Effects of Dredging on Anadromous Pacific Coast Fishes

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Charles A. Simenstad, Editor

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About the Workshop

This workshop on the effects of dredging on Pacific Coast anadromous fishes took place on September 8, 1988, at The Meeting Place in the Pike Place Market, Seattle Washington. A subsequent working group session of the invited technical experts followed on September 9. The workshop was organized by the Wetland Ecosystem Team, Fisheries Research Institute, and was co-sponsored by the Washington Sea Grant Program. Financial support for the workshop was provided by the U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, Mississippi, through the Dredging Operations Technical Support Program.

The purpose of this workshop was to review the state of knowledge about the effects of dredging-induced turbidity on the physiology, behavior, and survival of anadromous fishes in the Pacific Northwest. Dredging effects attributable to the toxicity (excluding biological oxygen demand) of the suspended sediments were considered *outside* the purview of our consideration. Although many perspectives were represented, at the base of the issue was the question: *Can anadromous fishes detect and behaviorally avoid suspended sediment concentrations that may be harmful to them?* Most papers presented (see Appendix A) either background information on the physiology and behavior of anadromous fishes in estuaries or their specific responses to suspended sediment fields. Several papers addressed more peripheral issues, such as dredging impacts due to entrainment with the dredged material or the use of dredged materials for habitat enhancement.

Although the focus was on salmonids migrating through estuaries, some papers included information on striped bass, smelt and other anadromous fishes, and on riverine effects. The objective was to assess the present issues surrounding dredging impacts, to re-evaluate strategies for minimal-impact dredging, and to recommend approaches for evaluating and quantifying the more significant impacts. Technical presentations and panel discussions were given by experts in various fields of fish biology and dredging operations. Interested personnel from appropriate federal and state management agencies were invited to observe and ask questions during the general presentations.

The working group session integrated these presentations into a general discussion about the biological basis for temporal and spatial limits on dredging activities that are presumed to impact anadromous fish populations in Pacific Northwest estuaries. A synopsis of these discussions is presented in the summary and conclusions.

Charles A. Simenstad
April 1990

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The tests described and the resulting data presented herein, unless otherwise noted, were obtained from research conducted under the Environmental Effects of Dredging Program at the U.S. Army Engineer Waterways Experiment Station. Permission was granted by the Chief of Engineers to publish this information.

Physical and Chemical Alterations Associated with Dredging: An overview

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Abstract: *Physical and chemical alterations associated with dredging include increased levels of suspended sediments and the potential for associated dissolved oxygen reduction and release of natural and industrially-derived chemicals. The magnitude and spatial extent of the suspended sediment field around any dredging operation is a function of the type of dredge used, the physical/biotic characteristics of the material being dredged (e.g., density, grain size, organic content) and site-specific hydrological conditions (e.g., currents, water body size/configuration). A generalized "worst case" field can be defined as having suspended sediment levels ≤500 mg L^{-1} at distances ≤500 m from the dredge, with maximum concentrations generally restricted to the lower water column within 50-100 m, decreasing rapidly with distance. Given the relatively small quantities of material being suspended by most dredging operations, reduction in dissolved oxygen and chemical release from suspended sediments should be minimal and short lived.*

INTRODUCTION

Dredge-induced physical and chemical alterations are the direct result of the suspension of sediments into the water column. The magnitude of any given alteration is determined by the nature and quantity of the material being suspended as well as site-specific hydrological conditions which affect dispersion. The goal of a dredging operation is to remove quantities of sediment from an aqueous environment. In the process, however, quantities of sediment are lost to the water column as a consequence of the operational mechanics of a given dredge type (e.g., hydraulic suction or mechanical grab), operational controls of the dredge (speed of operation, etc.), and the character of sediment being removed (e.g., grain size, organic content).

In addition to the suspension of sediments (often referred to as turbidity), other potential alterations include reduction in dissolved oxygen (resulting from the oxidation of anoxic sediment compounds), and mobilization of sediment-associated chemical compounds (both natural and industrially-derived). Each of these general classes of alterations must be viewed in terms of the magnitude, spatial extent, and duration of the potential impact around a dredging operation as well as how this compares to site-specific ambient conditions. This information, along with knowledge of the response or tolerance

Table 1. General characteristics of dredge-induced suspended sediment [SS] fields from selected studies.

Dredge Type Location	[SS] Field Characteristics	Reference
Bucket Dredge		
San Francisco CA	[SS] 50 m downstream generally <200 mg L^{-1}, averaged 30-90 mg L^{-1} relative to background of 40 mg L^{-1}. Plume length: 300 m-surface, 450 m-bottom (10 m).	USAE (1976) Bay, as cited by Barnard (1978)
Thames River Estuary, CT	Maximum [SS] of 68, 110 and 168 mg L^{-1} at surface, mid-depth (3 m) and near-bottom (10 m) within 100 m downstream. [SS] decreased to background of 5 mg L^{-1} within 300 m, surface and 500 m, bottom. Fine grain sands and silts.	Bohlen & Tramontano (1977) as cited by Barnard (1978)
	[SS] adjacent to dredge were 200-400 mg L^{-1}, approaching background within 700 m. Maximum [SS] within 300 m. Fine grain sands and silts.	Bohlen, Cundy & Tramontano (1979)
St. Johns River, FL	[SS] plume extends 300 m at surface and 450 m near-bottom. Average [SS] within 240 m were 106 mg L^{-1}, surface and 134 mg L^{-1}, near-bottom. Silts.	Raymond (1983)
	Comparison of open and enclosed buckets. Enclosed bucket reduced [SS] in upper water column (by as much as 56%) but increased [SS] in lower water column (by as much as 70%). Silts.	Hayes et al. (1984)
Cutterhead Dredge		
Yokkaichi Harbor, Japan	[SS] under low current ranged from 2-31, 1-16 and 1-4 g L^{-1} at distances of 1,2 and 3 m above the cutterhead, relative to background.	Yagi et a al. (1975)
James River, VA	Average [SS] (over 4-days) within 240 m ranged from background to 282 mg L^{-1} above background. [SS] >100 mg L^{-1} restricted to lower water column. Average [SS] for upper water column were 11.5 mg L^{-1} for flood and 37.5 mg L^{-1} for ebb tides. Clay.	Raymond (1984)
Savannah River, GA	[SS] above background within 480 m generally <200 mg L^{-1} in lower, <100 mg L^{-1} in middle, and 50 mg L^{-1} in upper water column. Silts.	Hayes (1986)

Table 1 (continued). General characteristics of dredge-induced suspended sediment [SS] fields from selected studies.

Dredge Type Location	[SS] Field Characteristics	Reference
Hopper Dredge		
San Francisco Bay, CA	Avg. [SS] (and maximum) behind dredge (3 sites) ranged from 65-210 (210), surface; 33-64 (110), mid-depth (5 m); and 58-743 (1,110) mg L^{-1}, bottom (10 m). Avg. [SS] (and maximum) 50 m at right angle (2 sites) ranged from 43-45 (60), surface; 46-55 (55), mid-depth; and 337 (2,600) mg L^{-1}, bottom. Silty clay.	Sustar et al. (1976)
Grays Harbor, WA	Elevated [SS] in absence of overflow restricted to near-bottom, 60 m wide and 1100 m long with maximum [SS] of 70 mg L^{-1}. With overflow surface plume 60-m wide and 1200 m long with maximum of 857 mg L^{-1} at 30 m behind dredge. Near-bottom plume >120 m wide and 2600-m long with maximum [SS] of 891 mg L^{-1} at 30 m and 460 mg L^{-1} at 60 m behind dredge. Silty clay.	Hayes et al. (1984)

of organisms to these alterations, can then be used to evaluate the potential for dredge-induced problems. The objective of this paper is to provide an overview of the existing knowledge concerning each of these classes of alterations around the major dredge types used in coastal habitats.

Suspended Sediment Fields

The three most commonly used dredges operating in coastal habitats are bucket or clamshell, cutterhead, and hopper. A fourth "type" of dredging operation (agitation dredging), accomplished by a number of means, is also included because of its use in maintaining small harbors and boat slips, and in some cases shoal removal. Descriptions of these and other dredge types and their operation are provided by Barnard (1978), USAE (1983), Raymond (1984), and Richardson (1984). Reviews of available information on dredge-induced suspended sediment fields are provided by Barnard (1978), Raymond (1984), Hayes et al. (1984), Richardson (1984), Hayes (1986), Lunz and LaSalle (1986), Havis (1988a), and McLellan et al. (1989). For each of these dredge types, the sources of turbidity, as well as the quantity of material suspended, can, in part, be directly attributed to the mechanics of operation (i.e., mechanical removal as in bucket dredging versus hydraulic removal as in cutterhead or hopper dredging).

Bucket Dredges

The turbidity generated by bucket dredge operations comes from four major sources: (1) sediment suspended by the impact and withdrawal of the bucket from the bottom; (2) washing of material from the top and sides of the bucket as it moves through the water column; (3) spillage of sediment-laden water out of the bucket after it breaks the water's surface; and (4) inadvertent spillage of material during barge loading or intentional overflow intended to increase a barge's effective load. In addition, a number of operational variables can affect the quantity of material suspended, including bucket size and type (open or enclosed), volume of sediment dredged per cycle, and hoisting speed.

On the basis of a number of earlier studies (Gordon 1973; Cronin et al. 1976; Sustar et al. 1976; Williamson and Nelson 1977; Yagi et al. 1977; Nakai 1978; Onuschuk 1982) and those summarized in Table 1, a typical bucket dredge operation can be described as producing a downstream plume which extends up to 300 m at the surface and 500 m near the bottom. Maximum suspended sediment concentrations in the surface plume are generally less than 500 mg L^{-1} in the immediate vicinity (100 m) of the operation, decreasing rapidly with distance due to settling and dilution. Average water column concentrations in this same area are generally less than 100 mg L^{-1}. The use of enclosed or covered buckets has been shown to reduce surface suspended sediment concentrations by as much as 56% (Hayes et al. 1984). Bottom concentrations, however, were shown to be increased by as much as 70% due to the effect of a pressure wave which precedes the enclosed bucket as it descends through the water.

