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**Oil Spill Impacts and the Biological Basis for Response
Guidance: An Applied Synthesis of Research on Three
Subarctic Intertidal Communities**

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Oil Spill Impacts and the Biological Basis for Response Guidance: An Applied Synthesis of Research on Three Subarctic Intertidal Communities

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CONTENTS

CHAPTER	PAGE
INTRODUCTION	v
ACKNOWLEDGEMENTS	vii
1 FUCAS COMMUNITIES	1
<i>Systematics and Description</i>	1
<i>Distribution and Habitat</i>	2
<i>Ecology</i>	3
<i>Reproduction, Recruitment, and Settlement</i>	4
<i>Age and Growth</i>	5
<i>Natural Disturbance and Recovery</i>	7
<i>Effects of Oil on Fucus Communities</i>	8
<i>Discussion</i>	10
<i>Approaches to Fucus restoration</i>	14
2 MUSSEL (MYTILUS SPP.) COMMUNITIES	19
<i>Systematics and Taxonomy</i>	19
<i>Distribution and Habitat</i>	20
<i>Mussel Community Structure and Associated Fauna</i>	21
<i>Recruitment and Settlement</i>	21
<i>Age Distribution and Growth</i>	22
<i>Natural Disturbance and Recovery</i>	23
<i>Effects of Oil on Mussel Bed Communities</i>	26
<i>Discussion</i>	29

CONTENTS continued.

CHAPTER	PAGE
3	SOFT SUBSTRATUM INTERTIDAL MACRO-INFAUNAL COMMUNITIIES37
	Natural Disturbance and Recovery37
	Effects of Oil on Infaunal Communities38
	Direct Mortalities39
	Uptake and depuration of oil41
	Cleanup effects42
	Reproduction, Recruitment, and Recovery Following Oiling43
	Discussion45
	Depth of oil penetration45
	Acute and chronic oil toxicity46
	Indirect oil impacts47
	Cleanup Impacts49
4	CONCLUSIONS55
5	REFERENCES AND ACRONYMS57

PLATES

Plates 1 and 2.	Two views of the <i>Exxon Valdez</i> and <i>Exxon Baton Rouge</i> during lightering operations at Blight Reef, March 1989.viii
Plate 3.	Heavy shoreline oiling during the <i>Exxon Valdez</i> oil spill, April 1989.ix
Plate 4.	Shoreline cleanup operations during the <i>Exxon Valdez</i> oil spill, 1989.ix
Plates 5 and 6.	Two views of the “Omni-Boom” System used for HPHW shoreline cleanup during the <i>Exxon Valdez</i> oil spill.x

CONTENTS continued.

PLATES continued

Plate 7.	Typical example of intertidal <i>Fucus gardneri</i> cover on a shoreline in Prince William Sound, Alaska	15
Plate 8.	<i>Fucus gardneri</i> plants on boulder substrate in Prince William, Alaska.	15
Plates 9 and 10.	Closeups of young <i>Fucus gardneri</i> plants in Prince William Sound, Alaska.	16
Plates 11 and 12.	Two views of <i>Fucus gardneri</i> plants in Prince William Sound, Alaska, in September 1989 following shoreline treatment with HPHW washing..	17
Plate 13.	Experimental test bed for evaluating biological effects of HPHW.	18
Plate 14.	Example of intertidal <i>Mytilus trossulus</i> bed on a gravel beach in plants in Prince William Sound, Alaska.	33
Plate 15.	Closeup of <i>Mytilus trossulus</i> mussels in bed shown in Plate 13.....	33
Plate 16.	<i>Mytilus trossulus</i> bed on a bedrock outcrop in plants in Prince William Sound, Alaska.	34
Plate 17.	<i>Nucella lamellosa</i> snails feeding on juvenile <i>Mytilus trossulus</i> in Prince William Sound, Alaska.	34
Plate 18.	Heavy oil coat on <i>Mytilus trossulus</i> mussels and other epifauna in 1990, one year after the Exxon Valdez oil spill in Prince William Sound, Alaska.	35
Plate 19.	Shoreline treatment using a chemical shoreline cleaner after the Exxon Valdez oil spill in Prince William Sound, Alaska.	35
Plate 20.	1994 mussel bed cleanup activity in Prince William Sound, Alaska. Oiled sediments have been moved to lower intertidal zone for tidal flushing of residual oil.	36
Plate 21	Mussel bed cleanup activity in 1994 in Prince William Sound, Alaska. Restored <i>Mytilus trossulus</i> bed alongside undisturbed bed.	36
Plates 22 and 23.	Two views of the surface of a sandy beach in Elkhorn Slough, California, showing burrow holed of <i>Phoronopsis viridis</i> , a tube-building infaunal worm.	51

CONTENTS continued.

PLATES continued

Plate 24.	Surface of an artificial muddy-substrate environment created for the study of benthic infauna, showing burrows of polychaete worms.	52
Plate 25.	Cross-section of muddy substrated showing polychaete (<i>dorvilleid</i> and <i>neroid</i>) tubes and burrows.	52
Plate 26.	Typical infaunal sample from Prince William Sound, Alaska. Field samples are pre-processed, preserved, then completely sorted and identified in the laboratory.	53
Plate 27.	Intertidal hard-shell clam (<i>Protothaca staminea</i>) sample from Prince William Sound, Alaska.	53
Plate 28.	February 1992 cross-section of surface sediments from intertidal mud flat in Saudi Arabian coastal zone following the 1991 Gulf War, showing penetra-tion of crude oil via marsh crab burrows.) sample from Prince William Sound, Alaska.	54

INTRODUCTION

The 1989 *Exxon Valdez* oil spill occurred in Prince William Sound, Alaska, a remote subarctic marine system characterized by a diverse and productive intertidal environment. The nearly 44 million liters of North Slope Crude oiled an estimated 5,000 kilometers of shoreline and resulted in the largest cleanup operation ever mounted for a spill.

Intertidal communities, because of their location at the interface between coastal waters and the shoreline where floating oil strands, are highly vulnerable to spill incidents. In addition, oil spill cleanup operations generally are targeted in the intertidal zone, subjecting the resident biota to another series of disruptive influences. For these reasons, NOAA's Hazardous Materials Response and Assessment Division chose to monitor the short- and long-term effects of the *Exxon Valdez* spill on the Prince William Sound intertidal environment. These are the living resources most likely to be directly affected by intimate exposure to oil and cleanup activities. Understanding the nature and time frame of their recovery will provide the basis for improved spill response in the future.

Intertidal ecosystems are subjected to a myriad of perturbations of varying degrees of intensity and effect, and an oil spill should be considered as one of many. Large storms, ice movements, unusual climatic extremes of heat or cold, and episodic oceanic phenomena such as El Niño also can be responsible for significant changes in biological communities along the shoreline. However, oil spills also engender a number of unique considerations not usually encountered in other disturbance events.

The degree and duration of damage from oil spills is a function of several factors:

1. the type of oil spilled,
- 2).. the quantity and duration of the spill,
3. seasonal, oceanographic, and meteorological conditions,
4. nature of exposed biota,
5. habitat and substratum,
6. geographic location, and
7. type of spill cleanup used (Clark and Finley 1977).

Recovery of intertidal communities from large-scale disruptions is believed to be slower in cold-water, sheltered areas such as Prince William Sound. There are several reasons why an oil spill in arctic or subarctic environments could be expected to result in more serious effects than a spill in warmer climates. These include:

1. Cold temperatures result in slower evaporation of aromatic fractions.
2. Bacterial degradation is slower.
3. Less daylight reduces photochemical oxidation.
4. Many biota are generally long-lived with low reproductive potential and limited dispersal stages.
5. Most food chains are relatively short, increasing vulnerability to disruption (Sanborn 1977).

Sheltered marine intertidal areas in the Pacific Northwest to subarctic Alaska can be characterized by the presence of several distinct communities that include *epifauna* and *infauna*. Epifauna are defined as those animals living on the sediment-water interface, while infauna live within the sediment (Barnes and Hughes 1982). The infauna are further distinguished by size groupings into megafauna, macrofauna, and meiofauna. Although there is no set size definitions for these groups, in general the macro and meiofauna are usually separated at size cutoffs of 0.5 or 1.0 millimeter (mm), i.e., meiofauna are those animals that will pass through a 0.5 or 1.0 mm screen while macrofauna are those animals retained by that size screen. Generally the most abundant groups found in the macrofauna are the polychaete worms, followed by bivalve mollusks, amphipod and decapod crustaceans, holothoroid echinoderms, and burrowing anemones (Gray 1981). Other groups of animals that are often seen within the infauna community include nemertean worms, sipunculids, nematodes, and harpacticoid copepods.

On rocky substrates, the alga *Fucus gardneri* is often the dominant epibiota. Where the substratum consists of silt, mud, sand, or cobble/gravel infaunal communities of mollusks, polychaetes, and crustaceans may be found in high numbers. Finally, mussel communities may become abundant in patches on the surface of both soft and rocky substrata.

Here, we present a synthesis of the natural history and ecology of *Fucus* spp., mussel (*Mytilus* spp.), and intertidal infaunal communities. We summarize research results on the effects of oil on these communities, and discuss recovery times for these communities

following perturbations. Our interest in this material is not purely academic: knowledge of the biota and ecological relationships within these communities is necessary in order to make intelligent response decisions following a spill. We present the available information in that context, with the hope that there may be useful insights for the response community at large.

This report is intended to be a companion document to a literature review of oil effects on shallow water, cold water, intertidal communities (Fukuyama and VanBlaricom 1997). Many of the citations contained within this report are abstracted within the literature review.

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Plates 1 and 2. Two views of the Exxon Valdez and Exxon Baton Rouge during lightering operations at Blight Reef, March 1989 (NOAA/HAZMAT photo).

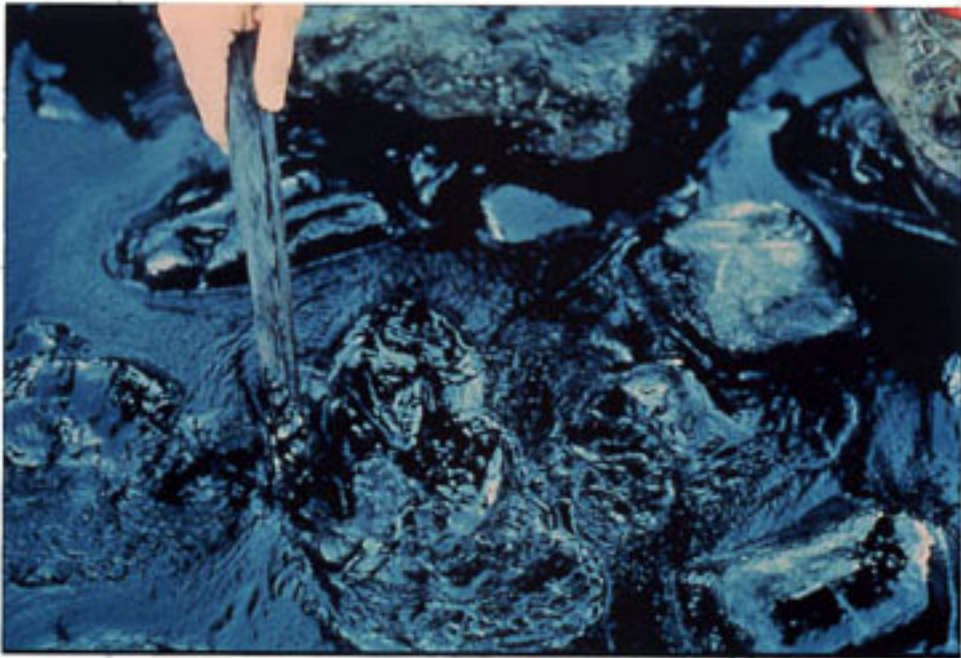


Plate 3. Heavy shoreline oiling during the Exxon Valdez oil spill, April 1989 (NOAA/HAZMAT photo).



Plate 4. Shoreline cleanup operations during the Exxon Valdez spill, 1989 (NOAA/HAZMAT photo).



Plates 5 and 6. Two views of the "Omni-Boom" system used for high-pressure, hot-water shoreline cleanup during the Exxon Valdez spill (NOAA/HAZMAT photo).

Chapter 1

Fucus Communities

The brown alga *Fucus* spp., commonly called rockweed, often dominates the intertidal zone in north temperate and subarctic areas. For example, *Fucus gardneri* comprises up to 90 percent of the biomass in the intertidal of Prince William Sound, Alaska (Stekoll and Deysher 1996). As a community dominant, *Fucus* plays a critical ecological role in primary production, as food for other organisms, and as habitat for many other biota (Plates 7-10). In the event of a major oil spill, it is likely that much of the intertidal where this alga is found will be impacted. Following the *Exxon Valdez* oil spill, an estimated 5,800 metric tons of *Fucus* biomass was destroyed due to oiling and subsequent cleanup of intertidal areas with high-pressure hot-water (HPHW) treatment (Mearns 1996). The resulting structural and functional alteration of intertidal areas in Prince William Sound remained evident at least 5 years after the spill.

Due to its widespread distribution in the temperate and subarctic intertidal worldwide, *Fucus* is likely to be one of the primary species adversely affected during oil spills that reach the shoreline. Political and public pressure to clean the shoreline may require decisions regarding cleansing or removal of oiled *Fucus* plants, as was done following the *Exxon Valdez* spill. In that case, *Fucus* plants were manually cut from large areas of the intertidal. HPHW washing of the intertidal scalded and removed *Fucus* from still larger areas of the intertidal (Plates 11 and 12). In some locations where HPHW treatment was employed (e.g., a NOAA monitoring site in Northwest Bay) large previously covered areas remain unvegetated (at this writing) with little recruitment and recovery of algal communities. Uncertainties exist with respect to the efficacy and advisability of HPHW treatment and manual removal of *Fucus* and the ability of the *Fucus* community to recover. Little is known, for example, about the vegetative regeneration capabilities of *Fucus gardneri*. The need for such information is likely to arise again following future spills in temperate and arctic environments.

Systematics and Descriptions

The rockweed, *Fucus gardneri* Silva, is the dominant algal species found in north temperate and subarctic intertidal areas. The previous name for this species was *Fucus distichus* L. emend. Powell. *Fucus evanescens* C. Agardh may also be considered synonymous (O'Clair et al. 1996). Distribution of this species is from the Kamchatka Peninsula through

the Bering Sea and Aleutian Islands to central California. *Fucus gardneri* is olive brown to dark brown and is regularly dichotomous with thalli 10 to 25 centimeters (cm) in length (Abbott and Hollenberg 1976).

