing intertidal communities as is habitat structure. Previously unrecognized subtle effects on subtidal fauna from anthropogenic stressors near the marine oil terminal in Port Valdez are identified. Demonstration of statistical methods (variogram estimation, repeated measures analysis of variance, and geostatistical modeling) for field studies with spatially and temporally correlated data should be useful to others seeking to establish new long-term studies or analyze previously collected, long-term field data. Investigation of the re-adjustment of benthic fauna from a large earthquake and ecosystem-level effects of salmon aquaculture are not readily available and this dissertation provides a reference point for any such future studies. Although re-adjustment from the large earthquake was a key process during the study period, salmon aquaculture appears to have a strong effect on the benthic ecosystem.

The model of detecting change is simple and adaptive and provides inputs for larger models and scientific investigations in marine ecosystems. Broad questions are developed through a long-term study of an ecosystem.

The hypotheses formulated are then evaluated and refined through retrospective analysis of long-term data and results can be used to refine larger models. This dissertation contributes rigorous, statistically bounded biological time series to regional monitoring programs by providing small-scale, ecological information necessary for larger models. As a result, the information provided in this dissertation should increase the accuracy of ecological models and aid in the management of marine resources.

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Chapter 1 Long-term Monitoring:
A Case Study of Port Valdez, Alaska

1. Introduction

As human populations grow, pressures on marine ecosystems from resource use have never been greater. Humans have consistently overused resources as demonstrated by temporal changes in fish stocks and marine mammal populations where exploitation has led to the collapse of commercially valuable populations (Jackson et al., 2001; Lewison et al., 2004; Caddy and Seijo, 2005; Scheffer et al., 2005). Damage is not limited to stocks of harvestable species but includes bycatch species, and incidental damage to marine habitat and associated fauna (e.g., damage due to trawling; Collie et al., 2000; Lewison et al., 2004; Brown et al., 2005; Scheffer et al., 2005). The overharvesting of fish and marine mammal stocks is not the only example of damage resulting from marine resource exploitation. Other examples of unsustainable uses include stress on coral reefs from human activities, development of coastal habitat, and point-source disposal of contaminants from cities and industries in critical habitats (Hall-Spencer and Moore, 2000; Zakai and Chadwick-Furman, 2002; Bax et al., 2003; Morrisey et al., 2003).

Effects from resource use in Alaska's coastal fringe have followed worldwide trends of overuse, chronic pollution, and environmental disturbance. Serial depletion of stocks (moving to new stocks as lucrative species are depleted) has been identified as a major contributor to collapses of commercially valuable crustacean populations in the Bering Sea and Gulf of Alaska including the red king crab (*Paralithodes camtschaticus*) (Orensanz et al., 1998). As a direct result of exploitation, sea otter (*Enhydra lutra*) and gray whale (*Eschrichtius robustus*) populations collapsed in the eastern Pacific leading to long-term, ecosystem-level effects (Estes et al., 1998; Springer et al., 2003). Sources of pollution in Alaska are similar to those in other urbanized coastal environments including global contaminants, chronic pollutants from vessel traffic, localized waste disposal, and runoff from urban developments (Naidu et al., 1997; Schiff et al., 2000; Blanchard et al., 2002; Blanchard and Feder, 2003; Dojiri et al., 2003; Morrisey et al., 2003). The Exxon Valdez oil spill was a large perturbation of the marine environment in Prince William Sound (PWS), damaging marine habitats, altering marine communities, and resulting in bird and marine mammal declines (Rice et al., 1996; Peterson et al., 2003). Other sources of anthropogenic disturbance in Alaska include marine dredging, dumping of sediment wastes, fish waste disposal from processing plants, and the Valdez marine oil terminal (Feder and Burrell, 1982; Johnson et al., 1998; Jewett et al., 1999; Kline and Stekoll, 2001; Blanchard et al., 2002 and 2003; Blanchard and Feder, 2003).

Effects from anthropogenic disturbance can be long-lasting. Exxon Valdez oil spill effects have persisted and 86% of randomly selected, previously oiled beaches had substantial quantities of oil 12 years later (Jewett et al., 2002; Short et al., 2004; Peterson et al., 2003). Benthic communities disturbed by dredging for gold off Nome, Alaska were still only partially readjusted 12 years after dredging stopped (Jewett et al., 1999). Interactions between multiple stressors can have unintended effects, as was observed at Nome when severe storm activity interacted with incomplete readjustment from dredging to disrupt the previously disturbed sediments (Jewett et al., 1999). Interactions between fish wastes discharge and vessel turbulence in Port Valdez, a fjord in Prince William Sound, Alaska delayed readjustment of infauna from sediment disposal and extended enrichment effects over a broader area (Blanchard and Feder, 2003).

Natural disturbance, including earthquakes, tsunamis, and storms, can have large effects on the marine environment (Coulter and Migliaccio, 1971; Haven, 1971; Hubbard, 1971; Bythell et al., 1993; Naidu and Klein, 1988; Jewett et al., 1999; Dreyer et al., 2005). A large earthquake in March, 1964 was centered in PWS and resulting tsunamis caused extensive disturbance to the marine environment in PWS and the Gulf of Alaska (Coulter and Migliaccio, 1971; Losey, 2005). Effects from uplifted land and the tsunami waves were large, resulting in the death of many intertidal organisms in uplifted and scoured sediments but some algae and invertebrates were quick to recolonize available substrates (Stanley, 1968; Haven, 1971; Hubbard, 1971). Pink and chum salmon were adversely affected by the loss of spawning habitat at uplifted stream mouths although the temporal persistence of the habitat losses is unknown (Stanley, 1968). Interactions among effects from the earthquake and later anthropogenic stressors in PWS may have occurred like those observed elsewhere in Alaska (Jewett et al., 1999).

It can be difficult to plan for, describe, or predict the effects and interactions of multiple change processes (natural variability, stressors and environmental gradients). Many subjective choices affect the power and results of statistical tests in observational studies and no single sampling design can be advocated for long-term monitoring though principles of experimental design are always applicable (e.g., Underwood, 1994; Eberhardt and Thomas, 1991; Ellis and Schneider, 1997; Kuehl, 1994). Disturbance investigations in marine environments rarely examine more than a single factor and are often spatially and temporally limited. This is due in part to the failure to recognize the importance of long-term research by resource managers and users, hesitation to adequately fund marine environmental research, and the difficulty in designing studies in systems open to many environmental influences. Research with inadequate spatial and temporal sampling (typically a result of financial constraints) often results in a failure to detect environmental damage due to low statistical power (Peterman, 1990; Caddy and Seijo, 2005). Most marine communities have already been affected by human activities, and it must be assumed that even within a very small area, changes due to anthropogenic factors have likely occurred (Steneck and Carlton, 2001).

Temporal and spatial readjustment of biota to environmental variation should be a primary concern of marine long-term monitoring studies. Any biological community affected by disturbance (disruption of biota and communities) responds through a series of successional stages over time. The resulting community may not be similar to the pre-disturbance community due to long-term change in environmental conditions as a result of disturbance, changes in larval supply and distribution, and recruitment and timing of colonizers (de Grave and Whitaker, 1999). This is readjustment rather than recovery where the latter implies a return to pre-existing conditions. If disturbance is spatially and temporally limited, recovery to the original faunal composition may occur. If disturbance is large, spatially extensive, or continues over time, communities may readjust to new environmental conditions rather than recover to pre-existing community structure. Thus, a goal of long-term monitoring should be to document how communities respond to anthropogenic stressors (factors causing stress or disturbance), natural sources of variability, and the scales at which they readjust.

Port Valdez, Alaska has been a focus for environmental investigations since construction of a marine terminal for the Trans-Alaska pipeline was proposed. Intertidal surveys began in 1968 and were followed by extensive, multi-disciplinary studies from 1971-1985 providing insights into physical, chemical, and biological aspects of the fjord (McRoy and Stoker, 1969; Hood et al., 1973; Colonell, 1980; Shaw and Hameedi, 1988) (Tables 1.1 and 1.2). The construction of the marine oil terminal marked the onset of multiple anthropogenic stressors, increasing pressure from urban growth, industrial development, and increased marine recreational activities. Shoreline developments include a salmon hatchery, a container dock, and a gas refinery. The release of millions of salmon fry and adult salmon returns (White, 2005) may have large, ecosystem effects on Port Valdez resulting in increased quantities of wastes as entrails from sport fishing and carcasses falling to the benthos and discharges from a processing plant. As a result of easier access and large fish returns, Port Valdez now has one of the largest recreational salmon fisheries in Alaska. The effects of many anthropogenic stressors on the fjord have not been measured but the largest stresses (tension or strain on biota, possibly leading to disturbance) appear to be related to the City of Valdez and the fish hatchery (Wiegers et al., 1997, 1998). Localized responses of benthic animals from the disposal of treated ballast waters at the marine oil terminal have been observed but the effects were ecologically minor (Blanchard et al., 2002 and 2003).

Environmental change processes can be single large events, continuous or one-time small events. When change processes are small, then single landscape elements (e.g., a beach) may be impacted. If, however, large environmental change occurs with effects increasing or decreasing over time, the result may be a gradient of change. With just a few sampling periods, distinguishing between a local change process and a larger gradient of change may not be possible. Initial efforts at understanding systems can erroneously assume that conditions observed are natural conditions but human impacts have likely occurred to some extent (Steneck and Carlton, 2001).

In Port Valdez, multiple gradients of change interact together to shape the marine biological communities. For example, expansion of sea otters into the fjord in the late 1970s and the growth of bird colonies represent substantial environmental change in Port Valdez and throughout Prince William Sound (Hogan and Irons, 1988). This expansion of the marine mammals and birds co-occurred and likely interacted with urban growth and industrial developments to shape the current marine communities (McRoy, 1988). Since the 1964 earthquake, no single year of "baseline" conditions can be identified due to readjustment of marine communities from changing environmental conditions and introductions of anthropogenic stressors (McRoy, 1988). What may be determined, however, is how biological communities vary as environmental conditions change over space and time and this is a goal of this dissertation. This retrospective analysis of intertidal and subtidal data for this fjord documents methods useful for identifying spatial and temporal variability for marine biological studies and the ecological responses to the long-term, environmental trends.

2. Environmental Setting

The physical and biological characteristics of Port Valdez are typical of sheltered fjords in the region (Hood et al., 1973; Colonell, 1980; Shaw and Hameedi, 1988). Port Valdez is a double-silled, glacial outwash fjord located in the northernmost corner of Prince William Sound (Fig. 1.1). The fjord is a U-shaped valley with steep rocky shores and a flat bottom and is about 5 km wide by 18 km long. Approximately 15% of the subtidal habitat in the fjord is 0-40 m in depth, and 57% is ≥ 200 m in depth (McRoy, 1988). The intertidal region is rocky in the west, grading to extensive mudflats in the eastern end of the fjord where glacially-fed rivers enter the fjord (Naidu and Klein, 1988; Blanchard and Feder, 1997 and 2000b). The tidal range is approximately 5.3 m (Colonell, 1980). Average monthly wind speeds usually range from 3 to 16 km h^{-1} but peak winds up to 64 km h^{-1} are commonly observed in winter and peak winds up to 145 km h^{-1} are occasionally observed (NOAA Weather Service, Valdez, AK).

Sediment characteristics in the study area are heavily influenced by glaciers. In the summer, the influx of sediment-laden water from melting glaciers in the surrounding mountains results in gradients in suspended sediments and sedimentation rates with higher values near the head (eastern end) of the fjord (Sharma and Burbank, 1973; Naidu and Klein, 1988). As a result, a sediment plume is apparent in surface waters over the eastern half of the fjord. Approximately 2.26×10^6 metric tons of sediments are transported to the region annually with a sedimentation rate of 1.67 cm yr^{-1} overall and 13 cm yr^{-1} at the head of the fjord (Sharma and Burbank, 1973). Silt and clays are the dominant sediment fraction in the basin sediments and organic carbon in sediments is low, ranging from 0.1% to 0.6% (Sharma and Burbank, 1973; Naidu and Klein, 1988). Early studies of the abundance of deep benthic fauna found a statistical separation between benthic communities in the east (under the summer sediment plume) and the west (Feder et al., 1973; Feder and Matheke, 1980).

The climate extremes in this Alaskan fjord can be great. Seasonal temperature differences are large as the mean air temperature ranges from 13°C in summer to -10°C in winter and surface-water temperature ranges from 16° C in summer to -2°C in winter (Hood et al., 1973; Jewett and Feder, 1977; Blanchard and Feder, 2000a). Mean annual precipitation is 158 cm. The fjord receives greatly decreased amounts of light in winter (less than seven hours of daylight from mid-November through February) with a very low angle of incidence but daylight in summer approaches 20 hours

(Alexander and Chapman, 1980).

Circulation variations in Port Valdez are seasonally large (Muench and Nebert, 1973; Colonell, 1980). The water column is stratified in summer due to solar warming and freshwater runoff. Estuarine circulation (with surface waters flowing out and subsurface water flowing in) is strong in the upper 20 m but inflows of mid-depth water are common as well (Muench and Nebert, 1973; Sharma and Burbank, 1973; Colonell, 1980). Reversals of flow, with surface waters moving in and deep water flowing out, are common. In winter the flow of freshwater into the fjord declines and the water column is mixed due to wind-driven mixing and reduced insolation, as shown by uniform temperature and salinity profiles. Movement of deep water (>120 m depth) from PWS into the fjord is limited by the outer sill (at about 120 m depth) but intrusions associated with storms and related weather events are frequent. Summer bottom-water temperatures range from 2-3° C and surface water temperatures from 2.5 to 16° C. In winter, the water column is nearly uniform ranging between 3-6° C although surface waters may be slightly cooler (-2° C). Surface salinity ranges from near 0‰-15‰ to 20 m depth in summer and >32‰ below 75 m in summer and all depths in winter.

Nutrient cycles and primary productivity in Port Valdez resembles other marine systems at similar latitudes (Goering et al., 1973). Nutrient concentrations are high in winter and early spring, fueling a phytoplankton bloom as sunlight increases and water warms. Nutrients are depleted after the bloom and primary production remains low throughout the summer in the stratified water column. In early fall, a smaller pulse of primary production may be observed with water column mixing by storms but the fall bloom is greatly restricted due to the lack of sunlight (Goering et al., 1973, Horner et al., 1973). High surface-water turbidity limits the vertical zone of primary production by decreasing light penetration, in some cases to as little as 1 m near the river mouths (Goering et al., 1973, Sharma and Burbank, 1973). Except for a narrow time horizon, the fjord appears inadequate for marine phytoplankton growth (Goering et al., 1973). In spite of these limitations, the annual net primary production is estimated as 150 g C m^2 with daily primary production rate of around 2 g C m^2 day^{-1} during the peak of the bloom (Goering et al., 1973).

Zooplankton in Port Valdez are a typical boreal assemblage similar to that of eastern Prince William Sound (Cooney and Coyle, 1988). Prince William Sound is considered a pelagic system with most food resources consumed in the upper water column and this is probably true for Port Valdez as well (Cooney and Coyle, 1988; Cooney, 1993). Food quantity reaching the benthos in deeper waters is limited by zooplankton grazing in the upper water column and dilution by glacial sediments leading to a low percent sediment organic carbon.

A small but ecologically significant saltmarsh occurs in the north eastern corner of the fjord. This small marsh supports a diverse assemblage of birds including gulls, terns, and migratory waterfowl many of which feed upon infauna particularly the bivalve *Macoma balthica* (Stout, 1999).

3. Objectives

The objective of this dissertation is to expand knowledge of spatial and temporal variability or marine communities and of monitoring methods through retrospective analysis of data from a long-term marine research program. The chapters of this dissertation provide examples of

1) analysis of spatially correlated data using geostatistical methods (Chapters 2, 4, and 5),

2) assessment of environmental gradients in a heterogeneous, spatially correlated intertidal community (Chapter 3),
3) subtle influence of treated effluents on marine, benthic assemblages (Chapter 4), and
4) long-term changes in deep subtidal benthic invertebrate communities reflecting readjustment from a large earthquake and anthropogenic stressors (Chapter 5).

In documenting these changes,

5) spatial and temporal scales of change, disturbance, and readjustment in marine communities are identified, and
6) statistical methods for long-term monitoring are assessed.

The importance of understanding long-term trends in marine communities, effects of environmental gradients of change, and anthropogenic impacts has never been greater. Lessons learned from existing long-term research programs, such as the Port Valdez project, can contribute greatly to understanding change in marine communities. Although a wealth of statistical literature exists, only a few examples of statistical methods for long-term studies are available in the marine literature (e.g., Burd, 2002; Diaz et al., 2003). Thus, demonstration of statistical methods useful for identifying spatial and temporal trends for different study designs is needed to further marine monitoring and is an important goal of this dissertation. This investigation of data from over 33 years of research using modern statistical methods provides an example useful for developing studies to assess disturbed environments elsewhere. Additionally, this work may provide reference points for ecological work in coastal environments of Alaska and for studies of effects from large earthquakes.

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Table 1.1. Summary of biological investigations in Port Valdez, Alaska. Studies are summarized by habitat and organism and the major contributions described.

Organism/habitat	Study Focus	Contributions	Citations
Intertidal flora and fauna (Rocky shores and low profile beaches)	The spatial and temporal distributions of intertidal organisms.	Investigation in 1969 observed petroleum products in the intertidal (observed both as hydrocarbons in subsurface sediments and as hardened tar or "pavement") as a result of the 1969 earthquake. This was not observed in following investigations.	McRoy and Stoker, 1969; Lees et al., 1979; Feder and Keiser, 1980; Feder and Bryson-Schwafel, 1988.
Microbial organisms	The distribution of bacteria and fungi and response to oil.	Numbers of hydrocarbon degrading bacteria are high in Spring.	Robertson et al., 1973; Feder et al., 1976.
Meiofauna (Mudflats)	Seasonal trends and response to crude oil.	Seasonal trends in meiofaunal abundance described. Harpacticoid copepod abundance unaffected by experimental exposure to oil.	Jewett and Feder, 1977; Feder and Paul, 1980; Feder et al., 1990.
The bivalve *Macoma balthica*	Hydrocarbon body burdens and population dynamics.	Variability in the distribution of the clam documented with possible effects related to road construction in 1976. Effects of continuous exposure to low hydrocarbon concentrations assessed. Bivalve responded negatively to oil exposure.	Shaw et al., 1976; Myren and Pella, 1977; Stekoll et al., 1980; Shaw et al., 1986; Naidu et al., 1993.
The mussel *Mytilus trossulus*	Shell growth, survival, reproduction, and distribution	Aspects of the ecology of *Mytilus trossulus* described. Effects on reproductive biology from hydrocarbons not observed.	Blanchard and Feder, 1997, 2000a, 2000b.

Table 1.1. Continued.

Organism/habitat	Study Focus	Contributions	Citations
	Hydrocarbon burdens.	Increased concentrations of hydrocarbons noted for mussels at both the Valdez Boat harbor and the marine terminal, relative to more distant locations. Recently, a pathway for exposure of intertidal mussels has been found through surface microlayer transport.	Shaw et al., 1986; Payne et al., 2001.
The barnacle *Semibalanus balanoides* (Rocky shores)	Shell Growth, survival, reproduction, condition index, and distribution.	Aspects of the ecology of the barnacle *Semibalanus balanoides* described. Reduced densities of juvenile barnacles described following a minor oil spill in the terminal area.	Rucker, 1983.
Limpets (Rocky shores and low profile beaches)	Shell Growth, survival, reproduction, condition index, and distribution.	Limpet distribution, reproduction, and growth trends described.	Feder and Keiser, 1980; Feder and Bryson-Schwafel, 1988; Blanchard and Feder, 2000 a.
Benthic infaunal communities (Deep subtidal)	Community composition, response to disturbance, and long-term changes.	Community composition, response to disturbance, and long-term changes documented. Response to a localized accumulation of degraded hydrocarbons within the terminal area described. The response noted as changes in total abundance and diversity for Station D25 and increases in opportunistic taxa (e.g.,	Feder et al., 1973; Feder and Matheke, 1980; Feder and Jewett, 1988; Blanchard and Feder, 2003; Blanchard et al., 2002, 2003.

Table 1.1. Continued.

Organism/habitat	Study Focus
Benthic infaunal communities (Continued.)	
Plankton	Seasonal dynamics, production.
Sea Otters (Open water)	Habitat use and interactions with human.
Pink Salmon (Open water and stream mouths)	Juvenile and adult salmon.

Contributions	Citations
Capitella capitata and *Axinopsida serricata*.). Following investigations demonstrated decreased densities of two tube-dwelling polychaetes (*Melinna cristata* and *Galathowenia oculata*) and increased densities of a number of species (including *Axinopsida serricata* and *Prionospio steenstrupi*) associated with changes in sediment hydrocarbon concentrations. Response to organic enrichment and dredging documented.	
Seasonal cycles of phytoplankton and primary production documented. The composition and seasonal fluctuations of zooplankton described.	Goering et al., 1973; Alexander and Chapman, 1980; Cooney and Coyle, 1988; Jewett and Blanchard, 1997.
The ecology of sea otters in Port Valdez described.	Hogan and Irons, 1988; Anthony, 1995.
Seasonal cycles of phytoplankton and primary production documented. The composition and seas	Merrel, 1988; Jewett and Blanchard, 1997.

Table 1.2. Summary of biological resources and anthropogenic stressors by habitat types in Port Valdez, Alaska. The habitat types and descriptions are from Wiegers et al., (1997, 1998)

Habitat	Description	Biological Resources	Anthropogenic Risks
Saltmarsh	Shoreline and mudflat characterized by marsh grasses and sedges.	Shorebirds, waterfowl, benthic organisms, and intertidal flora.	Oil spills, contaminated runoff from Valdez, erosion, and shoreline activities.
Mudflat	Tidal flats characterized by silt and clay sediments.	Shorebirds, waterfowl, fishes, sediment-dwelling (meiofauna and infauna) organisms.	Oil spills, runoff, construction, and shoreline activities.
Spit and low-profile beach	Flat shoreline or spits that consist of cobble or coarse sediments.	Birds, seals and sea otters, intertidal flora and fauna.	Oil spills, runoff, and construction.
Rocky Shoreline	Shoreline comprised of large rocks, boulders, or cliffs.	Birds, seals and sea otters, and intertidal flora and fauna.	Oil spills and shoreline activities.
Shallow Subtidal	Water column to benthic region less than 50 m deep.	Sea otters, diving sea birds, benthic flora and fauna, fishes.	Oil spills, runoff, fish waste, vessel traffic and construction.
Deep Benthic	Underwater area greater than 50 m deep, the sediment bottom.	Epibenthic and infaunal invertebrates, fishes.	Treated discharge, fish waste, vessel traffic and construction.
Open Water	Water column/pelagic zone.	Phytoplankton, zooplankton, salmon, marine mammals, sea birds.	Treated discharge, oil spills, runoff, vessel traffic and hatchery fish.
Stream Mouth	Intertidal entrances to streams and rivers.	Salmon spawning beds, other fishes, flora and fauna.	Oil spills, runoff, vessel traffic and hatchery salmon.

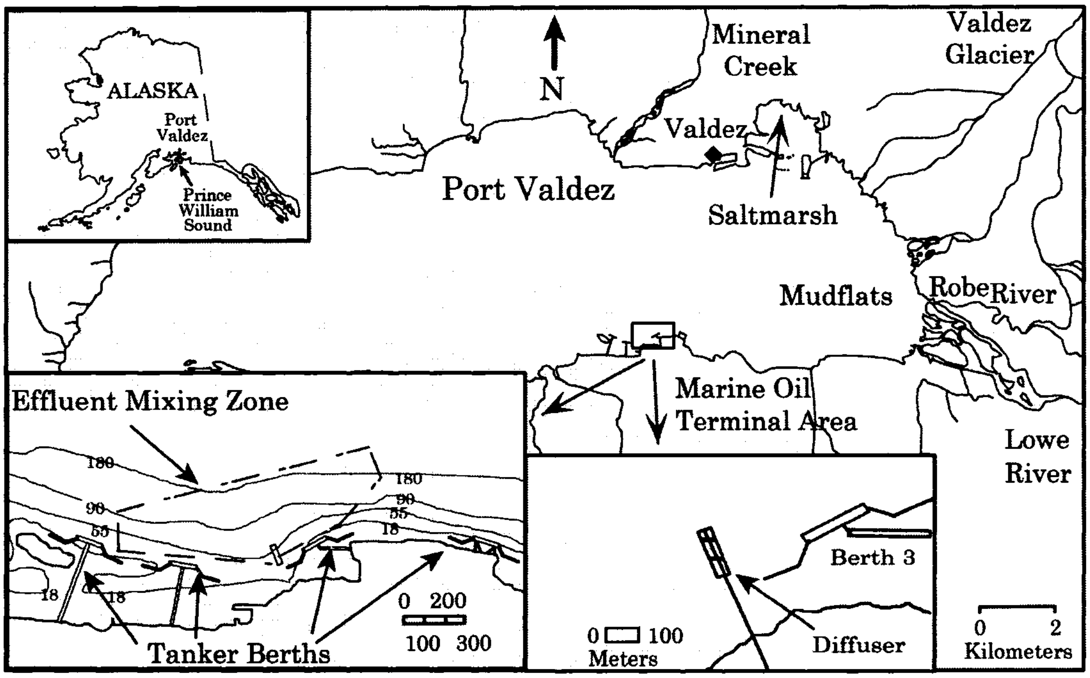

Figure 1.1. Map of Port Valdez, Alaska. Insets show the location of Port Valdez in Alaska, the mixing zone, and the location of the diffuser pipe. Depth contours in the mixing zone inset are in meters.

Chapter 2 Detecting Change in Spatial Variability of Intertidal Communities Through Variograms[1]

Abstract

Spatial variability of intertidal organisms was assessed through variograms as a potential tool for detecting anthropogenic stress. Geostatistical methods were applied to test whether variograms for three intertidal organisms are proportional to a common variogram among years differing only by the sill, as might be expected in the absence of environmental stress, or if they deviate in all parameters. Analyses suggest that the spatial variability of barnacles, mussels, and rockweed appears to be similar between years and proportional to a common variogram for each group. This is supported for mussels and rockweed by goodness-of-fit tests using likelihood ratio statistics. Simulated disturbance resulted in substantial changes in the spatial variation of the three organisms requiring separate variograms for affected years. Goodness-of-fit tests demonstrated moderate power for detecting a change in variograms with moderate to large simulated disturbance for rockweed and mussels but failed to detect a change for barnacles. The variogram nugget is associated with patch size, the range poorly reflects dispersal distance, and the sill reflects landscape variability. Use of variograms as a means for assessing variation of intertidal organisms to document disturbance appears to be a promising tool for marine environmental investigations.