Hydraulic Cutterhead Dredges

The turbidity generated by hydraulic cutterhead dredge operations is primarily due to the action of the rotating cutter and is directly related to the type and quantity of material being disturbed, but not picked up by the suction. A number of operational variables may also influence suspended sediment levels around the cutterhead, including the rate of cutterhead rotation, vertical thickness of the dredge cut, and the swing rate of the dredge.

The suspended sediment field generated around a typical cutterhead dredge operation is largely restricted to the immediate vicinity of the cutterhead, with little suspension in surface waters (Huston and Huston 1976; Markey and Putnam 1976; Smith et al. 1976; Sustar et al. 1976; Barnard 1978; Koba 1984; Koba and Shiba 1983; Kuo et al. 1985, and references in Table 1). Maximum levels of suspended sediment (on the order of grams per liter) are confined to within 3 m above the cutterhead and decline exponentially to the water's surface (Yagi et al. 1977). Near-bottom levels may be on the order of hundreds of milligrams per liter at distances of up to a few hundred meters laterally from the cutterhead. Levels in the upper water column are usually quite low or even undetectable, depending on water depth. Raymond (1984) reported that higher current speeds associated with ebb tidal flow can act to suspend material higher into the water column.

Hopper Dredges

The turbidity generated by a hopper dredge operation (exclusive of overflow) is due primarily to the action of the dredge's dragheads (near-bottom) as they are pulled through the bottom sediment (Smith et al. 1976; USAE 1976; and references in Table 1). Additional turbidity in surface waters results when overflow of sediment-laden water is practiced in an attempt to increase sediment solids content during hopper loading.

Suspended sediment concentrations may be on the order of a few grams per liter near the draghead and on the order of tens of grams per liter near the hopper overflow. Suspended sediment concentrations at the surface decrease exponentially with distance from the dredge. A plume, however, may be perceptible at distances in excess of 1200 m, primarily because the dredge is mobile. A comparison of hopper dredge operations with and without overflow (Hayes et al. 1984) suggested that, in the absence of overflow, a plume was not encountered at the surface or mid-depth levels.

Agitation Dredging

Agitation dredging can be accomplished by a number of means, the common objective being to suspend bottom sediments into a flowing current (e.g., ebbing tide) for transport away from the site. Compared to other dredging operations, therefore, the characteristics of a sediment plume from an agitation operation are determined primarily by the nature of the sediments being disturbed and the extent of turbulent mixing through the water column. Richardson (1984) reviewed a number of methods ranging from the use of simple devices, such as iron beams which are dragged along the bottom or the use of propwash from ships, to more elaborate devices, such as arrays of pipes laid within boat slips through which water or air is pumped to suspend bottom material.

The turbidity generated by agitation dredging operations is restricted largely to the near-bottom waters where concentrations can reach as high as a few hundred to a few thousand mg L^{-1} (Richardson 1984). As with other dredging practices, concentrations decrease sharply with distance from the dredging site. Steel beam dragging operations at various sites in Savannah Harbor, Georgia, elevated concentrations of suspended sediments by 180-260 mg L^{-1} near the bottom (9 m), 40-130 mg L^{-1} in mid-depths (4.5 m), and 20-80 mg L^{-1} near the surface. A propeller wash operation in Chinook Channel, Columbia River estuary, Oregon, produced suspended sediment concentrations up to 2,580 mg L^{-1}. Air-injected devices deployed at a ship terminal installation near the Chehalis River, Grays Harbor, Washington increased suspended sediment levels by averages of 13-25 mg L^{-1}.

Generalized Plume Characteristics

The suspended sediment fields generated around the three most commonly used dredge types (bucket, cutterhead, hopper) can be generally defined in terms of the range of sediment concentrations (surface, bottom) and the range of spatial (horizontal) distribution around the dredge (Table 2).In general, suspended sediment concentrations in both surface and bottom waters are highest for bucket dredges (up to 2 1/2 times that of other dredges) and about equal for hopper (without overflow) and cutterhead operations. The horizontal extent of the sediment plume is greatest for bucket and hopper dredges in both surface and bottom waters. Worst-case conditions for each of these dredge types are compared in Figure 1, and include the case of hopper dredging with overflow. As previously discussed for each type, observed differences among dredges are attributable to the mode of operation of the two major types of dredges (mechanical, hydraulic) and relate to the site of the operation as well as operational parameters. Suspended sediment fields associated with agitation operations have characteristics similar to those described above, except that propwash operations appear to suspend larger quantities (an order of magnitude) of material than other types of agitation operations. This difference is expected given the greater energy that can be generated by the propeller of a ship or tug along with its position relative to the bottom. This observation also points to the role of propwash from ship traffic in increasing suspended sediment levels within ship channels.

While there are differences in the quantity and spatial extent of suspended sediment fields around various dredge types, a generalized "worst-case" field could be described as having suspended sediment levels ≤500 mg L^{-1} at distances ≤500 m from the dredge. Maximum concentrations are generally restricted to the lower water column within 50-100 m from the dredge, decreasing rapidly with distance. Turbidity plumes are short-lived, dissipatinghave also been compared to other anthropogenic activities that generate comparable quantities of suspended sediment including shrimp trawling, in the range of 500-600 mg L^{-1} (Schubel et al. 1979), and ship traffic (Slotta et al. 1973) which affects a given channel year-round.

DISSOLVED OXYGEN REDUCTION

Direct measurements of dissolved oxygen (DO) concentrations around dredges come primarily from four studies: a bucket dredge operation in a highly industrialized channel in New York (Brown and Clark 1968); a cutterhead dredge operation in Grays Harbor, Washington (Smith et al. 1976); a hopper dredge operation in a tidal slough in Oregon (USAE 1982); and a bucket dredging operation in a widened portion of the lower Hudson River, New York (Lunz et al. 1988; Houston et al., in press). Dissolved oxygen was depleted in the New York channel by 16-83% in the mid to upper water column and by as much as 100% in near-bottom layers (Brown and Clark 1968). Periodic reductions of bottom water dissolved oxygen of up to 2.9 mg L^{-1} were observed in Grays Harbor (Smith et al. 1976). Reduction in DO (1.5-3.5 mg L^{-1}) at the Oregon site was limited to slack water periods in the lower third of the water column, lasting until tidal flow resumed (within two hrs). Oxygen levels were observed to increase by as much as 2 mg L^{-1} under flood tide conditions (USAE 1982). The effect of dredging on DO in the immediate vicinity (within 90 m) of the dredge at

Table 2. General characteristics of suspended sediment [SS] fields around three commonly used dredge types (compiled from Barnard 1978; Hayes et al. 1984; Raymond 1984; McLellan et al. 1989).

Dredge type	[SS] Conc. (mg L^{-1})		[SS] Plume Length (m)	
	Surface	Bottom	Surface	Bottom
Bucket	0-700	≤1100	100-600	≤1000
Hopper *	0-100	≤500	0-700	≤1200
Cutterhead	0-150	≤500	0-100	≤500

*Without overflow

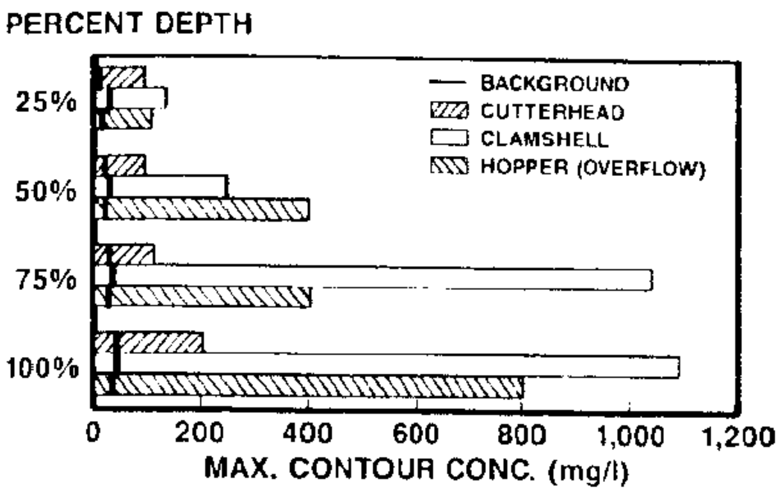

Figure 1. Maximum total suspended sediment concentrations measured around three commonly used dredge types (redrawn from Havis 1988a; data from McLellan et al. 1989).

the Hudson River site was minimal (generally <0.2 mg L^{-1}) and restricted to the lower water column (Lunz et al. 1988). At times, the range in DO at the reference site was greater than that observed near the dredge. Percent DO saturation bay-wide was also minimally reduced with maximal change (by 10%) corresponding to a 1 mg L^{-1} drop in DO (Houston et al., in press). Other studies have reported minimal or no measurable reduction in DO around dredges (Slotta et al. 1973; Sustar et al. 1976).

Another approach toward addressing potential dredge-induced DO reduction has involved prediction of DO depletion based on factors contributing to oxygen demand (Lunz and LaSalle 1986; Lunz et al. 1988). The basic premise of these models is that any reduction in dissolved oxygen (DO) around an operating dredge is the direct consequence of the oxidation of suspended anoxic sediment compounds. A review of the processes associated with DO reduction (Lunz and LaSalle 1986; Lunz et al. 1988) suggested that DO demand of the suspended sediment is a function of the amount of material placed into the water column, the oxygen demand of the sediment, and the duration of suspension.