Distribution and Habitat

The upper intertidal limit of several furoid algae, including *Fucus*, was found to be a function of their ability to tolerate prolonged desiccation during periods of low tide (Schonbeck and Norton 1978). Edelstein and McLachlan (1975) believed that *Fucus distichus* required constant immersion to survive. The lower intertidal limit of *Fucus*, on the other hand, was found by Gail (1918) to be defined by the high light requirements of the sporelings. However, Schonbeck and Norton (1980) concluded that the main factor controlling the lower limit of furoid algae was interspecific competition. They also found that furoid species required high levels of light to grow in culture. It is known that furoids can settle and grow in low intertidal zones, but laminarian algae are able to eventually exclude furoids from these zones due to the faster growth of the former (Kain 1975, Hruby 1976). Lubchenco (1980) concluded that competition from *Chondrus* limited distribution of *Fucus* into the lower intertidal zone. Total removal of *Chondrus* allowed *Fucus* to grow quickly and *Fucus* plants appeared healthy and reproductive. Competition between ephemeral algae and *Fucus* was postulated to occur during the first year of recruitment (Lubchenco 1983). With herbivores present, *Fucus* became abundant since herbivores grazed the ephemerals, but in the absence of herbivores, ephemeral algae were concluded to be competitively superior due to their faster growth and recruitment.

The distribution of *Fucus* in Maine was found to be related to degree of wave exposure and availability of substrata for attachment (Topinka et al. 1981). They noted that the highest biomass of *Fucus* was in areas that were considered "exposed" or "very exposed".

The northeastern shores of the Pacific apparently have only one *Fucus* species, *F. gardneri*. In contrast, the northwestern shores of the Atlantic have several *Fucus* species. In Nova Scotia, upper shore pools are dominated by *F. distichus*, upper emergent shores contain *F. spiralis*, and middle to lower zones contain a mix of *F. vesiculosus* and *F. evanescens*, although *F. evanescens* may be found in the lowest intertidal zones (Chapman and Johnson 1990). In a series of transplant experiments, Chapman and Johnson (1990) found that only *F. evanescens* was restricted to lower zones. This was due to physiological constraints. The other three species were found to be physiologically competent in zones other than those in which they were naturally found. They concluded that biotic

interactions of herbivory and/or competition accounted for the natural distribution of these *Fucus* species.

Ecology

Many ecological studies on *Fucus spp.* have occurred on the eastern coast of the United States and in Europe. Much of the previous ecological work on the northwestern Pacific species of *Fucus*, *Fucus gardneri* (= *F. distichus*), has dealt with recruitment, growth, and mortality estimates of populations [see Ang (1991a, 1991b, and 1992) and Ang and DeWreede (1992 and 1993)].

The importance of the *Fucus* canopy in the maintenance of community structure has been studied by various researchers. For example, McCook and Chapman (1991) found that the *Fucus* canopy was very important in structuring an assemblage of algae and invertebrates. The canopy served as shelter from waves, dessication, high temperatures, or freezing, as well as from predators. Cover of *Fucus* on intertidal shores in Britain over 7 to 10 years fluctuated from less than 10 percent to 75 percent (Hartnoll and Hawkins 1985; Crothers 1983). Much of this variability was related to changes in barnacle cover and abundance of limpets. The inhibition of barnacle settlement by fucoids is a generally accepted paradigm (Lewis 1964, Hawkins 1983).

It has been suggested that the large perennial seaweeds of intertidal areas such as the fucoids may be competitively inferior to sessile filter feeders like mussels and ephemeral algal species (Lubchenco 1983; Chapman and Johnson 1990). The maintenance of the fucoid assemblage is thought to be mediated through predation on the dominant filter filters and grazing on ephemeral algal species. However, other studies (McCook and Chapman 1991) suggest that the competitive interactions between *Fucus* and mussels may be less than suggested by previous work (Menge 1976).

Important grazers on *Fucus* include amphipods, isopods, littorines and other snails, chitons, and limpets. The limpet, *Patella*, feeds on the edges of the thallus and excavates tissue along the mid-rib (Jones 1948). Eventually the holdfast was eaten completely. The isopod, *Dynamene bidentata*, grazed on the surface of the thallus while the littorine, *Littorina littoralis*, fed on small areas of the thallus eating both external and internal cells (Viejo and Arrontes 1992). Littorines grazing on *Fucus distichus* were found to lower growth rates and delay reproduction of individual plants, though complete removal of individual plants was not seen (Van Alstyne 1990). Small *Fucus* plants are especially susceptible to grazing by littorines until they reach a size of about 3 to 5 cm, but germlings also find refuge from

grazing in rock crevices (Lubchenco 1980) or in crevices between barnacles (Lubchenco 1983). In some cases where abundances of littorines are especially high, *Fucus* may be prevented from settling at all (Menge 1975).

Reproduction, Recruitment, and Settlement

Several life history stages have been defined for benthic algae: spores or zygotes, germlings, juveniles, and adults (Vadas et al. 1992). However, the transition between germling stage to juvenile stage is subject to various interpretations. For example, juvenile stages of fucooids were defined as two-week old and 0.5 mm thalli (Chapman and Johnson 1990; Brawley and Johnson 1991). When thalli grew to sizes of 3 to 4 cm, the term germling was applied (Keser and Larson 1984).

The maximal period of reproduction of *Fucus spiralis* is between June and September (Niemeck and Mathieson 1976) on the east coast of North America and from July to September in Britain (Subrahmanyam 1961). The reproductive season of *Fucus distichus* occurs from early November to early June on the east coast of Canada (Edelstein and McLachlan 1975). They found that germination of sporelings extended over a period of 5 to 6 months.

A minimum size of *Fucus* is required before the onset of reproduction. Young *Fucus vesiculosus* were at least 15 to 20 cm in length while *F. serratus* were at least 18 to 25 cm long before forming receptacles (Knight and Parke 1950). Reproductive plants of *Fucus spiralis* (Niemeck and Mathieson 1976) and *F. distichus* were greater than 9.5 cm during the maximal period of reproduction (Ang 1991a). Although most of these plants were at least 2 years old, some one-year old individuals were found to be reproductive (Knight and Parke 1950, Edelstein and McLachlan 1975, Ang 1991a).

McConnaughey (1985) estimated density of settling *Fucus* gametes following the harvesting of *Fucus* plants in Bristol Bay, Alaska. He found densities were 0.42 to 2.5 million eggs/m²/yr. Sporeling densities ranged from 3500 plants/m² next to adult plants to 152 plants/m² at a distance of 10 m from the nearest adult plant.

Important factors affecting settlement of algal spores include substratum type, sediment, silt, scouring effects, water motion, dessication, temperature, nutrients, canopy effects, presence of turf, adult plants, and other invertebrates, grazers, and spatial and temporal refuges (Vadas et al. 1992).

The natural progression of algal settlement following a disturbance that creates space in the intertidal is the rapid settlement of spores of rapidly growing ephemeral species, followed by the settlement and subsequent domination of longer-lived algal species as ephemeral forms die off (Lee 1966; Kain 1975). In some cases, the initial settlement of ephemerals facilitates settlement of other algal species; for example *Enteromorpha* facilitated settlement of *Fucus* sporelings by providing protection from desiccation (Hatton 1938 in Kain 1975; Subrahmanyam 1961). Recruitment of *Fucus* was also found to be facilitated by the presence of ephemeral, blue-green algae (McCook and Chapman 1993). It was believed that the presence of ephemerals allowed attachment and subsequent protection and improved survival for the *Fucus* zygotes. The interactions between ephemeral algae and *Fucus* remain unclear since other studies have concluded that ephemeral algae did not facilitate settlement of *Fucus*, but instead was a competitor for space with *Fucus* (Lubchenco 1982, 1983, 1986).

Most sporelings of *Fucus* are found concentrated within cracks (Pyefinch 1943, Knight and Parke 1950, Edelstein and McLachlan 1975). High concentrations of sporelings were found in shaded areas in the high intertidal zone, while sporelings were absent underneath adult plants and in the deep bare areas of pools (Edelstein and McLachlan 1975). Recruits of *Fucus gardneri* were found to be most abundant where adult plants were within 1 m of the recruits (DeVogalaere and Foster 1994). The existence of a *Fucus* "germling bank" was hypothesized by Ang (1991a). He thought that this germling bank could survive without growing for a prolonged period and grow to larger sizes only when conditions were favorable.

The most important post-settlement factor affecting recruitment success of *Fucus spiralis* was intraspecific competition between adults and juveniles (Chapman 1989). Other studies have determined that grazing was the most important process affecting success of juvenile recruitment and settlement (Lubchenco 1980, 1983).

Age and Growth

Most *Fucus* spp. live for 2 to 4 years (Knight and Parke 1950; Subrahmanyam 1961). *Fucus spiralis* on the eastern U.S. coast usually live for about 2 years, though some plants were found to live for 4 years (Niemeck and Mathieson 1976). *Fucus distichus* was found to live for 2 to 3 years in western Canada (Ang 1991a, 1991b). There is good evidence from the NOAA long-term monitoring program in Prince William Sound that *Fucus gardneri* in that region lives for at least 4 years (Houghton et al. 1997).

In cultured samples, Schonbeck and Norton (1980) found that *Fucus serratus* and *F. spiralis* that were initially 10 to 16 mm in length grew from 1.2 to 2.2 cm in length per month and gained from 0.7 to 1.76 grams (g) of weight per month. Field measurements of growth in *F. spiralis* showed variable growth depending on the season, with maximal growth in the summer of 1.9 to 2.8 cm per month and minimum growth of 0.6 to 0.8 cm per month from November through March (Niemeck and Mathieson 1976). They attributed the faster growth during summer months to an increase in water temperature, light intensity, and day length. Another study by Subrahmanyam (1961) found that *Fucus spiralis* grew from 7.9 to 18.7 cm per year. Growth was slower at smaller sizes, but there was growth of the smaller plants throughout the year compared to larger plants that exhibited periodicity in their growth rates. No growth was found in larger plants from April to September. *Fucus distichus* grew about 0.4 cm per month in eastern Canada (Edelstein and McLachlan 1975) while *Fucus vesiculosus* and *F. serratus* had growth rates of about 2 cm per month in Britain (Knight and Parke 1950). Growth of sporelings following initial settlement was measured in *Fucus distichus* and found to be slow (< 1 cm) for the first couple of months (Edelstein and McLachlan 1975). They found that growth was slow during February to April, but was followed by a period of rapid growth in April and May. *Fucus distichus* on the Pacific coast showed higher growth rates in spring and summer with rates of 0.24 cm to 1.17 cm per month compared to growth in the winter of—0.5 to 0.4 cm per month (Ang 1991b). He also thought that growth rates were correlated with sizes. Growth of *Fucus distichus* in Bristol Bay, Alaska was variable depending on plant size, season, and plant density (McConnaughey 1985). Small plants surrounded by adult plants grew 0.13 g per month while small plants growing, without adjacent adult plants, grew 0.89 g per month.

Attrition of *Fucus spiralis* plants was found to be greatest during the late summer period of reproduction and during the winter when storms and ice scouring resulted in the loss of plants (Niemeck and Mathieson 1976). Most of the losses occurred during summer months due to the heavy weight of the receptacles and decay of the stipe by bacteria and fungus. Ang (1991b) found that smaller plants exhibited higher mortality during the winter and spring, while older plants had higher mortalities in the fall. Long exposures during summer daytime low tides were thought to be the main factor contributing to mortalities of larger plants. Smaller plants were protected by the canopy, so they did not exhibit the mortalities seen in older plants.

Natural Disturbance and Recovery

Following a disturbance, grazers such as limpets and littorine snails can negatively affect fucoids by feeding activities on spores and germlings (Jones 1948; Newell 1958). Spores and germlings are susceptible to removal by grazers until they reach a size of 30 to 50 mm (Knight and Parke 1950). The activities of these grazers may be so intensive that entire shores can be kept free of macroalgae (Lodge 1948; Southward 1962).

Several studies have examined the time it takes for an algal community to reach "climax conditions". Typical time frames for *Fucus* communities have ranged from 9 months in *Fucus spiralis* communities (Pyefinch 1943), more than 2 years for *Fucus* spp. communities on new concrete blocks (Rees 1940), and more than 4 years for *Fucus distichus* (= *gardneri*) communities following removal by burning (Dayton 1971). However, studies on complete clearings of *Fucus* from the substratum have shown that complete recovery takes a long time. Edelstein and McLachlan (1975) did not see recovery of *Fucus distichus* to its original status even after 4 years following removal and burning. Another clearing in an intertidal strip took over 10 years for recovery of algal species (Lodge 1948).

One factor limiting recovery of *Fucus* populations following disturbance is the limited ability of *Fucus* sporelings to disperse. Experiments on *Fucus* zygotes indicated that most zygotes did not disperse widely from where they were released (Burrows and Lodge 1950). *Fucus serratus* was found to have a low ability to disperse with new plants found within 2.5 m of the parental plant (Arrontes 1993). In addition, it was experimentally found that *F. serratus* only recruited on surfaces surrounded by other *Fucus* plants. An additional potential dispersal mechanism is drifting of detached fertile adult plants. This mechanism has not been studied in any detail.

Another factor important in algal settlement following a disturbance is substratum heterogeneity. *Fucus* was found to settle, but not survive, on smooth rocks following clearings when herbivores were present (Lubchenco 1983). If all herbivores were excluded, *Fucus* could become established on smooth rock. When crevices or barnacles were present, *Fucus* could survive only if herbivores were found in low densities.

Recovery of algal assemblages may be advanced if some parts of the plants are able to survive and regenerate tissue. The importance of regeneration and regrowth of tissue in the colonization of space in the rocky intertidal has been documented in several studies (e.g., Lubchenco and Menge 1978; Sousa 1979). Edelstein and McLachlan (1975) saw generation

of new tissue in old *Fucus distichus* plants. Any point along the frond could regenerate tissue. The newer tissue was yellowish to light-brown in contrast to the olive-green to dark-brown color of older tissue. They also noted that new fronds could be regenerated from basal discs. Most of the regeneration occurred in early spring, from May to June following damage from winter storms. Vegetative regeneration of two species found on the east coast of the United States, *Fucus vesiculosus* and *Fucus evanescens*, was studied by McCook and Chapman (1992). They found that 20 to 30 percent of experimentally damaged holdfasts regenerated adventitious shoots even when the basal tissue was cut to as little as 2 mm. The selective advantage of this strategy in the context of the individual or for the population was speculated as being very important for these species, especially in response to grazing pressure or in the event of large-scale disturbances.