Key words: Disturbance. Intertidal. Variogram. Barnacles. *Fucus gardneri*. *Mytilus trossulus*.

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1. Introduction

Effective long-term ecological monitoring requires an understanding of how communities vary in response to perturbations. To meet this need, methods for quantifying and characterizing natural ecosystem perturbations and detecting deviations from natural state are necessary (Ricotta and Avena 1999). Such studies must include assessment of small and large scale disturbances of different magnitudes and sources. Efforts at measuring change in marine intertidal communities usually focus on differences over space and/or time in univariate cover or abundance measures (e.g. Gilfillan et al. 1995, Highsmith et al. 1996). Variability of univariate measures increases with disturbance making it difficult to identify anything but large changes (Hawkins and Hartnoll 1983, Warwick and Clarke 1993). The variability of flora and fauna, therefore, is an important characteristic to understand.

Observations from field studies are commonly spatially and temporally correlated so methods for analyzing correlated data are necessary. Multiple-year sampling efforts are often made at similar times of the year at non-randomly selected locations to reduce the effect of spatial, seasonal, and annual variability (Green 1993). This regular sampling at fixed locations and times ensures spatially and temporally correlated data. Environmental factors resulting in spatial correlations include closeness to other organisms (both the same and other species), environmental gradients, and ecological processes influencing dispersal, growth, and success of flora and fauna. When environmental influences vary within bounds that do not disrupt normal ecological processes, gradients and ecological factors should predominantly control spatial correlation. As conditions change, whether from regional-scale sources of variability, localized extreme natural variability processes, or disturbance, the strengths of normally important ecological factors are reduced (Ricotta and Avena 1999). Stress large enough to disrupt ecological processes and biological communities (disturbance) should be apparent as changes in the structure of spatial correlation and variability. Spatial covariance can be measured through a statistical model of variability as a function of distance using the variogram, $\gamma(h)$ (Cressie 1993). Seasonal drought on Mediterranean terrestrial vegetation resulted in ecologically significant changes in the structure of spatial variation and variogram parameters (Ricotta and Avena 1999). Drought resulted in greater heterogeneity in the distributions of plants, and subsequently, clear changes in variogram parameters.

Ecologically significant stressors acting on intertidal communities should be detectable as a shift in variogram parameters. If environmental conditions are relatively similar between years and without large disturbances, the flora and fauna are likely to be distributed similarly with numbers differing because of variability in ecological factors. When data come from established populations with little stress, data for each time period should be proportional in total variance to a basic variogram, $\gamma(h)_i = p(t_i)\gamma(h)$ where $\gamma(h)$ is the common variogram and $p(t_i)$ is the proportionality factor reflecting temporal differences (Case 1, Stein et al. 1998). As stress is imposed on the system (e.g., anthropogenic disturbance, extreme weather, or long-term climatic change), the stressing environmental factor causes greater differences among adjacent components of the system leading to increased heterogeneity of the landscape elements (Ricotta and Avena 1999). It can be expected then, that the structure of spatial variability, as shown by the shape of the variogram, should be different as well. This should lead to the situation where different variograms are required, $\gamma(h)_i$, with separate estimates necessary for all model parameters, (Case 2, Stein et al. 1998). This was observed by Ricotta and Avena (1999) leading them to conclude that the approach of assessing changes in variogram parameters is useful for identifying ecologically important differences in biological populations.

The approach of comparing variograms between time periods may be a useful strategy for long-term monitoring studies of marine communities if there is an identifiable, common variogram reflecting the natural state (Ricotta and Avena 1999). Intertidal organisms are known for their patchy distributions and it may be that a stable variogram does not exist for any intertidal organism when comparing among time periods. In the latter case, the distance to which spatial correlations extend would vary between time periods and be unpredictable (Stein et al. 1998). If a common, stable variogram can be shown to exist, then demonstrating a shift in spatial variability may be useful for documenting stress. This approach needs to be tested, a goal to which this paper contributes. This study seeks to demonstrate empirically whether Case 1 (similar variograms) or Case 2 (different variograms) of Stein et al. (1998) is applicable for the "natural state" in a rocky intertidal community (collections made in the absence of known anthropogenic disturbance). Comparisons between variograms from data representing the natural state can then made against variograms from data with simulated disturbance to determine if a change from Case 1 to Case 2 is a possible means of detecting disturbance.

2. Methods

2.1 Sampling methods.

An investigation was initiated in 1988 to assess intertidal communities within Port Valdez for disturbance from oil terminal operations and to provide a reference point in the event of a major spill. Port Valdez (61°N, 146°30'W) is a glacial outwash fjord located in northeastern Prince William Sound, Alaska and is the site of the marine oil terminal for the

Trans-Alaska pipeline (Colonell 1980, Shaw and Hameedi 1988, Blanchard and Feder 1997, 2000a, b) (Fig. 2.1). The intertidal shoreline consists of rocky shores and low-profile cobble shores near stream mouths in the west with mudflats in the eastern end reflecting the high sediment load from glacial streams (Feder and Keiser 1980). The sampling frame is limited to safely accessible shores ranging from cobble shores to rock pinnacles. Two locations, Berth 4 and Saw Island, are at the marine terminal. One transect at each site was established at the highest occurrence of the black lichen *Verrucaria maura* with sampling points marked at 40 cm vertical intervals down to the low tide mark. An area of approximately 1.2 m^2 (40 cm x 3 m centered on marked sampling locations) was examined. A rapid-assessment, visual estimation method (Moyse and Nelson-Smith 1963, Nelson-Smith 1967, Feder and Keiser 1980, Cowell and Monk 1981) was used to classify the abundance and percent cover of intertidal flora and fauna at each sampling point along the transect. Sampling occurred in mid-summer from 1988 to 1992. The percent cover of a brown alga, the rockweed *Fucus gardneri* and the mussel *Mytilus trossulus* and abundance of adult barnacles (*Balanus glandula* and *Semibalanus balanoides*) (Table 2.1), three dominant taxon groups, are used for the spatial analyses in this paper. Difficulties arose with the classification scale used in 1988 for barnacles so data from that year for barnacles are not included.

2.2 Statistical methods

Geostatistical methods were used to estimate variograms for abundance scores of the three selected organisms. These analyses were performed with the statistical program R (www.r-project.org) and the library GeoR (Ribeiro and Diggle 2001). Parameters defining the shape of the spherical variogram model are the nugget, which is microscale variation and measurement error; the range, indicating the distance at which data are no longer correlated; and the upper limit of the model called the sill, which represents total variance (Cressie 1993). Empirical (semi)variograms were determined from the data as:

$$\hat{\gamma}(\mathbf{h}) = 1/(2\,|\,N(\mathbf{h})\,|\,)\Sigma[Z(\mathbf{x}) - Z(\mathbf{x}+\mathbf{h})]^2 \qquad 2.1$$

where $\hat{\gamma}(h)$ is the observed semivariance, **x** and **x+h** are data vectors separated by the distance vector **h**, Z(**x**) is a random process, and N(**h**) is the number of distance pairs (Cressie 1993, Ricotta and Avena 1999). The spherical semivariogram model fitted to the empirical (semi)variograms is:

$$\gamma(\mathbf{h};\,\theta) \quad \begin{cases} 0, & \text{for } \mathbf{h} = 0 \\ c_o + c_s\{3\,\|\mathbf{h}\|\,/2a_s - 1/2(\|\mathbf{h}\|\,/a_s)3\}, & \text{for } 0 < \|\mathbf{h}\| \le a \\ c_o + c_s & \text{for } \|\mathbf{h}\| \ge a, \end{cases} \qquad 2.2$$

where $\gamma(\mathbf{h};\,\theta)$ is the semivariance, **h** = vector of distances, $\theta = (c_o, c_s, a_s)'$, $c_o \ge 0$ is the nugget, $c_o + c_s \ge 0$ is the sill, and $a_s \ge 0$ is the range (Cressie 1993) (Figure 2). Distance west from the furthest east sampling point (km) and tidal height (m) were used as the spatial coordinates. Upper and lower tidal heights for which a particular taxon was absent or only rarely abundant (outside its range) were excluded to minimize problems with zero values. Preliminary assessments suggested that variograms did not differ by direction so isotropy was assumed. Models were limited to two-dimensional variograms due to the small number of data points. Variogram models were fit using a maximum likelihood approach with the geoR function likfit.

Variogram estimation was employed to test for the two special cases of space-time modeling suggested by Stein et al. (1998). These are the situations (Case 1) where variograms for each time period are proportional in total variance to a common variogram ($\gamma(h)_i = p(t_i)\gamma(h)$), and (Case 2) data require different variograms with unique range and sill estimates ($\gamma(h)_i$).

Variograms were estimated for the three taxa selected for each of the five years sampled.

Likelihood ratio statistics were evaluated as a goodness-of-fit test to determine if variograms fit Case 1 or Case 2. The null hypothesis for this goodness-of-fit test can be written as H_0: $\gamma(h)_i = p(t_i)\gamma(h)$ (the nuggets and ranges of all models are the same and only the sill varies by $p(t_i)$) and the alternative as H_a: $\gamma(h)_i \neq p(t_i)\gamma(h)$ (all parameters vary). This was tested by comparing the loglikelihood of the full model with each variogram having separate parameter estimates, $ln\mathrm{L}(\gamma(h)_i)$, to the loglikelihood of the models with the range and nugget fixed and only the partial sill varying, $ln\mathrm{L}(p(t_i)\gamma(h))$. Variograms were first fit to data separately for the five years using the geoR function likfit and averages calculated for the fitted model parameters for each taxon category. Then, the loglikelihoods of individual variograms with no fixed parameters (parameters equal to those estimated for each year) were calculated using the geoR function loglik.GRF. The loglikelihood of the full model, $ln\mathrm{L}(\gamma(h)_i)$, was determined by adding these individual loglikelihoods together. The loglikelihood of the reduced model, $ln\mathrm{L}(p(t_i)\gamma(h))$, was calculated as the sum of loglikelihoods for variograms with the (partial) sill parameter varying but the nugget and range fixed at the calculated average values. The loglikelihood ratio test statistics is calculated as

$$\mathrm{LRT} = -2[ln\mathrm{L}(p(t_i)\gamma(h)) - ln\mathrm{L}(\gamma(h)_i)], \qquad 2.3$$

which is distributed as χ^2 with degrees of freedom equal to the number of parameters in the full model – the number of parameters in the reduced model. A test statistic that is small indicates that the loglikelihoods of the full and reduced models are similar, the null hypothesis cannot be rejected, and thus, evidence suggests that the best fit is the model where total variance fluctuates around a common variogram. A test statistic that is large and significantly different at the 5% level of significance leads to the rejection of the null hypothesis providing strong evidence that individual estimates for all variogram parameters are necessary to model the spatial variation.

Simulated data were generated to model the effect on variograms of one agent of change, an oil spill, at the marine terminal. To simulate effects of oiling and cleanup following a moderate oil spill at the terminal (a simulated point-source pollution event), abundance and percent cover scores for two sites near the tanker berths in Port Valdez, Berth 4 and Saw Island, are reduced to 50% observed values for 1989 and 1990, and to 80% (a 20% reduction) in 1991 to simulate recovery. Values for 1988 and 1992 are not modified. Values for sites adjacent to the terminal but at a greater distance from the berths, Sawmill Spit and Terminal Rock, are adjusted to 80% in 1989 and 1990 and to 90% (a 10% reduction) in 1991. A severe disturbance was simulated by reducing scores at Berth 4 and Saw Island to 25% (a 75% reduction) in 1989 and 1990, 50% in 1991, and 90% in 1992. Scores for Sawmill Spit and Terminal Rock were reduced to 50% in 1989 and 1990 and to 80% in 1991. The values for these adjustments were selected to model the size of disturbance effects from oil contamination and beach cleanup described for intertidal communities following the *Exxon Valdez* oil spill (Gilfillan et al. 1995, Highsmith et al. 1996, Stekoll et al. 1996, Van Tamelen and Stekoll 1996, Peterson et al. 2001). If variograms are responsive to ecologically significant change, values of variogram parameters should be altered substantially from those of the natural state without disturbance. Thus, disturbance should be recognized as a switch from Case 1 to Case 2. Comparisons of variograms for the simulated disturbance provide information on the power of this approach for detecting environmental change from a point-source stressor.

3. Results

Fitted spherical variogram models for the observed data appear to reflect annual variation of the sill around a common variogram for each organism. The estimated average nugget for adult barnacles, representing microscale variability, is 0.46 from 1989-1992 (Table 2.2). The average value for the range of the variogram, the point where data are no longer correlated, is 2.01 distance units (based on km for distance west and m for tidal height). The average sill, the horizontal asymptote or total variance of the variograms, is 3.94. The likelihood ratio statistic $-2 \ln L = 3.06$ is not significant at the 5% level of significance ($p = 0.93069$), the null hypothesis cannot be rejected, and evidence suggests that total variance fluctuates around a common variogram (Case 1) (Table 2.3). A common variogram is supported graphically as it appears that the plotted variograms generally have the same form and differ largely by the sill (Fig. 2.3a). For rockweed, the average nugget is 0.63, the average range is 2.21, and the sill is 4.18 (Table 2.2). The likelihood ratio statistic for rockweed is not significant at the 5% level of significance with $-2 \ln L = 12.25$ ($p = 0.26835$) and the null cannot be rejected indicating no deviation from the hypothesis of a fluctuation around a common variogram (Table 2.3). Graphically, the variograms for rockweed appear similar but with a different sill for each year (Fig. 2.4a). For mussels, the average nugget is 0.29, the range is 2.73, and the sill is 3.39 (Table 2.2). The likelihood ratio statistic for mussels is not significant with $-2 \ln L = 10.15$ ($p = 0.42702$) and graphically, the variograms for mussels appear to be similar in form (Table 2.3 and Fig. 2.5a).

Variogram parameters demonstrate increased range and decreased sills with increasing disturbance. For barnacles with moderate disturbance 1989-1992, the average nugget of 0.46 is the same as the undisturbed data set while the range is increased to 2.33 and the sill is decreased to 3.56 (Table 2.2). With severe effects, the nugget is slightly decreased to 0.43, the range increased to 2.52, and the sill decreased to 3.18. The average nugget of 0.63 for rockweed with moderate disturbance is the same as without disturbance but the range is increased to 2.56 and the sill is decreased to 3.97. With severe disturbance, the nugget of 0.60 is about the same as with moderate disturbance, the range is 2.61 and the sill is decreased to 3.66. As with the above variograms, the average nuggets for mussels of 0.31 and 0.29 are relatively unchanged while the range is increased to 2.75 and 2.92 for moderate and severe disturbance, respectively. The sill decreased to 2.96 and 2.73 with moderate and severe disturbance, respectively.

Plots of fitted variograms confirm the trends observed in average parameter values. The increasingly greater ranges in the variograms for barnacles with increasing disturbance are reflected in the movement of the range (shown in the plots as the point where the curves level off) to the right of the vertical line (Fig. 2.3b and c). The decreasing sill is shown in the variograms by the drop in the heights of affected curves from the undisturbed state to moderate and severe disturbance. Changes in variograms for rockweed and mussels are similar to those of barnacles with the height of the curves (the sill) lower and the curvature extending further (the range) to the right with increasing disturbance (Figs. 2.4b and c and 2.5b and c). The exception is the variogram for 1991 for rockweed in which the sill is higher for both moderate and severe disturbance.

Likelihood ratio tests provide evidence that as disturbance increases, a common variogram is not applicable for rockweed and mussels. The likelihood ratio statistics for moderate and severe effects on barnacle data of $-2lnL = 2.17$ ($p = 0.97521$) and 2.18 ($p = 0.97502$), respectively, are not significant at the 5% level of significance providing no evidence of a departure from the common variogram model (Table 2.3). For rockweed, the likelihood ratio statistics for both moderate and severe disturbance provide moderate evidence (the 10% significance level) that a common model is not appropriate with $-2ln\mathrm{L} = 16.96$ ($p = 0.07523$) and 17.46 ($p = 0.06472$) for moderate and severe disturbance, respectively. Likelihood ratio tests for mussels provide no evidence of a departure from a common variogram model with moderate disturbance but strong evidence for separate variograms for severe disturbance with $-2ln\mathrm{L} = 13.88$ ($p = 0.17838$) and 19.62 ($p = 0.03304$) for moderate and severe disturbance, respectively. Thus, the likelihood ratio test appears to have reasonable power to reliably detect moderate and severe disturbance for rockweed, a severe disturbance for mussels, but inadequate power to detect either level of disturbance for barnacles although differences in variograms are shown graphically for all levels of disturbance for all three organisms.

4. Discussion

Analysis of spatial variability is useful for demonstrating stress-related changes in biological communities (Stein et al. 1998, Ricotta and Avena 1999). This approach requires that a common variogram exists for a particular organism reflecting a natural or unstressed state. If such a variogram does not exist then no comparisons can be made to conclude that spatial variability has significantly changed as every variogram for any time period will be different. It is also necessary to understand whether ecologically significant changes in flora and fauna will actually be evident as a change in spatial variability. The hypothesis of a common variogram among years with the same nugget and range value cannot be rejected for the selected organisms in the present study. The similarity of the variograms among years for barnacles, mussels, and rockweed is suggested by the similar forms of variogram plots and lack of significance in the goodness-of-fit tests for the observed data. Thus, there is a common variogram model, Case 1 of Stein et al. (1998). Now it needs to be determined if an ecologically important change in the biota can detected as a change in variograms, as suggested for other habitats (Ricotta and Avena 1999).

Variograms for barnacles, rockweed, and mussels show a change in the range and sill with increased disturbance. If it is accepted that the observed data represent the case where variograms are proportional to a basic model (Case 1, Stein et al. 1998), then the evidence presented here indicates that disturbance to biological communities is detectable as deviations for the range and sill for the spherical model (Case 2, Stein et al. 1998, Ricotta and Avena 1999). Disturbance of the type modeled (where values are depressed at a few sites creating a dip in percent cover and abundance) will increase heterogeneity between sites thus increasing the distance to which spatial correlations extend (Ricotta and Avena 1999). The sill, or total variance, will also decrease as a segment of the population has a much lower variance. The changes in range and total variance with disturbance are supported by the changes in the plots of variogram models for all three organisms and the significant likelihood ratio tests for mussels with severe disturbance and rockweed for moderate and severe disturbance. For all models, the nugget appears to be relatively unaffected by the simulated disturbance. This simulation of disturbance did not alter patch size as all data within the "disturbed" sites were equally depressed. Patch size will be altered if disturbance acts unequally within a site eliminating aggregations of organisms or organisms on the boundaries of aggregations. Disturbances of this type might include logs hitting a beach, grazing and predation on the edges of patches, or deposition of vegetation mats.

Disturbances acting on patches and individuals would differentially reduce the abundance or cover of organisms creating open patches of space and this should be reflected in nugget values.

The likelihood ratio test applied here has moderate power for detecting deviations from Case 1. Deviations from the null model (only the sill varies) are supported by the goodness-of-fit tests for moderate and severe disturbance for rockweed and for severe disturbance for mussels. In spite of the trends of decreased variogram parameters for barnacles with disturbance, goodness-of-fit tests for neither level of disturbance were significant. It may be that additional covariates are necessary to remove remaining trends in the spatial models for barnacles to have adequate power, although this did not appear to be necessary from diagnostic analyses. It may also be that the change in score estimation for barnacles applied to the 1989-1992 sampling did not fully correct the problem observed in 1988 and the apparent low power in the loglikelihood ratio tests may be a result. It appears that, in general, stress on intertidal organisms can be detected by changes in variogram parameters when considering the changes graphically and with moderate power, using likelihood ratio tests. Overall, the approach of comparing variograms is useful for documenting effects of environmental change on marine biota, as has been shown elsewhere (Ricotta and Avena 1999).

The variogram nugget (the measure of small-scale variation) will be related to patch-size and density of the organisms studied. Focusing on the two extremes, the comparatively high average nugget for rockweed of 0.63 reflects greater between-quadrat variability (variability at a scale smaller than the 1.2 m^2 sampling quadrat is lost) as compared to mussels with an average nugget of 0.29. Although rockweed cover can be dense, open space is frequent with plants occurring in clumps. Space unoccupied by rockweed may persist in this zone due to the inability of this algal species to actively select settlement sites or quickly occupy disturbed space and from biological and physical interactions (van Tamelen and Stekoll 1995, Menge and Branch 2001). Thus, the nugget for rockweed reflects variability slightly larger than the scale of the patch. In contrast to rockweed, mussels can be found in dense aggregations with regular cover (nearly 100% cover in dense beds) and the patch size is much bigger than the 1.2 m^2 quadrat used in sampling (Feder and Keiser 1980, Seed and Suchanek 1992, Lutz and Kennish 1992, Blanchard and Feder 1997). Movement of mussels of all ages may be possible contributing to the regular cover and density of mussel beds (Schneider et al. 2005). As a result of the dense cover, the nugget measures within-patch variability for mussels (which would be low for a regular distribution) rather than between-patch heterogeneity. Barnacles often occur in the study area as uniform bands of irregularly spaced individuals or bands of small aggregations but patch size is not clear. The nugget for barnacles suggests a patch size intermediate between mussels and rockweed.

The range of spatial models is associated with dispersal distance. Investigations of insect behavior have shown that the distance to which spatial correlations extend increases with dispersal distance (Williams and Liebhold 2000). This is poorly shown for the organisms in this study as the range for mussels, which has the greatest dispersal capability, is the largest but rockweed does not have the lowest range although dispersal distance is short (range = 2.73 for mussels vs. 2.21 for rockweed and 2.01 for barnacles). Mussels release gametes into the water column for wide dispersal and juvenile mussels actively select settling sites (Feder and Keiser 1980, Seed and Suchanek 1992, Lutz and Kennish 1992, Blanchard and Feder 1997, Schneider et al. 2005). The larger range for rockweed, as compared to barnacles, is unexpected based solely on dispersal distance. Rockweed has very short dispersal distances with most fertilized eggs falling within 20 cm of the adult plant and germlings having little opportunity to select preferred habitats during settlement (van Tamelen and Stekoll 1995). In contrast, barnacles are able to widely disperse and they actively select settling sites near other barnacles (Walker et al. 1987). The reversal of barnacles and rockweed in the ordering of the range may be due to difficulties with the visual estimation for barnacles. It may also be a result of the distance separating sampling locations as the shores are at a horizontal scale appropriate for assessing dispersal of barnacles and mussels but not for rockweed since this alga has a very short dispersal distance.

The variogram sill reflects heterogeneity among shores. The sill for spherical models decreases when values for a few individual landscape units are decreased, as shown above with simulated disturbance, so the lower sills for barnacles and mussels, 3.94 and 3.39, respectively, as compared to rockweed, 4.18, may reflect increasing heterogeneity between sites sampled. Variability in environmental factors, disturbance events, and predation will contribute to patchiness by creating open areas with random, variable sizes. This is particularly true for mussel beds which are known to experience disturbances to whole beds (Carroll and Highsmith 1996, Seed and Suchanek 1992). This heterogeneity will be of the form in which one or a few individual shores will have substantially lower abundance scores contributing less to total variability. This suggests that within the study area, rockweed patches are less variable between shores while mussel beds are the most variable, a prediction that can be tested experimentally. The cover of rockweed and abundance of mussels have been shown to be lower on cobble shores (Chapter 3).

The objective of this study was to determine if a common model describing spatial variability could be identified for intertidal organisms and if so, could disturbance be shown as a deviation from that model. The results of this work demonstrate that at least for three selected organisms in a glacial fjord, a common, stable variogram does exist among years with annual variability largely reflected in variation of the sill. Comparing between organisms, the variogram parameters reflect ecological aspects of the organisms sampled. The nugget reflects quadrat-scale variability for rockweed as compared to more within-patch variability for barnacles and mussels. The range reflects the larger dispersal distance for mussels but poorly reflects the dispersal distance for barnacles and rockweed, possibly the result of measurement error for barnacles or an inappropriate sampling scale for rockweed. The sill reflects landscape unit (between-beach) heterogeneity and suggests a more regular distribution of rockweed among shores, as compared to barnacles and mussels. With simulated disturbance, the spherical variogram models demonstrated an increase in the range and a decrease in the sill. There was a shift from a common variogram (varying by the sill) (Case 1) to separate variograms for each year (all parameters varying) with simulated disturbance (Case 2). This demonstrates that disturbance can be detected in variograms for dominant intertidal organisms. Power of the likelihood ratio test was adequate for demonstrating disturbance for rockweed and mussels but not for barnacles, indicating the need for additional covariates or possibly a difficulty with the estimation scale used for the latter taxon group. The geostatistical approach to assessing spatial variability applied here should be a useful tool for monitoring marine communities as it has been in terrestrial environments.

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Table 2.1. Abundance and percent cover scales for intertidal flora and fauna for Port Valdez, 1988-1992. The scores and associated category definitions are given for the three organisms studied.

Score	Rockweed % Cover	Barnacle Abundance 1989-1992	Mussel Abundance
0	Not Present	Not Present	Not Present
1	1-2 plants	$< 1\ m^{-2}$	$< 1\ m^{-2}$
2	Scattered plants, zone indistinct	$1\text{-}10\ m^{-2}$	Scattered individuals, no patches.
3	<5%, zone apparent	$10\text{-}50\ m^{-2}$	Small Patches, many scattered individuals.
4	5-30%	$50\text{-}100\ m^{-2}$	Large patches < 20% cover
5	30-60%	$1\text{-}10\ \ 0.1\ m^{-2}$	20-50% Cover
6	60-90%	$10\text{-}100\ \ 0.1\ m^{-2}$	50-80% cover
7	> 90%	$1\text{-}3\ cm^{-2}$	> 80% Cover

Table 2.2. Summary of variogram parameters for selected organisms from Port Valdez, 1988-1992. Parameters for fitted spherical models for barnacles, rockweed, and mussels are given for the raw data and two simulated levels of disturbance. Min. = minimum, Max. = Maximum, and SD = standard deviation.