Using various levels of suspended sediment concentrations known to ccur around a typical hydraulic dredging operation (100-500 mg L^{-1}), and stimates of low, moderate, and high levels (5, 20, 150 μl DO g^{-1} sediment dry weight hr^{-1}) of sediment demand, Lunz and LaSalle (1986) estimated a "worst case" DO depletion of no more than 0.1 mg L^{-1} (i.e., 500 mg L^{-1} suspended sediment). Estimates of DO demand used for this model were arbitrarily chosen evels over the range expected for estuarine sediments. Modifications of this pproach (Lunz et al.,1988) differed in that DO demand was based on direct neasurements of the most readily oxidizable chemical constituents in the ediment and that these reactions occurred rapidly (minutes), indicative of the hort duration of suspension.

One such model was based on measurements of the most commonly encountered reactive chemical components found in estuarine sediments (ferrous ron and free sulfides) which were used to calculate (using stoichiometric quivalents) the amount of DO needed to completely oxidize the material present. Jsing suspended sediment concentrations of 100 and 500 mg L^{-1}, predicted DO lepletion was 0.4 and 1.9 mg L^{-1}, respectively (Lunz et al. 1988). Results of ctual monitoring efforts in the immediate vicinity of the dredge (summarized previously for the Hudson River) showed suspended sediment levels no higher hat 150 mg L^{-1} and DO depletion to be on the order of 0.2 mg L^{-1}. Greater levels of DO depletion (by as much as 1 mg L^{-1}), measured as changes in percent saturation bay-wide, were suggested to be a function of DO demand from organic material acting over an extended period of time (days) (Huston et al., in press).

Given the relatively low levels of suspended material generated by dredging operations and considering factors such as flushing, DO depletion around these activities should be minimal. Other studies (Slotta et al. 1974; Smith et al. 1976; Markey and Putnam 1976) have also concluded that dredge-induced water quality alterations (including DO reduction) are short-term phenomena and do not cause problems in most estuarine systems.

Chemical Contaminant Mobilization

Much of the information concerning the release of naturally occurring (nutrients, sulfides, iron, etc.) and industrially-derived substances (heavy metals, organohalogens, etc.) comes from studies of dredged material disposal, reviewed by Lunz and LaSalle (1986). In general, most metals and other compounds are not readily available in a soluble form in the water column due, in large part, to reaction of these compounds with iron complexes and association with organic matter and clays (Windom 1972; May 1974).

Direct measurements of chemical releases around dredging operations are reported in Smith et al. (1976), Wakeman (1977), Tramontano and Bohlen (1984) and Havis (1988b). Wakeman (1977) reported significantly higher concentrations of four metals in San Francisco Bay. Average concentrations (filtered water) above background in surface samples were 0.16 mg L^{-1} for zinc, 0.01 mg L^{-1} for lead, 0.03 mg L^{-1} for chromium, and 0.01 mg L^{-1} for nickel;

bottom sample levels were 0.05 mg L^{-1} for chromium and 0.08 mg L^{-1} for nickel. Copper and mercury levels were unaffected by dredging. Smith et al. (1976) observed elevated concentrations of sulfides (range 3.9-1690 µg L^{-1}) in Grays harbor, with levels generally <50 µg L^{-1}. Tramontano and Bohlen (1984) observed elevated quantities of phosphate, ammonia, and silica in near-bottom waters within 180 m of the dredge and elevated amounts of manganese and copper within 12 m; cadmium levels were unaffected. While concentrations of these compounds in the immediate vicinity of the dredge (3-6 m) exceeded background levels by as much as 2-9 times, the absolute levels remained low: 17.1 µM L^{-1} for ammonia, 1.0 µM L^{-1} for phosphate, 14.5 µM L^{-1} for silica, 0.4 µM L^{-1} for manganese, and 0.1 µM L^{-1} for copper. These authors also suggested that, when compared with background levels of the whole system, dredging operations would increase these constituents by no more than 2% for ammonia, 1% for phosphate, 0.5% for silica, 0.1% for manganese, and 0.2% for copper.

The only other comparable information for these kinds of compounds involved dredging of contaminated sediments (Havis 1988b): bucket dredge operations in Black Rock Harbor, Connecticut and the Duwamish River, Washington, and a cutterhead dredge operation in the James River, Virginia. The chemical constituents measured and their average concentrations (mg L^{-1}, absolute values) were: for Black Rock Harbor, cadmium (0.001), zinc (0.03), lead (0.003), copper (0.01), mercury (0.0001), arsenic (0.01), chromium (0.001), nickel (0.01), manganese (0.12), and iron (0.7); for the Duwamish River, zinc (0.02), lead (0.007), and copper (0.002); for the James River, cadmium (0.003), zinc (0.002), lead (0.009), and copper (0.01). Relative levels for chemical species common between sites were similar.

SUMMARY

A generalized dredged-induced turbidity plume can be described as a "near-field" phenomenon (after Bohlen et al 1979) restricted in spatial extent to within a few hundred meters of a dredge and having levels of suspended sediment generally ranging up to 500 mg L^{-1} above ambient. The quantity of sediment suspended is, to a large extent, a function of the dredge type, while the spatial extent and duration of suspension is largely a function of the type of sediment being suspended. Given the relatively small quantities of sediment typically suspended, the short duration of suspension, and allowing for dilution during dispersion, the suspension of sediments (not containing highly toxic substances) around dredges is not likely to lead to appreciable reductions in dissolved oxygen or appreciable releases of chemical contaminants. This information, along with data on site-specific ambient conditions, can be used to place dredge-induced alterations in perspective with a given environment as well as evaluate potential impacts to aquatic resources.

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Migratory Behavior of Pacific Salmon in Estuaries: Recent Results with Ultrasonic Telemetry

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Abstract: *This paper briefly reviews the roles that estuaries play for anadromous salmonids and presents preliminary findings on salmonid migratory behavior in coastal and estuarine waters. Ultrasonic telemetry was used to study coho salmon smolts and adult sockeye salmon, chinook salmon and steelhead trout. Various hypotheses have been proposed to explain overall migratory orientation but the detailed patterns of vertical and horizontal movements are still poorly understood. There is a need for careful studies of the movement patterns of fishes under natural conditions before the impacts of estuarine development and activities by humans can be predicted.*

INTRODUCTION

Pacific salmon (*Oncorhynchus* spp.) pass through or reside in estuaries as smolts migrating to sea and as adults returning to spawn. Estuaries have been hypothesized to provide three main benefits for smolts: successful foraging, physiological transition from fresh to salt water and a refuge from predation (Simenstad et al. 1982; Macdonald et al. 1987). Estuaries are highly productive and smolts often grow rapidly there (Sibert et al. 1977; Healey 1982a; Myers and Horton 1982; Tschaplinski 1987). However, estuarine residence patterns vary among and within species and Healey (1982a) concluded that salmon use estuaries for foraging on an opportunistic basis. McInerney's (1964) evidence that smolts preferred increasingly saline water as the season progressed suggests that an estuary might serve as a zone of physiological transition from freshwater to marine conditions (Hoar 1976). Iwata and Komatsu (1984) subsequently demonstrated that chum salmon (*O. keta*) fry residing in estuarine water for at least 12 h showed improved osmoregulatory adjustment to seawater. The hypothesis that estuaries provide refuge from predators (e.g., McCabe et al. 1983) does not seem to have been critically tested. Mortality rates during early sea life are often high (Healey 1982b; Bax 1983) and it is not clear what role(s) estuaries play in predator avoidance.

The rapid migration of juvenile salmon through some estuaries (Jaenicke et al. 1984; Dawley et al. 1986; Groot and Cooke 1987) implies that the benefits of residence do not always outweigh the costs. However, Bax

(1982) argued that displacement by currents might explain some cases of rapid migration.

Estuaries also constitute habitat for adult salmon. Net migration rate often declines markedly in estuaries, relative to marine rates (e.g., Verhoeven and Davidoff 1962; Stauffer 1970; Barker 1979; Manzer et al. 1985). Some salmon may need a period of physiological adjustment prior to entry into freshwater. Maturing salmon may have ceased feeding by the time they reach estuaries and have relatively few predators. Therefore, foraging opportunities and predation risk are probably minor factors in determining estuarine residence patterns of adults.

Estuaries are often the focus of human activity as well as salmon usage. Shoreline development, domestic and industrial pollution and dredging are among the processes which may affect the use of estuaries by salmon. Dredging might affect salmon in various ways, including impacts on migratory behavior. However, it is difficult to predict the impacts of dredging on a behavior pattern such as migration which is not fully understood.

This paper summarizes results of four recent studies which employed ultrasonic telemetry to investigate salmon migration patterns in estuaries and nearshore waters. The studies had different specific objectives but they give a perspective on the problems and prospects of research on migration in estuaries.

TELEMETRY STUDIES

Coho Salmon Smolts in Grays Harbor, Washington

Methods: In 1988 and 1989 we studied the movement patterns of coho salmon (*O. kisutch*) smolts in the interface between the Chehalis River and Grays Harbor, a large estuary on the Pacific coast of Washington. Migrating wild smolts were captured in a trap on Scatter Creek, about 60 km upstream of the estuary, and brought to a pen in the lower Chehalis River. Each fish was anesthetized with MS-222 and an ultrasonic transmitter (33 x 9 mm) was inserted into its stomach. Relatively large smolts (at least 17 cm total length) were selected to minimize the effects of transmitters on fish behavior (Moser et al. in press). Each fish was given at least one day to recover before being released into the river. We followed the fish by boat using a directional hydrophone and receiver for about 12 h, recording our position every 15 min to estimate the fish's position. We abandoned the salmon at night but were usually able to relocate them the next day. Turbidity (Secchi disk), temperature, dissolved oxygen, salinity, pH and water current at 1 m were recorded every 30 min. Vertical profiles of temperature and salinity, recorded less frequently, revealed vertical, horizontal and temporal changes in water conditions within the study area (Fig. 1). Thus a fish could experience constant or fluctuating salinity, depending on its depth and movements over a tidal cycle.