In contrast, Ang (1991b) did not observe vegetative regeneration of tissue from the holdfasts of large *Fucus distichus* (= *F. gardneri*). Regeneration of *F. distichus* from the holdfast or stipe was only observed in very young plants. This *Fucus* population in western Canada was maintained primarily by periodic large recruitment pulses (Ang and De Wreede 1993). However, another study by Van Alstyne (1989) specifically examined adventitious branching in *Fucus distichus* following injury to the plant. She found that damage to plant apices produced branching and suggested that production of these branches was in response to grazing by herbivorous snails.

Effects of Oil on *Fucus* Communities

The effect of oil on Arctic macroalgal communities has been investigated in only a few studies, though many more studies have been conducted in subarctic and north temperate areas. Two macroalgal species in the Beaufort Sea were studied by Hsiao et al. (1978) and in situ primary production was significantly decreased by all oils tested. Cross et al. (1987) found that biomass of two of three algal species tested at Baffin Island, Canada were not affected by an experimental subsurface release of oil. The other species, *Dictyosiphon foeniculaceus*, showed an increase in growth in the year following the release of oil. Unfortunately, the study did not investigate effects on *Fucus* or *Laminaria* spp, though populations of both species were abundant in nearby areas.

There are conflicting reports about the direct effects of oil and oil spills on *Fucus* communities. O'Brien and Dixon (1976) found *Fucus* to have some tolerance to crude oil. Ganning and Billing (1974) determined that *Fucus vesiculosus* is resistant to light and heavy exposure to fuel oil in laboratory experiments. This may have been due to the mucilage layer

found in *Fucus* (Stebbing 1970). Anecdotal experiences from the *Exxon Valdez* spill (see, for example, Houghton et al., 1993) were consistent with these observations, in that *Fucus gardneri* stands at so-called "setaside" sites (areas that were oiled and by interagency agreement were allowed to remain uncleaned) survived initial oiling in relatively good condition. In contrast, oil removal operations employing high-temperature and high-pressure washes killed large areas of *Fucus*.

Low concentrations of oil may reduce fertilization in *Fucus* spp. (Derenbach and Gereck 1980). Concentrations as low as 0.2 ppb were found to inhibit germination of *Fucus edentatus* gametes and zygotes were completely killed at concentrations of 20 ppm (Steele 1977). There may also be a reduction in growth of *Fucus* zygotes and in adhesion of settling sporelings (Johnston 1977). Field experiments on recruitment of *Fucus* following the *Exxon Valdez* oil spill concluded that recruitment of algae was reduced or did not occur on rocks with tar (DeVogalaere and Foster 1994; Duncan and Hooten 1996).

Studies following oil spills also show contrasting results of effects of oil on *Fucus* spp. Following the oil spill from the *Tsesis*, no measurable effects were found on *Fucus vesiculosus* (Linden et al. 1979; Teal and Howarth 1984). No irreversible injury to *Fucus vesiculosus* was found in the first winter following a release of medium and heavy fuel oil (Notini 1978). No effect on *Fucus* growth was found following the *Eleni V* fuel oil spill (Blackman and Law 1980). Experimental oiling of the rocky intertidal with weathered oil by Nelson (1982) found no effects on *Fucus* and he concluded that *Fucus* was resistant to mortality from the oil.

However, immediately following the *Chryssi P. Goulandris* spill in Wales, a decrease in *Fucus vesiculosus* was observed (Crapp 1971). He also found during the "recovery" phase that *Fucus* decreased in abundance as populations of the limpet, *Patella vulgata*, increased. Damage to *Fucus serratus* was seen following the crude oil spill of the *Universe Leader* (Cullinane et al. 1975). They noted that algae were discolored or dislodged and that much of the damage was done by sea-borne detergents. The vertical distribution of *Fucus* spp. was dramatically reduced following the *Arrow* spill (Thomas 1977). Recolonization of *Fucus* continued from lower intertidal to upper intertidal, but no recolonization was seen in the high-tide zone, which he speculated was due to long-term toxicity of the oil. Similar to the results of Thomas, Stekoll, and Deysher (1996) also found that the upper boundary for *Fucus gardneri* populations in areas oiled by the *Exxon Valdez* had not recovered to equivalent heights of those in unoiled areas 5 years post spill. They did not speculate as to

the cause of the difference in height of the upper limit, but inferred that cleanup may have played a role.

Several studies have shown that the *Exxon Valdez* oil spill and subsequent cleanup treatment had dramatic effects on *Fucus* populations. Coverage of *Fucus* was found to be significantly less at oiled sites than at unoiled sites in many areas immediately after the spill (Stekoll and Dreyser 1996, Highsmith et al. 1996). A study focusing on *Fucus* populations at Herring Bay, Alaska found reduced numbers of large plants at oiled sites for at least 3 years post spill (van Tamelen and Stekoll 1996). They also found lower reproduction potential in plants from oiled upper intertidal areas. In many cases, the reduction in abundance of *Fucus* plants may be attributed to effects from hydraulic washing of intertidal areas following oiling (Houghton et al. 1996; Lees et al. 1996).

Following the *Torrey Canyon* spill, areas that were bare from toxic effects of the oil and dispersant were initially colonized by ephemeral green algae, which was followed in succession by *Fucus vesiculosus* and *F. serratus* (Southward and Southward 1978). This pattern of succession following oil and dispersant pollution had also been documented by other studies (e.g., Crapp 1971; Nelson-Smith 1972). As grazing limpets increased in abundance, they began to reduce the *Fucus* canopy until the substratum became bare again (Southward and Southward 1978).

Coverage of *Fucus gardneri* averaged <1 percent at oiled and cleaned sites versus 80 percent at unoiled sites 18 months following the *Exxon Valdez* oil spill. (DeVogalaere and Foster 1994). This pattern persisted even after 30 months. They also found that recruitment of *Fucus* was lowest at the intensively cleaned sites.

Discussion

The literature on *Fucus* and its recovery from major perturbations in the intertidal suggests that oil spills can affect *Fucus* communities in several ways that may not be immediately obvious to spill responders or resource managers working to minimize the impacts from an incident and its cleanup. In addition, there are distinct differences in plant physiology across different species of *Fucus* that would substantially affect the cleanup approach that might be considered during a spill.

Acute and chronic toxicity of oil to adult and immature plants

Adult *Fucus* plants apparently have some resistance to toxic effects of oil, and existing communities may survive at least a moderate amount of oiling without further human

intervention. However, an oil exposure non-toxic to adults may reduce fertilization success in reproductive plants, and earlier *Fucus* life stages appear to be much more sensitive. A given spill scenario, therefore, may present a number of conflicting tradeoffs for responders who are concerned about potential effects on *Fucus* communities. That is, the relatively high tolerance to direct oil exposure and known adverse reaction to the commonly used mechanical cleanup approach of high-temperature pressure washing may argue for minimal cleanup; on the other hand, by taking this approach, longer-term viability of the community may be affected by reduced reproductive and recruitment success. The latter concerns may be reduced in their significance if the spill occurs during a period of limited reproductive activity. Unfortunately (for response-minded individuals), the literature suggests that different *Fucus* species have very different reproductive periods: as noted previously, middle to upper intertidally dominant *F. spiralis* on the eastern coast of North America reproductively peaks between June and September, while the upper intertidal to supratidal *F. distichus* reproduces between November and June. Perhaps more frustrating to those response-minded personnel would be the observation of Ang (1991a), who found *Fucus gardneri* in British Columbia reproductively peaked in winter, in contrast to another researcher who found it peaked in nearby Puget Sound in June.

Fucus resistance to high-pressure hot water cleanup techniques

In contrast to the varying degree of sensitivity to the toxic effects of oil in different life stages of *Fucus*, the limited empirical evidence on the effects of elevated water temperature on *Fucus* seems consistent: the plants are relatively intolerant to hot water. During the *Exxon Valdez* response, wide expanses of *Fucus* cover washed with high-temperature seawater were killed, with exposed plants showing a distinctively unnatural color (bright orange).

A recent pilot study by Environment Canada has sought to determine temperature and pressure thresholds for adverse effects in intertidal biota, including *Fucus gardneri*, subjected to hydraulic cleanup techniques (Plate 13). Preliminary results, reported by Mauseth et al. (1996), noted the following impacts with increasing temperature and pressure:

- *Fucus* plants changed color from dark olive to light or emerald green and produced a viscous mucus-like material beginning at 40°C and becoming pronounced at 60° and 80°C. Analysis of the data showed that a significant difference in mortality occurred between 40°C and 80°C.

- ❑ *Fucus* plants began to tear at 60 kPa pressure (produced from a nozzle 15.2 cm from the substrate). However, increasing pressure did not cause significant mortality except at the highest pressure tested (344.1 kPa).
- ❑ Oil removal percentages increased most rapidly between 40° and 60°, and between 18.6 and 60 kPa.

Although these are preliminary results from a small-scale study, they are consistent with the *Exxon Valdez* experience. Both suggest that pressure and especially temperature levels where the use of hydraulic cleanup methods begin to show effectiveness at oil removal are also the levels where acute impacts to *Fucus* become pronounced. While this may indicate that HPHW is inappropriate for cleaning *Fucus* beds, it also implies that the lower temperature water deluges often employed during shoreline cleanup are relatively benign in terms of their impact on *Fucus* plants.

Oil and cleanup effects on competing plants and animals or predators

It is difficult to generalize about features of the nominally undisturbed intertidal environment, but a consistent portrayal from ecology-oriented studies is that of a highly dynamic system of action and reaction. Members of the intertidal community compete for a number of resources, including food, substrate, light, and other less obvious biophysical parameters. In the case of *Fucus*, several plant and animal competitors co-exist, with one of the commonly targeted resources being the limited substrate space in the intertidal. Under normal conditions, a dynamic equilibrium exists among species. However, a major perturbation like an oil spill and cleanup provides atypical opportunity for some and impediment to others. *Fucus*, which has a fairly slow rate of growth and a restricted set of conditions (possibly related to substrate rugosity, zygote dispersal, and protection from dessication) under which it can recruit, is at a competitive disadvantage with species that are more fecund or robust when substrate becomes available. Among the plants, green algae can be very opportunistic and often dominate the intertidal following an oil spill. Prolific invertebrate reproducers like barnacles and mussels also compete with furoid algae for substrate space and can preclude the recolonization of an area. Especially in the absence of predators and grazers, as is frequently the case following a spill, other intertidal species can be given a competitive advantage over *Fucus*.

Although it is not completely clear what organisms feed on *Fucus* in Prince William Sound, grazers documented in other regions on other furoid species represent a wide range of invertebrates. As mentioned previously, these include amphipods, isopods, snails,

chitons, and limpets: all organisms at risk and likely to be affected by exposure to oil and cleanup activities. Substantive reduction in numbers of grazers on *Fucus* should ostensibly encourage growth of the latter; however, it is likely that *Fucus* grazers additionally target other algal species that can more readily take advantage of the reduced grazing pressure to quickly recruit and fill in available substrate space, as described above. For example, the use of cleanup methods that cause widespread mortality of grazers like limpets or predators like drills may permit otherwise limited species like green algae or mussels to "bloom," thereby delaying or altogether preventing recruitment, recolonization, and recovery of *Fucus*.

The practical implications of these competitive relationships between *Fucus* and other intertidal constituents are that oil spill and response impacts that may not directly influence *Fucus* at all may still exert profound indirect effects by facilitating or inhibiting its competitors and grazers.

Cutting oiled Fucus as a potential cleanup approach

The presence of a *Fucus* bed in the intertidal zone greatly increases the surface area for stranded oil to potentially cover. Although as a rule, oil does not tightly adhere to *Fucus* because of the presence of a mucilaginous coating on plant fronds, oil will settle on the exposed surfaces of the plants and removal of oiled *Fucus* can represent one approach to spill cleanup in an affected intertidal area. Heavy *Fucus* cover can effectively shield the underlying biota and substrate from heavy exposure, and it is possible that "harvesting" the *Fucus* would remove a substantial amount of oil from the intertidal zone. The attractions of this approach as a cleanup method include fairly rapid oil removal with little need for specialized equipment beyond cutting tools, collection bags, and personal protective gear. The obvious question for a resource manager and the tradeoff to be evaluated by an on-scene decisionmaker would be, what are the longer-term environmental consequences of this cleanup? Specifically, how would the affected *Fucus* community respond to removal of large portions of the stand?

Beyond the short-term question of whether the removal of *Fucus* cover places associated animals and plants at greater risk by eliminating the shelter and refuge from potential re-oiling (a common occurrence during larger oil spills), probably the most important consideration for long-term recovery is the speed with which removed *Fucus* stands will regrow. The available literature on the subject of vegetative *Fucus* regeneration suggests that the species found on the eastern coast of North America—*F. distichus*, *F. vesiculosus*, and *F.*

evanescens—regenerated new growth from both cut fronds and holdfasts. In contrast, the *F. gardneri* found on the Pacific coast of North America appears to have very limited regenerative capabilities. Trial studies in Prince William Sound on *F. gardneri* have documented almost no vegetative regeneration (in three years) in plant stands where the *Fucus* was cut about 5 cm above the holdfast (Fukuyama, Hoff and DeVogalaere, unpublished data).

For spill response, this could result in distinct regional differences in the kinds of guidance that might be given to operational staff during a major oil spill. That is, on the Atlantic coast of North America, the approach of large-scale cutting of *Fucus* plants to remove oil and reduce long-term exposure would seem to be a viable response option because the plants could be expected to regrow from established holdfasts. On the Pacific coast, the advisability of such an approach would be much reduced, based on the research results we have reviewed and undertaken. *Fucus* communities that are removed in such fashion would recover only through recruitment, which can be a lengthy process. At least one NOAA long-term monitoring site in Prince William Sound, a rocky site that was stripped bare by aggressive HPHW cleanup showed only initial stages of re-establishing the heavy *Fucus* cover known to exist at the time of the spill 8 years after the initial impact.

Approaches to *Fucus* restoration

In the event of a spill, are there techniques that might be employed to promote a more rapid recovery of *Fucus* than would otherwise take place? Stekoll and Deysher (1996) investigated this question during their studies of *Fucus gardneri* related to the *Exxon Valdez* spill. In 1993, they deployed coconut-fiber erosion-control fabric at a site that had been heavily oiled in 1989. The fabric ostensibly served two purposes: first, it provided a substrate for *Fucus* germlings; second, it would protect the germlings from excessive heat and reduce desiccation. Half of the deployed mats were also inoculated with *Fucus* zygotes. By the next summer, the surfaces of all the fabric supported heavy growth of *Fucus*. Uncovered control plots with no fabric remained barren. Artificial inoculation did not provide sufficient improvements in promoting germling growth to justify the labor and other costs associated with it.