Organism	Parameter	Effect	Average	Min	Max	SD
Barnacles	Nugget	Normal	0.46	0.36	0.62	0.11
		Moderate	0.46	0.36	0.54	0.08
		Severe	0.43	0.36	0.52	0.07
	Range	Normal	2.01	1.60	2.48	0.40
		Moderate	2.33	2.04	2.77	0.31
		Severe	2.52	2.27	2.92	0.30
	Sill	Normal	3.94	3.24	4.65	0.58
		Moderate	3.56	2.72	4.65	0.86
		Severe	3.18	2.41	4.56	1.01
Rockweed	Nugget	Normal	0.63	0.14	0.97	0.32
		Moderate	0.63	0.14	0.94	0.33
		Severe	0.60	0.14	0.86	0.32
	Range	Normal	2.21	2.09	2.47	0.15
		Moderate	2.56	2.20	3.45	0.51
		Severe	2.61	2.20	3.42	0.47
	Sill	Normal	4.18	3.56	5.05	0.63
		Moderate	3.97	2.64	5.62	1.21
		Severe	3.66	2.25	5.02	1.16
Mussels	Nugget	Moderate	0.29	0.06	0.63	0.22
		Normal	0.31	0.08	0.63	0.23
		Severe	0.29	0.05	0.63	0.22
	Range	Normal	2.73	2.35	3.17	0.30
		Moderate	2.75	2.35	3.17	0.31
		Severe	2.92	2.38	3.17	0.35
	Sill	Normal	3.39	2.93	3.73	0.35
		Moderate	2.96	2.05	3.73	0.76
		Severe	2.73	1.80	3.56	0.81

Table 2.3. Summary of goodness-of-fit tests for variograms of selected organisms from Port Valdez, 1988-1992. Likelihood ratio statistics and p-values are given for tests of the null hypothesis of a common variogam model with variation around the sill versus the alternative that individual variograms are required.

		Raw Data		Moderate Disturbance		Severe Disturbance	
Organism	df	-2lnL	P-value	-2lnL	P-value	-2lnL	P-value
Barnacle	8	3.06	0.93069	2.17	0.97521	2.18	0.97502
Rockweed	10	12.25	0.26835	16.96	0.07523	17.46	0.06472
Mussels	10	10.15	0.42702	13.88	0.17838	19.62	0.03304

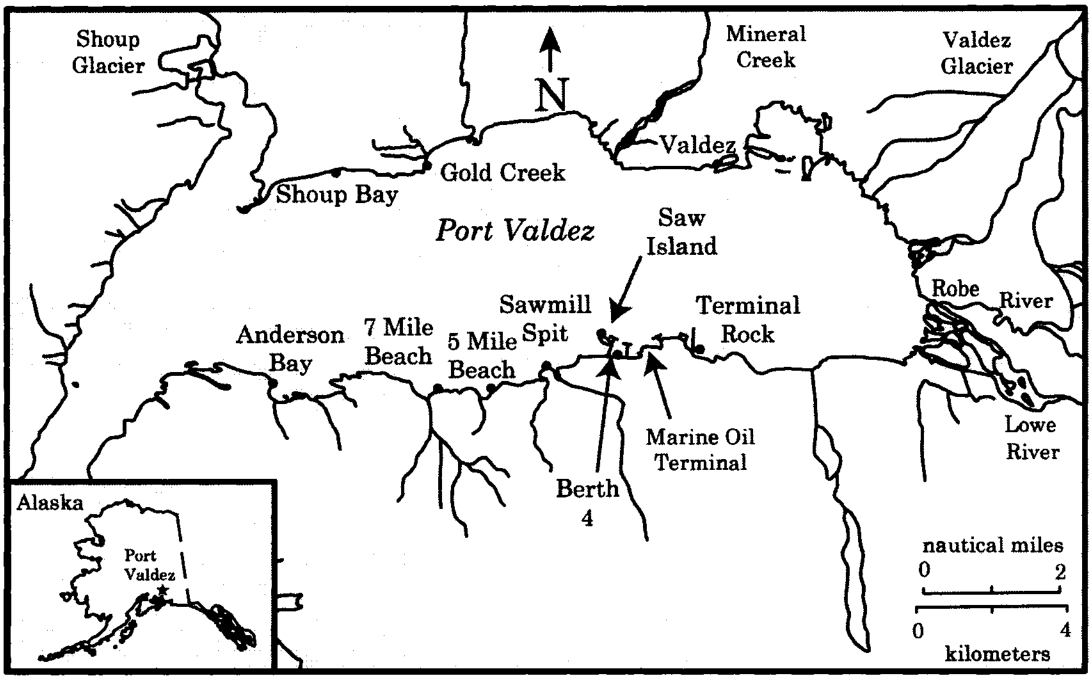

Figure 2.1. Map of intertidal sampling locations in Port Valdez, 1988-1992.

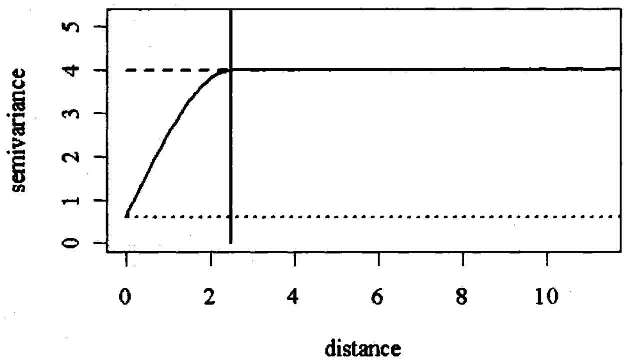

Figure 2.2. Example of a spherical variogram. The curve is the variogram model, the vertical line represents the range, the dotted line the nugget, and the dashed line the sill.

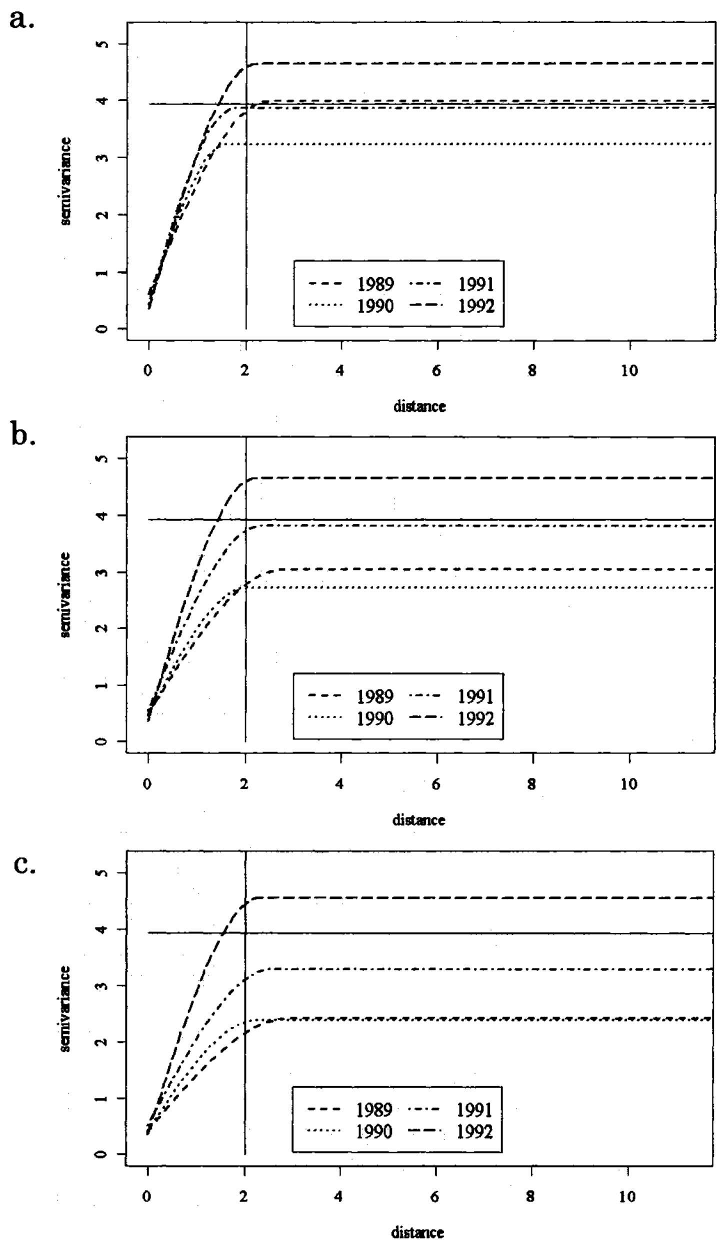

Figure 2.3. Spherical variogram models for adult barnacles for Port Valdez, 1989-1992. The vertical lines mark the average range of the spherical variogram model and the horizontal line marks the average sill from the model without simulated disturbance. Variograms include a) raw abundance scores, b) simulated moderate disturbance (1989-1991, 1992 unchanged), and c) simulated severe disturbance (1989-1992).

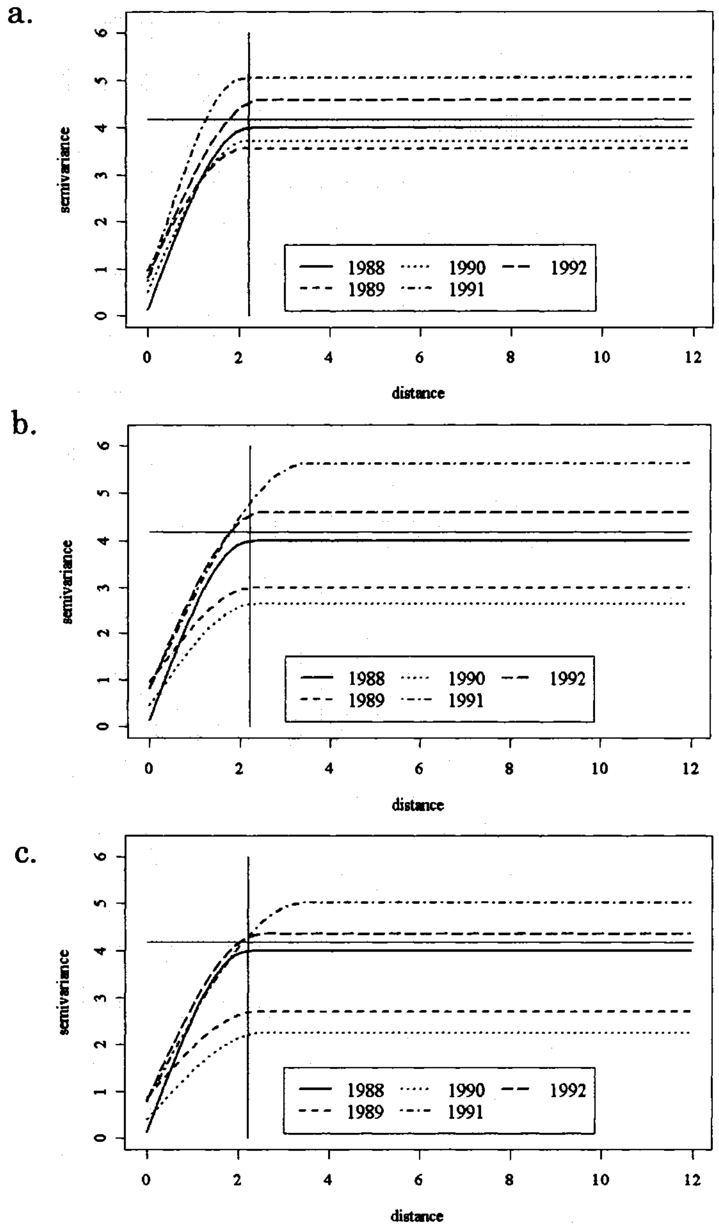

Figure 2.4. Spherical variogram models for rockweed for Port Valdez, 1988-1992. The vertical lines mark the average range of the spherical variogram model and the horizontal line marks the average sill from the model without simulated disturbance. Variograms include a) raw abundance scores, b) simulated moderate disturbance (1989-1991, 1992 unchanged), and c) simulated severe disturbance (1989-1992, 1988 unchanged).

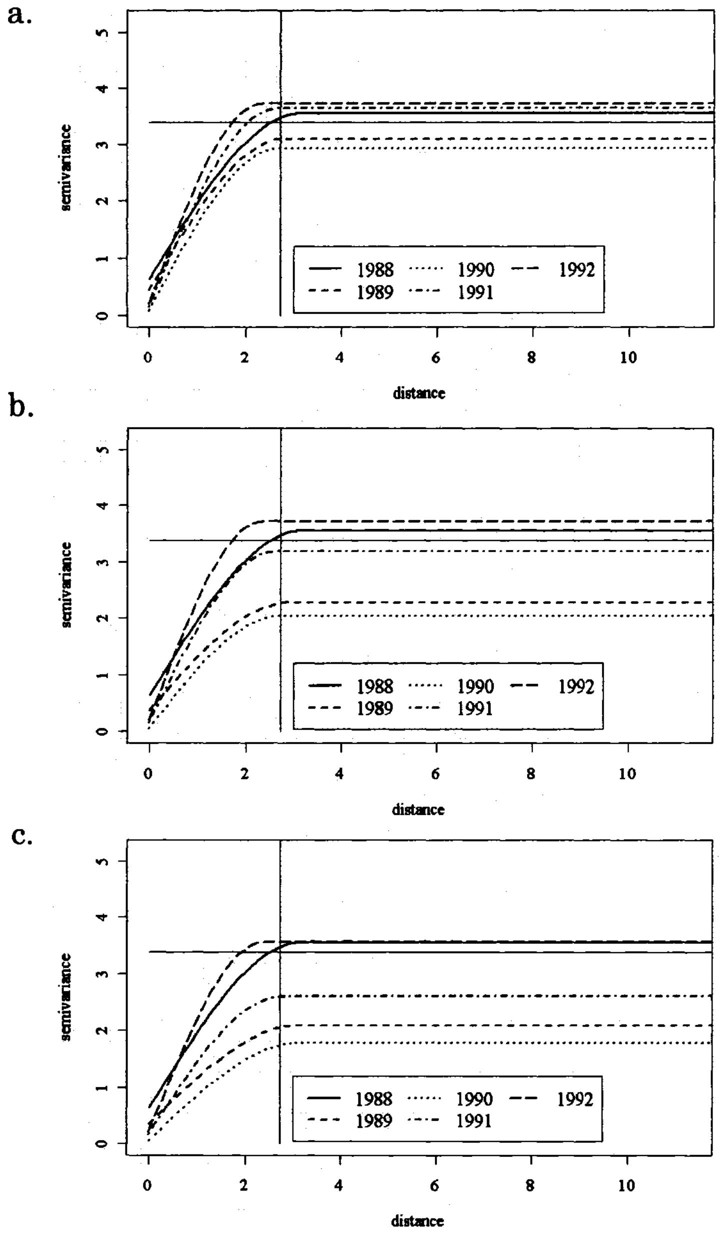

Figure 2.5. Spherical variogram models for mussels for Port Valdez, 1988-1992. The vertical lines mark the average range of the spherical variogram model and the horizontal line marks the average sill from the model without simulated disturbance. Variograms include a) raw abundance scores, b) simulated moderate disturbance (1989-1991, 1992 unchanged), and c) simulated severe disturbance (1989-1992, 1988 unchanged).

Chapter 3 Associations of an Intertidal Assemblage with a Salinity Gradient and Habitat Structure in a Glacial Fjord[1]

Abstract

The distribution of intertidal organisms was investigated in an Alaskan glacial fjord. Seasonal environmental conditions affecting intertidal rocky shores in fjords include exposure to estuarine conditions in summer due to melting of glaciers and fully marine conditions in winter. Salinity and habitat structure were most closely associated with an ordination of abundance and percent cover. Low salinity appears to have a direct effect on the assemblage by excluding invertebrate predators and to a lesser extent, grazers at sites with very low salinity. Salinity effects on algal species are less clear as mechanisms may include direct effects through physical intolerance to low salinity and indirect effects through reduced grazing at sites with very low salinity, an altered competitive balance, and interactions of these mechanisms. Intertidal flora and fauna in the fjord are generally euryhaline species in contrast to an expected gradation of fauna from marine to freshwater organisms in estuaries. Habitat structure appears to have a large association with an ordination but relationships with common flora and fauna are not clear. Small, chronic disturbance at cobble beaches may contribute to the separation of sites by habitat structure in the ordination. Salinity appears to be the dominant factor associated with changes in the intertidal community of the fjord with habitat structure playing a secondary role.

Key words: Estuary. Marine Ecology. Port Valdez, Alaska. *Fucus gardneri*. *Mytilus trossulus*.

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1. Introduction

Tolerance to physical extremes is an important mechanism controlling distributions of intertidal organisms. In general, flora and fauna able to withstand desiccation and air temperature extremes can survive higher on the shore (Menge and Branch, 2001). Salinity tolerance is similarly an important mechanism directly affecting distributions of organisms vertically and horizontally, particularly within estuaries where freshwater exposure occurs (Ritter et al., 2005). Exclusion of intolerant marine organisms may indirectly influence the importance of other mechanisms structuring intertidal communities in brackish water. For those marine organisms able to tolerate low salinity, tradeoffs include slower growth rates, negative, synergistic interactions with other stressors, and changes in competitive balances (Depledge, 1987; Cheung and Lam, 1995; Fong et al., 1996; Martins et al., 1999).

The key characteristic of temperate to subarctic estuaries, including fjords, is a salinity gradient associated with freshwater inflow from rivers (Perkins, 1974; Nybakken, 2001). Freshwater influx at the head of an estuary results in low salinity values grading to marine conditions as distance increases from riverine inputs. Physical tolerance to salinity along the gradient is a dominant mechanism controlling biotic assemblages in soft-sediment estuaries resulting in the gradation of estuarine flora and fauna from freshwater to marine organisms as salinity increases (Perkins, 1974; Hardwick-Witman and Mathieson, 1983; Menge and Branch, 2001; Nybakken, 2001; Ritter et al., 2005). When low salinity occurs irregularly, it can be viewed as an environmental perturbation and intertidal assemblages in soft sediments may be maintained in a state of early to intermediate succession (Ritter et al., 2005). The length of exposure to low salinity and salinity variability are critical as short-term exposure to low values and higher salinity variability may result in much smaller effects (Kramer and Fong, 2000; Murphy et al., 2003). Biological interactions, including competitive balances and predator-prey relationships, may be altered by low salinity resulting in potentially large, indirect effects (Cimberg, 1982; Stickle et al., 1985; Feder and Christensen, 1966; Carroll and Highsmith, 1996; Fong et al., 1996; Feder et al., 2003). Investigations of northern fjords found low, surface-water salinity (< 15‰) to be an important factor in community differences within and outside fjords, partially a result of exclusion of invertebrate predators (Feder and Keiser, 1980; Cimberg, 1982; Feder and Bryson-Schwafel, 1988).

Soft sediment habitats are common in estuaries due to the high sediment loads in river runoff (Perkins, 1974; Nybakken, 2001). Sediment-laden runoff entering estuaries leads to higher sediment accumulations, reduced light penetration, and creation of soft sediment habitats. Excessive sediment accumulation within mussel beds is known to damage mussels through burial, suffocation, and weakening of byssus thread attachment (Seed and Suchanek, 1992). High suspended sediment loads and sedimentation rates are common wherever glacial sediments are introduced (Hoskin et al., 1978; Alexander and Chapman, 1980; Farrow et al., 1983; Naidu and Klein, 1988; Seed and Suchanek, 1992). In deep fjords, intertidal soft-sediments comprise a lower proportion of habitat compared to other estuarine environments. Habitats of fjords range from mudflats near the head to rocky shores (Perkins, 1974). Hard substrate habitats in fjords range include cobble beaches, cliffs, and moderately sloped rock (Feder and Bryson-Schwafel, 1988).

Fjords with active glaciers are characterized by strong, seasonal environmental gradients. During summer months, the influx of cold, sediment-laden, glacial melt-waters in fjords leads to classic estuarine circulation with low salinity water flowing out at the surface and saline waters in subsurface water layers flowing in (Hoskin et al., 1978, Colonell, 1980; Farrow et al., 1983). Substantial freshwater flow often begins in late May, peaks in August, and ends in October as freezing conditions occur. When the decrease in freshwater flow occurs, estuarine circulation stops and the water column becomes well-mixed resulting in marine salinity levels, a condition common among fjords along the Alaskan and British Columbian coastlines (Hoskin et al., 1978; Colonell, 1980; Farrow et al., 1983). As a result, intertidal assemblages on hard substrates in glacial fjords are seasonally exposed to freshwater, a combination of habitat and environmental conditions not common in estuaries or on rocky shores. Intertidal communities in temperate to boreal-arctic regions can be also exposed to severe freezing temperatures (Carroll and Highsmith, 1996; Blanchard and Feder, 2000a). Geographic temperature changes can limit distributions of organisms and alter lifestyles and behavior (Barry, 1995; Blanchard and Feder 1997 and 2000a). Within a local area, climatic extremes can cause mass mortality of intertidal organisms (Carroll and Highsmith, 1996). This study assesses associations between intertidal assemblages and environmental gradients and habitat characteristics for five years (1988-1992) in an Alaskan fjord. The goal of this study is to determine the mechanisms by which environmental factors in a glacial fjord may influence intertidal assemblages. The effects of interactions between environmental factors and biological mechanisms structuring intertidal communities are discussed.

2. Methods

Port Valdez (61°N, 146°30'W) is a glacial outwash fjord located in northeastern Prince William Sound, Alaska (Figure 3.1). It is approximately

22 km long and 5 km wide. The intertidal shoreline consists of steep, rocky shores and low-profile cobble beaches (near stream mouths) with extensive mudflats in the eastern end (Feder and Keiser, 1980). The tidal range is 5.3 m (Colonell, 1980). In the summer, runoff and glacial meltwaters are entrained in a surface plume up to 10 m deep resulting in strong salinity and suspended sediments gradients (Colonell, 1980). Dominant intertidal flora and fauna of the rocky shores in the fjord include the alga *Fucus gardneri*, ulvoid algae, the mussel *Mytilus trossulus*, six limpet species (*Acmea mitra, Lottia borealis, L. pelta, Tectura fenestrata, T. persona*, and *T. scutum*), littorine snails (*Littorina scutulata* and *L. sitkana*), and barnacles (*Semibalanus balanoides* and *Balanus glandula*) (Feder and Keiser, 1980; Cowell and Monk, 1981; Rucker, 1983; Feder and Bryson-Schwafel, 1988; Blanchard and Feder, 1997, 2000a, b). The highest tidal heights are occupied by the lichens *Verrucaria* spp. The mid to high tidal zone is dominated by *F. gardneri*, littorine snails, and barnacles. Dominant organisms in the low to mid-zone are ulvoid algae, mussels, and barnacles. Limpets occur throughout the intertidal zone up to the edge of the lichen zone.

An investigation was performed to assess intertidal communities within Port Valdez for disturbance from oil terminal operations and to provide a reference point in the event of a major oil spill. Non-destructive sampling was used to estimate abundance and percent cover of all flora and fauna. The sampling frame was limited to safely accessible beaches ranging from cobble beaches to rock pinnacles. Sampling locations included a rock pinnacle surrounded by a mudflat located at the western edge of the mudflats, three cobble beaches, and five rocky shores (Fig. 3.1). Two locations, Berth 4 and Saw Island, are at the marine terminal. The first location is approximately 6 km and the last is 17 km from the head of the fjord. One transect at each site was started at the highest occurrence of the black lichen *Verrucaria maura* with sampling points marked at 40 cm vertical intervals and re-sampled in mid-summer from 1988 to 1992. An area of approximately 1.2 m^2 (40 cm x 3 m centered on marked sampling locations) was examined. A rapid-assessment, visual estimation method (Nelson-Smith, 1972; Feder and Keiser, 1980; Cowell and Monk, 1981) was used to classify the abundance and cover of intertidal flora and fauna at each sampling point along the transect. The percent cover and abundance of all algae and animals were categorized as scores ranging from 1 (rare) to 7 (extremely abundant) using criteria determined for five general groups (Table 3.1). Difficulties arose with the classification scale for barnacles so data for this group is not included here.

Physical and climatic variables were determined from pre-existing data and climatic information. Distance from the head of the fjord (km) was estimated for each station from maps. Salinity and suspended sediments were estimated from environmental data collected in Port Valdez from sampling in early August, 1972 (Sharma and Burbank, 1973). Salinity and suspended sediments data from the 1972 sampling were fit to nonlinear equations with distance from the head of the fjord as the explanatory variable. Values were then estimated for each intertidal site of the present study from the fitted equations. One site, Gold Creek, has additional source for glacial runoff and data for that site were taken from the nearest sampling location. Although sampled in a different time period, the summer environmental gradients are recurrent in Port Valdez, as has been shown for other glacial outwash fjords (Sharma and Burbank, 1973; Hoskin et al., 1978; Colonell, 1980; Feder and Keiser, 1980; Cimberg, 1982; Farrow et al., 1983). Additional environmental variables were determined from available climatic data for the City of Valdez (National Weather Service, http://www.arh.noaa.gov/). These variables include average annual air temperature (°C), annual minimum and maximum air temperatures (°C), annual precipitation (cm), maximum winter wind speed (km hr^{-1}), number of days over 22°C, and prior summer's number of days over 22°C.

Community data were analyzed using ordination techniques. Abundance and percent cover scores were averaged for each taxon category by site and year and a Euclidean distance matrix calculated on square-root transformed data. The square-root transformation was applied to reduce effects of dominant species on the ordination. Ordination of the distance matrix was performed using nonmetric multidimensional scaling (MDS). Correlations between environmental variables and the biotic ordination were assessed to determine which environmental variables may strongly influence intertidal community structure. The BIOENV routine of Primer was used to determine associations of the biotic ordination to environmental variables by correlating the biotic distance matrix to distance matrices derived from combinations of environmental variables. Climatic and physical variables are highly correlated so a multivariate method is needed . The indirect gradient approach taken here will provide evidence as to which of the variables are most closely associated with intertidal community structure (Clarke and Ainsworth, 1993). Analyses were performed with the Primer software package (Clarke and Gorley, 2001).

3. Results

Ordination of the percent cover and abundance data suggests groupings of stations associated with habitat structure and salinity. The strongest pattern in the MDS ordination was the separation of stations along the horizontal axis from the head to the mouth of the fjord (Fig. 3.2). The MDS plot also suggests a separation by habitat structure as rocky locations were positioned to the top of the plot and cobble beaches and the mud and pinnacle site in the lower half of the plot. Overlays of environmental variables demonstrate that salinity had a strong gradient along the horizontal axis of the MDS ordination (Fig. 3.3). The variables distance from head of the fjord and suspended sediments show a poorer association with the ordination plot but distances generally increased and suspended sediments decreased from the right to the left in the MDS overlays. Two sampling points appear separated from their respective habitats, Seven Mile Beach 1989 and Terminal Rock 1992, but were not associated with extremes in environmental variables while a third, Gold Creek 1990 in the upper right corner, was associated with a low minimum air temperature (Figs. 3.2 and 3.3). Correlations of distance matrices using BIOENV demonstrate that the matrix based on salinity had the highest correlation to the biotic similarity matrix of all variables (Table 3.2). Among the single variable correlations, distance to the head of the fjord had the second highest correlation. The multiple variable combination of salinity and annual air temperature had the second highest correlation overall.