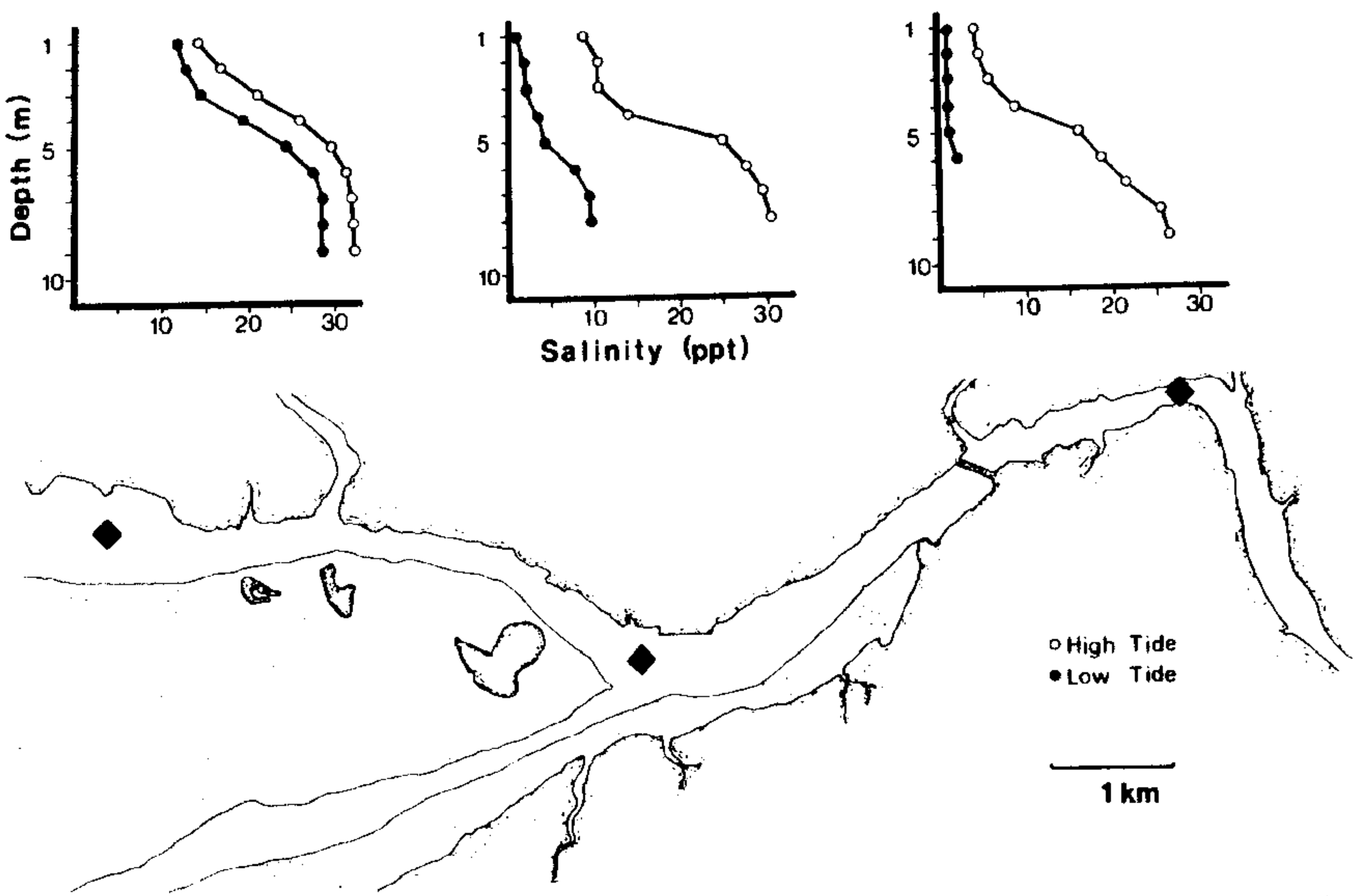

Figure 1. Vertical profiles of salinity at high and low tides at three locations in Grays Harbor, Washington, 1988, where coho salmon smolts were tracked.

Results: To date only a preliminary analysis of data from 1988 is available (Moser et al. 1989). Fifteen coho smolts were tracked intermittently for a total of 1032 h. Three basic behavior patterns are indicated by the data: moving, holding and disappearing. When released, the fish generally moved in or near mid-channel for several hours. Smolts generally moved in the direction of tidal flow. Analysis of smolt movements and observed current speeds indicated that the salmon were not drifting but rather were swimming into (i.e., against) the currents. On average, coho swam 34.4 cm/sec against ebbing currents (average current = 64 cm/sec) and 10.8 cm/sec against flooding currents (average current = 26 cm/sec).

While all of the fish moved to some extent, 12 of the 15 fish spent more than 70% of the time holding, defined as staying in one location over a tide change. On average, the fish initiated holding 6.4 h after release. Salmon held in areas of low current velocity, usually around docks, pilings and other structure, often for several days. Although fish were often relocated on successive days in the same general location, they made local movements of about 100-200 m.

One important question was, "How long do the smolts stay in the lower river?" We attempted to address this by searching the river with the hydrophone to relocate as many fish as possible. Most of the fish were relocated at least once but all eventually disappeared. Four fish disappeared after the first day of tracking, three after the second day, one after 3 d and one after 4 d. Two fish disappeared after 6 d, one after 8 d, two after 11 d and one after 12 d. Disappearance might be attributed to: (1) the fish being in an area which hampered signal detection, (2) transmitter failure, or (3) departure of the fish from the approximately 10 km section of river which was routinely searched. However, we were confident of our ability to locate fish in the study area because fish were seldom lost and subsequently relocated on the same day. Transmitter failure would preclude relocation but one of the transmitters was functioning 12 d after release and all were designed to last at least 6 days (Vemco, Ltd., Nova Scotia). If we assume that disappearance corresponded to emigration, then the fish generally emigrated after 1-4 days but four resided in the lower river for over a week.

Adult Sockeye Salmon Approaching the Fraser River

Methods: In 1985 and 1986 a large-scale, ultrasonic tracking program was conducted on Fraser River sockeye salmon (*O. nerka*) migrating between Vancouver Island and the British Columbia mainland. Sockeye were captured by purse seine, anesthetized and a pressure-sensitive ultrasonic transmitter (74 x 16 mm) was inserted into the stomach of each fish. Thirty-three salmon were tracked in hydrographically distinct regions in an effort to determine the relationship between vertical stratification and depth of travel (Quinn and terHart 1987).

Results: In general, there was a distinct difference in the depth of travel from one hydrographic region to another (Quinn and terHart 1987). In the relatively cool, oceanic waters of Queen Charlotte Strait and Johnstone Strait, the modal depth of sockeye was 3-4 m, though fish spent time in deeper waters, often to 20-30 m (Quinn et al. 1989). However, in the stratified waters of the Strait of Georgia, the modal depth was about 10-20 m. For example, a sockeye tracked in the plume of the Fraser River for 8.33 h swam at an average speed of 1.9 km/h. It spent little time in the upper 6 m and little time below 22 m (Fig. 2) and had a modal depth just below the thermocline/halocline. However, other fish tracked in similar hydrographic regimes swam up and down through steep gradients with no apparent hesitation.

In the nearshore waters, sockeye predominately oriented SE, which would generally lead them to the mouth of the Fraser River. When salmon encountered bays or islands, they tended to mill around and then swim in the reverse direction for several hours but would eventually turn around and resume SE orientation. Orientation seemed to be generally independent of depth. The salmon seemed to avoid the warmest, least saline water (>18°C, <26 ppt). However, there was a great deal of individual variation in vertical movements in the stratified waters of the Strait of Georgia, indicating that there was not a simple relationship between depth of travel and thermocline or halocline. Sockeye salmon also seemed to slow down at night in all regions, though only six fish were tracked at night.

Adult Chinook Salmon in the Columbia River Estuary

Methods: In 1987, adult fall chinook salmon (*O. tshawytscha*) were captured in the lower Columbia River near Astoria, Oregon. Each day, a very brief (ca. 5 min) gillnet set was made to minimize stress to the fish and the healthiest fish was selected for tracking. A pressure-sensitive transmitter was inserted into the fish's stomach and the fish was transported to a common release site. We periodically measured the water's turbidity (Secchi disk) and vertical profiles of temperature and salinity along the fish's path during the track.

Results: The movement patterns of these nine chinook salmon are presently being analyzed in detail (Olson and Quinn, in prep.) but several general patterns were evident. Several fish spent most of their time in the relatively warm, brackish surface waters rather than the cooler and more saline water below (e.g., Fig. 3). Other fish also spent time in the surface waters but descended to the bottom or swam at mid-depths periodically. The salmon often moved rapidly through steep vertical gradients in their ascents and descents.

The salmon generally swam actively and ground speeds averaged 2-3 km/h but made little net progress upriver. Indeed, two of the fish swam seaward over the river's bar and were abandoned heading west. The salmon seemed to swim into the current in most instances and milled during periods of slack water. In general, the salmon did not follow the shorelines but rather were in open water.