Although these experiments showed that it is possible to promote a more rapid recovery of *Fucus* cover than would otherwise take place, implementation of the methods described by Stekoll and Deysher required an investment of time (to ensure a sufficiently long life span in the intertidal) and materials (stainless steel screws in addition to the coconut-fiber

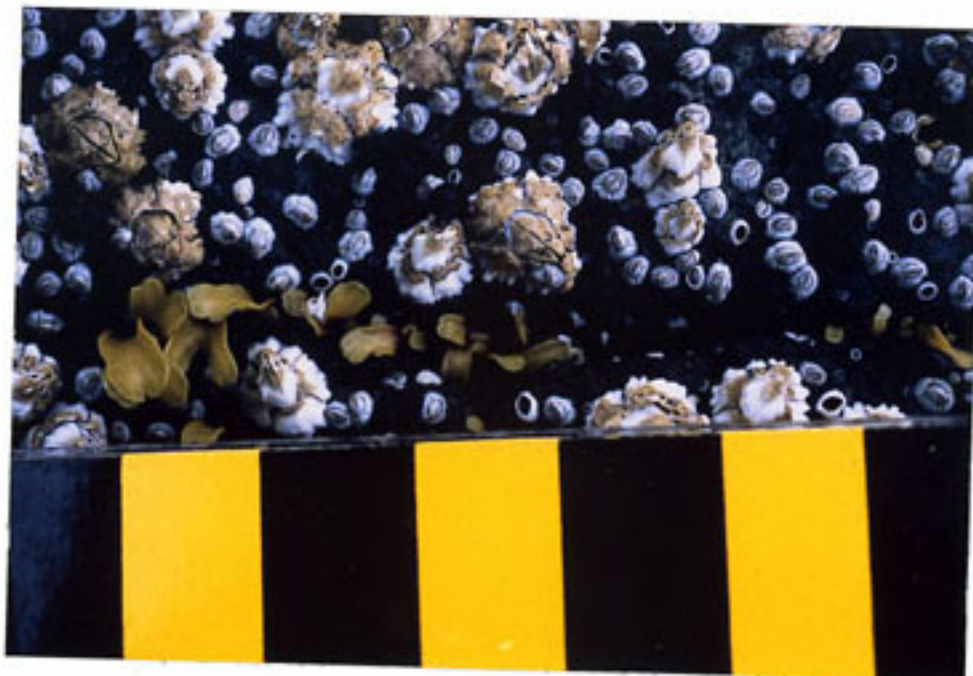
matting itself) that might not be appropriate for a large-scale restoration project. Nevertheless, they demonstrated its feasibility and suggested that prevention of desiccation to young plants is the most important factor to consider when encouraging a robust *Fucus* population.



Plate 7. Typical example of intertidal *Fucus gardneri* cover on a shoreline in Prince William Sound, Alaska (Photo by G. Shigenaka, NOAA)



Plate 8. *Fucus gardneri* plants on boulder substrate in Prince William Sound, Alaska (Photo by G. Shigenaka, NOAA)



Plates 9 and 10. Closeups of young *Fucus gardneri* plants in Prince William Sound, Alaska
(Photo by G. Shigenaka, NOAA)



Plates 11 and 12. Two views of *Fucus gardneri* plants in Prince William Sound, Alaska, in September 1989, following shoreline treatment with HPHW washing (Photo by A. Fukuyama, University of Washington).



Plate 13. Experimental test bed for evaluating biological effects of HPHW washing. Environment Canada project implemented by BEAK Environmental at Bamfield Marine Station, British Columbia (Photo by G. Shigenaka NOAA)

Chapter 2

Mussel (*Mytilus* spp.) communities

Mussels are common inhabitants of the intertidal zone in temperate and subarctic areas around the world. Their beds can provide physical structure and habitat, as well as sources of food for a number of other species including humans. The genus *Mytilus* is widely distributed in both the northern and southern hemispheres and has been discussed frequently in the literature related to both environmental monitoring and general intertidal ecology.

Systematics and Taxonomy

The Pacific coast of North America is known to have at least two to four species of *Mytilus*. *Mytilus edulis* and *M. californianus* are the most common species, however *M. trossulus* and *M. galloprovincialis* have been identified by some workers as well. *M. galloprovincialis* was probably introduced accidentally to southern California and *M. trossulus* is found from northern California to Alaska (McDonald and Koehn 1988). However, Gardner (1992) evaluated electrophoretic and mitochondrial DNA evidence and concluded that *M. galloprovincialis* should be considered as subspecies of *M. edulis*. Debate continues over the number of species found in the Pacific and most studies have been done on what have been known as *M. edulis* and *M. californianus*. These two "species" have been primarily used in the following studies.

Mytilus edulis and *Mytilus galloprovincialis* have generally been separated based on external shell characteristics. The shell of *M. galloprovincialis* is higher and flatter than in *M. edulis* (Gosling 1992). The anterior end of the shell is distinctly beaked or incurved in *M. galloprovincialis* whereas in *M. edulis* it is snub-nosed. The color of the mantle edge is purple-violet in *M. galloprovincialis* and yellow-brown in *M. edulis*. However, Gosling (1992) concludes that no single morphological character can be used to reliably separate mixed populations of the two mussel species. She also states that single morphological characters cannot be used to separate allopatric stocks of *M. edulis*, *M. galloprovincialis*, and *M. trossulus*. For now, the electrophoretic evidence has pointed to the existence of at least these three species, as well as *M. californianus*, along the northwest Pacific coast of North America.

Distribution and Habitat

Mytilus californianus is usually found on exposed rocks and wharf pilings in the middle to upper intertidal zone. Its distribution is from the Aleutian Islands, Alaska to southern Baja California. This mussel generally grows to about 130 mm long although specimens have been found to about 250 mm. The shell is typically thick with strong radial ribs. Byssal threads provide a strong attachment to rocks to prevent dislodging by waves. *Mytilus edulis* is generally found in more protected areas from the Arctic Ocean to Baja California. The *Mytilus edulis* species-complex is circumpolar in its distribution and is found in both the northern and southern hemispheres. *M. edulis* grows to about 100 mm long and has a much thinner shell than *M. californianus*. Byssal threads are much weaker in this species, but *M. edulis* is also more motile than *M. californianus*. If dislodged, *M. edulis* can form new threads and reattach within a few minutes if solid surfaces are available.

The upper and lower distribution limits of mussels are generally regarded to be influenced by abiotic factors for upper limits and by biotic factors for the lower limits. The upper limits of mussel distribution are known to be mainly due to physiological intolerance to temperature extremes and to desiccation (Seed and Suchanek 1992). Although *M. edulis* can withstand freezing temperatures (Williams 1970, Aarset 1982), *M. californianus* is known to be intolerant of freezing conditions and substantial mortality of *M. californianus* has been described by Suchanek (1985) due to winter freezes. High temperatures are also known to adversely affect mytilids and act in conjunction with desiccation to set the upper limit of mytilid distribution in the intertidal. Sudden and massive mortalities in mytilids have been documented in the intertidal by Suchanek (1978; 1985) and Tsuchiya (1983).

The lower limit for mussels (and many other organisms) in the intertidal zone is predominantly determined by biological factors such as predation and competition (Connell 1972; Paine 1974). Major predators affecting mussel distribution include seastars, gastropods (Plate 17), crabs, fishes, shorebirds, sea ducks, and sea otters. Competition from other sessile organisms may also affect mussel distribution. For example, where *M. edulis* and *M. californianus* co-occur in exposed rocky intertidal areas, *M. californianus* tends to out compete and dominate (Suchanek 1978, 1981). This is although young *M. edulis* are known to settle in the exposed situations favored by *M. californianus* (Kozloff, 1983).

There are instances where abiotic factors were found to be the primary factor affecting lower distributional limits. Littler et al. (1983) reported that periodic sand burial limited mussel distribution in lower intertidal areas of southern California.

Salinity conditions determine the distribution of mussels into an estuary. *M. trossulus*, however, is exceptional in that it can tolerate both full oceanic salinity of 35 parts per thousand (ppt), as well as very low salinity conditions of 6 to 7 ppt (Gosling, 1992).

Mussel Community Structure and Associated Fauna

Mussels occur in a diverse group of habitats including hard rocky shores, gravel/cobble substrata, and soft sediment shores in protected habitats (Plates 14-16). The aggregative behavior of mussels results in the formation of secondary substrata and a complex network of structures that allow many other organisms the opportunity for settlement and refuge. Suchanek (1979) has described the mussel matrix as consisting of:

1. A physical matrix of living and dead mussel shells of up to 5 to 6 mussel layers deep.
2. A bottom layer of sediments, feces and pseudofeces, organic detritus, and shell debris.
3. Diverse assemblage of associated flora and fauna.

Several studies have dealt with the variety of organisms associated with the mussel complex [see Shelford et al. (1935), Newcombe (1935), and Ricketts and Calvin (1939) for early accounts and Suchanek (1979, 1980, 1985); Kanter (1978, 1980); Paine and Suchanek (1983); Tsuchiya (1979); Tsuchiya and Bellan-Santini (1989) for more recent accounts of the associations]. The diversity of organisms within this mussel complex is very high. For example, Suchanek (1979, 1980) found a maximum of about 135 taxa associated with a sample of 30 x 30 cm and a total community richness of over 300 taxa of macroalgae, fishes, and invertebrates. The predominant taxa found were arthropods, mollusks, bryozoans, and annelids. The associated taxa may be divided into several categories: epizoans are sessile forms that utilize mussel shells directly as a substratum (e.g., algae, barnacles, bryozoans); motile fauna move throughout the complex (e.g., crabs, amphipods, shrimps, isopods, some polychaetes, gastropods); and infauna found directly within the sediments associated with the mussel layers (e.g., some polychaetes, other bivalves, sipunculids, ophiuroids).

Recruitment and Settlement

Larvae of *Mytilus edulis* are known to preferentially settle on filamentous algae following a period of pre-competent development (i.e., time preceding larval ability to react to external cues for settlement) (Bayne 1965, 1971; Seed 1969). Larvae are also known to

settle onto byssal threads and directly into adult mussel beds (Eyster and Pechenik 1987). Following primary settlement, *Mytilus edulis* are capable of migration and secondary settlement into adult mussel beds (deBlok and Geelen 1958; Bayne 1964). Peterson (1984) found that *Mytilus edulis* and *M. californianus* preferentially settled on different substrata; *M. edulis* preferred to settle on red algae and avoided settling in beds of adult *M. californianus* while *M. californianus* settled on filamentous algae, other algal species, and adult mussel beds of both species. An explanation for these behaviors is that *M. californianus* is competitively superior to *M. edulis* so there is little evolutionary pressure for *M. californianus* to select preferred substrata, whereas the competitively inferior *M. edulis* has evolved a strategy to settle in areas where the likelihood of survival is enhanced.

Although competition for space exists between the two species, both species coexist in the intertidal. *M. edulis* rapidly colonizes open patches in intertidal areas (Dayton 1971; Suchanek 1979) and is considered a classic fugitive species that matures early and has one massive spawning per year (Suchanek 1981). The competitively superior *M. californianus* eventually displaces *M. edulis* from areas where both species are found together (Harger 1970; 1972; Paine 1974).

Age Distribution and Growth

Mussel growth has been reviewed by Seed (1980) and Seed and Richardson (1990). The most appropriate measurement of growth in mussels is as a rate of change in biomass according to Seed and Suchanek (1992), but this measurement can only be made by removing the animal from its shell. Thus shell length is the most common measurement used for growth.

In many animal populations, size-frequency distributions are used to examine individual size classes. Changes in the mode of size classes are examined to measure growth of the population. In mussels, extended recruitment over the year results in merging of age / size classes, so this method has limited value (Seed 1976, Kautsky 1982).

Growth in *Mytilus* varies according to water temperature, food availability, population density, size, and age of the mussel. Growth variation is also a function of genotype, though this is minor compared to environmental conditions. Trevelyan (1991) found that growth in *M. edulis* individuals can vary as much as tenfold under heavy settlement. *M. edulis* reaches sizes of 60 to 80 mm within 2 years under optimal conditions, but reaches lengths of 20 to 30 mm after 15 to 20 years in the upper intertidal (Seed 1976). *M. californianus* grows faster and reaches body lengths of up to 250 mm (Paine 1976; Suchanek 1981; Yamada 1989).

Position within a mussel patch has an effect on individual growth. Mussels along an edge of a bed were found to be significantly larger by Svane and Ompi (1993). They also found that mussels in small isolated patches were even larger due to more food, less competition, and differences in predation and recruitment.

VanBlaricom (1987) found that variations in mortality, rather than site-specific variation in growth rates, account for differences in size structure of mussel populations in Prince William Sound. He also found that sex- and age-specific predation by sea otters also affected mortality rates. Higher mussel growth rates occurred when density was lower.

Several abiotic and biotic factors affect growth in mussels. These include temperature, salinity, food availability, tidal exposure, light, competition, crowding, and genotype. The most important factor is probably food supply since availability of food directly provides energy for growth. Tidal elevation is also important in growth since mussels are able to feed only when submerged under water and able to filter food particles from the water column. Mussels are efficient filter feeders removing particles as small as 2 to 3 micrometers (μm) with 80 to 100 percent efficiency (Mohlenberg and Riisgard 1977). A study by Baird (1966) of intertidal mussel distribution found that the point of zero growth in mussels coincided with a tidal level corresponding to 56 percent aerial exposure.

Water temperatures above 20° C or below 3° to 5° C slow growth of mussels (Almada Villela et al. 1982). The optimum temperatures for growth are between 10° and 20° C. (Coulthard 1929). Over the normal temperature range experienced by mussels, temperature is not as important a factor as food availability. Lowered salinity has a detrimental effect on mussel growth and may also lead to death (Almada Villela 1984; Gruffydd et al. 1984). Light may effect mussel growth, with continuous darkness, wavelengths below 600 to 700 nanometer (nm), and photo periods of 7 hours or less reducing growth rate in *M. edulis* (Stromgren 1976a, 1976b).

Natural Disturbance and Recovery

As defined by Paine (1994), the term “gap” is used by terrestrial biologists to describe openings or discontinuities within a canopy and the terms “gap” and “patches” are considered synonymous. Various studies have reported gaps found in the mussel matrix that have been caused by either physical forces (Dayton 1971; Levin and Paine 1974, 1975, Paine and Levin 1981; Sousa 1984 for examples), or due to biological processes (Paine 1966, 1969, 1974; Dayton 1971, 1973; Suchanek 1978, VanBlaricom 1988 for examples).

The mechanisms of gap formations are primarily due to the actions of log battering, wave disturbances, fouling, and hummocking (Seed and Suchanek 1992).