Overlays of percent cover and abundance on the MDS ordination suggest some associations between intertidal flora and fauna and environmental characteristics. Flora and fauna presented included organisms identified as contributing to dissimilarities between habitats by the analysis of similarities (SIMPER) procedure. Percent cover for *Fucus gardneri*, *Halosaccion glandiforme* and ulvoid algae were higher on rocky shores (the top of the MDS plot) although ulvoid algae were high on the mud and pinnacle habitat as well (Fig. 3.4). The algal family Rhodomelaceae demonstrated highest percent cover on rocky beaches with high salinity. The percent cover of the alga *Leathesia difformis* increased in association with higher salinity whereas percent cover for *Achrochaetium* sp., *Enteromorpha* sp. and *Ulothrix* sp. decreased with higher salinity values (Fig. 3.5). Limpet abundance scores appeared nearly constant among beaches (Fig. 3.6). Abundance scores for *Littorina sitkana* suggest an association with very low salinity as scores were low only where salinity values were very low. The abundance of the predatory gastropod *Nucella lamellosa* was relatively high only where salinity values were high in the left side of the ordination and was low throughout the rest of the study area. Where the abundance scores for *N. lamellosa* were highest, the sampling points furthest to the left in the ordination, scores for the mussel *Mytilus trossulus* were reduced. *Mytilus* scores were also reduced on cobble beaches with medium salinity in the lower middle portion of the MDS plot. The gastropod grazer *Margarites pupillus* was abundant only at one site in the upper right of the plot, a sampling point associated with lower minimum temperature, lower percent cover of *Achrochaetium* and *F. gardneri*, lower abundance of *L. sitkana*, and *M. trossulus*, and higher percent cover of *Enteromorpha* and *Ulothrix*.

4. Discussion

Low salinity can have large effects on intertidal assemblages directly through physical tolerance or indirectly through other mechanisms. Salinity appears to directly affect invertebrate predators and grazers in the glacial fjord studies but its influence on algae is less clear. Effects were partially predictable as invertebrate predators in Port Valdez and Prince William Sound, *Nucella lamellosa* and sea stars, are known to be sensitive to low salinity (Feder and Christensen, 1966; Stickle et al., 1985; Feder and Bryson-Schwafel, 1988; Carroll and Highsmith, 1996; Menge and Branch, 2001) (Table 3.3). The exclusion of predators may allow the mussel *Mytilus trossulus* to flourish as suggested in this study by the negative association with high abundance of *N. lamellosa* (Feder and Keiser, 1980; Cimberg, 1982; Feder and Bryson-Schwafel, 1988; Hunt and Scheibling, 1998). *Mytilus trossulus* is heavily preyed upon and low-salinity refugia may be important for establishing and maintaining mussel populations in areas with many predators (Cimberg, 1982; Carroll and Highsmith, 1996; Feder et al., 2003). The distributions of intertidal grazers appear to be less affected by low salinity showing reduced values for only one of the grazers, *Littorina sitkana,* where salinity values were lowest. Another littorine, *L. littorea,* is known to be tolerant to low salinity so some tolerance by *L. sitkana* is expected but the low salinity values in the most eastern site may exceed the tolerance of this grazer (Taylor and Andrews, 1988).

It is not clear whether effects of salinity on flora in this fjord are direct. Distributions of algae may be influenced directly through physical tolerance, indirectly through reduced grazing and competition, or combinations of these and other factors (Petraitis, 1987; Menge and Branch, 2001) (Table 3.3). Some algal species may be intolerant to and may be from excluded at sites with low salinity making more space available for other algae. It is also possible that some algae become superior competitors at low salinity (*Enteromorpha* sp. and *Ulothrix* sp.) and are able to occupy more space (Fong et al., 1996). Alternatively, associations of flora with the salinity gradient may be related to changes in grazer populations. The reduced grazing at sites with very low salinity may allow flora normally grazed down to compete for space. This may be the case for *Achrochaetium* sp. where percent cover is low at sampling points with high salinity. Yet, this alga also has low percent cover where herbivorous gastropods (*L. sitkana* and *Margarites pupillus*) are abundant confounding a conclusion as to which factor, salinity or grazing pressure, controls its distribution. Indirect effects of salinity through increased competition and reduced grazing both seem plausible explanations (Menge and Branch, 2001). It appears then, that for some algae found in the study area, altered biological interactions caused by low salinity may be as or more important a mechanism than physical tolerance whereas tolerance appears to be most important for invertebrates.

Habitat structure is strongly associated with the MDS ordination but its relationships with dominant flora and fauna are not clear as only a few algae show a habitat preference. A few organisms, including *F. gardneri*, Ulvoid algae, and *M. trossulus*, demonstrated lower abundance on cobble beaches but others appeared to be more associated with the salinity gradient or other factors (*Achrochaetium*, *Enteromorpha* and Rhodomelaceae). The lack of association between most overlays of abundance and cover for dominant organisms and habitat structure suggests habitat structure has a weak to small effect on the dominant members of the intertidal assemblage. The separation of sampling points by habitat in the ordination, however, suggests a much larger effect. Habitats sampled include cobble beaches and a pinnacle surrounded by mud. The cobble habitat is likely to be frequently disturbed by moving rocks and the mud pinnacle habitat reflects higher suspended sediments. Frequent disturbance, as from rocks overturning on cobble beaches, can be an important factor in community development (Sousa, 2001; Menge and Branch, 2001; Ritter et al., 2005). High suspended sediments can reduce productivity and accumulations in mussel beds can cause increased mortality (Goering et al., 1973; Seed and Suchanek, 1992). Thus, natural disturbance, stress from wave action, and sediment accumulations (where suspended sediments are high) may all have a role in separating sites (Seed and Suchanek, 1992; Sousa, 2001).

Most of the species in the fjord are euryhaline, as opposed to a gradient from marine to freshwater organisms expected in estuarine conditions. The flora and fauna of estuaries commonly change from marine to freshwater organisms as salinity decreases towards the head where riverine influences are strong (Perkins, 1974; Hardwick-Witman and Mathieson, 1983; Menge and Branch, 2001; Nybakken, 2001; Ritter et al., 2005). In contrast, euryhaline flora and fauna, including *F. gardneri* and *M. trossulus*, are found on rocky shores at the head of fjords in Port Valdez and Prince William Sound (Feder and Keiser, 1980; Cimberg, 1982; Feder and Bryson-Schwafel, 1988; Stout, 1999). It is possible that the marine conditions present much of the year prevent establishment of brackish water species on hard substrates in the fjord (Colonell, 1980; Feder and Keiser, 1980; Feder and Bryson-Schwafel, 1988). It is also likely that the measured low salinity values do not reflect the complete range to which an organism is exposed over the day or week as salinity variations arise from nearshore turbulence, tidal change, wave action, and wind mixing. Although surface-water salinity measurements were low, intertidal flora and fauna were likely to be frequently exposed to higher salinity values offsetting effects of temporary exposure to freshwater (Kramer and Fong, 2000).

In summary, salinity appears to have large direct and indirect effects on biota and their interactions while habitat structure plays a lesser role. Salinity tolerance directly affects invertebrate predators in the study and possibly, to a lesser extent, grazers. Biological interactions may be indirectly altered by direct influence of the summer salinity gradient on grazing rates and some algae, resulting in shifts in competitive balances. Although habitat structure is associated with a large separation of sites in the MDS ordination, large differences in the abundance and cover of many common organisms are shown in the MDS overlays. Overall, the salinity gradient appears to be important in explaining horizontal variability in intertidal assemblages along the rocky shoreline of the glacial fjord studied, possibly by indirect effects from physical tolerance of grazers and predators to low salinity, and habitat structure (probably reflecting disturbance from wave action, rolling rocks, and other environmental factors) plays a secondary role.

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Table 3.1. Abundance and percent cover scales for intertidal flora and fauna for Port Valdez, 1988-1992. The scores and category definitions are given for all flora and fauna sampled.

Score	Lichens % Cover	Algae % Cover	Mussel % Cover	Limpets, Littorine Abund.	Other Gastropod Abund.
1	1-2 small patches	1-2 plants	1-2 individuals	1 ind. 2m^2	< 1 ind. 2m^2
2	Widely scattered, small patches	Scattered plants, zone indistinct	Scattered individuals, no patches.	1 ind. m^2	1 ind. 2m^2
3	Large scattered patches, <1% cover.	<5%, zone apparent	Small Patches, many scattered individuals.	> 1-10 m^2	1 ind. m^2
4	1-20%	5-30%	Large patches < 20%	10-50 m^2	> 1-10 m^2
5	20-50%	30-60%	20-50%	50-100 m^2	10-50 m^2
6	50-80%	60-90%	50-80%	100-200 m^2	50-100 m^2
7	> 80%	> 90%	> 80%	> 200 m^2	> 100 m^2

Table 3.2. Summary of matrix correlations from the BIOENV procedure for intertidal data from Port Valdez, 1988-1992. Spearman's rank correlation (ρ) was used to determine correlations between the biotic similarity matrix and distance matrices for environmental variables. # Vars = number of variables.

	# Vars	ρ	Variables
Single Variables	1	0.568	Salinity (‰)
		0.432	Distance (km)
		0.374	Suspended Sediment (mg l^{-1})
		0.022	Prior Summer #Days > 22° C
		0.017	Winter Maximum Windspeed (km hr^{-1})
		0.014	Minimum Air Temperature (C°)
		-0.010	Prior Summer Maximum Temperature (C°)
		0.032	Maximum Air Temperature (C°)
		-0.041	Number Days > 22° C
		-0.043	Annual Precipitation (cm)
		-0.049	Annual Air Temperature (C°)
Best fit with multiple variables.	1	0.568	Salinity
	2	0.555	Salinity, Annual Air Temperature
	2	0.503	Salinity, Distance
	3	0.502	Salinity, Distance, Annual Air Temperature
	3	0.486	Salinity, Distance, Maximum Air Temperature

Table 3.3. Hypothesized interactions and mechanisms controlling flora and fauna in Port Valdez, Alaska.

Organism	Evidence	Hypothesis	Mechanism
Nucella lamellosa	Abundant only at west end of fjord.	Low salinity excludes *Nucella* from most of the fjord.	Tolerance. Direct effect.
Littorina sitkana	Slight decrease with decreasing salinity, low abundance with lowest salinity.	Appears to be moderately sensitive to salinity but may be excluded at lowest values.	Tolerance. Direct effect.
Margarites pupillus	Abundant only where other grazers are depleted.	May be poor competitor but able either to reproduce quickly or hide subtidally to respond to open space.	Competition. Direct effect.
Achrochaetium sp.	1) Percent cover increases with decreasing salinity.	Better tolerance to low salinity allows these algae to occupy more space.	Tolerance. Direct effect on *Achrochaetium*. Indirect effect through other algae. Competition. Direct effect.
	2) Cover higher where abundance of grazers lower.	May be unable to tolerate grazing.	Grazing. Direct effect.
	3) Particularly low at a disturbed site where it should otherwise be abundant.	May have been removed during winter disturbance. May be sensitive to physical disturbance.	Tolerance. Direct effect. Life history trait.

Table 3.3. Continued.

Organism	Evidence	Hypothesis	Mechanism
Enteromorpha sp., *Ulothrix* sp.	1) Percent cover increases with lower salinity.	Better tolerance to low salinity allows these algae to occupy more space. Superior competitive ability	Tolerance. Indirect effect through other algae. Competition, Direct effect.
	2) Percent cover increases as abundance of grazers decreases.	These turf algae may be a preferred food source for grazing gastropods.	Grazing. Direct effect.
Leathesia difformis	Percent cover decreases with decreasing salinity.	Sensitive to low salinity.	Tolerance. Direct effect.
Porphyra sp.	Highly abundant only at a disturbed site.	Poor competitor.	Competition. Direct effect.
Rhodomelaceae	Abundant on rocky beaches with high salinity.	Sensitive to low salinity and possibly to disturbance.	Tolerance. Direct effect. Habitat Preference.

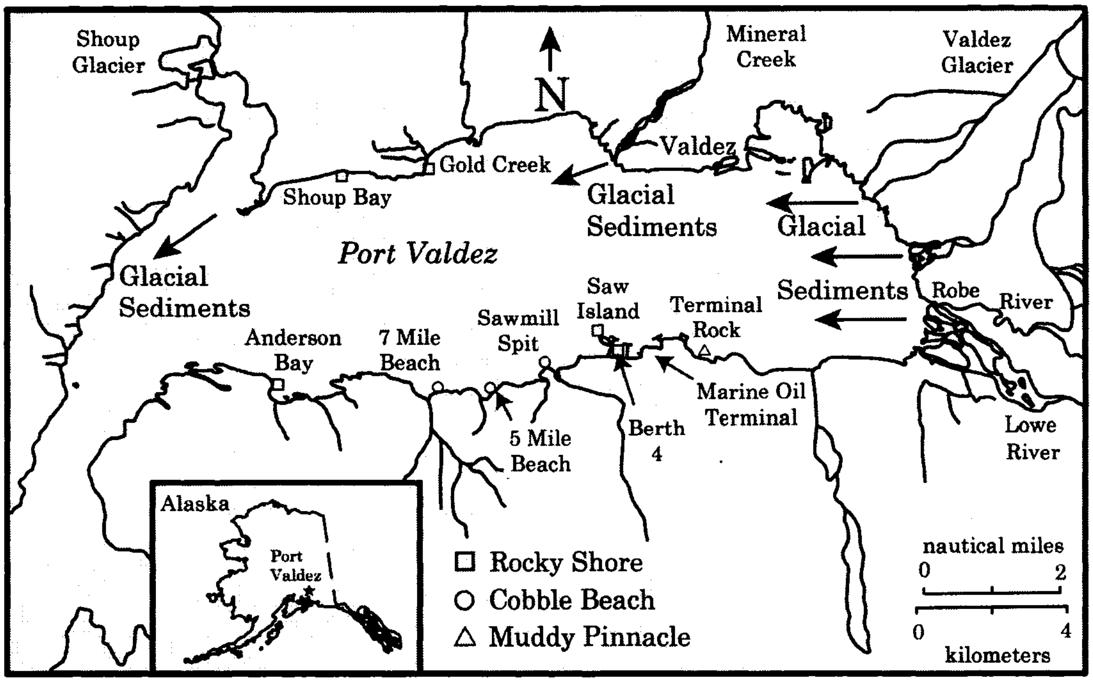


Figure 3.1. Map of intertidal sampling locations in Port Valdez, Alaska, 1988-1992. Arrows denote the seasonal flow from streams with glacial meltwaters.

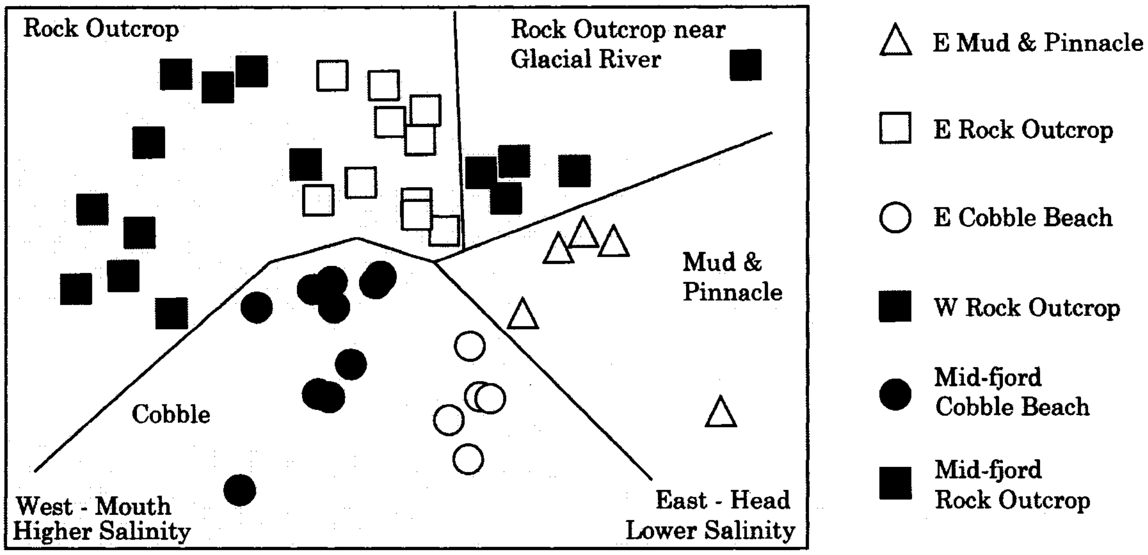


Figure 3.2. MDS ordination of intertidal sites by year for Port Valdez, 1988-1992. The ordination is based on square-root transformed abundance and percent cover scores. The relationship to salinity and habitat structure are shown.

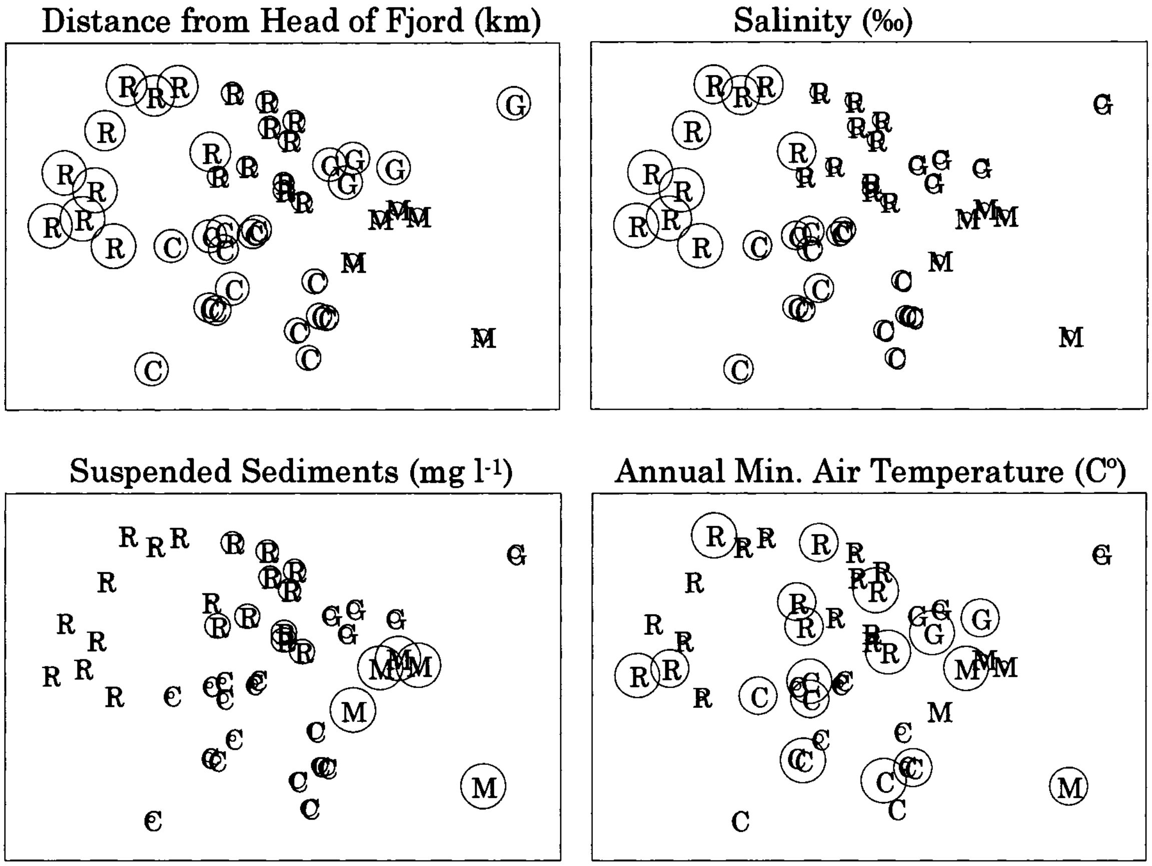
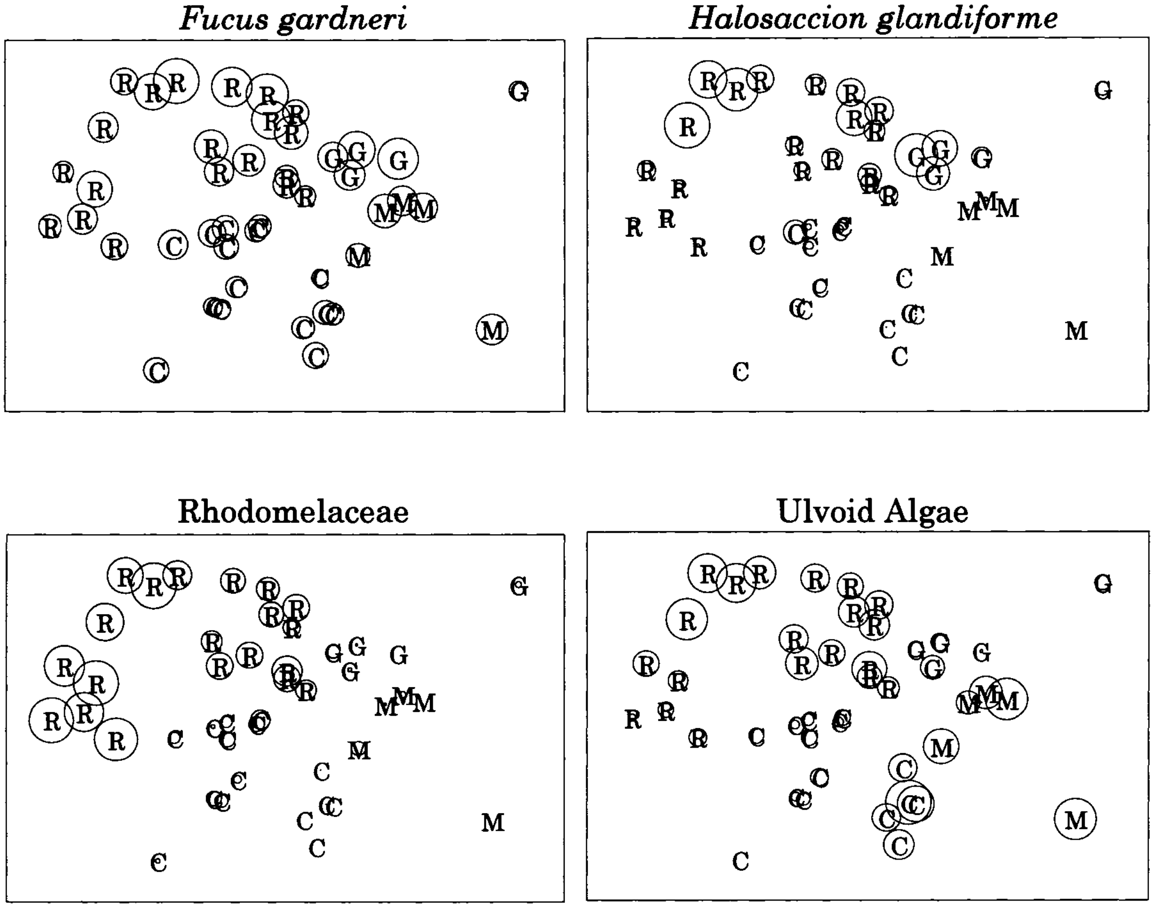


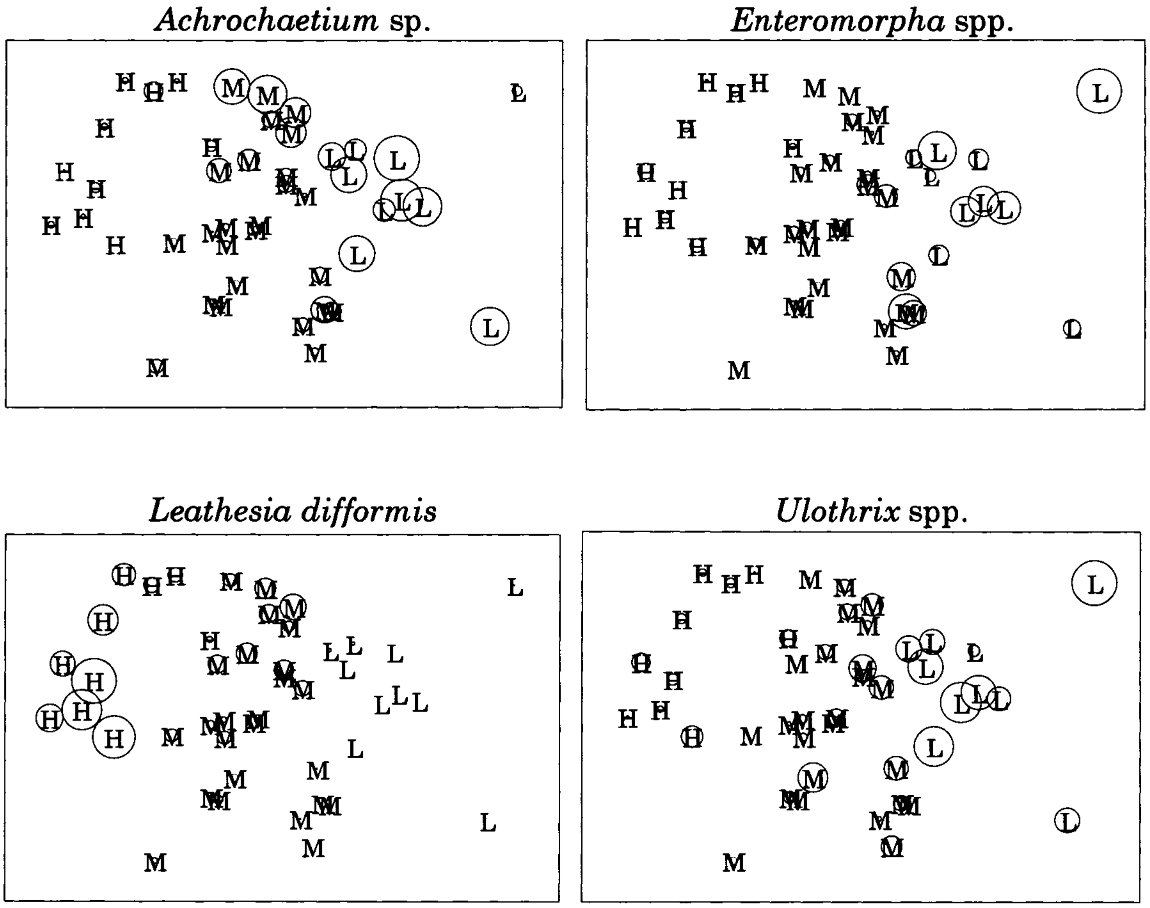
Figure 3.3. Overlays of selected environmental variables and habitat categories on the MDS ordination for Port Valdez, 1988-1992. Circles are proportional to variable values. Habitat codes are C = cobble beach, G = rocky beach with glacial influences, M = mud and pinnacle, and R = rocky beach.