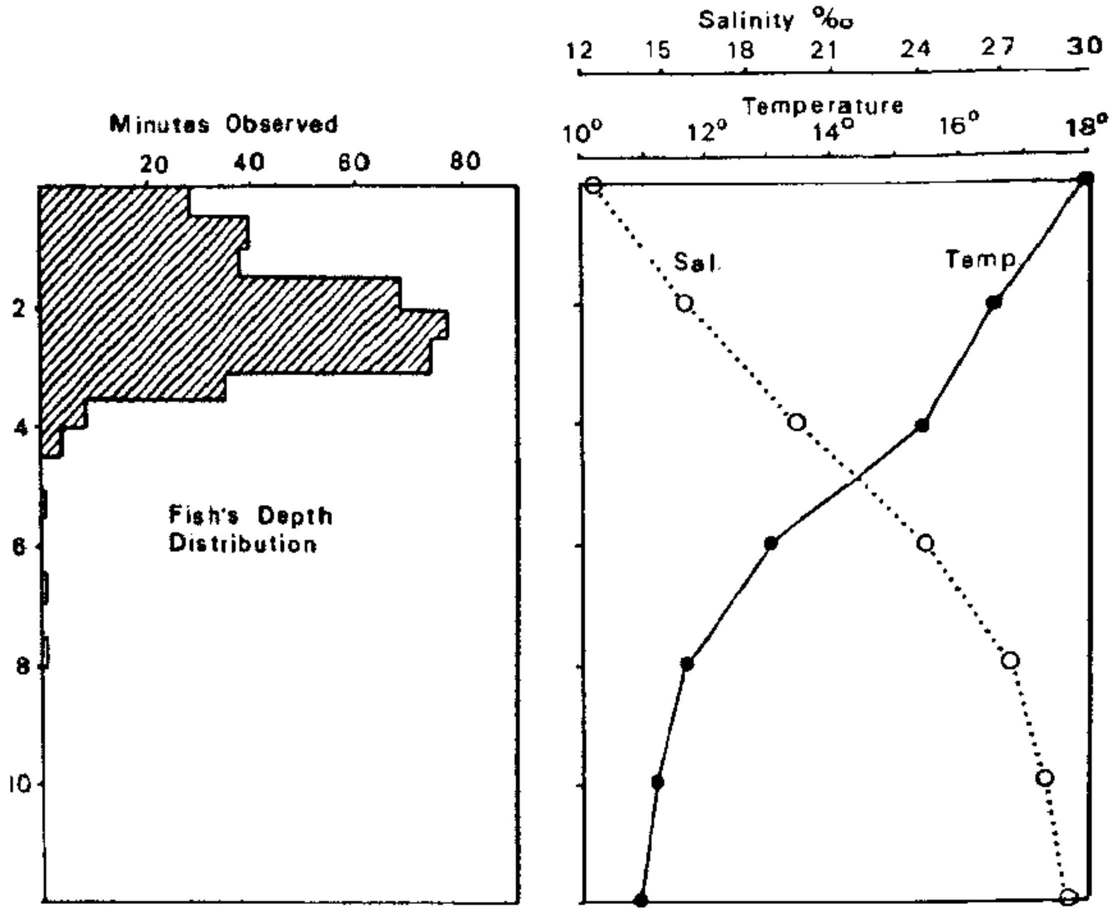

Figure 2. Number of 1-min observations of an adult sockeye salmon in 2-m depth increments and the vertical profiles of temperature and salinity (solid lines = mean, dashed lines = standard deviations). The salmon was tracked in the plume of the Fraser River on 13 September, 1986 and corresponds to fish #8616 of Quinn (1988) and Quinn et al. (1989).

Adult Steelhead Trout in Coastal British Columbia

Methods: In July 1988, six adult summer steelhead (*O. mykiss*—formerly *Salmo gairdneri*) were captured by purse seine in Fisher Channel, part of the complex fjord of Fitzhugh Sound in the central coast of British Columbia (Ruggerone et al., in press). These fish were presumed to be returning to the Dean River, as this river produces an estimated 90% of the steelhead in this region. As with the sockeye and chinook, the fish were anesthetized and a pressure-sensitive transmitter was inserted into the stomach. The steelhead were tracked continuously for 21-50 h in deep, stratified waters.

Results: The steelhead displayed a strong affinity for the surface; five of the six fish spent over 70% of the time within 1 m of the surface. This affinity placed them in the warmest (ca. 14°) and least saline water available. Depending on distance from the Dean River, surface salinities varied from about 20 ppt (123 km away) to 2 ppt (4 km from the river's mouth). Daytime travel rates averaged 2.2 km/h but the steelhead moved markedly slower at night (1.1

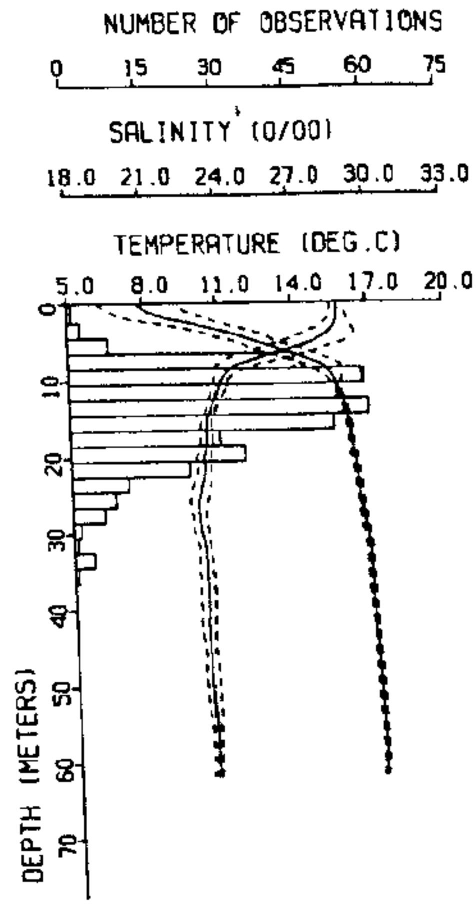

Figure 3. Depth distribution of an adult chinook salmon and vertical profiles of temperature and salinity. The salmon was tracked in the lower Columbia River on 28 August, 1987 (Olson 1989).

km/h). In fact, the fish appeared to virtually stop swimming at night and most of their displacement could be attributed to drift. The steelhead swam in open water but also showed a greater tendency than the sockeye or chinook to follow shorelines.

DISCUSSION

The reliance of many juvenile salmonids on estuaries for feeding, physiological transition and perhaps predator avoidance seems likely but the details of fish movements have not been adequately explained. The coho results reviewed here represent the only tracking study conducted on Pacific salmon smolts in an estuary. The functional significance of the observed coho smolt behavior patterns is tentatively interpreted as follows. Smolts swam into the current after release. The tendency to swim faster against ebbing currents would tend to minimize seaward displacement. The smolts eventually located regions of weak currents and commenced holding. Some combination of foraging, osmoregulatory adaptation or responses to water currents might result in holding behavior but it is unclear what triggers emigration from the holding areas.

Comparable studies on Atlantic salmon (*Salmo salar*) have not yielded consistent results. Smolts accomplished much of their migration by drift in Maine's Penobscot River estuary (Fried et al. 1978; LaBar et al. 1978). Studies in two Scottish estuaries revealed contradictory results. In one estuary, hatchery-reared smolts all emigrated within one tidal cycle but wild smolts in another estuary never left within a cycle and some remained as long as 108 h (Tytler et al. 1978). Unfortunately, differences in behavior might be attributed to either the genetic background of the populations (Clarke 1981), characteristics of the estuaries or the rearing history (hatchery or wild) of the fish (Levings et al. 1986).

The patterns of smolt migration in estuaries may be affected by biological characteristics such as the species, its rearing history, physiology and health, and by physical characteristics of the estuary, variations in river discharge or oceanography (Bax 1982), lunar phase, etc. There is too little descriptive information on smolt movements and too few quantified relationships linking physical factors with estuarine residence and emigration to reliably predict the impacts of activities such as dredging on smolts. The natural variation in behavior seems to be considerable and is poorly understood at present.

Adult salmon movements have been much more intensively studied than those of smolts. Groot et al. (1975) tracked sockeye salmon in the estuary of the Skeena River, B.C. and observed fish drifting with ebbing and flooding tides and thus displaced seaward. Stasko (1975) observed similar behavior in some Atlantic salmon in the Miramichi River but other salmon drifted upstream with flooding tides and stemmed ebbing tides, thus achieving net upriver progress. Ultrasonic tracking by Brawn (1982) revealed Atlantic salmon residing in Sheet Harbour, Nova Scotia for weeks and even months during the summer. In light of these studies on sockeye and Atlantic salmon in estuaries, it is not surprising that chinook salmon tracked in the Columbia River estuary did not swim directly upriver.

In virtually all tracking studies, adult salmon appear to swim at relatively efficient (least cost) speeds rather than faster but more energetically demanding speeds. Typical ground speeds are 0.5-1.0 body lengths/sec or about 2 km/h (Madison et al. 1972; Stasko et al. 1973; Stasko et al. 1976; Westerberg 1982a). When currents speeds were recorded and used to estimate actual swimming speed, salmon were still found to be swimming at about 0.5-1.5 lengths/sec (Smith et al. 1981; Quinn 1988).

The homeward migration of salmon in coastal waters seems to occur in three phases which blend into each other. First, salmon migrate from distant locations to the estuary or vicinity of their home stream. This migration seems to be initially rather rapid in sockeye, chum and pink salmon but declines as fish near their home river (e.g., Verhoeven and Davidoff 1962; Anderson and Beacham 1983). On the other hand, the coastal movements of coho and chinook salmon seem to be generally slow (e.g., Fisher and Pearcy 1987). Following coastal migration, salmon of all species generally hold for a period of varying duration in the estuary or waters near the home river. Finally, upriver migration is initiated and can be as fast as 40 km/day (e.g., Milligan et al. 1985).

In terms of vertical movements, the Columbia River chinook tolerated warm water and rapid changes in temperature and salinity. The steelhead were also close to the surface but the water was cooler than that experienced by the chinook. In contrast to these findings, the Fraser River sockeye salmon apparently avoided the warm, fresh surface waters. There may also be population-specific depth preferences or temperature tolerances and this possibility should be considered in future studies of migratory behavior.

It has been hypothesized that vertical movements by salmon in coastal waters (e.g., Ichihara and Nakamura 1982; Westerberg 1982b) reflect the fish's need to gather olfactory and rheotactic information from the microstructure at the halocline (Westerberg 1982b; Doving et al. 1985). Johnsen (1987) hypothesized that vertical excursions in estuaries and horizontal zig-zagging in rivers facilitate orientation at steep concentration gradients of odors. Use of vertical hydrographic structure for orientation must be integrated with the need for a physiologically suitable environment. Thus depth of travel and vertical excursions may represent tradeoffs between regulation of temperature and salinity and the detection of chemical gradients. Adult salmon movements in estuaries may thus reflect compromises between orientation, optimization of the external environment, physiological adjustment to freshwater and the process of sexual maturation.