Log battering and wave action cause gap formation primarily during the winter season. Paine and Levin (1981) found typical rates of loss of *M. californianus* of about 0.4 to 5.4 percent of the mussel cover per month and 89 percent of the disturbances affecting <25 percent of the mussel beds during each winter. During the summer, gaps were smaller and from 5 to 10 times slower. The sizes of the mussel gaps range from the size of an individual mussel to about 57.5 m² (Dayton 1971; Suchanek 1979; Paine and Levin 1981) with subsequent enlargement of about 5000 percent increase in size (Dayton 1971). The reason for such enlargement seems to be due primarily to weaker byssal attachments by this species in the center of the beds where gaps were initiated, rather than in the edges of the beds (Witman and Suchanek 1984).

The primary biological processes that initiate disturbances in mussel beds are related to predation by crabs, seastars, sea otters, or to epizoism by algae such as *Postelsia* or *Fucus* (Seed and Suchanek 1992). VanBlaricom (1988) found that sea otter predation on mussels may result in loss of up to 20 percent of the *M. californianus* cover with a mean gap size of 0.25 m². He also found that about one quarter of the gaps enlarged before they recovered to previous densities, which took about 2 years to recover. This process resulted in having portions of the mussel bed in different successional stages, which in turn, resulted in a complex mixture of various species to occupy primary space.

Mussel beds are known to be resilient to the effects of physical and biological disturbances and full recovery usually occurs. Their ability to regain lost space has been described as a deterministic process by Paine (1974, 1984), i.e., the eventual outcome is to return to a state of predominantly mussels within the community. The most important factors that affect this process were larval settlement, tidal height of mussel bed, substratum type and angle, size of initial gap, and season when gap was initiated (Seed and Suchanek 1992). Scale of gaps was important in determining recovery. When gaps were small, recovery was almost instantaneous (0.2 cm per day) due to collapse or intrusion of the surrounding mussel bed (Paine and Levin 1981). They found that gaps < 3.0 m² recovered at a rate of about 0.05 cm per day and, for large gaps, recovery depended on settlement of larval mussels into the center of the gaps. Other mechanisms for recovery into larger gaps included reattachment of dislodged adult mussels or settlement of mussel larvae onto byssus threads left when mussels were displaced.

Suchanek (1979) and Paine and Levin (1981) found a predictable series of biological events that occurred following a disturbance event in the middle intertidal of *M. californianus* beds. Remnant byssus threads left behind remain on the substratum for about 10 months. Colonization occurred by diatoms and filamentous algae initially along with barnacles and *M. edulis* larvae. *M. edulis* may be found as a fugitive species within these gaps for up to 3 years, which allowed them time to reproduce (Suchanek 1978). *Nucella* (a carnivorous gastropod) settled into these areas at densities of about 50 to 75 m² and eliminated most of the *M. edulis* population. Drill holes were found in about 85 percent of dead *M. edulis* shells (Suchanek 1978). These patterns also occurred in *M. californianus* beds in California, with the exception that the *M. edulis* population usually did not reach reproductive size (Sousa 1984).

Settlement by *M. californianus* usually occurred 20 to 26 months after the disturbance and the mussels increased their coverage to over 80 percent after about 36 months. Total areal coverage by *M. californianus* occurred after 60 to 80 months (Seed and Suchanek 1992).

Season of disturbance is important to mussel recovery as well as to persistence of associated fauna. Winter months slowed the rate of recovery of mussels and gaps healed more quickly during May to September mainly due to mussel movement (Seed and Suchanek 1992). Winter gaps also allowed barnacles such as *Balanus glandula* and *Semibalanus cariosus* and some algae to colonize free space (Suchanek 1978, 1981; Paine and Levin 1981; Sousa 1984). Following a winter settlement, some barnacles were able to persist on the substratum within the mussel matrix throughout the life of the mussel bed (Suchanek 1979).

Patch size and angle of substratum affected recovery of mussel beds. Gap sizes influenced ability of motile grazers to reach and feed on organisms that colonized gaps, which in turn influenced the pattern of succession into these gaps. Angle of substratum affected rate of recovery of mussel beds from disturbances. In areas of low wave exposure, mussels easily colonized vertical surfaces, but in high wave exposure areas the opposite was true (Seed and Suchanek 1992).

Recovery of mussels from a major disturbance is a long-term process and depends on the species (Seed and Suchanek 1992). *M. edulis*, which is considered a fugitive species, appears to recover more quickly than other species. It typically recovered lost space in intertidal zones in 1 to 3 years (Hoshiai 1964; Tsuchiya 1983; Paine 1986), although gaps in a study area in southeast Alaska had not recovered after 7 years (Suchanek and Duggins, unpublished data, cited in Seed and Suchanek 1992). In *M. californianus* beds in

Washington, some gaps in upper intertidal areas were projected to take hundreds of years for full recovery based on slow recovery rates seen in artificially created gaps (Suchanek and Duggins, unpublished data, cited in Seed and Suchanek 1992). Recovery was found to vary depending on exposure to wave action and tidal height, with higher and less exposed sites recovering more slowly than mid and low intertidal and more exposed sites. This was also found by Lewis (1977) where upper and sheltered sites showed the least amount of change over time. On exposed Oregon coasts, major disturbances to *M. californianus* beds resulted in slow recovery of the mussel community (Castenholz 1967). He found that although lower algal species recovered after about 2 years, mussels and associated fauna had not recovered after at least 6 years. A minimal period of 6 to 7 years is required for mussel beds off the Washington coast to recover (Paine and Levin 1981).

Seed and Suchanek (1992) believe that the most important factor in recovery of mussel populations is larval availability, larval settlement, and recruitment by mussels. An emerging paradigm (at least for *M. californianus* beds) is that mussels are primary space occupiers and recover quickly from disturbances in lower and middle intertidal areas with high exposure (Paine 1984), but mussels in upper and vertical intertidal areas show variable recovery, with alternative communities possibly becoming established (Suchanek and Duggins in Seed and Suchanek 1992). Other organisms may eventually dominate the upper intertidal, as shown by the establishment of barnacles (Cimberg cited in Seed and Suchanek 1992) or the alga *Fucus gardneri*.

Effects of Oil on Mussel Bed Communities

Research on the effects of oil on mussels has yielded a range of impacts that can seem contradictory. That is, some studies indicate that crude oil is only slightly toxic to mussels, while others suggest a greater sensitivity.

M. edulis survived in 1,000 milligrams per liter (mg/l) crude oil for 24 hours (Smith 1968) while *M. galloprovincialis* could survive up to 2 percent oil in seawater (Alyakrinskaya 1966). The tissues of *M. edulis* were found to contain 77 to 103 milligram per gram (mg/g) fuel oil following an oil spill by Zitko (1971). Zitko (1971) also noted that aromatic fractions of oil were more toxic than aliphatic compounds. The sensitivity of mussels and other animals was tested to the water soluble fractions of Cook Inlet crude oil and No. 2 fuel oil by Rice et al. (1979). They found that mussels were relatively tolerant to crude oil with a 96-hour TLM (median tolerance limits) of >8.97 mg/l "total" aromatic hydrocarbons, but were more sensitive to fuel oil (TLM of >1.25 mg/l total aromatic hydrocarbons).

Kanter (1974) exposed *Mytilus californianus* to Santa Barbara crude oil under laboratory conditions and found that larger animals suffered higher mortalities than smaller animals. He speculated that the reason for this result was that smaller animals were pumping less water (and accordingly, oil fraction) through their mantle cavities than larger animals. An intuitively obvious result was that oil concentrations of 1×10^5 ppm resulted in higher mortalities in most mussels than lower concentrations! The most obvious pattern seen was a period of 3 to 4 weeks of survival in these mussels, followed by a rapid and continuous period of mortalities. A study by Straughan (1980) examining mussel communities associated with natural oil seeps in Santa Barbara found no differences in number of species associated with mussel communities whether oil seeps were present nearby or not. She also concluded that natural chronic exposure to petroleum did not influence the community composition. Following the Santa Barbara oil spill of 1969, assessments were made on the effects of the oil on intertidal and kelp-bed organisms. Although some mussels were found gaping and dead, generally mussels were found to have suffered little damage (Foster et al. 1971).

These previously cited studies suggest a fairly high short-term tolerance of adult mussels and mussel beds to oil pollution. Other studies, however, indicate that earlier developmental stages of mussels are more sensitive. For example, Wu and Zhou (1992) found that while concentrations of 1.25 and 2.50 mg/l had little effect on mussel veliger development, higher concentrations not only affected growth, but caused increased rates of developmental deformities that also increased mortalities in exposed organisms. The trochophore stage of mussels was found to be strongly susceptible to abnormalities attributed to oil exposure. Bayne (1976a) found that the cleavage and early embryonic stages of mussels have a limited tolerance to environmental change in general, and observed differences effects in oil between adult and juvenile mussels are consistent with this.

Sublethal concentrations of hydrocarbons accumulate in lipids of mollusks (Stegeman and Teal 1973). Lee et al. (1972) found that the hydrocarbons [^{14}C heptadecane and (^{14}C) naphthalene] were concentrated in the alimentary canal after uptake by gill tissues in *M. edulis*. They also determined that mussels exposed to clean seawater after exposure to hydrocarbons depurated 80 to 90 percent of the hydrocarbons that had accumulated after 72 to 336 hours. The primary source of uptake of aromatic hydrocarbons was through the aqueous phase, even when the source of hydrocarbons was from sediments (Pruell et al. 1986; McLeese and Burrige 1987). Experiments were done on blue mussels (*Mytilus edulis*) exposed to contaminated sediment collected from Narragansett Bay, Rhode Island (Pruell et al. 1986). They found that both polycyclic aromatic hydrocarbons (PAH) and

polychlorinated biphenyls (PCB) were rapidly accumulated by the mussels exposed to the contaminated sediment. After the mussels were transferred to control seawater, individual PAHs were depurated with half-lives ranging from 12 to 30 days.

Scope for growth is one measure of whether enough energy is available for growth and reproduction or whether external stresses are adversely affecting the animals. It is the energy balance of an animal under specified conditions. Scope for growth is positive when energy is in surplus and available for growth and reproduction or it may be negative resulting in loss of weight due to use of energy reserves (Bayne 1976b). When mussels were exposed to increasing concentrations of crude oil from 0.12 mg/l⁻¹ to 6 mg/l⁻¹, reduced scope for growth occurred (Gilfillan 1975). Mussels exposed for five months to a concentration of 30 mg liter⁻¹ total hydrocarbons showed a significant reduction in scope for growth (Widdows et al. 1982). The mussels in this study had reduced clearance rates and elevated excretion and respiration rates. A field study involving transplants of mussels over a pollution gradient in Narragansett Bay was performed by Widdows et al. (1981). They found that there was a decline in scope for growth with increasing concentrations of contaminants. Since environmental variables such as salinity, temperature, and dissolved oxygen were relatively constant among the sites, it was concluded that differences were caused by anthropogenic stressors rather than natural environmental variables.

The nature of sublethal effects of oil exposure was further investigated by Lowe and Pipe (1985). They determined that diesel oil exposure had a deleterious effect on the nutritional storage cells in *M. edulis* that subsequently led to reduced fecundity and oocyte degeneration. However, following a period of depuration in clean seawater, recovery was noted in both reproductive and storage cell systems. The overall ability of mussels to survive hydrocarbon exposure was in part dependent on the state of nutrient reserves at the time of exposure.

Carr and Reish (1978) observed that exposure to oil compromised the ability of *Mytilus edulis* to produce byssal threads, which the organisms use to anchor themselves to the substrate. In laboratory studies on *Mytilus trossulus*, Babcock et al. (1994) found a relation between oiling history of sites in the area affected by the *Exxon Valdez* spill and the ability of mussels to form byssus threads. However, other factors unrelated to oil spills, such as temperature variation, can also inhibit the byssal-production ability of mussels (Glaus, 1968).

A number of sophisticated “biomarkers” have been used to gauge oil spill effects in mussels. A biomarker has been defined by the National Research Council (1987) as “...a xenobiotically induced variation in cellular or biochemical components or processes, structures, or functions that is measurable in a biological system or sample.” Molecular biomarkers can be used to link a contaminant source with a decrease in animal condition. Solé et al. (1996), for example, assessed the effects of the *Aegean Sea* oil spill on such mussel (*M. edulis*) biomarkers as biotransformation enzymes, oxidative stress, and DNA adducts in digestive gland. They found that tissue levels of PAHs increased with proximity to the spill, as did levels of induced biotransformation enzymes like cytochrome P450. They also found evidence of oxidative damage in the digestive gland that could be linked to oil exposure.

Early versions of detergents and oil emulsifiers used on oil spills were found to be toxic to marine species. The effects of various surfactants were tested on *M. edulis* with non-ionic surfactants found to be more toxic than anionic compounds, especially on eggs and veliger stages (Swedmark et al. 1971). They postulated that surfactants accumulated on gill surfaces and impaired gas exchange and ionic regulation.

Qualitative studies (Shigenaka et al., 1995) on a tropical mussel species (*Brachiodontes exustus*) during a heavy fuel oil spill in Puerto Rico suggested that exposure to oil and a chemical shoreline cleaner mixture may have caused a transient decrease in the ability to produce byssal threads. No such effect was observed after mussels were permitted to recover in clean seawater for 24 hours.

Discussion

As we are learning for any resource of interest, the effects of an oil spill on mussel communities must be assessed in several ways that include both direct and indirect potential impacts.

Acute and chronic toxicity of oil to adult and immature mussels

A number of investigations into the toxicity of oil to mussels have indicated that adults can be relatively tolerant to exposure (Plate 18). However, it is also clear that outright mortality—or lack thereof—during a spill incident should not be taken as the sole indicator of potential effects to a mussel community. Cooley (1977), for example, cautioned that

...while oil can be lethal to mussels in large doses, equally devastating results (decreased feeding, cessation of reproduction, etc.) occur at much lower levels of contamination. Areas suspected of pollution should be monitored for effects other than death of marine organisms.

Several studies have demonstrated that chronic oil exposure affects the health of mussels, impacting several physiological aspects of the organisms, including nutrient metabolism and reproduction. It would be reasonable to expect that an oil spill not killing a mussel bed outright could have significant impacts ultimately resulting in the same net mortality, but over a longer time. Igic (1988) studied the factors affecting mussel (*M. galloprovincialis*) colonization from the perspective of biofouling on ships' hulls, and notably, presence of oil pollution ranked among the most unfavorable conditions for recruitment and growth of mussels.

Another subtle sublethal effect of oil exposure that nonetheless could result in the increase in mussel bed gap size or even in the erosion of entire beds is the inhibition of mussels' ability to produce byssal threads following exposure to oil. By avoiding potential detrimental cleanup effects and leaving some level of oil in contaminated mussel beds, it is possible that the chronic exposure may cause physiological changes rendering the bed susceptible to dislodging by physical forces that ordinarily would be inconsequential.