Fucus gardneri | *Halosaccion glandiforme*

Rhodomelaceae | Ulvoid Algae

Figure 3.4. Overlays of percent cover of selected algae and habitat categories on the MDS ordination for Port Valdez, 1988-1992. Habitat codes are C = cobble beach, G = rocky beach with glacial influences, M = mud and pinnacle, and R = rocky beach. Size of circle is proportional to average percent cover score.



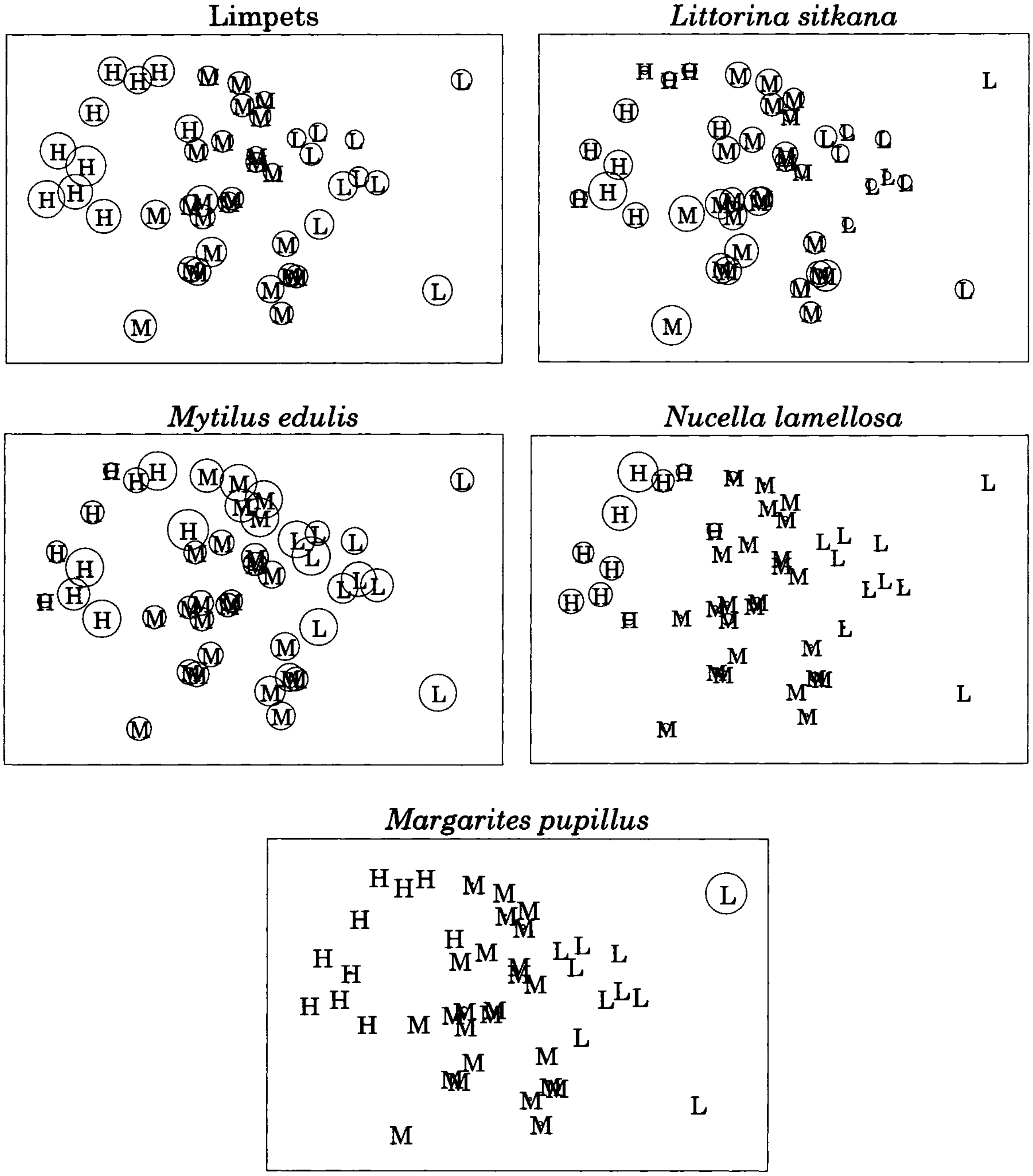
Achrochaetium sp.

Enteromorpha spp.

Leathesia difformis

Ulothrix spp.

Figure 3.5. Overlays of percent cover of dominant algae and salinity categories on the MDS ordination for Port Valdez, 1988-1992. Salinity codes are H = high, M = medium, and L = low. Size of circle is proportional to average percent cover score.



Limpets

Littorina sitkana

Mytilus edulis

Nucella lamellosa

Margarites pupillus

Figure 3.6. Overlays of the abundance of selected fauna and salinity categories on the MDS ordination for Port Valdez, 1988-1992. Salinity codes are H = high, M = medium, and L = low. Size of circle is proportional to average abundance score.

Chapter 4 Statistical Methods for Assessing Subtle Effects in Marine Benthic Communities[1]

Abstract

Relationships between sediment hydrocarbon concentrations and infaunal communities are evaluated to describe subtle effects from the disposal of treated ballast-water effluents from a marine oil terminal. Statistical methods for correlated data provide an integrated approach for understanding how sediment hydrocarbon concentrations and infaunal communities have changed over time. Repeated measures ANOVA and geostatistical modeling indicate that a small, localized but persistent alteration of the infaunal assemblage associated with total aromatics (TARO) occurred underneath a mixing zone over the study period. Analyses suggest that as TARO declined over the study period, the percent abundance of two sensitive polychaete worms (*Galathowenia oculata* and *Melinna cristata*) and to a lesser extent, infaunal abundance, increased. In contrast, multivariate analyses do not suggest an association between TARO and infaunal community structure. It appears that multivariate methods may not be effective for documenting subtle effects. Multiple statistical tools should be considered when designing studies to avoid missing small effects.

Keywords: Infauna, geostatistics, hydrocarbons, disturbance, *Melinna, Galathowenia.*

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[1] Blanchard, A. L. 2006. Statistical methods for assessing subtle effects in marine benthic communities. Prepared for publication in Marine Environmental Research.

1. Introduction

Invertebrate community data from soft sediments are widely used to survey marine environments for anthropogenic effects. Infauna are excellent indicators of environmental conditions as they are not highly mobile and move small distances compared to the scale of anthropogenic stressors (Clarke, 1999). Measurement of change in infaunal communities is particularly useful when assessing effects from the point-source disposal of pollutants and sources with defined spatial limits. Effluent outfalls are stressors of this type including outfalls associated with treatment of oily ballast waters from oil tankers (Blanchard, Feder, & Shaw, 2002, 2003). Documented responses of infaunal communities to sediment hydrocarbons include increases in the abundance of the small, opportunistic polychaete families Capitellidae and Spionidae and elimination of some tube-dwelling worms (Seng et al., 1987, May & Pearson, 1995, Blanchard et al., 2002, 2003).

Multivariate statistics have been successfully applied to a range of anthropogenic disturbance in marine environments (Kenny & Rees, 1996; Lindegarth & Hoskin, 2001; Je, Belan, Levings, & Koo, 2004; Blanchard & Feder, 2003; Morrisey, Turner, Mills, Williamson, & Wise, 2003; Hewitt, Anderson, & Thrush, 2005). Associative evidence from such studies can be difficult to interpret but when effects are moderate to large, an integrative weight of evidence approach using multivariate methods for pattern recognition can lead to a correct conclusion that a disturbance is present (Morrisey et al., 2003). When effects are small, however, these statistical methods may not lead to detection of disturbance (Blanchard et al., 2002; Morrisey et al., 2003). Confounding of variables and the presence of unmeasured environmental gradients may obscure small responses of infauna to contaminants in field studies (Morrisey et al., 2003). Failure to recognize the limitations of statistical methods may lead to understating anthropogenic effects or concluding recovery of a system when effects still linger and this can have large, negative impacts on management decisions (Peterman, 1990; Peterman & M'Gonigle, 1992). Increased sediment hydrocarbons coincided with increased numbers of opportunistic taxa, lower abundance of sensitive taxa, and higher average infaunal abundance in a small area at an oil terminal in Alaska (Blanchard et al., 2002, 2003). Petrogenic hydrocarbon accumulations in sediments in the latter study were derived from the disposal of treated ballast-water effluents and a short-term rise in aromatic hydrocarbon concentrations was observed at one sampling location in 1995 through 1997 (Blanchard et al., 2002). Although large effects associated with the increased sediment hydrocarbons in 1995 to 1997 were readily apparent using widely accepted multivariate statistical procedures, it remained unclear how the infaunal assemblage had adjusted to the presence of petrogenic hydrocarbons over a longer period of time.

A common feature of marine monitoring studies is model-based rather than design-based selection of sampling locations. This is particularly true when monitoring point-source pollutants in marine environments as regulatory agencies often require fixed sampling locations selected on expectations of where disturbance might occur and distance from source (model-based studies: Oslgard & Gray, 1995; Ellis & Schneider, 1997; Ellis. Schneider & Thrush, 2000), rather than by probabilistic designs (design-based studies: Underwood, 1994; Van der Meer, 1997). Advantages of fixed sampling locations include higher power compared to random effects (Van der Meer, 1997) and, for repeated sampling, higher confidence in conclusions from the same sites sampled over time. Disadvantages of fixed sampling locations are that inferences are restricted to sampling locations in analysis of variance, the data are spatially and temporally correlated, and statistical methods for correlated data (which are less familiar to marine ecologists) are required. A geostatistical approach is appropriate for correlated data from model-based studies and has wide applicability in marine monitoring studies (Cressie, 1993; Stein, van Groenigen, Jeger & Hoosbeek, 1998; Ricotta & Avena, 1999).

Here, retrospective analysis of data from a long-term environmental study using repeated measures analysis of variance (rm ANOVA), geostatistical analysis, and multivariate methods is performed to clarify the extent of small spatial and temporal variations of a soft-sediment infaunal community and associations with low sediment hydrocarbon concentrations. Observations of a small disturbance event in 1995 (Blanchard et al., 2002) led to addition of sites to the sampling plan in 1998 to understand the spatial and temporal extent of faunal adjustments (Blanchard et al., 2003). The hypothesis that subtle effects have persisted as a result of the disposal of treated ballast-water effluents was developed from initial results using the expanded sampling plan and is tested here. The ability of the different statistical tests to detect disturbance is discussed.

2. Methods

Sampling for this study occurred in Port Valdez, Alaska (Figure 4.1). Port Valdez is a glacial outwash fjord in Prince William Sound and the site of a marine oil transfer facility (Shaw & Hameedi, 1988; Blanchard et al., 2002). Oily ballast water from tankers is treated onshore and discharged into the fjord at 60 to 80 m depth through a diffuser pipeline into a permitted mixing zone (USEPA NPDES permit AK-002324-8). Sediment was sampled annually adjacent to the diffuser pipeline within and outside the permitted mixing zone (Blanchard et al., 2002, 2003). Sampling consisted of the collection of sediment for hydrocarbon concentration determinations and infaunal identifications at fifteen locations in the fall of each year from 1989-2004 (analytical methods are described in Blanchard et al., 2002). Sampling locations were selected based on the expected patterns of dispersal of treated effluents. These locations form a transect from the western most sampling location to the eastern site (roughly 2 km). The sum of concentration of 18 hydrocarbon species in sediments (TARO ng g^{-1}) identified as priority aromatic hydrocarbons by the U.S. Environmental Protection Agency and infaunal abundance were analyzed in this study. Total aromatics represents a proxy for trends in hydrocarbons, rather than an absolute measure, since concentrations of additional hydrocarbon species, the alkyl homologs and other aromatic compounds, were not determined. Empirical evidence from earlier studies indicated a strong, negative association between TARO and two tube-dwelling polychaete worms, *Galathowenia oculata* and *Melinna cristata*, suggesting sensitivity to hydrocarbons (Blanchard et al., 2002, 2003). Thus, the percent abundance of two sensitive species (*G. oculata* and *M. cristata*) was included as well.

Repeated measures ANOVA was applied to discern temporal trends in TARO, infaunal abundance, and the percent sensitive species. Repeated measures ANOVA is appropriate because the same sites are repeatedly sampled (Kuehl, 1994). Due to unequal replication for some years, rm ANOVA was applied separately to data for each station with time as the repeated factor. Repeated measures ANOVA analyses were performed using SAS (SAS Institute, Cary, NC). Multiple contrasts for these analyses are the difference of means for each year from the grand mean. Results are summarized in plots of contrast values and 95% confidence intervals for four stations; stations 80 and 82 located at the west and east ends of the transect and outside the effluent mixing zone, respectively, and D25 and D33 just to the west and east of the diffuser pipeline, respectively, and under the mixing zone. Data for TARO are *ln*(X) transformed, abundance *ln*(X+1), and the percent abundance of sensitive taxa fourth-root transformed to meet the assumptions of normality for ANOVA.

Spatial and temporal predictions of transformed average TARO, abundance, and percent sensitive species are modeled using geostatistical methods. The geostatistical approach used here makes predictions for hydrocarbon and biological variable values in the presence of linear trends (distance and time) while accounting for spatial correlations (universal kriging; Cressie, 1993). The benefits of applying geostatistical methods are numerous but for this study, the greatest benefit is the confidence that inferences made from plots of predicted values generated by this method are statistically valid. Additionally, statistical procedures are often applied assuming the only source of error is a random process but this may overlook small-scale effects that might be important (Cressie, 1993). The goal of this approach is to assess hydrocarbon and biological data for small-scale effects missed in prior analyses (Blanchard et al., 2002, 2003).

Geostatistical analyses were used to model trends in TARO, abundance, and percent sensitive species. Analyses were performed with the statistical program R (www.r-project.org) and the library GeoR (Ribeiro & Diggle, 2001). Parameters defining the shape of the variogram models are the nugget, which is microscale variation and measurement error; the range, indicating the distance at which data are no longer correlated; and the upper limit of the model called the sill, which represents total variance (Cressie, 1993). Empirical (semi)variograms were determined from the data as:

$$\hat{\gamma}(h) = 1/(2\,|\,\mathrm{N}(\mathbf{h})\,|\,)\Sigma[\mathrm{z}(\mathbf{x}) - \mathrm{z}(\mathbf{x}+\mathbf{h})]^2 \qquad 4.1$$

where $\hat{\gamma}(h)$ is the observed semivariance, **x** and **x+h** are data vectors separated by the distance vector **h,** Z(**x**) is a random process, and N(**h**) is the number of distance pairs (Cressie, 1993; Ricotta & Avena, 1999). The empirical variogram for total aromatics (TARO in ng g^{-1}) was modeled by the exponential model (Cressie, 1993):

$$\gamma(\mathbf{h};\theta) = \begin{cases} 0, & \text{for } \mathbf{h} = 0 \\ c_0 + c_e\{1 - \exp(-\|\mathbf{h}\|/a_e)\} & \text{for } \mathbf{h} \neq 0 \end{cases} \qquad 4.2$$

where $\gamma(\mathbf{h};\theta)$ is the semivariance, $\mathbf{h}$ = vector of distances, $\theta = (c_o, c_e, a_e)'$, $c_o \geq 0$ is the nugget, $c_o + c_e \geq 0$ is the sill, and $a_e \geq 0$ is the range (Cressie, 1993). For the exponential model, data are correlated at all distances and 3a represents a distance at which correlations are reasonably low (Webster and Oliver, 2001). The spherical model fitted to the empirical variograms for abundance and percent sensitive species is:

$$\gamma(\mathbf{h};\theta) = \begin{cases} 0, & \text{for } \mathbf{h} = 0 \\ c_o + c_s\{3\|\mathbf{h}\|/2a_s - 1/2(\|\mathbf{h}\|/a_s)3\}, & \text{for } 0 < \|\mathbf{h}\| \leq a \\ c_o + c_s & \text{for } \|\mathbf{h}\| \geq a, \end{cases} \qquad 4.3$$

where $\theta = (c_o, c_s, a_s)'$, $c_o \geq 0$ is the nugget, $c_o + c_s \geq 0$ is the sill, and $a_s \geq 0$ is the range (Cressie, 1993). Inclusion of a linear model for year and distance removed significant trends making the models isotropic (varying similarly in all directions). Data were transformed as described above for rm ANOVA. Contour plots of back-transformed predicted values are presented.

Nonmetric multidimensional scaling (MDS) was applied to the hydrocarbon and infaunal datasets. Here, a second-stage MDS approach for testing matrix associations was used to determine if a change in environmental conditions has occurred over time (Somerfield & Clarke, 1995). Assuming sampling has occurred for a number of years, distance or similarity matrices can be calculated for each year based on a repeated sampling of the same locations within each year. Items within the matrices will be distance coefficients between pairs of sites. When environmental conditions among years are similar, differences between site pairs will be similar from year-to-year and thus, matrices will be similar. That is, if the community has not varied substantially over time, then community differences between any two site pairs will be approximately the same (with some error) and the distance or similarity coefficient between these two sites will be similar from year-to-year. If correlations are calculated among pairs of matrices and these correlations are then used as the distance matrix for MDS, the degree of separation between any two years in the ordination reflects their similarity. If two year matrices are not close in the ordination, this indicates that differences between site pairs have changed and thus, underlying environmental conditions have likely changed. In the case of disturbance, if a perturbation has affected sites within a year or a set of years, they will be positioned remotely from other entities in the MDS plot.

A second-stage analysis for a disturbed infaunal community is presented as an illustration of the method since this approach has not been widely applied in disturbance investigations. Disturbance of infauna from dredge spoils disposal occurred at a few locations as a dock was being built in Port Valdez (Blanchard & Feder, 2003). Sampling was initiated in Spring 1994 before dredging occurred and was performed again in Fall 1994, 1995, and 1996. Spearman's correlations between the similarity matrix for Fall 1994 (the sampling period following sediment disposal when fauna were disturbed) and matrices for Spring 1994, Fall 1995, and Fall 1996 ($\rho = 0.29$, 0.23, and 0.23, respectively) are low. The second-stage ordination shows Fall 1994 to be well separated from the other years (Fig. 4.2). The faunal composition of sites sampled in Fall 1995 indicated partial readjustment and this is reflected in the second-stage analysis by the closeness of the sampling period Fall 1995 with Spring 1994 in the ordination ($\rho = 0.89$). Following an interaction between other sources of stress, the infaunal community of Fall 1996 diverged from the community sampled in Spring 1994 as reflected in the plot ($\rho = 0.66$). This second-stage analysis effectively summarizes the patterns observed by Blanchard & Feder (2003).

The second-stage analysis method was used in the present study to determine whether changes in sediment hydrocarbon concentrations over time match changes in faunal community structure. This approach was required due to the large number of individual sampling locations and years contained in the temporal database for this study. A second-stage analysis was performed by calculating Euclidean distance matrices (years) for sediment hydrocarbon concentrations (18 priority aromatic hydrocarbons) and Bray-Curtis similarity matrices for infaunal abundance values for each year (Bray and Curtis, 1957). A Spearman's correlation matrix was calculated between pairs of distance matrices (years) for the hydrocarbon data and separately, for biotic similarity matrices. A second-stage analysis was applied to the hydrocarbon and then, to the biotic similarity correlation matrices. If any one year or sets of years were positioned remotely and in the same way in both the hydrocarbon and infauna analyses, that provides evidence that sediment hydrocarbons may be altering infaunal communities within the study area. To test the overall similarity of the matrices, a correlation was calculated between the final second-stage analysis matrices for hydrocarbons and infauna. A high correlation here would indicate similar trends within analyses, again suggesting an association between hydrocarbons and infaunal community structure. A conclusion that hydrocarbons were affecting infauna must also be supported by high hydrocarbon values and associated changes in abundance in the rm ANOVA and geostatistical analyses.

3. Results

Analyses of hydrocarbon concentrations indicate significantly higher TARO concentrations near the point of effluent disposal in some years (Fig. 4.3). The dominant trend in the contour plot of predicted TARO is a decrease over time from 1989 to 2004. Total aromatics (TARO) were higher in all years at distances of about 0.9 to 1.2 km on the y-axis corresponding to the location of diffuser pipeline (located at roughly a distance of 1 km). Contrasts from rm ANOVA also suggest that TARO values decreased over time. Mean TARO tended to be higher at all four sites over the period 1989-1992 and through 1997 for D25 and D33 but decreased overall from 1998-2004. The significantly high mean TARO values in 1994 and 1995 at D25 and D33 were not matched by values observed at 82 which decreased but were reflected in slight increases at 80. The significantly high TARO concentration at D25 in 1997 was not matched by high values at the other three stations.

Some association between TARO and infaunal abundance values are apparent. Values in the contour plot of predicted values for abundance are relatively low at distances from 0 km to 1 km over most of the study, higher to the east, and slightly higher after 2002 at most distances (Fig. 4.4). The higher abundance after 2002 shown in the contour plots is reflected in the rm ANOVA contrasts for stations D25, 80, and 82 as the increased abundance after 2002 occurs in years with lower TARO concentrations (Figs. 4.3 and 4.4). The significantly higher abundance value at D25 in 1995, lower abundance at D25 in 1997, and lower abundance at D33 in 1994 occur in years with significantly high TARO concentrations (Figs. 4.3 and 4.4). Similarly, significantly high TARO at 80 in 1991 was associated with low abundance. This was also true for 80 in 1995 where TARO was high but not outside the 95% confidence interval.

Trends in the percent abundance of sensitive species are inversely associated with trends in TARO. In the contour plot for predicted percent abundance of sensitive species, values are low at all distances between 1989-1994 and at a distance of > 1 km through 2004 coinciding with the locations and years when TARO values are generally highest (Fig. 4.5). Percent sensitive species are highest from about 1996-2004 at a distance of 0 km to 0.7 km where TARO values are lowest. Contrasts from rm ANOVA do not clearly follow the trends in TARO for D25 and D33 as values were significantly high when TARO is also high. However, the percent sensitive species are very low at D25 and D33 in the middle of the transect, as compared to stations 80 and 82 at the west and east ends of the transect, respectively so small increases would be significant (Fig. 4.5). The contour plot for 80 suggests that sensitive species increased in abundance over time. Percent sensitive species at 82 also demonstrated an increasing trend from the significantly low value in 1989 to the high value in 1994. The increases in percent sensitive species at 80 and 82 match the decline in TARO at these sites (Figs. 4.3 and 4.5).

Multivariate analysis of the correlation matrix for the 2nd stage MDS approach does not clearly identify associations among ordinations reflecting increased hydrocarbons and disturbance of infauna. Based on results from analysis of TARO concentrations, it would be expected that stations could group into sets with high TARO, 1989-1992, 1994, 1995, and 1997, and those with low TARO, 1998-2002 (Fig. 4.3). The second-stage analysis of the hydrocarbon data partially reflects the trends in the geostatistical and rm ANOVA analyses (Fig. 4.6). The years 1989, 1991, and 1997 which are years with high TARO, are grouped separately and of the years with low TARO, 1999 and 2004 are the only ones positioned remotely in the ordination. The second-stage analysis for the infaunal data does not reflect patterns observed in the analyses of TARO as years with high TARO are grouped with years of low TARO and other years are mixed. The year 1995 is slightly separated in the infaunal ordination and this year has high TARO providing the only case where a high TARO year ordinates differently based on infauna. The correlation between the 2nd stage ordinations is very low ($\rho = 0.02$) indicating no similarity in the ordinations.

4. Discussion

The variability of marine systems, study designs with low power, limited sampling, and failure to account for spatial correlations generally limit detection of disturbance to large-sized effects, potentially missing small but important small effects (Peterman, 1990; Cressie, 1993; Morrisey et al., 2003). Multivariate statistical tools are now widely accepted as the standard for analysis of ecological community data, including cluster analysis, multidimensional scaling, and correspondence analysis (McCune & Grace, 2002). These tools are successful at detecting large changes in multivariate compositional data but the range of conditions under which these methods are useful has not been fully investigated.

Prior studies at a marine oil terminal in Alaska documented a localized effect on infauna from petrogenic hydrocarbons (Blanchard et al., 2002, 2003). Increased TARO values within a small area at the oil terminal, D25 in 1995 to 1997 (mean = 2,174 ng g^{-1} with a maximum of 6,248 ng g^{-1} at site D25 in 1997), coincided with increased numbers of opportunistic taxa, lower abundance of sensitive species, and higher average infaunal abundance reflecting an acute stress event (Blanchard et al., 2002). The exclusion of tube worms (*Galathowenia oculata* and *Melinna cristata*) appeared to be associated with increased sediment hydrocarbons surrounding the diffuser pipeline (Blanchard et al., 2002, 2003). It remained unclear how the infaunal assemblage adjacent to the diffuser adjusted to treated effluents over a longer period of time.

It appears from this study that higher sediment hydrocarbon concentrations were associated with lower percentages of sensitive species over the whole study period and that as hydrocarbons declined, effects on infauna declined, and the percent sensitive species increased. Overall, the evidence suggests a subtle response of the infaunal assemblage to hydrocarbon concentrations that are generally much lower than agency-imposed sediment quality criteria, with the exception of 1995 to 1997 at site D25 (Long, Field & McDonald, 1998; Blanchard et al., 2002, 2003). Exposure of infauna to hydrocarbons may be higher than reflected by TARO since this measure does not fully encompass all aromatic hydrocarbon species present in oil. Some marine invertebrates and fishes are intolerant of very low levels of hydrocarbons so it is not surprising that responses by infauna might occur at the low levels shown for TARO (Stekoll, Clement, & Shaw 1980; Peterson et al., 2003). The low TARO concentrations make it possible that other environmental factors could confound and be confused with effects from increased sediment hydrocarbon concentrations. Ultimately, experimental studies will be necessary to clarify the extent to which petrogenic hydrocarbons contribute to the distributions of the sensitive polychaetes. For the present, however, the subtle variations of infauna in association with the temporal trends in TARO should not be discounted. This is particularly true when for a fixed sampling locations, the biota shift towards an undisturbed assemblage as the strength of the stressor, in this case TARO, decreases. In addition to changes in sediment hydrocarbons, salmon aquaculture may be enhancing benthic biomass within the fjord through carcasses of returning adult salmon (Chapter 5). Thus, salmon carcasses may be enhancing the percentage of sensitive species in this study, except near the diffuser where these polychaetes remain low, representing a potential interaction between the oil and salmon aquaculture industries.