The variety of factors affecting vertical and horizontal movements of salmon in estuarine and coastal waters makes it difficult to demonstrate the effects of human activities or man-made structures on migration. However, migratory routes are a form of habitat and factors delaying salmon or forcing them through sub-optimal conditions may have deleterious effects on survival or subsequent reproductive success.

Before we can predict the effects of dredging activities on migration and residence, we need to combine thorough field studies with a realistic model of salmon migratory behavior in estuaries. Once constructed, this model should not only be used to consider direct effects of dredging such as suspended material but should also evaluate the cumulative impact of diverse activities on fish movements. For example, dredging and forest practices can combine to alter the patterns of large woody material in estuaries. In light of the evidence that coho salmon smolts hold in low current areas near structure, the importance of such woody material should be examined.

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Effects of Chronic Turbidity on Anadromous Salmonids: Recent Studies and Assessment Techniques Perspective

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Abstract: Effects of turbidity on anadromous fish can be classified as behavioral, sublethal or lethal (e.g., changes in activity, reduced performance, and death). Sigler et al. (1984) concluded low levels of turbidity affect young steelhead trout and coho salmon. A review of articles relating to various aspects of the effects of turbidity or suspended sediment was completed and data related to research previously completed on steelhead and coho salmon. The difficulty of transposing or extrapolating data from one drainage to another or from one fish species to another is discussed with respect to differences in turbidity measurement techniques and the lack of consistent relationships between optical and physical properties of turbidity causing materials.

INTRODUCTION

Suspended sediment concentration (turbidity) from a variety of land management practices, above a threshold level is the principal impact on fishes. Sigler et al. (1984) found that low levels of turbidity reduce feeding effectiveness, alter migration habits and modify social behavior of the species of salmonids studied, while higher levels of turbidity produce sublethal stress levels which may result in less productive or less fit fish in an exposed population. Lethal effects may also occur in eggs, larvae and juvenile life stages. The predominant cause of mortality is suffocation. Sediments may also indirectly cause mortality by reducing resistance to diseases. Results however are difficult to interpret due to a number of inconsistencies which exist in the literature regarding effects of turbidity (or suspended sediment) on selected species of freshwater, marine or anadromous fishes.

Sediment can be classified as contaminated (by various organics, metals or other toxic materials) or uncontaminated. It can be further described with respect to origin and physical and chemical nature.

The effects of suspended sediments on fish can be discussed with respect to duration of exposure and timing of exposure (in annual cycles or in life stages of the fishes of concern). Effects can be further discussed as behavioral, sublethal or lethal. Behavioral effects are here defined as those changes in actions, movements, or position. Sublethal effects are here defined as those changes which may be physiological in nature and lead to reduced fitness of adults or progeny. Lethal effects are here defined as those which directly result in mortality of the life stage of concern. Newcombe (1986) proposed a system of classification of impact severity based on "pollution intensity," which he defines as the product of concentration of pollutant (including sediment) in water and the duration of exposure and for which numerical values could be assigned. Past research (Sigler 1980, Sigler et al. 1984) focused on specific effects within a set of known physical and chemical values. Behavioral, sublethal and lethal effects were addressed. This paper discusses these effects in relation to a specific suspended sediment source, dredging.

DEFINITION AND MEASUREMENT

Definition and measurement of turbidity, or suspended sediment, can both be accomplished with a variety of instrumentation or techniques. The lack of uniformity with respect to definitions and measurement techniques for "turbidity" and/or "suspended sediment" complicates both the actual measurements and the usefulness of data reported in the literature. Kirk (1983) and Bruton (1985) provided a series of useful definitions. Suspensoids (read "suspended sediments") are solid or colloidal particles which are held in suspension in a liquid. They may be organic or inorganic. Turbidity is defined as an optical property of a liquid causing light scatter and absorption rather than transmittal. Non-specific relationships between measurement devices contribute to the lack of a uniform "turbidity" measurement device. Secchi disk transparency is inaccurate with respect to turbidity due to the inconsistent nature of the curvilinear relationship between the two variables (light factors and physical factors). Dry weight of suspended material does not account for optical properties such as refractory index that may modify the effects of the material on fish. Optical measurements do not assess physical characteristics such as particle angularity. Comparisons between various techniques on the same turbidity agent may not be transferred to other agents due to differences in either refractory or physical characteristics. Even closely related optical techniques such as nephelometric turbidity units (NTU) and Jackson Turbidity Units (JTU) are difficult to relate consistently. These inconsistencies dictate that a multivariate approach to defining and measuring suspended sediment is necessary to adequately describe both the level and the mechanism of fish habitat impacts.

PAST EFFORTS

Data reported to cause behavioral, sublethal, or lethal effects, as defined above, is summarized below for selected recent efforts.

Behavioral Effects

Suspended sediment has been shown to affect avoidance responses, territoriality, feeding and homing behavior (Sigler et al. 1984). Wildish and Power

(1985) report on the avoidance of suspended sediments by rainbow smelt (*Osmerus mordax*) and Atlantic herring (*Clupea harengus harengus*). Apparent threshold for avoidance response by smelt was approximately 20 mg L^{-1} of suspended sediments; herring demonstrated the avoidance response at approximately 10 mg L^{-1}.

Berg and Northcote (1985) reported that short-term pulses of suspended sediments in the 30-60 NTU range caused breakdown of the dominance hierarchies of coho salmon, accompanied by more frequent gill-flaring activity and territorial defense cessation. Return to lower turbidities (0-20 NTU) allowed reestablishment of social organization. Reaction distance to prey and capture success were reduced in 30-60 NTU waters. These behavioral modifications may denote impairment of the fitness (sublethal effects) of salmonid populations exposed to short-term low level suspended sediments.

Whitman et al. (1982) reported that adult male chinook salmon (*Oncorhynchus tshawytscha*) tested for responses to suspended volcanic ash exhibited a significantly reduced preference for home water contaminated with ash. The ash reduced upstream movement in the testing apparatus at test levels of approximately 650 mg L^{-1}. Ash did not alter the proportion of fish that homed.

Sublethal Effects

Kirn et al. (1986) and Emmett et al. (this volume) document the changes in food habits (and subsequent potential growth reductions) of chinook salmon in the Columbia River estuary as a result of suspended sediment from the 1980 eruption of Mount St. Helens. Amphipods, a primary dietary constituent in March, April, and June of 1979 diminished in importance following the eruption, probably a result of both turbidity (affecting feeding) and siltation (affecting amphipod communities).

Bruton (1985) reports a range of effects from adverse (reduced food availability and potential reduced growth/maturation rates) to positive (predator protection and navigational aides) for various levels of turbidity in lakes.

Redding et al. (1987) reported on sublethal effects of topsoil, kaolin clay, and volcanic ash on yearling coho salmon and steelhead trout. Feeding rates were reduced at high exposure concentrations (2-3 g L^{-1}) for both species. While the exposure levels used did not cause mortality, it induced elevation of plasma cortisol levels and blood hematocrits. Reduced tolerance to infection was also reported. This type of physiological stress is sublethal but reduces performance capability.

Sigler et al. (1984) , using small oval and large raceway channels, reported statistically significant reductions in growth (mean daily length or weight increases), density (fish m^{-2}) or grams m^{-2})) and increased rates of out migration (a behavioral effect) for coho salmon (*O. kisutch*) and steelhead trout (*O. mykiss*, formerly *Salmo gairdneri*). Turbidities for all tests ranged from 22 to 265 nephelometric turbidity units (NTUs).

Lethal Effects

Morgan et al. (1983) reported that the survival of eggs and larvae of white perch (*Morone americana*) and striped bass (*M. saxatilis*) could be reduced when exposed to high concentrations of suspended sediments. Hatching success (percent) of white perch eggs was not significantly affected between 50-5,250 mg L^{-1} of suspended sediments. Above sediment concentrations of 1,500 mg L^{-1}, developmental rates were significantly lower (a sublethal effect). Percent hatch of striped bass eggs was not significantly affected by 20-2,300 mg L^{-1} suspended sediments. Above 1,300 mg L^{-1}, development was significantly slowed.

Concentrations of suspended sediments in the 1,626-5,380 mg L^{-1} range caused 15-19% mortality of white perch larvae during one-day exposures and 23-49% mortality during two-day exposures. Concentrations of suspended sediments in the 1,557-5,210 mg L^{-1} range caused 20-34% mortality of striped bass larvae during one-day exposures and 25-57% mortality during 2-day exposures. Covering eggs with 0.45 mm of sediment did not influence hatchability but 100% mortality resulted when white perch eggs were covered with sediment layers greater than 2 mm thick (1.2 mm over the egg).

DISCUSSION

Test results from Sigler et al. (1984), provided a relatively clear pattern of deleterious effects of turbidity on the anadromous fishes studied. Growth was significantly reduced for fish exposed for as little as 14 days over a wide range of turbidities reflective of natural conditions. While the degree to which growth was reduced may have been affected by the life stage (measured in days subsequent to hatch), it is apparent that all test levels of turbidity and length of exposure had adverse effects. In instances where significant differences developed in either size (weight or length) (Tables 1, 2, 3, and 4) or in density (numbers or weight per unit of area) (Figures 1, 2, and 3) reflecting reduced individual ability to compete for resources (food/space) and subsequently impacting the population through lessened reproductive potential, the long-term implications for individual fish stocks or populations could be critical.

Information presented on effects of suspended sediment conflicts with data from some earlier research which addressed the effects of various sediments on older (e.g., mature) fish. Much of the earlier research focused on observed lethal effects and did not address the more subtle behavioral and/or sublethal effects.