Toxicity of shoreline cleaning chemicals to mussels

Early versions of dispersants and shoreline cleaning chemicals were essentially modified versions of kerosene. As happens with most highly aromatic hydrocarbon mixtures, these first generation cleaners were very toxic to exposed marine life. Perhaps the most stunning example of this is the now infamous 1967 *Torrey Canyon* spill experience, in which vast quantities of chemicals were sprayed at sea and on the shorelines near Lands End, England, to combat the spill. The acute toxicity of the chemicals used caused widespread intertidal mortalities that profoundly changed the existing community structure (Smith, 1968).

Since the time of the *Torrey Canyon*, reformulations of shoreline cleaning chemicals have substantially reduced the inherent toxicity of the mixtures (Plate 19). Von Westernhagen and Dethlefsen (1982) exposed mussels to a dispersant/shoreline cleaner (Corexit 7664) and examined the effects on cadmium uptake. They found that a concentration of 50 $\mu\text{g}/\text{l}$ had no effect on cadmium uptake and no other acute impacts of exposure were observed. Because some shoreline cleaners may disperse oil into the water column and increase oil concentrations experienced, there is a possibility that mussels could experience an increased exposure to oil as a result of the use of a chemical shoreline cleaner.

Mussel sensitivity to high-pressure hot water cleanup techniques

Natural occurrences of high temperatures have been shown to set the upper limits of intertidal distribution for mussels, and incidences of unusually high ambient air temperatures cause sudden and massive mortalities (Seed and Suchanek, 1992). Tsuchiya (1983) documented the effects of abnormally high (34°C) air temperatures on a bed of *Mytilus edulis* in Japan. Tissue temperatures rose to greater than 40°C and killed about 50 percent of the intertidal portion of the bed.

Unfortunately, in the controlled study of the effects of HPHW washes on intertidal organisms sponsored by Environment Canada (Mauseth et al. 1996), mussels were too limited in number at the test site to permit inclusion in the analysis. However, the temperature sensitivity of mussels documented by Tsuchiya coupled with the need for high-temperature washes to mobilize oil as described by Mauseth et al. would lead us to infer that an inherent incompatibility exists between mussels and hot-water washing. This was strongly supported by data collected during tests of the Omni-Barge hydraulic washing system in Prince William Sound in 1989 (Houghton et al, 1993), in which observed dead *M. trossulus* increased from 5.4 percent cover in study plots before washing to nearly 33 percent after washing.

The temperature at which adverse effects to mussels begin to occur would be useful guidance for responders. There is anecdotal evidence that reduction of wash water temperatures in an explicit attempt to minimize impact does work (Houghton et al. 1998); however, specifics of the temperatures used are not available. Continuation of the Environment Canada studies may yield such guidance. In the interim, the available information indicates that temperatures that would be considered as mild for removal of oil (e.g., ~40°C) would not be well tolerated by mussels under conditions of extended direct exposure, as could be expected during a large-scale application of HPHW. Mediation of adverse effects through liberal use of cold-water header-hose-type flushing is a possibility and should be considered if hydraulic washing methods are employed.

Oil and cleanup effects on competitors or predators

Mussel predators include seastars, gastropods, crabs, fishes, shorebirds, sea ducks, and sea otters. Many of these organisms have been shown in laboratory studies or observed during spill incidents to be susceptible to oil exposure. The level of sensitivity varies with organism and oil, but as a rule the ability of bivalves like mussels to isolate themselves from

their immediate environment for extended periods (as they do when the tide is out) is a capability lacking in most predators. Moreover, differences in the nature of toxic effects affecting mussels and predators suggest an increased sensitivity in predators, particularly the vertebrates: birds and otters rely on the physical integrity of feathers and fur to maintain body temperatures and are impacted by exposure to much lower concentrations of oil than are mussels.

The consequences of this differential toxicity of oil spills are that mussels surviving the initial impacts may enjoy a temporary advantage in the intertidal zone while predators undergo a more substantial recovery phase. The combination of reduced predatory pressure and the mortality of plant and animal competitors for space may permit mussels to undergo a "boom" phase that could be expected to be attenuated as time passes and natural controls are restored in the system.

Mussel bed cleaning as a potential remediation approach

Generally, established mussel beds provide a much greater surface area than underlying substrate. When oil washes ashore, a larger amount can be retained in the mussel communities than would be the case in their absence. Once oil has penetrated into the interstices between mussels or created by byssus thread mats, it is much more sheltered from the natural weathering processes that normally reduce concentrations over time. As a result, there is an increased potential for long-term contamination of both the substrate as well as the mussel community itself.

In the *Exxon Valdez* example, a number of *Mytilus* beds were left uncleaned or only moderately cleaned in 1989 and 1990 out of concern for doing more harm than good to the mussels themselves. However, subsequent worries about these beds as a source of chronic contamination and as a source of hydrocarbons to the food web in Prince William Sound began to surface in 1991, and in 1992 the *Exxon Valdez* Restoration Program funded research to assess the extent of this perceived problem (see Babcock et al., 1994).

In 1994, a number of mussel beds in Prince William Sound that had been found to be contaminated with residual oil 5 years after the *Exxon Valdez* spill were treated in a straightforward but labor-intensive manner (Plates 20-21). Mussels overlying contaminated sediments were removed and placed in the lower intertidal to depurate, as were the underlying oiled sediments. After a period of cleansing in the lower intertidal zone, mussels and sediments were replaced in their original location. Analysis of monitoring results from these remediations is not complete at this writing, but initial indications were that the

procedure was successful in reducing the sediment contamination and mussel tissue concentrations of residual oil. The larger question of whether the technique reduced exposure to wildlife resources that potentially use the mussel beds as a food source has not been addressed, nor have the cost-benefit issues associated with this labor-intensive approach.



Plate 14. Example of intertidal *Mytilus trossulus* bed on a gravel beach in Prince William Sound, Alaska (Photo by G. Shigenaka, NOAA)



Plate 15. Closeup of *Mytilus trossulus* mussels in bed shown above in Plate 13 (Photo by G. Shigenaka, NOAA)



Plate 16. *Mytilus trossulus* bed on a bedrock outcrop in Prince William Sound, Alaska (Photo by G. Shigenaka, NOAA)



Plate 17. *Nucella lamellosa* snails feeding on juvenile *Mytilus trossulus* mussels in Prince William Sound, Alaska (Photo by G. Shigenaka, NOAA)



Plate 18. Heavy oil coat on *Mytilus trossulus* (center and center left) and other epifauna in 1990, one year after the Exxon Valdez oil spill in Prince William Sound, Alaska (Photo by G. Shigenaka, NOAA)



Plate 19. Shoreline treatment using a chemical shoreline cleaner after the Exxon Valdez oil spill in Prince William Sound, Alaska (NOAA/HAZMAT photo)



Plate 20. 1994 mussel bed cleanup activity in Prince William Sound, Alaska. Oiled sediments have been moved to lower intertidal zone for tidal flushing of residual oil (Photo by A. Mearns, NOAA)

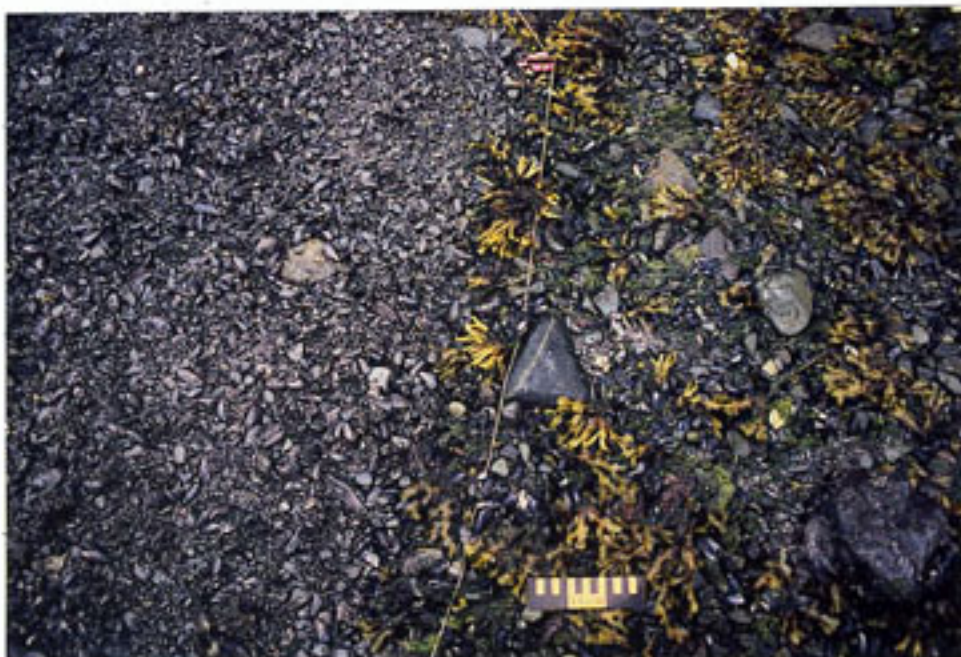


Plate 21. 1994 mussel bed cleanup activity in Prince William Sound, Alaska. Restored *Mytilus trossulus* bed (left) alongside undisturbed bed (right) (Photo by A. Mearns, NOAA)

Chapter 3

Soft Substratum Intertidal Macro-infaunal Communities

In contrast to the other members of the intertidal community we have discussed, macro-infauna is a category including many invertebrate taxa with little in common beyond their habitat. As defined in the introductory section of this report, infauna live in the sediment and commonly are represented by such organisms as polychaete worms, bivalve mollusks, amphipod and decapod crustaceans, holothoroid echinoderms, and burrowing anemones. Other groups of animals that are often seen within the infauna community include nemertean worms, sipunculids, nematodes, and harpacticoid copepods.

Soft substrata areas account for a large portion of the marine environment. Unlike the rocky shorelines where biological communities tend to be restricted to two dimensions, soft substrata permit a complex three-dimensional structure that is often hidden from easy viewing. Most infauna are usually not seen by the casual observer unless unusual or disruptive events have occurred that would cause them to emerge from the sediment (Plates 22-26). Effects of oil spills on soft-substratum infaunal communities are usually noted only if large numbers of dead organisms wash up onto shorelines (Sanborn 1977). As a result, it is therefore more difficult to assess impacts from an incident like an oil spill. This is particularly true for sublethal impacts in which the organisms may remain hidden in sediments, yet still be affected by oil or cleanup.

Natural Disturbance and Recovery

Natural disturbances to infaunal communities may be either physical or biological effects. Physical disturbances include storms and waves, ice scouring, flooding, and landslides. Biological processes include red tides, predation, and activities related to predation, such as digging or displacing sediments that indirectly affect infaunal communities (e.g., Thistle 1981; VanBlaricom 1982). Perturbations, whether physical or biological in nature, affect the community on several levels from the individual to the population to the community (Hall 1994).

Early studies on natural disturbance to infaunal communities reported changes to a sand beach community following a red tide kill of fish and subsequent anoxia to the sediment

that killed 97 percent of the invertebrates. The early recolonization sequence was led by the polychaetes *Polydora* and *Nereis*, *Eteone*, and *Paraprionospio* (Simon and Dauer 1972; 1977).

Several studies have examined recolonization into defaunated sediment within containers. The polychaete *Capitella capitata* initially recolonized boxes of defaunated sediment in the shallow subtidal of Massachusetts and increased in numbers for about 12 days before abruptly declining at 2 months (Grassle and Grassle 1974). Another study showed similar patterns with initial increases in numbers of the polychaetes *Capitella* and *Streblospio*, followed by increases in the amphipod *Ampelisca abdita* (McCall 1977).

There are several potential explanations for the patterns observed following natural disturbances in infaunal communities. One suggestion is that species are responding to an occurrence of favorable resources following a disturbance (Thistle 1981). VanBlaricom (1982) found that the species that became abundant following ray-induced disturbances were responding to an increase in organic matter concentrated within the pits. An alternate explanation is that early colonists are responding to a reduction in competitors that normally occupy the space (Thistle 1981). Understanding of the natural history of the individual species studied as well as the nature of ecological interactions among them is necessary to understand the patterns that are observed.

Effects of Oil on Infaunal Communities

Infauna inhabiting soft-substratum areas are usually considered very susceptible to oil pollution (Sanborn 1977). Oil may affect infaunal communities in several ways: either directly by killing benthos or indirectly by affecting behavior that may subsequently result in mortalities. Other effects of oil may be chronic and may have subtle effects on reproduction, growth, and recruitment of invertebrates. Relevant experimental work on several infauna species has focused on mortality, growth rates, effects on physiology, bioaccumulation of hydrocarbon components, and depuration rates of hydrocarbons from tissues.

Although we consider infauna as a group to be sensitive to oil, this is not to say that they cannot co-exist in its presence. Comparisons of infaunal communities at a natural petroleum seep and a nearby area without petroleum in the sediments were made by Davis and Spies (1980). They found common fauna at both sites, indicating that the sites were part of the same community. The most abundant group at the seep was deposit feeders, especially oligochaetes, which were rare at the unoiled site. The authors speculated that the seep community was trophically enriched by the presence of bacteria, which promoted the

increase in deposit feeders. Amphipod crustaceans were found in low numbers at the seep, while deposit-feeding bivalves were abundant.

Direct Mortalities

Oil in the intertidal may have adverse effects on the infauna due to smothering or toxic effects of the oil. These effects are mitigated by several factors, including rapid weathering of the oil and by tolerance of the animals that are adapted to live in these already stressful environments (Michael 1977).

The *Florida* oil spill in Buzzards Bay, Massachusetts resulted in immediate mortalities of intertidal and subtidal benthic fauna, documented by Sanders (1978). He found that the opportunistic polychaete, *Capitella capitata*, initially settled into disturbed areas at densities up to 200,000 individuals per m², but after about 7 months, densities of this species dropped as other species replaced it. Oiled areas showed large fluctuations in abundances and species composition while unoiled areas were relatively stable over time.

Following the Searsport spill of No. 2 fuel oil and JP5 jet fuel, small *Mya arenaria* were initially killed followed by mortalities of larger clams as the oil penetrated soft sediments (Dow and Hurst 1975). Production of *Mya arenaria* from the oiled area was reduced by 20 percent in 2 years, while production in unoiled areas increased by 250 percent. Following this observation, soft-shell clams were transplanted to oiled sites and growth and histopathology were studied in these clams compared with a control area (Dow 1975). Results of the transplant experiment confirmed that there was slower growth and survival in oiled areas. An examination of growth curves of soft-shell clams exposed to oil was made by Appeldoorn (1981) and he was able to show decreases in growth of the clams following exposure to oil. Where oil was no longer present, normal growth rates were seen.