The rm ANOVA and geostatistical approaches provide complementary evidence for the small, temporally persistent associations between community characteristics and aromatic hydrocarbons. Both statistical tools are appropriate for analysis of correlated data arising from the resampling of identical locations over time (Kuehl, 1994; Cressie, 1993; Ricotta & Avena, 1999). Analysis of variance provides an indication of where statistically significant differences occur but overall trends can be obscured when focusing on significant multiple comparisons. The geostatistical approach was applied with the desire to visualize trends in the data while maintaining confidence that inferences would be statistically valid. Thus, the two approaches provide an integrated approach: rm ANOVA provides information on specific differences among years and plots of geostatistical-based predictions demonstrate overall trends in the data. Application of the methods is necessary since the assumption of random error in the presence of spatial correlation may obscure small but potentially important trends (Cressie, 1993). Prior to this study, conclusions about the long-term effects of treated ballast-water disposal on infauna were equivocal as multivariate analyses were not clear (Blanchard et al., 2002). The results of this study suggest that the subtle effects of hydrocarbons on the fauna are limited to a very small area surrounding the diffuser. A benefit of documenting these small effects is the increased confidence that larger effects on benthic fauna are not occurring.

In contrast to rm ANOVA and the geostatistical analysis, the second-stage MDS analysis fails to demonstrate a relationship between hydrocarbons and infauna. The second-stage MDS plots of the biological and hydrocarbon matrices show little similarity and the correlation is very low. Other investigations have likewise concluded that multivariate tools may not always be effective when effects are small or variability is high (Blanchard et al., 2002; Morrisey et al., 2003). This study demonstrates that multivariate tools may, at times, underestimate effects from anthropogenic stressors. It can be assumed that low power for detecting subtle (but yet potentially important) trends may be a widespread problem in studies of benthic communities, as it has been in fisheries research (Peterman, 1990).

In conclusion, small, localized effects on soft-sediment fauna associated with low-levels of sediment hydrocarbons are present at a marine oil terminal. The spatially and temporally persistent, inverse association between the polychaetes *G. oculata* and *M. cristata* and aromatic hydrocarbons occurs at low TARO concentrations, generally below sediment quality criteria. As sediment hydrocarbon concentrations decrease over time, infaunal abundance and the percent sensitive taxa increase as shown using statistical methods for correlated data. Analyses using methods for correlated data confirm the minor and spatially limited nature of effects from disposal of treated ballast-water effluents. Multivariate methods widely accepted among ecologists were not effective at demonstrating these subtle effects. Investigators must be careful to not rely too heavily on associative methods when making statistical inferences and should design studies so that multiple statistical approaches can be applied.

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Philosophy degree at the Institute of Marine Science, University of Alaska Fairbanks by A. L. Blanchard.

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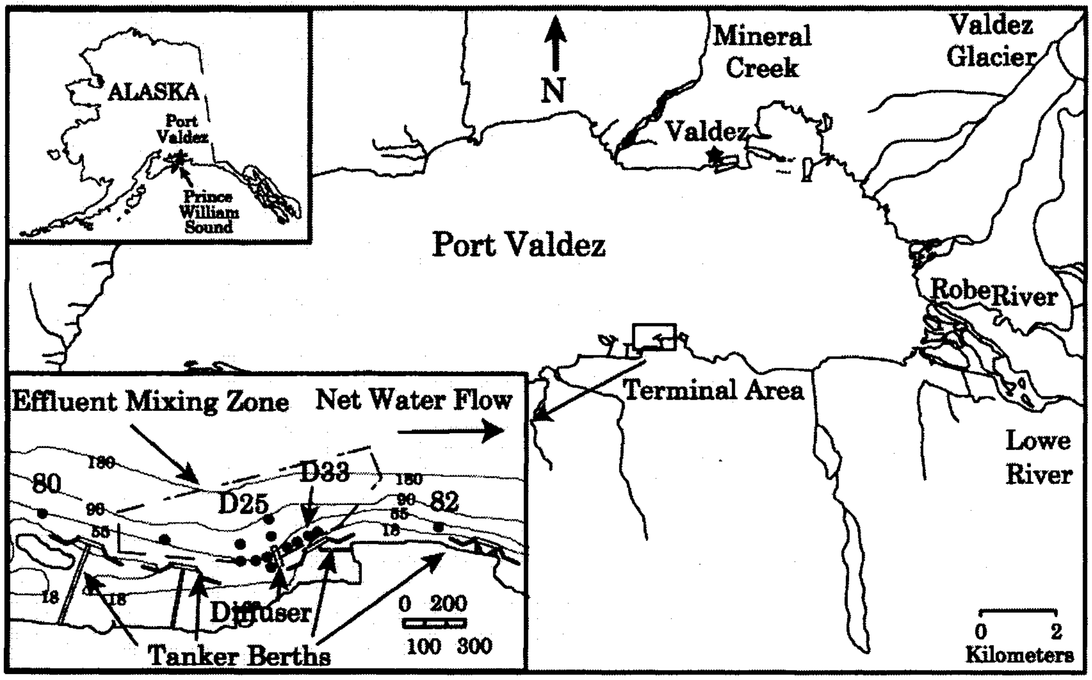

Figure 4.1. Map of sampling locations in Port Valdez, Alaska, 1989-2004. The inset in the bottom left shows the boundaries of the effluent mixing zone and locations of sampling stations. The four sites identified are used for rm ANOVA.

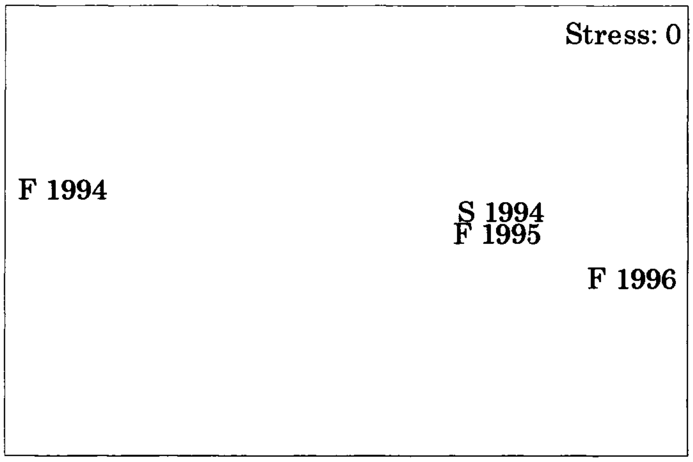

Figure 4.2. Second-stage nonmetric multidimensional scaling ordination for sites impacted by dredging in Port Valdez, 1994-1996. Sites were sampled to assess the effects of dredge-spoils deposition on infauna. Data are *ln*(X+1)-transformed infaunal abundance data from Blanchard and Feder (2003).

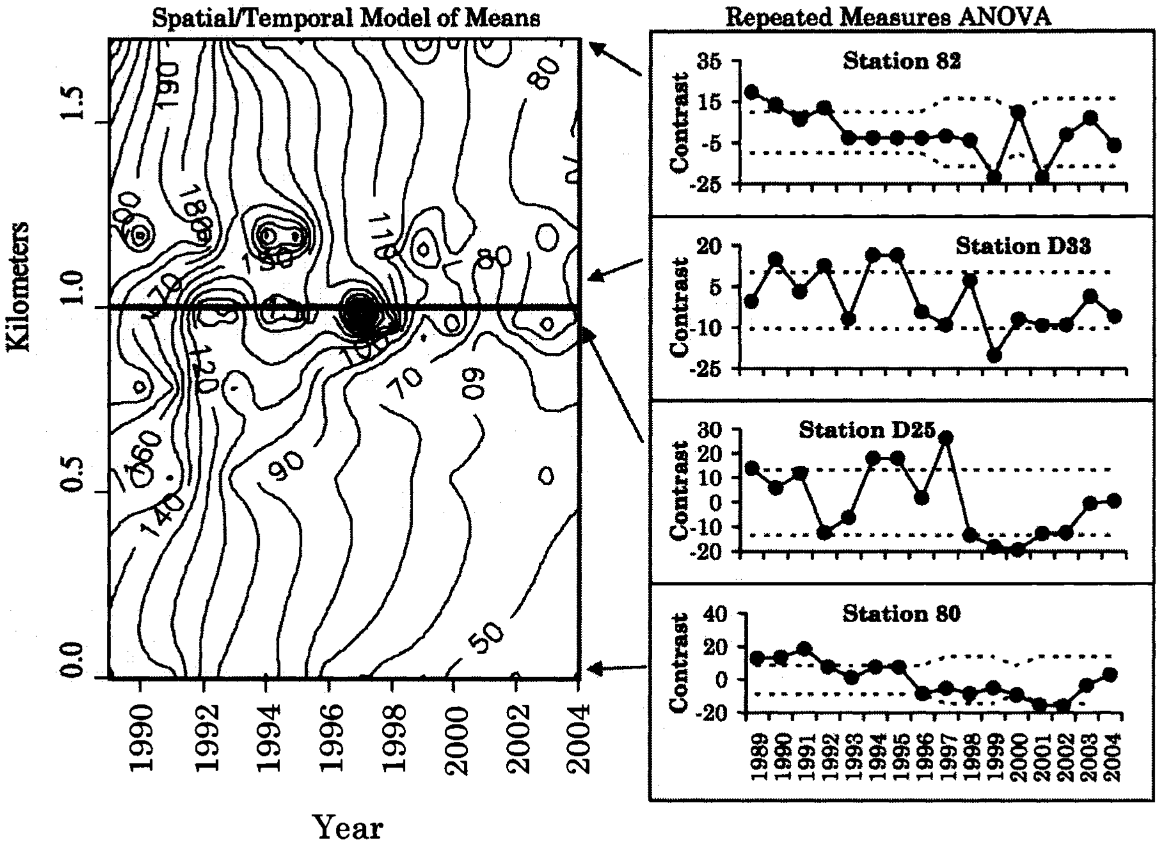

Figure 4.3. Contour plot of predicted values for TARO (ng g^{-1}) and contrasts from repeated measures ANOVA from Port Valdez, 1989-2004. The point of disposal is located at approximately 1 km along the y-axis (west to east transect distances) marked by the black line. Contrast values plotted are deviations of means from the grand mean and represent the multiple comparisons for the rm ANOVA. Dashed lines in the contrast plots are 95% confidence intervals and values outside the confidence limits provide evidence for a significant difference.

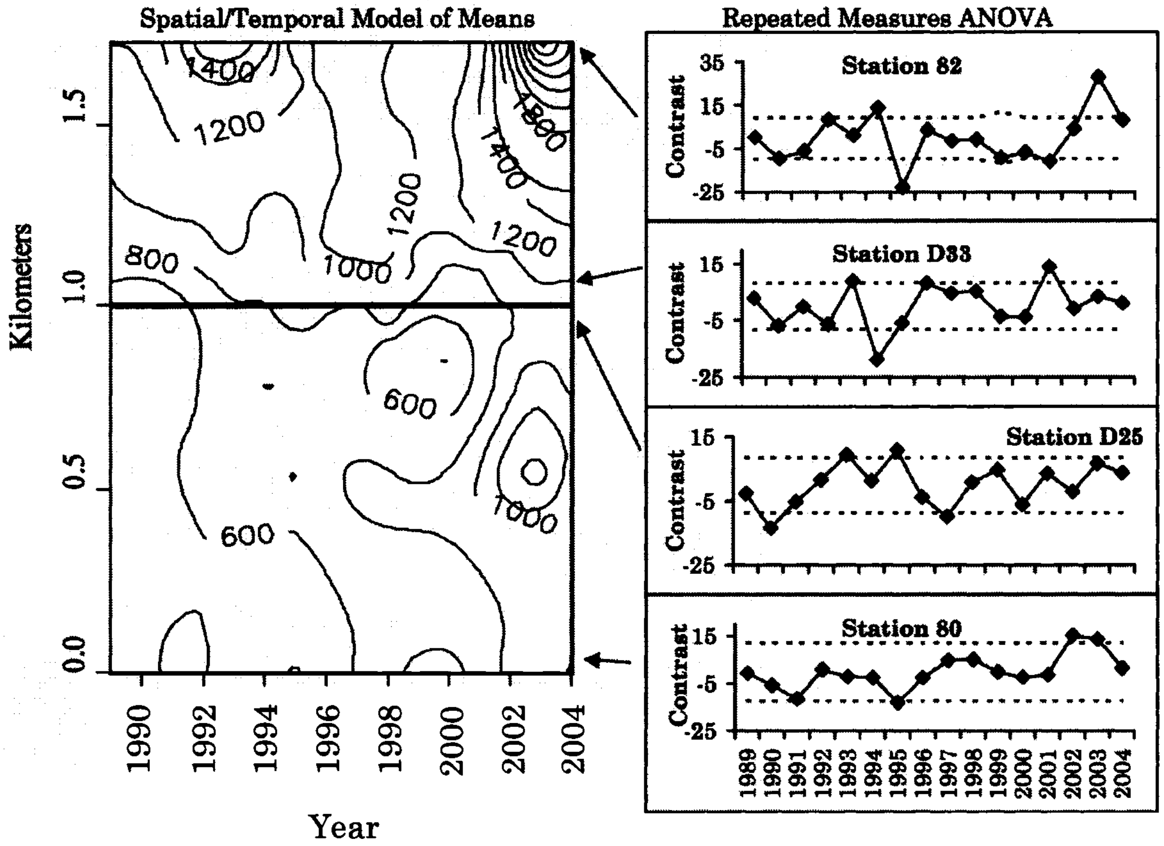

Figure 4.4. Contour plot of predicted values for abundance (individual m^{-2}) and contrasts from repeated measures ANOVA from Port Valdez, 1989-2004. The point of disposal is located at approximately 1 km along the y-axis (west to east transect distances) marked by the black line. Contrast values plotted are deviations of means from the grand mean and represent the multiple comparisons for the rm ANOVA. Dashed lines in the contrast plots are 95% confidence intervals and values outside the confidence limits provide evidence for a significant difference.

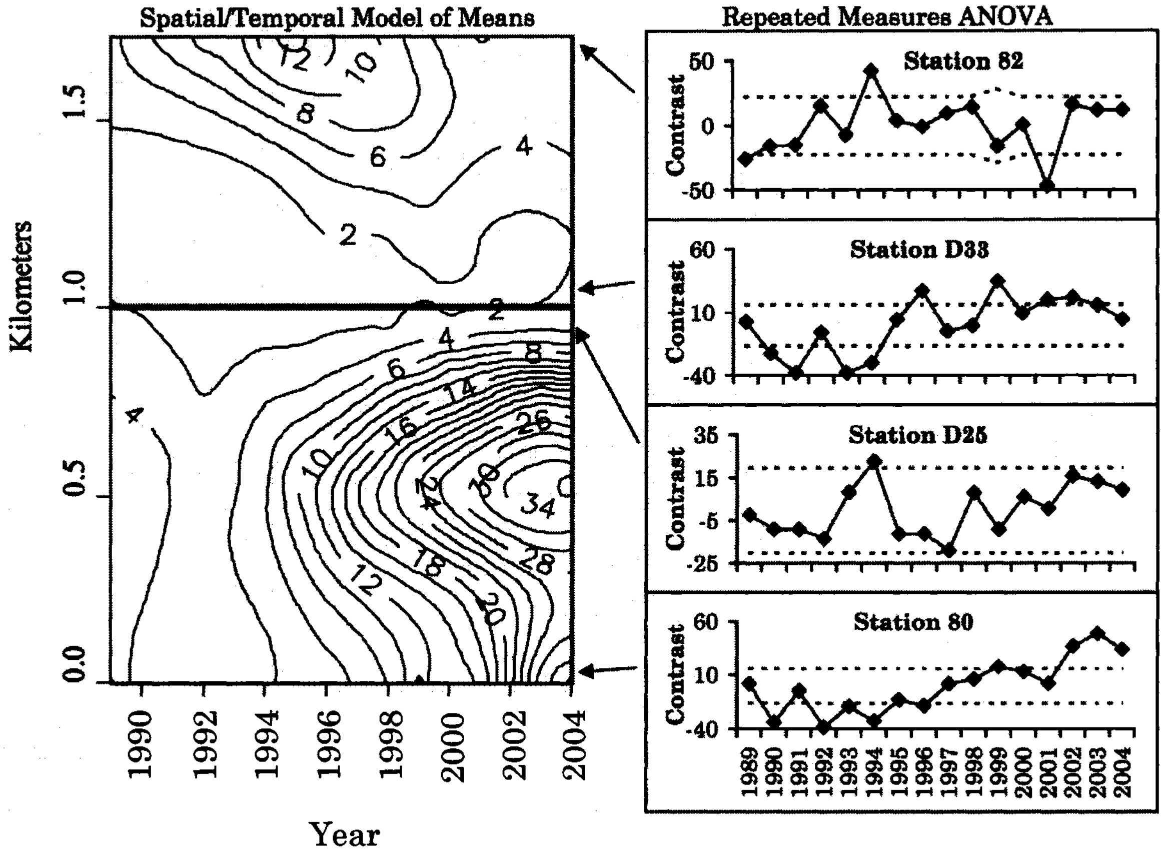

Figure 4.5. Contour plot of predicted values for percent sensitive species and contrasts from rm ANOVA from Port Valdez, 1989-2004. The point of disposal is located at approximately 1 km along the y-axis (west to east transect distances) marked by the black line. Contrast values plotted are deviations of means from the grand mean and represent the multiple comparisons for the rm ANOVA. Dashed lines in the contrast plots are 95% confidence intervals and values outside the confidence limits provide evidence for a significant difference.

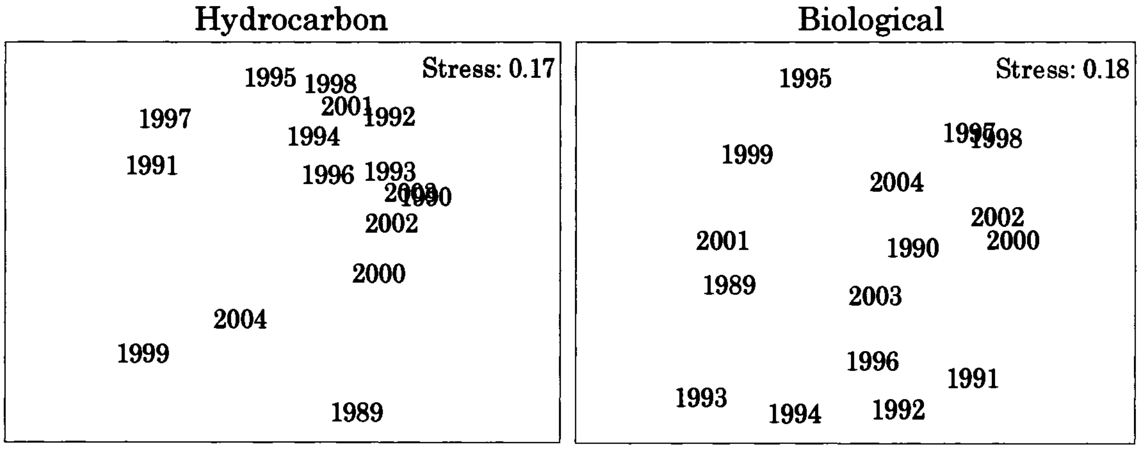

Figure 4.6. Second stage nonmetric multidimensional scaling ordinations for Port Valdez, 1989-2004. Each year noted in the ordination plots represent a matrix of distance/similarity coefficients for repeatedly sampled locations. Data are *ln*(X)-transformed hydrocarbon concentrations and *ln*(X+1)-transformed infaunal abundance data.

Chapter 5 Temporal Variability of Deep Benthic Communities in a Glacial Fjord Affected by a Large Earthquake and Anthropogenic Stressors, 1971-2003[1]

ABSTRACT: Long-term changes in deep subtidal faunal assemblages were assessed with respect to changes in environmental conditions in an Alaskan glacial fjord. In general, fauna were redistributed within the deep basin and faunal abundance was higher in 2002 as compared to 1971. Most infaunal families were present throughout the study period as indicated by the low β diversity values between 1971 and 2002. In 1990, the benthic community stabilized from the more variable state of earlier years. A magnitude 9.2 earthquake in 1964 caused catastrophic disturbance to benthic organisms and readjustment to this perturbation was a dominant trend with the deep benthos up to 1990. Another source of environmental change is the release of salmon fry from a fish hatchery starting in 1981 and subsequent returns of adult salmon. The large returns of adult salmon, on the order of 10 million fish annually, have likely altered nutrient pathways by importing oceanic nutrients into the system, possibly enhancing growth of benthic organisms and communities. A gradient in the faunal assemblage associated with higher sedimentation at the head of the fjord observed in earlier studies has weakened in later years, possibly reflecting final readjustment from the earthquake or enhancement of biomass from fish remains. Effects from the large earthquake and salmon returns appear to be the sources of change with the greatest potential to influence the benthos in the fjord. Effects on deep benthic communities from the oil terminal were

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negligible in comparison to those from the earthquake and the salmon aquaculture industry.

Keywords: Alaska, Benthos, Marine Ecology, Port Valdez, Long-Term Monitoring.

1. Introduction

Assessment of disturbance in marine communities is complicated by multiple stressors acting and interacting over small and large spatial and temporal scales (Blanchard and Feder, 2003; Hewitt et al., 2005). Preferably, undisturbed community characteristics and natural sources and directions of change should be identified (Nichols, 1985). This is increasingly difficult, however, due to global resource use and spread of contaminants. Assumptions that a region has not been disturbed may be misguided as all marine communities are adjusting to long-term change in natural and anthropogenic stressors (e.g., global climatic change, increasing urbanization, and increasing resource use) (Steneck and Carlton, 2001; Dojiri et al., 2003; Hewitt et al., 2005). Organisms are influenced by spatial or temporal gradients forcing change over much larger landscapes than the local scale at which sampling usually takes place, and often sampling cannot include the full range of spatial and temporal gradients (Armonies, 2000).

Investigations of infaunal communities over long periods of time are necessary to understand influences from natural and anthropogenic stressors. Variations in pelagic production and zooplankton communities can have demonstrable effects on benthic communities by altering the food resources reaching the bottom (Pearson and Barnett, 1987; Austen et al., 1991; Josefson et al., 1993; Pearson and Mannvik, 1998). Temporal trends following anthropogenic disturbance of benthic communities demonstrate that human activities can have large, long-lasting effects on soft-bottom communities (Nichols, 1985; Pearson et al., 1986; Burd, 2000; 2002; Perus and Bonsdorff, 2004). Gross organic enrichment and eutrophication from carbon loading in coastal areas, including effects from aquaculture, can lead to substantial alterations of marine communities (Josefson and Rosenberg, 1988; Maurer et al., 1993; Bakan and Büyükgüngör, 2000; Schiff et al., 2000; Conlan et al., 2004; Perus and Bonsdorff, 2004; Modica et al., 2006). Benthic communities may require many years to readjust when stressors are removed (Burd, 2000).

Depth, sediment grain-size, sedimentation rates, sediment stability, and sediment organic matter content are important influences for subtidal benthic communities in glacial fjords (Okey, 1997; Holte and Gulliksen, 1998; Wlodarska-Kowalczuk and Pearson, 2004). Fine sediment fractions are contained in runoff from melting glaciers in summer that accumulate on mudflats and in deep basins. The great volumes of sediments entering glacial fjords in summer result in high sedimentation rates at the head of fjords and increased sediment instability (Sharma and Burbank, 1973; Naidu and Klein, 1988). Factors influencing particulate organic carbon accumulations in sediments include consumption of a large proportion of primary production in the upper water column by pelagic grazers and dilution and burial of organic particles by the high volume of inorganic sediments (Sargent et al., 1983; Görlich et al., 1987; Cooney, 1993; Holte et al., 1996; Holte, 1998). As a result, food resources are commonly low in sediments of fjord basins and faunal assemblages are often impoverished (Wlodarska-Kowalczuk and Pearson, 2004). Additionally, sills restrict circulation in fjords sometimes resulting in hypoxic conditions and anoxic sediments (Jewett et al., 1996). With respect to environmental variability, deep basins of fjords are less variable environments with low hydrodynamic energy (Wlodarska-Kowalczuk and Pearson, 2004). Natural disturbance to deep benthic communities in fjords, including earthquakes and sediment slumping, are infrequent but can have large effects on the physical structure of the basin and on the biota (Coulter and Migliaccio, 1971; Sharma and Burbank, 1973; Feder and Jewett, 1988; Okey, 1997).

A 30-year investigation of deep subtidal infauna in an Alaskan glacial fjord provides the opportunity to examine the temporal variability of benthic organisms. A large earthquake in 1964 and increasing anthropogenic stress influenced the fjord during the study period (1971-2003) (Feder and Jewett, 1988; McRoy, 1988; Wiegers et al., 1998). The objective of this retrospective study is to examine trends in deep subtidal infaunal assemblages in the fjord from 1971 to 2003 to gain insights into temporal variability in the infaunal community of the deep basin and the influence of disturbance and environmental stressors.

2. Study area

Sampling occurred in Port Valdez, a glacial outwash fjord in the northeastern corner of Prince William Sound (Hood et al., 1973; Colonell, 1980; Feder and Jewett, 1988; Shaw and Hameedi, 1988; Blanchard et al., 2002) (Figure 5.1). The deep basin of the fjord has a flat bottom varying between 230-250 m depth with limited water exchange due to the presence of two sills (sills are located at about 120 and 200 m depth). Melt-waters from glaciers in the surrounding mountains carry silt and clay fractions to the fjord and these fine fractions dominate the sediments. The sedimentation rate at the eastern end of the fjord, near the head, is estimated to be 13.5 cm yr^{-1} but is < 1 cm yr^{-1} at the western end (Naidu and Klein, 1988). Sediment from rivers results in outwash deltas on the sides and extensive mudflats at the head of the fjord. Sediments of the deep basin have low organic carbon and infaunal communities demonstrate relatively low numbers overall, suggesting reduced food resources (Feder et al., 1973; Feder and Matheke, 1980; Feder and Jewett, 1988; Blanchard et al., 2002).

Large changes in environmental conditions in the study area resulted from a large earthquake in March, 1964. The magnitude 9.2 earthquake caused catastrophic disturbance to all marine communities in Prince William Sound (where the epicenter was located), throughout the Gulf of Alaska, and beyond (Haven, 1971; Hubbard, 1971; Stanley, 1968). Port Valdez was impacted by catastrophic sediment slumping and scouring by tsunamis (Coulter and Migliaccio, 1971; McRoy, 1988; Sharma and Burbank, 1973; Naidu and Klein, 1988; see also Okey, 1997 concerning effects of sediment slumping). Slumped sediments were deposited throughout the fjord basin, directly impacted benthic organisms through burial and resuspension with disturbed sediments.