Sigler (1980) determined the relationships between physical and optical properties of the turbidity causing agents utilized in those experiments. While relationships could be established within acceptable statistical limits, no attempt was made to provide a mechanism for transposition of these conversions to additional turbidity agents or to additional test situations because of the inherent inconsistencies noted above. The lack of consistent reporting protocols to express the levels of turbidity being tested or observed prevented comparisons with some

Table 1. Results of turbidity tests on steelhead trout in oval channels.

Test duration turbidity (NTU)	Mean daily length increase (mm)	Mean daily weight increase (g)	Density at end of test	
			fish/m^2	g/m^2
Test A (21 days)				
Clear water	0.42	0.019	3.8	3.2
Turbid water (80)	0.36	0.016	4.0	3.1
Test B (15 days)				
Clear water	0.19	0.002	1.3	0.3
Turbid water (72)	0.15	-0.001	0.3	0.1
Test C (19 days)				
Clear water	0.53	0.014	1.0	0.4
Turbid water (51)	0.25	0.000	0.3	0.1
Test D (17 days)				
Clear water	0.56	0.017	0.7	0.6
Turbid water (59)	0.04	0.002	5.3	3.6
Test E (19 days)				
Clear water	0.19	0.010	1.7	1.6
Turbid water (45)	-0.01	0.000	2.5	1.8

Small numbers of fish remaining at the conclusion of the tests precluded statistical analysis in this series. There is some inconclusive evidence of slower growth in turbid water channels.

published studies. Papers reviewed for the present effort indicate that this problem still requires resolution. Without inclusion of more complete physical and chemical data in published reports on turbidity or suspended sediments, it is not possible for researchers to make direct or even speculative comparisons among studies. Use of a set standard of measurement would facilitate future comparisons.

The approach suggested by Newcombe (1986) has merit with respect to standardizing reported turbidity values to noted effects. What Newcombe's proposed protocol lacked was a mechanism to address the differences in optical and mechanical properties of the material of interest. A standardized reporting format for turbidity which incorporates levels and duration of exposure as Newcombe proposed falls short of completely describing the causative agent and therefore loses some usefulness. The optimal reporting protocol would allow incorporation of a series of modifying values which assess optical differences (e.g., light penetration) and mechanical differences (e.g., angularity) in a usable fashion.

Table 2. Results of turbidity tests on steelhead trout in raceway channels.

Test duration turbidity (NTU)	Mean daily length increase (mm)	Mean daily weight increase (g)	Density at end of test fish/m²	Density at end of test g/m²
Test A (14 days)				
Clear water				
Upper channel	0.39	0.012	17.1	7.0
Lower channel	0.49	0.015	8.4	2.1
Turbid water (48)				
Upper channel	0.23	0.002	8.5	3.9
Lower channel	0.39	0.011	0.2	0.1
Test B (19 days)				
Clear water				
Upper channel	0.41	0.018	23.8	13.4
Lower channel	0.43	0.021	4.0	1.4
Turbid water (38)				
Upper channel	0.22	0.007	14.5	8.9
Lower channel	0.25	0.008	0.8	0.3
Test C (17 days)				
Clear water				
Upper channel	0.66	0.024	18.4	11.3
Lower channel	0.62	0.023	9.9	3.6
Turbid water (49)				
Upper channel	0.39	0.009	22.2	12.9
Lower channel	0.35	0.009	9.5	3.3
Test D (19 days)				
Clear water				
Upper channel	0.52	0.046	33.3	48.0
Lower channel	0.46	0.040	5.8	4.7
Turbid water (42)				
Upper channel	0.22	0.020	21.8	29.0
Lower channel	0.22	0.019	9.7	9.0

Differences in growth and density between fish in clear and turbid water were statistically significant for combined upper and lower channels: final weight ($F = 31.67$, $P =$).003), final length ($F = 36.64$, $P = 0.0002$), and mean daily length increase ($F = 46.61$, $P = 0.0001$). See also Figure 1.

Table 3. Results of turbidity tests on coho salmon in oval channels.

Test duration turbidity (NTU)	Mean daily length increase (mm)	Mean daily weight increase (g)	Density at end of test fish/m^2	Density at end of test g/m^2
Test A (14 days)				
Clear water	0.37	0.007	9.2	4.2
Turbid water (86)	0.16	-0.005	0.5	0.2
Test B (13 days)				
Clear water	0.38	0.013	7.0	4.0
Turbid water (45)	0.27	0.007	2.7	1.3
Test C (11 days)				
Clear water	0.36	0.020	3.5	2.6
Turbid water (22)	0.16	0.011	2.7	1.7
Test D (14 days)				
Clear water	0.31	0.006	7.3	6.9
Turbid water (31)	0.24	0.011	6.5	5.6
Test E (15 days)				
Clear water	0.31	0.009	5.3	3.7
Turbid water (23)	0.13	0.000	6.7	3.9

Differences in growth between fish reared in clear and turbid water were statistically significant: weight ($F = 31.52$, $P = 0.005$), length ($F = 35.09$, $P = 0.004$), mean daily weight increase ($F = 30.87$, $P = 0.005$), mean daily length increase ($F = 35.18$, $P = 0.004$). See also Figure 2.

Even with that achievement, comparison between different river drainages and different fish species at different life history stages would be tenuous. A reporting system which incorporates all aspects of turbidity would be useful. However, without a mandate from field level researchers, it is unlikely that a system will emerge.

A standard measurement protocol which incorporates a mathematically manipulated version of physical characteristics (e.g., angularity) and optical properties can be developed. The protocol could be as simple as adding or multiplying the two values together, or separating them by a decimal point (e.g., 500.6) where 500 reflects a standardized optical measurement (e.g., NTU) and the 6 represents a physical measurement of angularity for a sample processed using an accepted standard method.

Table 4. Results of turbidity tests on coho salmon in raceway channels.

Test duration turbidity (NTU)	Mean daily length increase (mm)	Mean daily weight increase (g)	Density at end of test	
			fish/m^2	g/m^2
Test A (14 days)				
Clear water				
Upper channel	0.49	0.022	7.3	5.5
Lower channel	0.45	0.020	4.1	2.5
Turbid water (11-32)				
Upper channel	0.38	0.011	4.0	2.9
Lower channel	0.30	0.007	3.1	1.7
Test B (31 days)				
Clear water				
Upper channel	0.50	0.040	15.8	27.7
Lower channel	0.61	0.050	12.7	15.0
Turbid water (41)				
Upper channel	0.29	0.021	6.6	13.7
Lower channel	0.35	0.025	5.3	7.0
Test C (21 days)				
Clear water				
Upper channel	0.35	0.021	36.4	32.4
Lower channel	0.32	0.023	27.6	14.9
Turbid water (49)				
Upper channel	0.06	0.004	12.9	11.9
Lower channel	0.07	0.007	11.7	6.9

Differences in growth between fish reared in turbid and clear water were statistically significant as were densities (biomass): weight ($F = 16.33$, $P = 0.006$), length ($F = 19.91$, $P = 0.004$), mean daily length increase ($F = 38.54$, $P = 0.001$), density in numbers were not significantly different ($F = 1.01$, $P = 0.35$), but density as biomass was ($F = 7.21$, $P = 0.036$). See also Figure 3.

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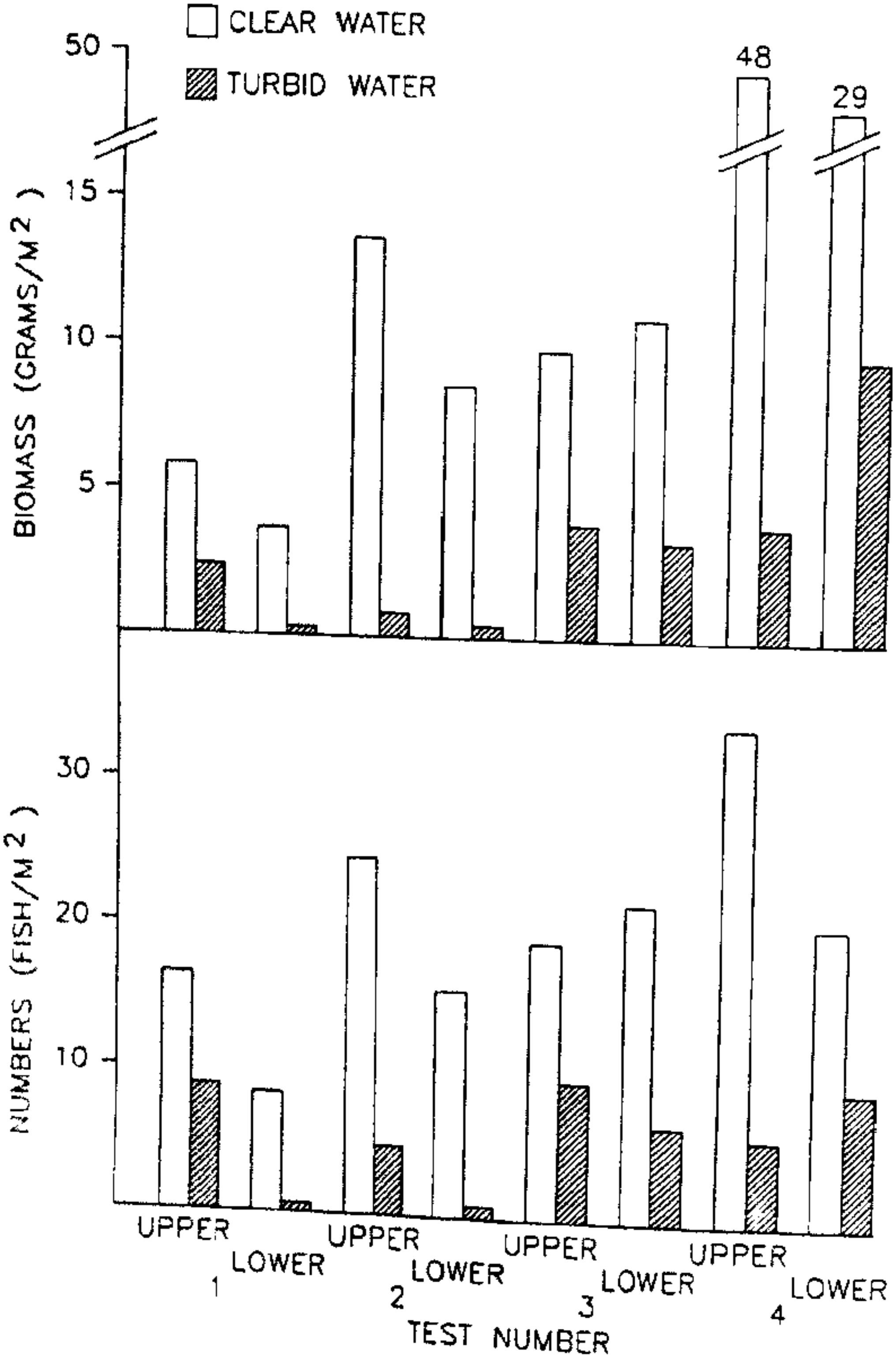

Figure 1. Densities (fish/m^2 and biomass/m^2) of steelhead trout from four tests in the large channels.