Smothering of clams by oil was the primary cause of mortality following the spill at Chedabucto Bay of Bunker C oil (Thomas 1973). A field experiment to examine mortalities of clams following an oil spill was conducted on Port Valdez, Alaska mudflats (Shaw et al. 1976). They found that the bivalve, *Macoma balthica*, was significantly affected by the applications of oil and hypothesized that the mortalities were primarily due to the feeding habits of the clam in the upper 1 cm of sediment where much of the oil was stranded. Since *M. balthica* is primarily a deposit feeder (Brafield and Newell 1961), it is expected that oil deposits on surface sediments would become concentrated by the clams.

Limpets, bivalves, and peracarid crustaceans were immediately impacted following the *Amoco Cadiz* oil spill (Conan 1982). Polychaetes and large crustaceans were less affected. The author speculated that three to six generations were necessary before stable age distributions would be reached.

In contrast to the above results, Hartwick et al. (1982) found that crude oil had relatively little effect on the littleneck clam, *Protothaca staminea* (Plate 27). However, the dispersant Corexit 9527 and a mixture of the dispersant and crude oil had significant impact on adult clams. The increase in mortality was attributed to an increase in availability of hydrocarbons when the dispersant was present, or due to a synergistic toxic effect when they were mixed together. After the *Tsesis* oil spill, no decreases in biomass of *Macoma* spp or a priapulid were observed, indicating little direct effects from the oil on these invertebrates (Elmgren et al. 1980).

Indirect effects

Although direct mortalities from oil may comprise much of the loss of infauna following oil spills, animals that are able to survive the spill may be subjected to other threats such as predation. For example, exposure to oiled sediment affected the behavior of the littleneck clam, *Protothaca staminea* (Pearson et al. 1981). They found that exposure to oiled sediments resulted in shallower burial of clams, which resulted in more effective predation by crabs. Similar results of a change in burrowing depth with oil concentration were found in *Mercenaria mercenaria* (Olla et al. 1983). Exposure of *Macoma balthica* to water-soluble and oil-treated sediment fractions of Prudhoe Bay crude oil resulted in movement of clams closer to the sediment surface (Taylor and Karinen 1977). Individual *Macoma* spp. collected a week after the *Tsesis* spill burrowed into clean sediments significantly more slowly than clams from control areas (Elmgren et al. 1980). This behavior would have increased the risk of the clams to predation in the field. Exposure to oiled sediments also resulted in abnormal burrowing and escaping behaviors in the fiddler crab, *Uca pugnax*, which could have accounted for lower population abundances (Krebs and Burns 1977).

Chronic exposures of oil-in-seawater were made on *Macoma balthica* (Stekoll et al. 1980). They found behavioral, physical, physiological, and biochemical changes in clams after 180 days of exposure and concluded that concentrations as low as 0.03 mg l^{-1} would lead to population decreases.

A comparison of selected invertebrates from oiled and unoiled areas was made by Thomas (1978) following a spill in Nova Scotia. He found that length and weight of soft-

shell clams, *Mya arenaria*, were significantly lower at oiled stations, but the opposite was true for the littorine snail, *Littorina littorea*. He attributed these results to a restriction of growth in *Mya* due to the oil, and possibly due to decreases in macroalgal composition and increases in microflora, which subsequently affected *Littorina* growth.

Recruitment of benthic invertebrates may be affected for many years following an oil spill. For instance, following the *Amoco Cadiz* oil spill, recruitment of bivalves varied in intensity from year to year; this may have been due to natural fluctuations. Conan (1982), however, hypothesized that it may have also been due to an unstable age distribution in the parental population following the spill.

Uptake and depuration of oil

Soft-shell clams, *Mya arenaria*, exposed to No. 2 fuel oil showed patterns of accumulation and depuration of hydrocarbons over time (Stainken 1977). He found that clams accumulated most of the hydrocarbons within a week after the initial exposure and depuration occurred as hydrocarbon exposure in water decreased. The accumulation and discharge of hydrocarbons was determined to have been via a mucus-oil binding mechanism. *Mya arenaria* also exhibited significant differences in respiratory rates when exposed to low concentrations of oil (Stainken 1978). Lowest concentrations of oil used (10 ppm) caused a doubling of respiratory rates in these clams. Gilfillan and Vandermeulen (1978) compared growth and physiology of *Mya arenaria* from oiled and non-oiled lagoons following the *Arrow* spill. They found that the oiled population had lower abundance and lower numbers of mature adults, a lag of 1 to 2 years in tissue growth, lower shell-growth rates, and reduced assimilation rate of carbon.

Roesijadi et al. (1978) measured uptake of Prudhoe Bay crude oil by the deposit feeders, *Macoma inquinata* and *Phascolosoma agassizii* (a sipunculid worm) and the suspension feeder, *Protothaca staminea*. They found that uptake of aliphatic and diaromatic hydrocarbons was greater in the deposit feeders, but that the clams showed a continual increase in uptake of hydrocarbons over the test period, while the sipunculid reached an equilibrium early during the uptake period.

Sato et al. (1992) examined depuration of PAHs in the hard-shell clam, *Mercenaria mercenaria*, in laboratory experiments. They found that PAHs of higher molecular weights were eliminated at slower rates than PAHs of lower molecular weights. An average of 90 percent of the total PAHs was depurated after 50 days.

Cleanup effects

One of the few studies to explicitly examine effectiveness and biological effect of hydraulic (i.e., washing) methods on soft substrate beaches is that of Howard and Little (1987a). They applied a thin (<4 mm) layer of oil to fine intertidal sand and then used low-pressure seawater flushing as a cleanup technique. The seawater was delivered from a 2-inch outlet at 2 liters per second (l/s) for 5 minutes. They obtained high (93 and 78 percent) oil recovery rates with negligible oil incorporation into the washed plots. The authors noted that the ready mobilization of the oil was probably attributable to the high water table, and that oil would otherwise likely have been incorporated into the sediments.

Biological measurements included cast counts of the polychaete lugworm (*Arenicola marina*), and analyses of infaunal community structure. There were no statistically distinct differences in infaunal parameters between oiled and oiled-treated plots, but the number of *Arenicola* casts in oiled only plots was lower than in treated plots.

This study identified a number of the advantages and disadvantages of hydraulic washing techniques. Among the advantages:

- Rapid and effective reduction in sediment hydrocarbon contamination.
- No evidence of oil incorporation into sediments from washing action.
- Rapid return to conditions favorable for biological colonization.

Among the disadvantages noted:

- Erosion of the surface sediment (approximately 4 cm).
- Exposure of buried infauna.

Howard and Little give pragmatic advice on situations where the technique would be ill-advised, and conversely suggested situations where it may be most successful. Their listed criteria for optimal results were viscous oil untreated with chemicals; thin (< 10 mm) continuous slicks; good shoreline access and minimized response disturbance; firm substrate surface; gently sloping beach with poor to moderate drainage; local water supplies to reduce salinity stress to organisms; low-pressure delivery (~2 l/s) with no nozzle; containment and collection possible downshore from washing.

Reproduction, Recruitment, and Recovery Following Oiling

Early life history stages are generally regarded as being the most sensitive in the development of marine benthic invertebrates (e.g., Obrebski 1979; Watzin 1983), with the establishment and structuring of benthic communities following a perturbation largely determined by recruitment and settlement events.

Recolonization of intertidal infauna was experimentally investigated in the field using five different concentrations of oil (from 100 to 10,000 ppm wet weight) mixed in sediment by Christie and Berge (1995). Although all the common species were found at all concentrations of oil, they found that most species were found in decreasing abundances with increasing concentrations of oil. Densities of meiofauna, especially ostracods, were significantly lower in oil greater than 400 ppm. Macrofauna abundances were affected at concentrations greater than 1,000 ppm.

Colonization trays with and without covers and large enclosures were used to study effects of water-soluble fractions of hydrocarbons on recruitment of benthos (Bonsdorff et al. 1990). The main invertebrate settlers were amphipods and polychaetes. Addition of low levels of hydrocarbons did not affect polychaete settlement, but had a negative effect, either directly or indirectly, on settlement of juvenile amphipods, *Corophium bonelli*.

The effects of effluents from a petrochemical plant on intertidal infaunal communities were studied by McLusky (1982). He found no infauna at distances up to 250 m from the center of the effluent, low abundance and species diversity in an area of severe pollution (to 500 m away), high numbers of a restricted number of species at distances up to 1.5 km away, then beyond this an area of higher number of species. An area in which effluents were no longer discharged showed rapid recovery of the infaunal community.

Recolonization of benthos into experimental boxes with and without oil in eutrophicated and non-eutrophicated fjords was studied over one season (Berge 1990). He concluded that restoration of benthos after oil contamination took longer in a non-eutrophicated area than in an eutrophicated area. He believed that the reduced densities of infauna were caused by toxic response to oil directly or by secondary effects leading to mortality rather than due to reduced settlement. Restoration to the eutrophicated area was seen by 3 months due to a spring/summer settlement by opportunistic polychaetes (*Polydora* spp.).

Following the *Amoco Cadiz* oil spill, Glemarec and Hussenot (1982) studied recolonization of infauna in two coastal inlets. They found an immediate disappearance of the infaunal community followed by initial colonization of fauna able to take advantage of the enriched organic matter. Subsequently, a community of pollution-tolerant species settled, replacing the opportunistic species. After 3 years, most communities had not reached equilibrium.

Teal and Howarth (1984) summarized the effects of oil on benthos, and they noted that massive deaths among sensitive species could be expected following an oil spill. This is followed by settlement by opportunistic species, and eventual recovery in soft sediments estimated as taking from 6 to 12 years. Suchanek (1993) similarly noted that in an oil spill, initial heavy mortalities are likely in invertebrate communities, especially in polychaetes, amphipods, and bivalves. Opportunistic species settle first and populations of these species undergo large fluctuations in abundance before dampening as other species replace the opportunistic species. In general, a decrease in diversity of species is seen, but with an increase in abundance of a few species. Some opportunistic species observed to settle into newly disturbed areas include the tubificid oligochaete, *Tubificoides brownae* (Diaz 1980), the polychaetes *Steblospio benedicti*, *Capitella capitata*, *Mediomastus ambiseta*, and *Polydora ligni* (Grassle and Grassle 1974), and the amphipod *Ampelisca abdita* (McCall 1977).

An important factor in recovery of benthos, especially in low-energy environments, is the role of secondary contamination from the oil (Clark, 1982). Weathered oil may still be trapped in soft sediments and may affect infauna for extended periods following the spill. Another important factor affecting recovery of benthos following a spill is the type of shoreline treatment used. For example, HPHW washing of soft sediments was employed at many locations within Prince William Sound following the *Exxon Valdez* oil spill. Impacts to infauna were expected from the washing due to:

1. thermal shock,
2. hydraulic flushing,
3. dislodgement and exposure,
4. burying organisms,
5. forcing oil deeper, and
6. spreading oil into the lower intertidal areas (Driskell et al. 1996).

Results of the study indicated that some recovery occurred at sites by 1992, but recovery of infaunal communities at sites that were HPHW washed lagged significantly behind unwashed sites.

Although many definitions of recovery following oil spills and other perturbations have been used, one of the most common is the return of the ecosystem to within the limits of natural variability, including alternative or substitute species (Ganning et al. 1984). Recovery by organisms is initiated as soon as conditions are no longer acutely toxic to the colonizing organisms, but the time to "full recovery" varies widely. Many factors affect recovery of organisms including season of the spill, whether potential recolonizing stages of organisms are present, and water circulation patterns (Ganning et al. 1984). A summary of estimates of recovery times from various environmental disturbances was made by Mann and Clark (1978). Their estimates included a recovery time of about 10 to 20 years for an oil spill in temperate waters. Other recovery approximations within certain habitats include >4 years for a soft subtidal community in the Baltic Sea (Elmgren et al. 1981), and >5 years in a soft subtidal environment in Massachusetts (Michael et al. 1975).

Discussion

Assessing the effects of oil on infaunal communities involves many of the same considerations we ponder for epibiota like *Fucus* and *Mytilus*, with the added complexities of a third dimension of potential impact and much less available information on the basic biology of the communities likely to be affected. Moreover, because resource managers tend to focus on selected infaunal species (like clams) that can be of commercial value or represent major prey items for other managed wildlife resources, infauna may be overlooked when spill and cleanup impacts are evaluated.

The habitat characteristics that define infauna also mean that infauna can also be overlooked in a literal sense. That is, epibiota like *Fucus* and *Mytilus* are readily visible and adverse impacts from an oil spill can be gauged in much easier fashion than is the case with infauna. Unless affected infauna either move to the surface of sediments or are exposed during cleanup, they may never be seen by those with an interest in assessing either short- or long-term spill impacts.

Depth of oil penetration

Influence of grain size and degree of water saturation

Depth of penetration into beaches is largely determined by physical factors of grain size and water saturation. That is, oil is likely to penetrate more deeply into shorelines of larger grain size and larger interstitial spaces; and if the state of the tide or water retention of the beach is such that sediments are well saturated, oil is less likely to penetrate because it will float on the surface of the water. A wet and muddy shore would therefore experience less oil penetration than a dry gravel beach. In the former example, a stranded oil slick may lift off and refloat at the subsequent high tide, reducing the possibility of localized impacts. In the latter example, oil may penetrate deeply into the beach where it would remain until physically exposed by human or storm activity.

Biological avenues for increased penetration

Even if the natural substrate characteristics and state of water saturation prevent substantial penetration by an oil slick, it is possible and perhaps even common for biological activities to facilitate oil movement deep into a beach. Bioturbation by near-surface infauna may mix oil into sediments at the water-sediment interface, and burrows of infauna can allow liquid oil to penetrate to great depths where it is likely to be sheltered from many natural weathering processes. For example, in the intertidal area of Saudi Arabia affected by the 1991 Gulf War oil spill, large amounts of oil flowed down burrows created by marsh crabs (*Cleistostoma dotilliforme*) where it remained largely unweathered for at least 2 years (Hayes et al. 1993) (Plate 28).

Howard and Little (1987b) explicitly examined the subject of burrows facilitating oil penetration. They found that all burrows down to 1 mm in diameter increased depth of oil movement, and that the oil affected the behavior of infauna responsible for the burrows. Hydrocarbon residues were found in the casts of some polychaete worms (*Arenicola marina*), and the authors inferred that such biological reworking of oiled sediment could help to promote more rapid degradation of residues. The fact that the relatively viscous oil (medium fuel oil mousse) rapidly penetrated the test sediments suggested that in areas with a substantial burrowing infauna community, spill response must also be rapid to minimize the impact of oil movement.