Anthropogenic sources of environmental change in the study area have increased greatly over the 30-year period. A major marine oil terminal became operational in 1977 and construction activities may have been a substantial disturbance to the fjord (Feder and Jewett, 1988). Known environmental stress from the oil terminal in the fjord is currently limited to a small area and is related to the disposal of treated ballast water effluents but due to the risk of large oil spills in the fjord, the potential exists for a large disturbance (Wiegers et al., 1998; Blanchard et al., 2002, 2003; Chapter 4). A fish hatchery is located on the south side of the fjord and commercial operations begun in 1982 with the first return of adult salmon in

1983. Over 200 million salmon fry are now raised and released every year in the fjord (White, 2005). Information on annual returns of natural salmon to the fjord are poor but Merrell (1988) suggests that escapement pink salmon was about 70,000 (the dominant species) in 1985, whereas returns of hatchery fish are now ranging beyond 10 million fish (Merrell, 1988; White, 2005). Thus, the release of millions of salmon fry and subsequent returns of adult fish in the fjord represent a very large environmental change potentially with ecosystem-wide effects from fry foraging for food and from nutrients recycled into the system from returning adults (Kline et al., 1990, 1993). Aquaculture has led to very large effects on coastal ecosystems elsewhere (Modica et al., 2006).

3. Methods

3.1 Sampling methods

Subtidal fauna were first sampled in Port Valdez in 1971, seven years after the earthquake. Sampling locations in this manuscript are organized as stations comprising fjord-wide sampling (1971 and 2002) and those comprising a repeatedly sampled transect (1971 to 2003). Three replicate samples were collected from locations throughout the fjord in September, 1971 and these locations were resampled in the late summer (Late August/Early September) of 2002 with one sample collected at each site (Fig. 5.1). The 32 sites sampled in both 1971 and 2002 provide a temporal comparison of faunal communities. Six stations sampled over the study period comprise a longitudinal transect down the center of the fjord with three replicate samples collected at sites on the transect in September 1971 and five from late summer in 1976-1977, 1980 to 1982, 1985, 1987 and 1989 to 2003. Sediment samples were collected with a van Veen grab (0.1 m^2), washed over a 1.0 mm screen, and preserved in 10% formalin. Invertebrates were identified to the lowest taxonomic level practical, counted, and weighed. Data are aggregated to family level in this study to account for irreconcilable taxonomic differences over the study period. Most families contain one or two dominant species and these are noted when applicable. Faunal measures calculated include average abundance (individuals m^{-2}), wet weight biomass ($g\ m^{-2}$), and number of taxa (family or higher) for each location and year but biomass estimates were not available for the 2002 single replicate collections as animals were not weighed due to financial constraints.

3.2 Statistical methods

Multivariate procedures were used to elucidate community-level changes over time. Bray-Curtis similarity matrices (Bray and Curtis, 1957) were calculated for $ln(X+1)$ transformed abundance data, to which cluster analysis was applied. Abundance data from 1971 and the resampled grid in 2002 were analyzed to compare trends over the 30 year period. Abundance data were also averaged for each year for each site on the longitudinal transect from 1971 to 2003 to determine differences among years. Abundance rankings from the SIMPER routine of PRIMER were used to determine taxon differences among groups. Multivariate analyses were performed using PRIMER software package (Clarke and Gorley, 2001).

β diversity was calculated between 1971 and 2002 based on the two fjord-wide sampling events and for the transect of stations for each year. Whittaker's β diversity was calculated as $\beta_w = (S/(\Sigma S_s/n))-1$ where S = total number of species within a study area, S_s = number of species in each sample, and $\Sigma S_s/n$ = the average number of species in the samples

(Magurran, 2004). The value of β_w approaches 0 as $\Sigma S_s/n$ approaches S so values near 0 indicate similar numbers of species within samples. Values for β_w near 1 indicate higher turnover. If only two locations are compared, a value for $\beta_w = 0$ indicates the same number of taxa occur at both sites and a value of 1 indicates there are no taxa in common.

Geostatistical modeling was used to analyze the benthic data from the longitudinal transect for trends in abundance and number of taxa over the study period. The statistical program R (www.r-project.org) and the spatial tools in the library GeoR (Ribeiro and Diggle, 2001; http://cran.r-project.org/) were used. The spherical model best described the spatial covariance in all measures in this study (Cressie, 1993). Parameters defining the shape of the spherical variogram model are the nugget, which is microscale variation and measurement error; the range, indicating the distance at which data are no longer correlated; and the upper limit of the model called the sill, which represents total variance (Cressie, 1993). Empirical (semi)variograms were determined from the data as:

$$\hat{\gamma}(\mathbf{h}) = 1/(2\,|\,N(\mathbf{h})\,|\,)\Sigma[Z(\mathbf{x}) - Z(\mathbf{x}+\mathbf{h})]^2 \qquad 5.1$$

where $\hat{\gamma}(h)$ is the observed semivariance, **x** and **x+h** are data vectors separated by the distance vector **h,** Z(**x**) is a random process, and N(**h**) is the number of distance pairs (Cressie, 1993; Ricotta & Avena, 1999). The spherical model fitted to the empirical variograms is:

$$\gamma(\mathbf{h};\theta) = \begin{cases} 0, & \text{for } \mathbf{h} = 0 \\ c_o + c_s\{3\,\|\,\mathbf{h}\,\|\,/2a_s - 1/2(\|\,\mathbf{h}\,\|\,/a_s)3\}, & \text{for } 0 < \|\mathbf{h}\| \leq a \\ c_o + c_s & \text{for } \|\mathbf{h}\| \geq a, \end{cases} \qquad 5.2$$

where **h** = vector of distances, $\theta = (c_o, c_s, a_s)'$, where $c_o \geq 0$ is the nugget, $c_o + c_s \geq 0$ is the sill, and $a_s \geq 0$ is the range (Cressie, 1993). Abundance values were *ln*(X+1)-transformed and biomass *ln*(X)-transformed to meet the assumption of normal errors and data were detrended by modeling trends as a linear function of distance and year. Contour plots of back-transformed values are presented.

Correlations were determined between biological indices and measures of anthropogenic stressors. Faunal measures used include average abundance and biomass, total number of taxa per year, and β diversity. Measures of anthropogenic stress include average total aromatics (TARO: 1989-2003), salmon fry releases (1982-2003), and adult salmon returns (1983-2003). The data for TARO were from the long-term monitoring study in Port Valdez and data on salmon fry releases and adult returns were from annual reports to the State of Alaska (Blanchard et al., 2002; Alaska salmon enhancement program annual reports available through Alaska Dept. Fish and Game, Juneau, AK; White, 2005). Pearson's coefficient was used to determine correlations between biotic variables and the measures of anthropogenic stress although Spearman's rank correlation coefficient was applied for a few comparisons due to a lack of normality. No measures of stress related to effects from the earthquake were available.

4. Results

4.1 Comparison between 1971 and 2002.

A total of 11 phyla encompassing a total of 75 families or higher taxonomic categories were sampled in both years with 63 recognized in 1971 and 65 in 2002. Polychaetes and bivalves were dominant among the deep subtidal benthos comprising 67% and 20% of the total numbers of individuals, respectively, among the fjord-wide locations for 1971 and 2002 combined. Average abundance values ranged from 13 to 1,093 individuals

m^{-2} with an average of 323 individuals m^{-2} in 1971, and 230 to 8,620 individuals m^{-2} with an average of 749 individuals m^{-2} in 2002. β diversity between 1971 and 2002 (based on family or higher taxonomic categories from a reconciled taxon list) was low with a value of 0.2.

Multivariate analysis of the 1971 and 2002 abundance data suggests station groupings by depth and distance from the head of the fjord (Fig. 5.2). Cluster analysis and MDS ordination resulted in six groups, Group I: two 1971 nearshore stations near the head of the fjord, Group II: a mix of nearshore (some deep) and shallow stations from 1971 and 2002, Group III eastern (fjord head to mid distance) basin stations 1971, Group IV: western (mid-distance to fjord mouth) basin stations 1971, and Group V: eastern (fjord-head), nearshore stations 2002, and Group VI: basin stations 2002 with no east to west division (Fig. 5.2). Six nearshore stations from 1971 and 2002 did not join one of the station groups. Group I, sites near the head of the fjord, was characterized by very low abundance values for most organisms and is dominated by polychaete worms of the family Capitellidae (mostly *Heteromastus filiformis* but also *Barantolla americana*, *Capitella capitata*, and *Mediomastus* sp.), Lumbrineridae, and Nephtyidae (*Nephtys punctata*) (Table 5.1). Fauna numerically dominant in Group II in 1971 and 2002, sites near the margins of the fjord, include the polychaete families Ampharetidae, Nephtyidae, and Oweniidae (largely *Galathowenia oculata* but absent in 1971). The 1971 eastern stations in Group III demonstrated an absence of the polychaete family Oweniidae, low abundances of Maldanidae, Sternaspidae (*Sternaspis scutata*), and the bivalves Thyasiridae (mostly *Adontorhina cyclia* and *Axinopsida serricata*), and higher abundance values for Capitellidae (Table 5.1). Characteristics of the 1971 western stations in Group IV include the absence of Oweniidae, low numbers of

Maldanidae, Sternaspidae, and Thyasiridae, and higher abundances of the cumacean family Leuconidae (mostly *Eudorella emarginata*). The stations near the head of the fjord from 2002 in Group V demonstrated high numbers of the polychaetes Ampharetidae (*Amphicteis scaphobranchiata* and *Melinna cristata*) and the bivalves Thyasiridae. Deep stations from 2002 in Group VI were characterized by relatively high numbers of the polychaetes Capitellidae, Cirratulidae, and Paraonidae.

Some changes in species distributions were readily apparent between 1971 and 2002. For Groups III and IV, basin stations in 1971, no Oweniidae are found, there were few Maldanidae, Paraonidae, Sternaspidae, and Trichobranchidae (*Terebellides stroemi*), but higher abundances of Leuconidae and the brittle star family Ophiuridae (*Ophiura sarsi*) were found in the western end of the deep basin (Fig. 5.3). In 2002, the polychaete families Oweniidae, Maldanidae, Sternaspidae, and Trichobranchidae were ubiquitous whereas the brittle star family Ophiuridae was absent – only the ophiuroid family Amphiuridae (*Diamphiodia craterodmeta*) was present at a few nearshore sites. The bivalve family Thyasiridae was much more abundant in the nearshore areas in 2002 than in 1971. Overall, the plots show increased abundances from 1971 to 2002.

4.2 Trends in the longitudinal transect, 1971 to 2003

A total of 11 phyla encompassing 77 families were identified from the deep transect of stations from 1971 to 2003. Polychaetes and bivalves comprised 63% and 21% of the total numbers of individuals, respectively, for all years combined. The highest abundance value is for 1987 with an average of 633 individuals m^{-2} while the highest biomass value was for 1980 with 31.33 g m^{-2} and the greatest number of taxa across the transect occurred in 1997 with an average of 30 taxon categories (Fig. 5.4). The lowest abundance value was for 1980 with 188 individuals m^{-2}, lowest biomass was in 1985 with 7.02 g m^{-2}, and the lowest number of taxa was found in 1971 with an average of 16 taxa identified. β diversity for each year ranged from about 0.4 (1987) indicating similar numbers of taxa to 1.2 (1981) indicating large differences over the transect. Looking at individual years, β diversity is more variable prior to 1989 with a coefficient of variation of 28% (mean = 0.85, SD = 0.24) as compared to a coefficient of variation of 16% from 1989 to 2003 (mean = 0.70, SD = 0.11). When calculated for all years together, β diversity was 1.1 indicating large differences in the number of taxa over time.

Contour plots of predicted values from geostatistical modeling for faunal measures show long-term increases in abundance, biomass, and the number of taxa. Contour plots of predicted values for abundance, demonstrate initially high values at the station closest to the head (east end) of the fjord (distance = 0 km on the y-axis) in 1971-1978, which decrease towards the mouth of the fjord (distance = 14 km on the Y-axis) (Fig. 5.5). Abundance values were low through 1985 but increased into the 1990s. Values for biomass were particularly low in the period 1983-1990 in most of the plot but tended to be high at the mouth of the fjord and overall, from 1994-2003. The number of taxa was higher in general at stations farther from the head of the fjord and higher overall in the period 1990-2003.

Cluster analysis of average abundance values for each year showed three groupings. The cluster dendrogram supports the groups 1976-1977, 1981-1989 and 1990-2002 (Fig. 5.6). The years 1971, 1980 and 2003 did not join a group. Differences of the dominant families over time include the absence of polychaete worms of the family Oweniidae through 1980, the absence of Maldanid worms from the basin transect in 1971, and lower abundances of Thyasirid bivalves compared to later years (Table 5.2). Capitellid polychaetes were high in 1976-1977 and Cirratulids in 2003.

4.3 Correlation Testing

Correlations analysis suggest some relationships between infauna and anthropogenic stress. Variables used for correlation analysis include four biological variables (average abundance, biomass, and number of taxa and β diversity) and three environmental measures of anthropogenic stress (salmon fry releases, adult salmon returns, and total aromatic hydrocarbon concentrations in sediments (TARO)). The highest correlations observed were between salmon fry releases and adult salmon returns to infaunal biomass ($r = 0.467$ and 0.341, respectively) (Table 5.4). A small correlation ($r = 0.266$) was observed between releases and the number of taxa. Correlations between TARO and biotic indices were mostly negligible (from $\rho = -0.102$ to 0.072) although a small correlation was observed between TARO and β diversity ($\rho = 0.227$).

5. Discussion

Rarely does the opportunity present itself to assess influences from large natural or anthropogenic perturbations on marine benthic communities. Large disturbances are infrequent and investigations of communities in the deep benthos following large perturbations can be inconclusive due to a lack of temporal collections (Kingston et al., 1995; Feder and Blanchard, 1998; Lay et al., 2005). Thus, there is a need to understand how marine communities may vary over long periods of time following large disturbances and the biological studies in Port Valdez Alaska provide such an opportunity.

Faunal readjustment from the magnitude 9.2 earthquake in Prince William Sound March, 1964, was likely a key process over the study period. Evidence suggests that the magnitude of the disruption was so large as to have impacted every marine community in Prince William Sound by topographic change, scouring by tsunamis, or sediment slumping (Coulter and Migliaccio, 1971; NRC, 1971). In Port Valdez, the deep benthos was greatly disrupted by mass slumping and deposition of sediments over the bottom (Coulter and Migliaccio, 1971; Sharma and Burbank, 1973; Feder and Jewett, 1988; Naidu and Klein, 1988). Long-term investigation of a benthic community in a fjord from British Columbia demonstrated a lengthy readjustment period following the cessation of anthropogenic disturbance (> 20 years) so a lengthy readjustment from a large earthquake in Port Valdez would be expected as well (Burd, 2002). Stratification of the water column and restriction of deep water movement by the sills in Port Valdez possibly contributed to the slow readjustment by hindering transport and recruitment of benthic organisms to the deep basin (Muench and Nebert, 1973; Colonell, 1980).

Long-term trends in faunal distributions and summary measures for the study area demonstrate a readjusting marine ecosystem. This state of variability is demonstrated by the change in multivariate groupings for basin stations in the fjord-wide samples, changes in the distributions of taxa from 1971 to 2002, and the geostatistical modeling of faunal measures over the study period. In spite of the large time span and substantial shifts in the distributions of some taxa, it is remarkable that the turnover diversity, β diversity, is low between 1971 and 2002 for the 32 sites sampled throughout the fjord. The value of 0.2 for β diversity suggests that most families found in the fjord in later years were already present by 1971. Changes in diversity for the longitudinal transect give a slightly different picture as overall, β diversity was high when comparing all years together and β diversity was more variable prior to 1989 as compared to later years. Thus, although most fauna found in 2002 were present in the fjord by 1971, shown by the low β diversity, a stable community (in terms of a diversity gradient) had not been reached along the deep basin transect until 1990 when variability of β diversity decreased. This is also reflected in the multivariate analyses where year groupings reflect introductions and redistributions of organisms on the deep transect. Overall, it appears the abundance increased as fauna redistributed themselves into their current habitats.

Sediment-related factors are important influences for benthic communities in glacial fjords. The large quantities of fine glacial sediments dilute particulate organic matter in the water column reducing food availability (particulate organic carbon, POC), can form unstable sediments when settled, and are unsuitable for many infauna (Görlich et al., 1987; Holte et al., 1996; Okey, 1997; Holte, 1998; Holte and Gulliksen, 1998; Wlodarska-Kowalczuk and Pearson, 2004). A depauparate benthos resulting from poor food supply is characteristic of glacially-influenced fjords. The apparent east-west separation of benthic fauna and gradient in sediment factors noted for Port Valdez in earlier studies are consistent with expectations for glacially-influenced fjords (Feder and Matheke, 1980; Feder and Jewett, 1988; Naidu and Klein, 1988; Holte and Gulliksen, 1998; Wlodarska-Kowalczuk and Pearson, 2004). A gradient in fauna does exist along the fjord in later years, as suggested by the moderately high β diversity values for 1990-2003, but apparently, it was too small in 2002 for the multivariate methods to detect as the cluster analysis showed no difference among deep basin stations. The reduction of magnitude of this gradient likely reflects the final adjustment of communities from the earthquake and possibly, new food resources from carcasses of returning hatchery salmon (see below). Infaunal assemblages on the slope up to the head of the fjord or close to the shoreline remain distinct from the assemblage in the bottom of the basin, reflecting shallower depths, proximity to the mouths of the glacial rivers and streams, different food supplies, and associated covariates (Feder et al., 1973; Feder and Matheke, 1980; Naidu and Klein, 1988; Feder and Jewett, 1988).

The marine ecosystem of Prince William Sound, Alaska is pelagic. Most phytoplankton in the water column are grazed by zooplankton in pelagic systems and only a small amount of utilizable organic carbon reaches the benthos (Cooney, 1993). Port Valdez is likewise a pelagic system, as are some glacial fjords elsewhere (Sargent et al., 1983; Cooney, 1993; Cooney and Coyle, 1988). The pelagic nature of the system and dilution by glacial sediments results in a carbon-poor benthic system (Feder and Jewett, 1988). Thus, carbon sources from the salmon hatchery in the fjord have the potential to release the benthos from food-deprivation. Effects of raising salmon fry include gross organic enrichment under rearing pens, often leading to depletion of oxygen in sediments (Weston, 1990; Gao et al., 2005; Islam, 2005). Feeding of the large number of released salmon fry may have a substantial impact on the overall balance of the ecosystem by feeding on zooplankton (allowing more ungrazed phytoplankton to reach the bottom). The largest effects for fauna in the deep basin, however, may be from returning adult salmon, ranging from 5 to 16 million salmon over the last 10 years. Adult salmon affect the nutrient-deprived deep benthos primarily through deposition on the bottom of dead salmon carcasses, remains of fish caught recreationally, and wastes from local fish processing plants with local to ecosystem-level effects (Kline et al., 1990, 1993; Blanchard and Feder, 2003; Ray, 2005). Salmon carcasses provide important contributions to freshwater nutrient pools connecting oceanic, estuarine, and freshwater systems (Kline et al., 1990, 1993; Islam, 2005; Ray, 2005). Thus, fry releases and returns of adult salmon may have altered nutrient pathways in Port Valdez by providing increased volumes of nutrients along paths that were minor prior to hatchery operations. Correlative evidence of moderate relationships between fry releases, adult returns and benthic biomass suggests that this source of change may be having measurable effects.

The construction and operation of the marine oil terminal on the south side of Port Valdez ushered in a period of development and growth at the head of the fjord impacting a number of marine habitats (Feder and Jewett, 1988; McRoy, 1988). Direct effects from the oil terminal include maintenance activities, small oil spills, and disposal of treated ballast water (Feder and Jewett, 1988; Blanchard et al., 2002, 2003; Chapter 4). Effects from the disposal of treated ballast water appear to be limited to the shelf where the marine oil terminal is located (Feder and Jewett, 1988; Blanchard et al., 2002, 2003; Chapter 4). Concentrations of aromatic hydrocarbons have generally been low in sediments of Port Valdez and have declined due to improvements in the ballast water treatment facility and declining oil shipments (Blanchard et al., 2002). Most documented variations associated with terminal operations have been localized consisting of changes in the distributions of intertidal barnacles at the terminal following a small oil spill and adjustment of benthic fauna to treated ballast water effluents on the shelf at the terminal (Rucker, 1983; Payne et al., 2001; Blanchard et al., 2002, 2003). Effects on the benthos related to the disposal of treated ballast water effluents appear to be decreasing in magnitude as the volume of ballast water treated declines (Chapter 4). As shown here, there is weak to no correlative evidence for effects from terminal operations on the deep benthos (Blanchard et al., 2002).

In summary, temporal trends in benthic communities of this glacial fjord suggest a lengthy readjustment to large natural and anthropogenic stresses. Overall, abundance increased from 1971 to 2002 with most fauna found in 2002 present in the fjord in 1971 but redistributed in later years. Multivariate analysis and geostatistical modeling provide evidence that infaunal assemblages were more variable to 1990 after which, the benthic assemblage appears to stabilize. Both natural and anthropogenic sources for environmental change may explain for increased variability in Port Valdez. Catastrophic disturbance from a large earthquake in 1964 had a large influence on biotic communities in the fjord, and was a dominant factor associated with faunal differences over time. Returns of adult salmon from a local hatchery possibly contribute to temporal variability within the fjord by importing oceanic carbon and nutrients to the fjord altering nutrient pathways and enhancing benthic biomass. Effects on deep benthic communities from the oil terminal appear to be negligible in comparison to those from the earthquake and salmon aquaculture industry.

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Table 5.1. Abundance of dominant taxa in cluster groupings, 1971 and 2002. Presented are the average abundance (individuals m^{-2}) values of numerically dominant taxa within station groups from the cluster analysis of the 1971 and 2002 abundance data.

Taxa	I	II	III	IV	V	VI
Rhynchocoela	7	7	10	6	1	4
Annelida: Polychaeta						
Ampharetidae	15	145	13	0	154	2
Capitellidae	33	55	71	39	46	61
Cirratulidae	0	29	35	4	27	67
Lumbrineridae	25	57	15	19	21	22
Maldanidae	0	29	0	2	14	25
Nephtyidae	50	78	39	19	78	37
Nereidae	0	31	0	0	1	0
Orbiniidae	0	13	2	2	1	1
Oweniidae	0	162	0	0	81	16
Paraonidae	0	12	9	1	38	43
Spionidae	10	28	31	19	25	18
Sternaspidae	0	56	1	0	6	14
Terebellidae	0	6	1	0	0	1
Trichobranchidae	0	11	0	0	6	11
Mollusca						
Aplacophora						
Chaetodermatidae	8	15	11	3	6	14
Bivalvia						
Cardiidae	3	41	0	1	0	0
Nuculanidae	3	56	1	2	24	5
Tellinidae	0	54	0	0	10	0
Thyasiridae	13	50	14	7	528	13
Arthropoda: Crustacea						
Cumacea						
Leuconidae	0	21	7	33	16	5
Amphipoda	0	8	0	1	7	6
Sipuncula						
Golfingiidae	0	2	0	0	0	0
Echinodermata						
Ophiuridae	0	2	1	6	0	0

Table 5.2. Abundance of dominant taxa for year groupings from 1971-2003. Presented are the average abundance (individuals m^{-2}) values of numerically dominant taxa within year groupings determined from cluster analysis for the longitudinal transect.

Year Groupings	1971	1976-77	1980	1981-89	1990-2002	2003
Rhynchocoela	9	0	1	2	6	6
Annelida: Polychaeta						
Ampharetidae	4	4	1	2	4	6
Capitellidae	47	164	10	23	27	57
Cirratulidae	17	0	1	3	25	104
Goniadidae	3	4	5	11	12	7
Lumbrineridae	16	45	15	13	24	27
Maldanidae	0	18	10	12	34	46
Nephtyidae	23	19	12	13	23	36
Nereidae	0	0	0	0	0	8
Oweniidae	0	0	0	21	18	19
Paraonidae	4	16	1	3	18	53
Spionidae	28	3	0	4	5	35
Sternaspidae	1	1	2	8	19	9
Terebellidae	0	8	1	2	3	1
Trichobranchidae	0	19	18	30	12	3
Mollusca						
Aplacophora						
Chaetodermatidae	6	20	11	9	12	3
Bivalvia						
Nuculanidae	3	10	20	36	15	3
Thyasiridae	6	25	19	70	62	62
Gastropoda						
Cylichnidae	4	1	0	1	1	1
Turridae	4	7	3	13	6	1
Arthropoda: Crustacea						
Cumacea	20	23	3	7	6	11
Leuconidae	9	0	1	2	6	6

Table 5.3. Biological and environmental data used for correlation analysis from Port Valdez, Alaska 1982-2003. Biological variables are average biomass (g m^{-2}), average number of taxa, average abundance (Abun; individuals m^{-2}), and b diversity (β). Environmental variables representing anthropogenic stress are number of salmon fry released (millions), adult salmon returns (millions), and total aromatic hydrocarbons (TARO, ng g^{-1}).

	Biological				Environmental		
Year	Abun	Biomass	Number Taxa	β	Fry Releases	Adult Returns	TARO
1982	353	14.7	26	0.8	8		
1985	211	7.0	17	1.1	54	0.6	
1987	586	7.3	28	0.4	63	5.6	
1989	231	9.6	23	0.8	132	3.5	148.0
1990	272	20.1	27	0.8	126	11.1	215.6
1991	355	22.5	29	0.7	134	6.1	65.6
1992	476	20.9	28	0.5	91	2.2	56.0
1993	4506	18.3	30	0.6	160	1.8	24.8
1994	352	18.5	25	0.7	156	13.3	42.2
1995	387	24.5	27	0.8	208	6.8	42.2
1996	404	15.9	27	0.8	225	7.8	26.0
1997	458	14.4	29	0.7	190	7.1	20.1
1998	336	13.3	26	0.6	197	4.7	21.7
1999	311	20.0	24	0.8	216	14.6	19.9
2000	260	26.2	25	0.7	197	12.4	19.5
2001	429	20.6	27	0.8	204	16.0	22.0
2002	348	21.8	28	0.7	204	5.2	19.3
2003	534	13.6	24	0.5	208	16.4	35.9

Table 5.4. Summary of correlations between environmental variables and benthic measures from Port Valdez, Alaska 1982-2003. Releases = salmon fry releases from the Solomon Gulch fish hatchery, Returns = adult hatchery salmon returns, and TARO = concentrations of total aromatic hydrocarbons.

	Environmental Variables		
Benthic Variables	Releases	Returns	TARO
Biomass	0.467	0.341	-0.102*
Number Taxa	0.266	-0.203*	-0.094*
Abundance	0.053	0.102	0.011*
β Diversity	0.097	-0.100	0.227*

* Nonparametric test applied due to violation of assumptions.