Acute and chronic oil toxicity

The preceding literature review summarizes a substantial body of evidence documenting the toxic effects of oil to many infaunal species. In most cases, the overlying sediments appeared to offer only minimal protection from toxicity. Sublethal exposures also resulted in significant secondary impacts from increased predation and decreased health of the affected communities.

The varied and complex nature of infaunal communities would suggest that oil spills will result in a range of impacts dependent on the type of oil and depth of penetration, the inherent sensitivity of specific organisms, the life history stage being affected, seasonal considerations linked to general activity level and reproduction, burrowing habits, and other organism- and community-specific parameters. However, basic biological information on infaunal species is scarce and concentrated on the more visible community members like clams and polychaete worms. Even less research has targeted oil impacts. For these reasons and because "infauna" encompasses so many individual species, it is more difficult to generalize about potential effects than is the case for the selected epibiota we have discussed previously.

Indirect oil impacts

Smothering and other physical effects

An oil spill can injure a resource not only through the inherent toxicity of its chemical constituents, but also through indirect physical effects. Infauna, though they spend most of their existence buried in intertidal sediments, obviously depend on an intimate association with nearshore waters. An oil slick can interrupt that association by providing a physical barrier between an organism and the sediment-water interface, even if the oil confers no outright toxicity. A large oil cover that blankets a section of shoreline effectively isolates the infauna (and in flora) and prevents impedes if not prevents normal exchange processes. The biological communities, in other words, are smothered.

Even a lighter cover of oil that would not necessarily result in smothering could conceivably cause adverse changes in the infaunal habitat. For example, oiled surface sediments that may not be toxic to resident infauna could absorb much more heat from exposure to sunlight than natural beach sediments, potentially resulting in temperature extremes not tolerated by resident organisms.

Opportunistic species

As was the case for epibiota, large-scale environmental perturbations such as oil spills may also represent opportunities for some infaunal species. It appears that polychaete worms, oligochaetes, and other species that can take advantage of enriched organic conditions are the first colonizers of a spill-affected area. These are succeeded by organisms that are able to tolerate the degraded or altered environment resulting from the pollution, with a return to conditions approximating normality generally taking a period of years.

Cleanup Impacts

Temperature effects

Anyone who has dined on a bucket of steamed clams understands—at a gut level—the effects of hot water on selected infauna. As was the case with the other intertidal species that we have considered, the important question is the temperature experienced by the organisms in question. From this perspective, the sediments that represent habitat for infauna can also serve to insulate the community from the immediate effects of high-temperature water. If these sediments are also water-saturated, the degree of thermal protection conferred to infauna increases.

However, as we will discuss in Chapter 4, washing the shoreline also physically alters the vertical structure of a beach. That is, the shelter of overlying sediments can be removed, permitting infaunal species to be exposed to warmer water (as well as more oil) than they may be able to tolerate.

Houghton et al. (1992) observed oiling and cleanup impacts to *Protothaca staminea* clams during the *Exxon Valdez* spill response. They attributed short-term mortalities to oil exposure, burial by washing (*Protothaca staminea* have relatively short siphons and are found closer to the substrate surface than many other infaunal species), and removal of surficial sediments normally covering the organisms.

There is little direct guidance information on threshold temperatures for adverse impact in the intertidal environment. We can infer from the *Exxon Valdez* experience and its wealth of anecdotal information that the 60°C temperatures typically used were above the threshold for many species; and it would be a reasonable assumption that most of the soft-bodied invertebrate infauna would be susceptible at much lower temperatures and pressures than have been identified in actual measurements such as those by Environment Canada.

Physical disruption of substrate

By design, any shoreline cleanup operation will focus some level of energy and effort on modifying conditions on the shoreline. The obvious intent is to remove stranded oil or facilitate its removal, but the tenacity with which the spilled product may adhere to the substrate can require substantial amounts of physical force for removal. Even the most basic cleanup techniques can be highly disruptive to the intertidal environment where infauna reside, and compounding the problem is the perturbation caused by the cleanup crews themselves as they access a contaminated site. On rocky substrate beaches, these physical impacts may be minimal. However, in muddy or wetland environments, the resulting "collateral damage" may be severe and even in heavily oiled areas may preclude familiar forms of shoreline protection and treatment.

Although Howard and Little (1987a) had good success testing low-pressure flushing as a response method in a fine-grained sediment area, they did caution that the technique would work best in a specific set of circumstances and did point out potential problems with erosion of the substrate and exposure of buried biota.

Several "common-sense" precautions can help reduce overall impacts in a sensitive environment affected by an oil spill. For example, cleanup crews can carefully monitor pressure and temperature to restrict washing activities below apparent threshold levels of obvious adverse effect (i.e., where herding or pressure washes begin to mobilize sediments). Establishing "boardwalks" of plywood or other similar materials accomplishes two goals: first, it distributes the weight of cleanup workers across a larger surface area and minimizes the trampling of oil down into a soft-substrate environment; and second, it designates a permissible area for walking and restricts foot and equipment traffic away from areas that might otherwise be disturbed. A similar approach to the latter would be to physically mark an area to which traffic would be restricted, thereby "sacrificing" this portion of the affected environment but hopefully minimizing the collateral damage to the remainder.

The preceding techniques were employed during the response to a pipeline spill near the Texaco refinery in Anacortes, Washington, in early 1991. A small salt marsh was heavily oiled, and cleanup relied primarily on vacuum trucks augmented with low-pressure seawater flushing. One corner of the marsh was used for access by cleanup crews and the vacuum trucks and the substrate was heavily trampled and the marsh community was highly disrupted; however, access elsewhere was restricted to a plywood walkway. Recovery of the marsh was extensive 2 years post spill. (Hoff et al. 1993).

Alteration of grain size and other physical substrate characteristics

There is ample evidence in the literature that many infaunal species require specific physical conditions of grain-size structure or organic content to successfully recruit to or inhabit an intertidal area. If these conditions are altered by shoreline cleanup methods, then subsequent recovery from an oil spill may be significantly delayed. It is empirically obvious from nearshore and aerial observations of hydraulic washing activities during spill responses that this method in particular has the potential for altering grain size structure by washing away finer-grain material. An example illustrated in Shigenaka et al. (1997) shows a site in Prince William Sound oiled by the *Exxon Valdez* being washed by a crew using fire hoses, with the aerial perspective also clearly showing a large sediment plume mobilized into the adjacent nearshore region. In a marine environment like Prince William Sound where deposition of such fine material is fairly restricted, the process of natural restoration of grain-size structure may be a lengthy one. Houghton et al. 1998 postulated that these indirect effects of the spill and its cleanup have been responsible for the lagging recovery of infaunal populations (clams) known to have been resident at some sites prior to shoreline treatment.

As pointed out by Shigenaka et al. (1997), the tradeoffs faced by responders in such situations are those of rapid removal of oil from the intertidal/infaunal environment vs. potential long-term impacts of habitat modification. In areas with greater depositional inputs, the latter consideration may be much reduced in significance and the advantages of oil reduction through the use of washing become more obvious. However, this example should also illustrate that spill responders and resource managers should think strategically in structuring a response and should attempt to anticipate the unanticipated consequences of cleanup methods.

The reality of oil spill response is that it will be hard to argue for the potential of adverse hard-to-see infaunal impacts during an incident if a competing consideration is the set of potential impacts to more tangible, more commercially important, or more photogenic resources. That is, those arguing the interests of polychaete worms vs. those of sea otters are bound to lose nearly every time. However, this does not mean that consideration of direct and indirect infaunal impacts cannot be factored into an overall response strategy; and in fact, in the preceding example, the possibility of damage to infaunal communities that provide a significant food source for those sea otters may represent a two-pronged and nonconflicting rationale for a specific course of action or response method.



Plates 22 and 23. Two views of the surface of a sandy beach in Elkhorn Slough, California, showing burrow holes of *Phoronopsis viridis*, a tube-building infaunal worm (Photos by A. Fukuyama, University of Washington).



Plate 24. Surface of an artificial muddy substrate environment created for the study of benthic infauna, showing burrows of polychaete worms (Photo by A. Mearns, NOAA)

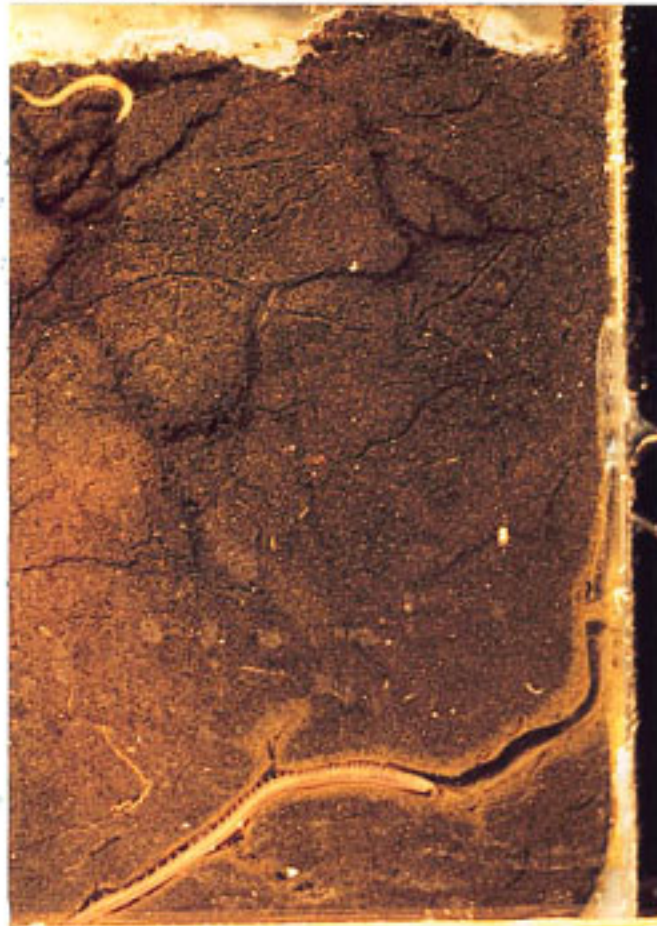


Plate 25. Cross-section of muddy substrate showing polychaete (dorvillid and nereid) tubes and burrows (Photo by A. Mearns, NOAA)



Plate 26. Typical infaunal sample from Prince William Sound, Alaska. Field samples are pre-processed (sieved), preserved, then completely sorted and identifies in the laboratory (Photo by A. Mearns, NOAA)



Plate 27. Intertidal hard-shell clam (*Protothaca staminea*) sample from Prince William Sound, Alaska (Photo by G. Shigenaka, NOAA)



Plate 2B. February 1992 cross-section of surface sediments from intertidal mud flat in Saudi Arabian coastal zone following the 1991 Gulf War, showing penetration of crude oil via marsh crab burrows (Photo by M. Hayes, Research Planning, Inc.)

Chapter 4

Conclusions

The *Exxon Valdez* oil spill was not only the largest such incident in United States waters and the largest response and cleanup ever mobilized, it also resulted in the largest oil spill research effort in history. Much of the latter was targeted on litigation-related issues, but the fact remains that a substantial and unprecedented scientific effort was directed toward understanding both the physical and biological effects of oil and cleanup methods.

In the NOAA long-term monitoring of the spill and recovery process, the primary goal has been to derive a scientifically supported basis for structuring oil spill response in subarctic regions such as Prince William Sound and extrapolate the results as widely as the results will reasonably support. Many of the lessons learned we have introduced and discussed in this manuscript are valid in a broader regional context. At the same time, clearly the specifics of biological details concerning life history, physiology, toxicology, and behavior (to name only a few considerations) are key to formulating an informed and effective response strategy that factors in the realities and peculiarities of a given intertidal community.

Oil and oil spills are toxic to each of the three intertidal community components we targeted in this report. The inherent and direct toxicity may not be as great as we might have intuitively expected, and organisms may be able to tolerate a relatively substantial exposure with few long-term effects. Conversely, our well-intentioned cleanup efforts may inflict considerable damage to intertidal communities, with much of the toll attributable to subtle or indirect changes in conditions necessary for robust recruitment and survival.

As happens on Wall Street, disaster for some translates into opportunity for others. In the intertidal, a competitive disadvantage for one community due to an oil spill or cleanup may reduce grazing or predatory pressure for another, or simply open up more physical space for a normally out-competed species. On Wall Street and in the intertidal, an individual or community able to tolerate a wide range of conditions will perform better in an unstable environment than a specialist, with all its metaphorical eggs in one basket. However, given the right set of conditions, that specialist can grow in explosive fashion and prosper—until the conditions change again.

For the reasons above, a lifeless intertidal, under any conditions of environmental degradation, is unlikely (although probably not impossible). *Something* should find a given environment to be acceptable, favorable, or even ideal. Recently discovered “extremophile” organisms living in deep ocean vents, mineral hot springs, or other physically challenging environments have expanded our concepts of what living communities can tolerate. We mention this because it is relevant to the operational definition of “recovery” from a spill incident and just what constitutes a “healthy” intertidal community. That is, a post-spill community may meet or exceed pre-spill measures of abundance or diversity, yet is likely to be different in the specifics of structure and species composition. Is that community “impacted”? Some students of chaos theory suggest that a disturbed system can never return to its pre-disturbance state. If we accept this concept, it would seem to preclude definition of recovery as a return to pre-spill conditions.

All of this would be interesting philosophical and quasi-academic musing, were it not that tangible definitions of recovery are necessary both for determining when a cleanup should end, and—a related consideration—to what point a responsible party can be held responsible for paying for a cleanup or restoration action. The implications are great, but addressing them appropriately is clearly beyond the comparatively modest scope of this document.

Returning in conclusion to the focus of this report: for all three of the important community components we have discussed, the basic biology of the plants and animals suggests that there are avenues available for reducing both short- and long-term effects of oil spills and oil-spill response. That is, it may be possible to use the specifics of life history or physiology to our advantage during an incident to reduce overall environmental impact. Information, however, can vary both in terms of its availability and its applicability, and so the fine art of extrapolation remains a key skill for those providing advice during an oil spill.

Chapter 5

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Acronyms

cm	centimeter
g	gram
HAZMAT	Hazardous Materials Response and Assessment Division (NOAA)
HPHW	high-pressure, hot-water
in	inch
kPa	
l	liter
l/s	liters per second
mg/g	milligrams per gram
mg/l	milligrams per liter
mm	millimeter
nm	nanometer
PAH	polycyclic aromatic hydrocarbons
PCB	polychlorinated biphenyls
ppm	parts per million
ppt	parts per thousand
TLM	medium tolerance level
$\mu\text{g/l}$	micrograms per gram