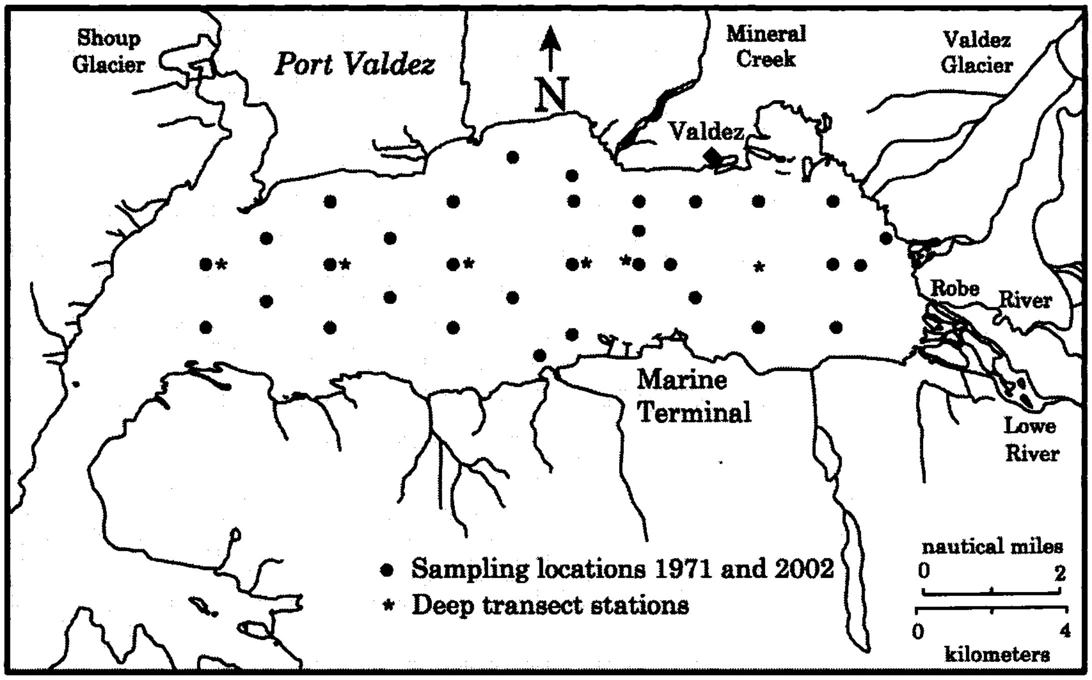

Figure 5.1. Sites sampled for benthic fauna in Port Valdez, Alaska, 1971-2003. Thirty-two locations are included in the fjord-wide sampling in 1971 and 2002. Six deep transect stations were sampled on a longitudinal transect from 1971-2003. An asterisk by a solid circle indicates that the location is included in the transect and basin-wide sampling.

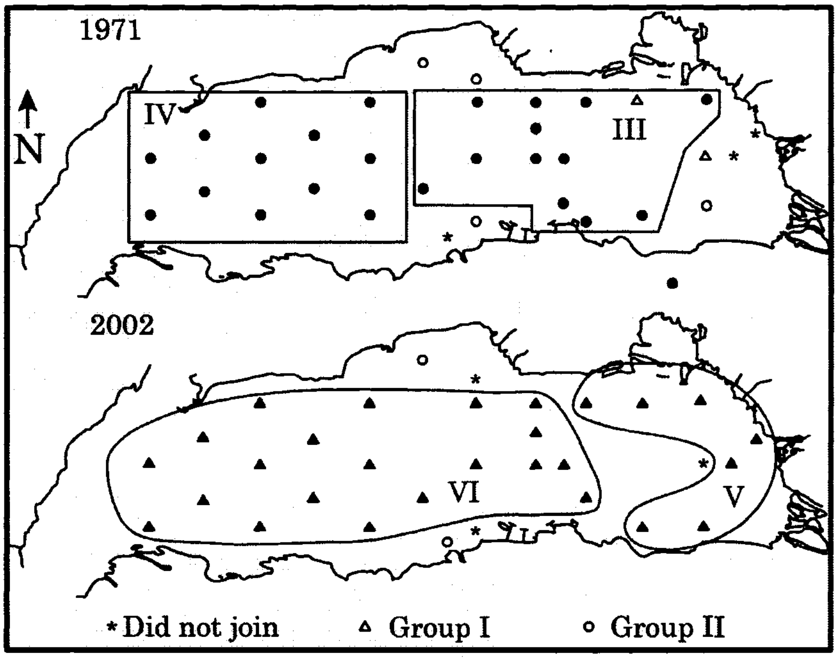

Figure 5.2. Summary of cluster analysis for stations sampled in Port Valdez in 1971 and 2002. Large cluster groupings are enclosed by lines. Small groupings of 6 or fewer stations are given symbols. Abundance data were *ln*(X+1)-transformed. Stations marked with an asterisk did not join a group in the cluster analysis.

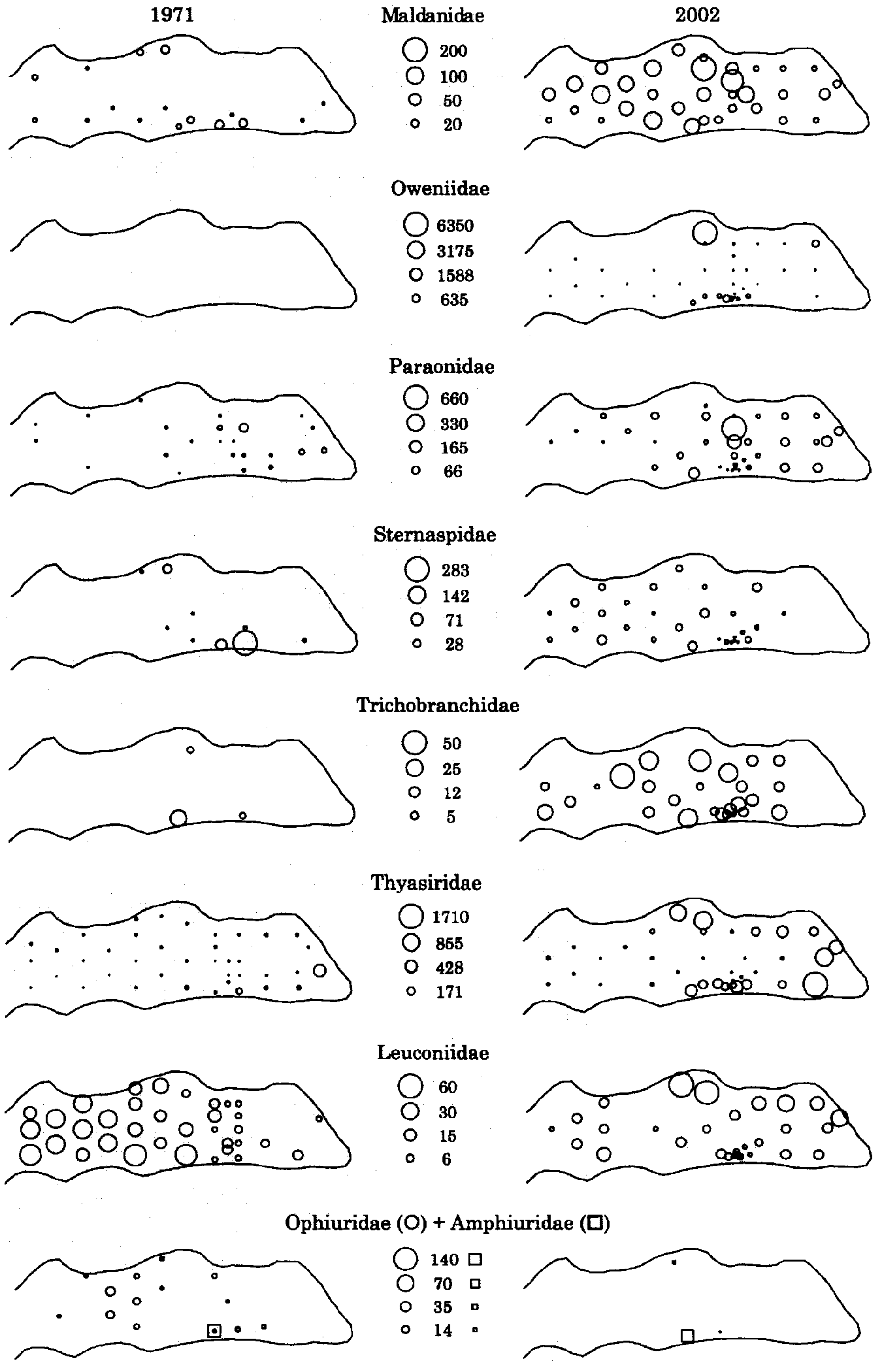

Figure 5.3. Bubble plots of the abundance of select species from Port Valdez, 1971 and 2002.

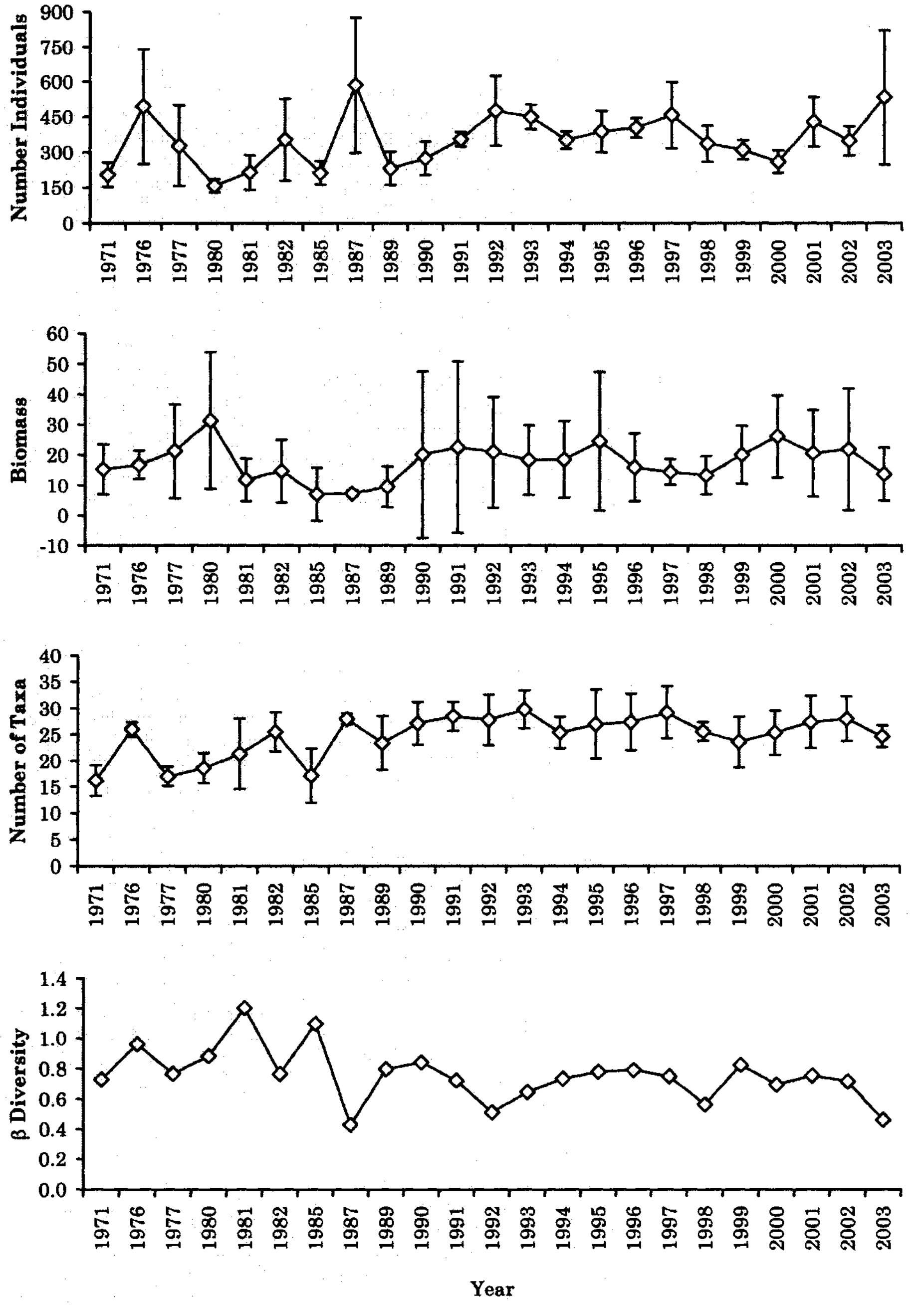

Figure 5.4. Profile plots of biotic measures from Port Valdez, 1971-2003. Data are averages of abundance (individuals m^{-2}), biomass (g m^{-2}), and number of taxa for the transect stations for each year. β diversity is calculated among the transect stations for each year. Error bars are ± 1 standard deviation.

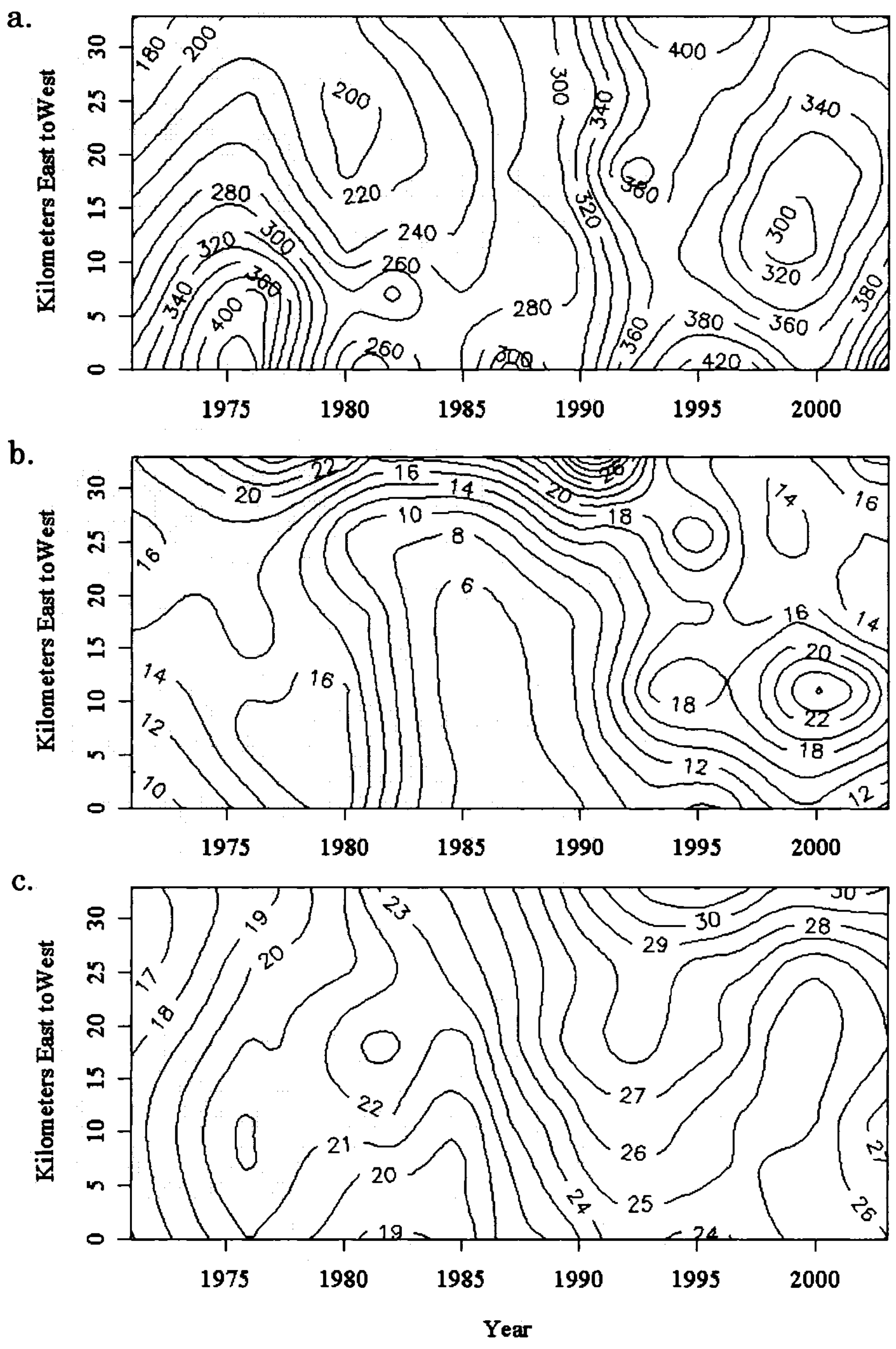

Figure 5.5. Contour plots of predicted biotic measures for transect stations from Port Valdez, 1971-2003. Values for the plots are predictions from spatial modeling for a) average abundance (individuals m^{-2}), b) biomass (g m^{-2}), and c) total number of taxa in the sample.

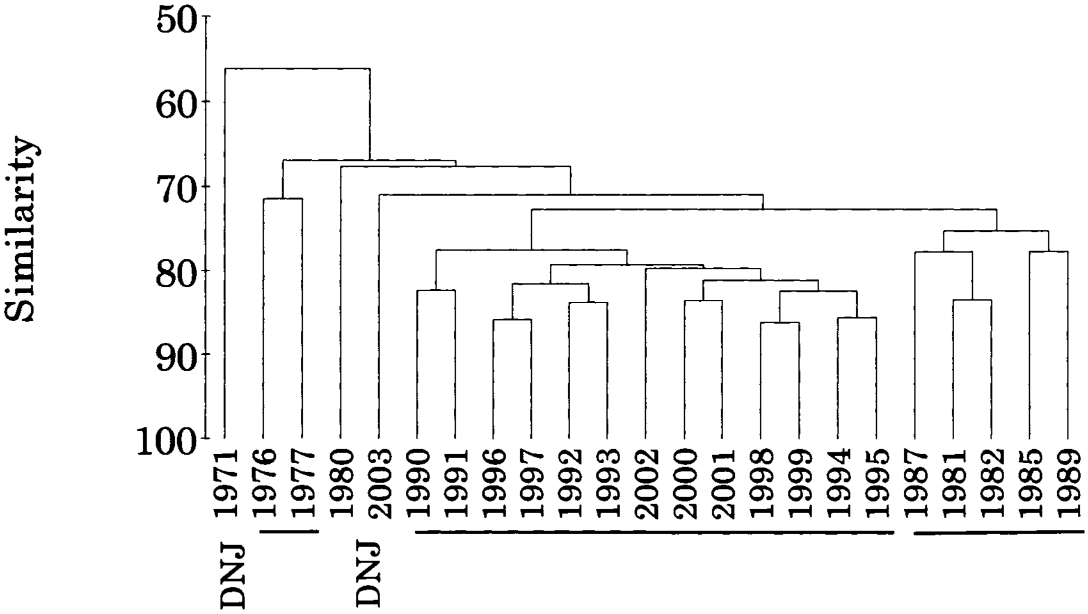

Figure 5.6. Cluster analysis of abundance data from Port Valdez, 1971-2003. Data are *ln*(X+1)-transformed average abundance of infauna (individuals m^{-2}) from the stations along the transect. DNJ indicates that a year did not join a cluster group.

Chapter 6 Conclusions

1. Introduction

Efforts to understand anthropogenic effects within the Port Valdez study area provide an overall model for developing and refining hypotheses that demonstrates how to measure the structure of and detect change from multiple environmental drivers in nearshore and benthic habitats. Anthropogenic drivers of change detected by this study are the oil transportation and salmon aquaculture industries. A natural driver of change was the 1964 earthquake. The methods used to determine responses of benthic communities are independent of the nature of the drivers, requiring only that the drivers are identified. Thus, the approach presented here can be widely applied.

The conceptual model for developing and refining hypotheses of change in marine habitats is simple and adaptive. Monitoring can be viewed as a platform for further research. From that perspective, mining of databases with long temporal records can guide development of new research hypotheses. As questions are developed through the long-term study of ecosystems, hypotheses can be evaluated through retrospective analysis of the accumulated data. Evaluating broad hypotheses leads to refined hypotheses that can then be tested experimentally and provide feedback to further refine the original broad hypothesis. Feedback may result in adapting the research to address managerial or scientific needs. For example, the pelagic/benthic split hypothesis provides a broad hypothesis for understanding processes through which nutrients reach the benthos in PWS (Eslinger et al., 2001). Six salmon hatcheries are operating in Prince William Sound and their adult salmon returns may now be so large as to possibly alter this pattern through carcasses settling on the bottom (http://www.cf.adfg.state.ak.us/; White, 2005; Chapter 5). It is possible that the increased carbon reaching the benthos as fish carcasses may eventually contribute to eutrophication, anoxic conditions, and degradation of benthic infaunal communities in PWS, particularly in carbon-poor, silled-fjords where circulation is restricted (Meunch and Nebert, 1973; Feder and Jewett, 1988; Jewett et al., 1996; Rosenberg and Nilsson, 2005). Thus, based on the small-scale, ecological information provided by retrospective analyses of data from Port Valdez, the larger model for PWS can be refined.

The information provided in this dissertation should increase the accuracy of ecological models and aid scientific endeavors in Prince William Sound and Port Valdez. This dissertation should assist investigators by providing information about long-term trends in flora and fauna in Port Valdez and Prince William Sound (PWS), thereby guiding future research efforts and permitting deviations related to anthropogenic stressors and environmental extremes to be more readily modeled and identified. Demonstration of associations of intertidal assemblages with salinity and habitat structure in Port Valdez are critically important to a wide variety of environmental management activities under federal and state statutes. Until now, effects from fish hatchery salmon fry releases and returns of adult hatchery salmon had not been considered as a potential source of change in the Port Valdez ecosystem. Ecosystem models for PWS can be updated to include the connection between the North Pacific Ocean, PWS, and Port Valdez through salmon returns (Oosterhout, 2005; Ray, 2005).

Three monitoring programs have been established that encompass Prince William Sound and adjacent waters in the Gulf of Alaska. These are the U.S. Environmental Protection Agency's Environmental Monitoring and Assessment Program (EMAP), Exxon Valdez Oil Spill Trustee Council's Gulf of Alaska Ecosystem and Monitoring program (GEM; Mundy 2005), and the

Prince William Sound Ocean Observing System (PWSOOS; http://www.aoos.org/ppt/Schoch%20_PWSOOS.pdf). PWSOOS is part of the Alaska Ocean Observing System, (AOOS; www.aoos.org), a regional implementation of the U.S, Integrated Ocean Observing System, (IOOS; www.ocean.us). This dissertation will contribute rigorous, statistically bounded biological time series to the observing systems and the small-scale, ecological information necessary to provide input and boundary conditions for larger models (Hansen, 1995; Oosterhout, 2005). Such biological benchmarks are essential for differentiating the effects of anthropogenic forcing factors from natural environmental forcing provided by oceanographic and atmospheric factors. Most importantly, the information in this dissertation provides a basis for developing and refining hypotheses that explain how benthic and nearshore ecosystems are structured and how they change. These data can also be used to define Essential Fish Habitat (EFH) required for fishery management under the Magnuson-Stevens Act, to advise regulatory activities under the federal Coastal Zone Management Act, and to define Critical Habitat (CH) under the federal Endangered Species Act.

2. Directions for Future Research

Research hypotheses generated by this investigation include the following.

- Direct effects of salinity tolerance on marine grazers and predators indirectly effects biological interactions and intertidal community structure on rocky shores in glacial fjords. The mechanisms and magnitudes of the effects need to be clarified.
- Habitat structure of rocky shores within glacial fjords influences intertidal assemblages through disturbance-related factors. The

importance of chronic, small disturbances by rock movement on intertidal community organization should be determined.

- Food availability has a large influence on the distribution of benthic fauna in glacial fjords. The influence of food availability versus sediment grain-size preferences of infauna needs to be examined.
- Carbon imported by returning hatchery salmon is influencing the benthic ecosystem. The magnitude of the change brought by adult returning salmon appears to be large but needs further investigation.
- Multivariate tools widely accepted by marine ecologists do not accurately represent small, subtle differences among species assemblages that may be important. Other situations where these techniques fail should be identified.
- The space-time modeling and test of Case 1, a common variogram model, vs. Case 2, separate variogram models, and the likelihood ratio test should be extended to additional environments.

Other areas of research suggested by this study include:

- Various study designs for long-term monitoring of effects of contaminants on marine biota should be evaluated.
- The effect-sizes observed here can be incorporated into decision criteria for long-term environmental assessments.

3. Recommendations for Monitoring

- Monitoring must be adaptive and allow feedback to revise monitoring efforts. Feedback can be provided regularly during permit renewal periods or irregularly as events of interest occur.
- Long-term trends must be expected and planned for in the sampling design.

- Baseline conditions are not determined from sampling a few sites over a few years. Spatial and temporal trends must be examined and research should be designed to address both with high power.
- Sampling needs to be performed over many years as effects of interest may not be apparent until after 10 years of data collection.
- Collection of data from a variety of habitats is important. Investigations should include a range of habitats that may provide additional records of long-term change.

4. Conclusions

The work presented in this dissertation contributes in three areas. These are 1) marine ecology (demonstration of environmental factors influencing flora and fauna in glacial fjords), 2), statistics and long-term monitoring (development of statistical methods for long-term monitoring and demonstration of subtle and large changes in marine communities), and 3) ecosystem disturbance (demonstration of faunal changes following large disturbance).

Conclusions resulting from this dissertation reflect the strong glacial influences in Port Valdez. Seasonally low salinity, high suspended sediment loads, and subsequent high sedimentation rates are key characteristic of glacial fjords including Port Valdez. Direct and indirect effects from physical tolerance to low salinity and factors associated with habitat structure appear to be important in organizing intertidal communities within these fjords. Demonstration of statistical methods for field studies applied here, including geostatistical modeling, may be useful to others seeking to establish new long-term studies or analyze field data. Previously unrecognized subtle effects from anthropogenic stressors near the marine oil terminal in Port Valdez are identified using analytical methods for correlated data. Studies following readjustment of fauna from a large earthquake and ecosystem-level effects of salmon aquaculture are nonexistent and this dissertation provides a reference point for any such future studies. Although readjustment from a large earthquake was a key process during the study period, salmon aquaculture appears to have a strong effect on the benthic ecosystem. The apparent relationship between the salmon aquaculture industry and the benthic ecosystem demonstrates the connectedness of the fjord to the ocean and has implications for managing local and regional ecosystems (Mundy, 2005; Ray, 2005). These results demonstrate a model for statistical assessments of long-term data, provide inputs for larger models, and should advance monitoring and managing of marine ecosystems.

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