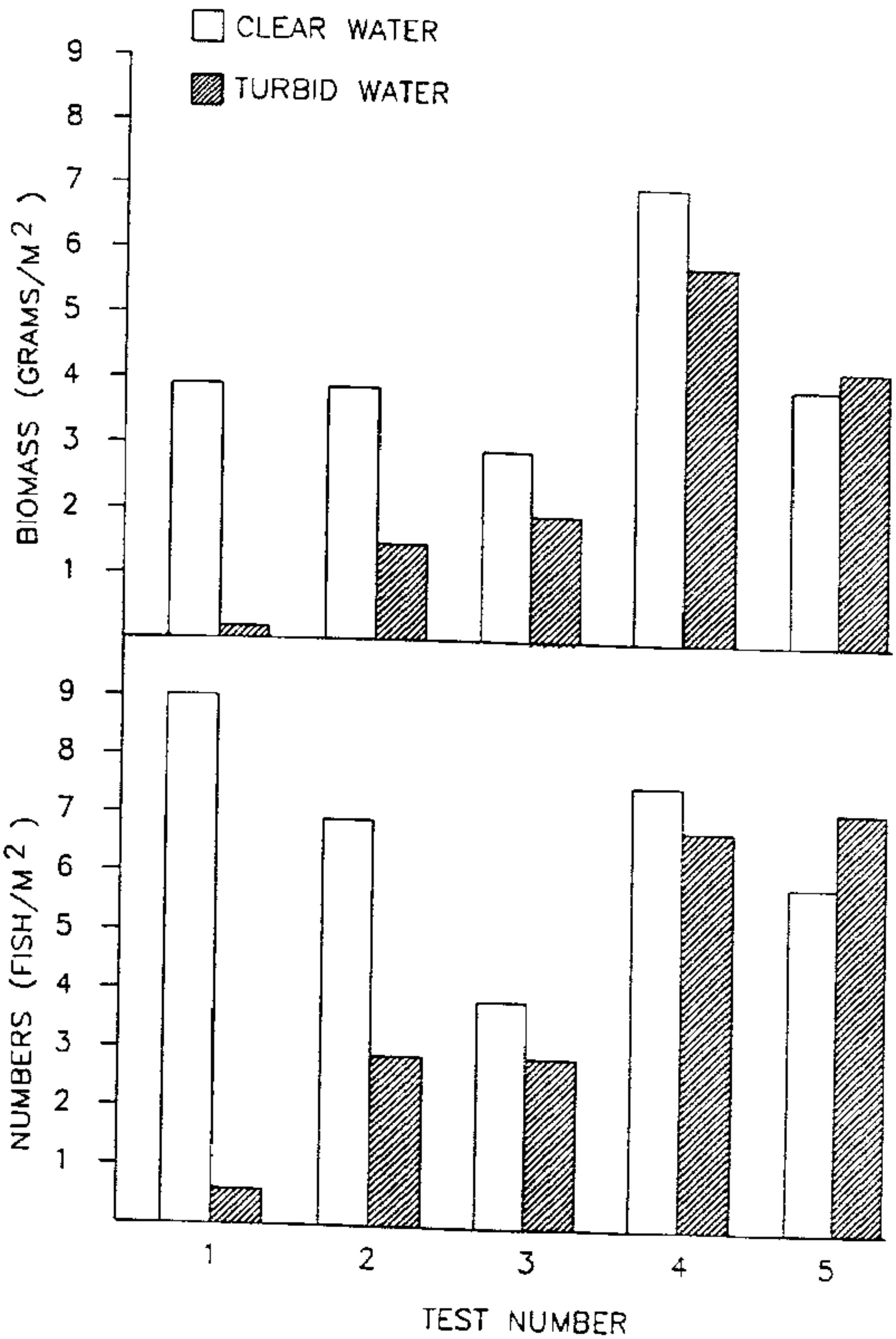

Figure 2. Densities (fish/m^2 and biomass/m^2) of coho salmon from five tests in oval channels.

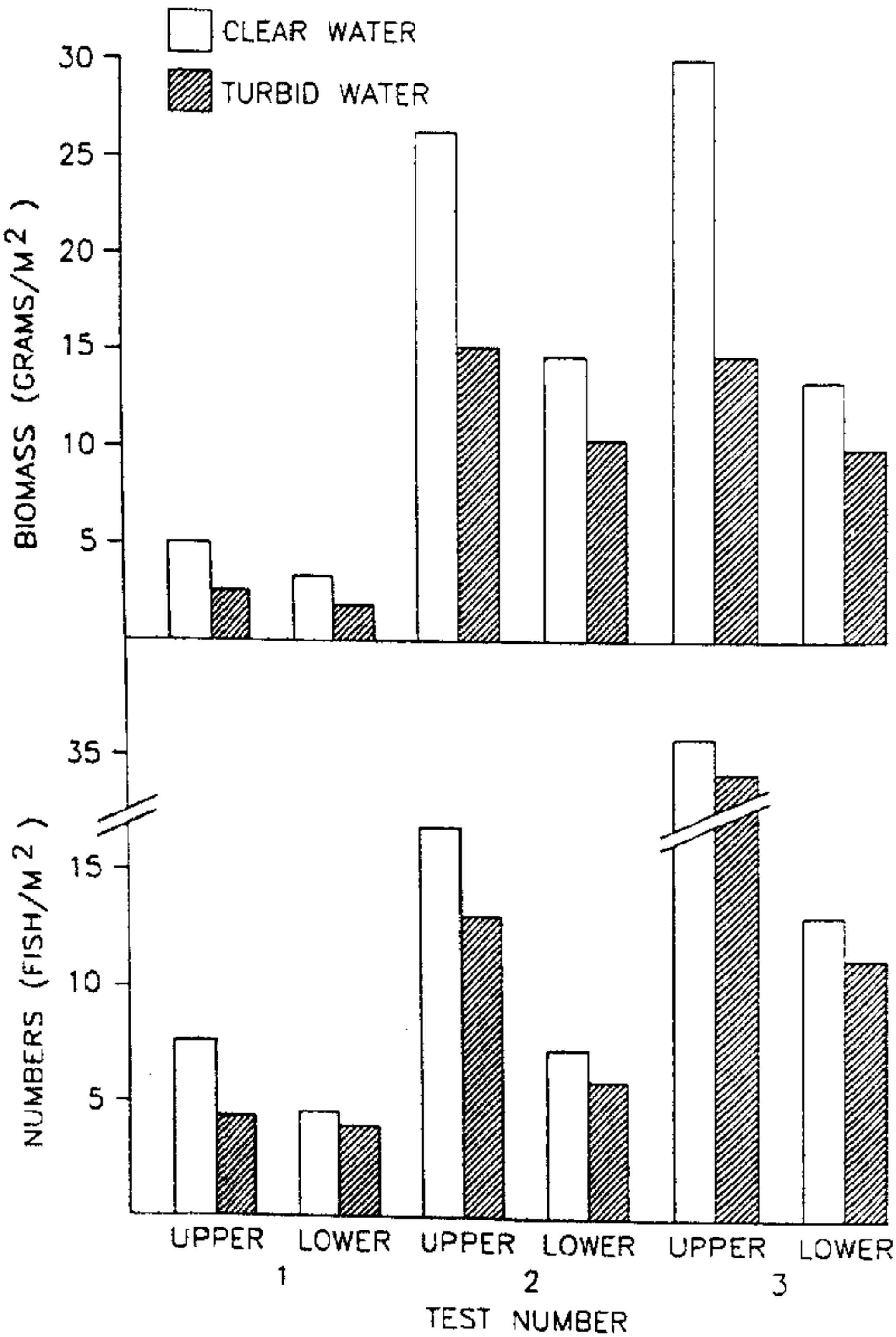

Figure 3. Densities (fish/m^2 and biomass/m^2) of coho salmon from three tests in the large channels.

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Potential Effects of Dredging on Early Life Stages of Striped Bass (*Morone saxatilis*) in the San Francisco Bay Area: An overview

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Abstract: Potential relationships between exposure to increased suspended sediment concentrations and striped bass hatching success, larval foraging, and adult migration and spawning in San Francisco Bay and the Sacramento-San Joaquin Delta were examined. Very little documented, quantitative information is available on the potential effects on striped bass resulting from exposure to increased turbidity and suspended sediment concentrations associated with either dredging activity or dredge spoil disposal practices. The limited information that is available suggests that striped bass are not affected adversely by exposure to increased suspended sediments at the concentrations encountered. This conclusion is consistent with the observation that striped bass have been able to establish an abundant population in San Francisco Bay and the Sacramento-San Joaquin Delta system—an environment characterized by high naturally occurring suspended solids concentrations and high turbidity conditions.

INTRODUCTION

Striped bass have become a prominent fish since being introduced into the San Francisco Bay and Sacramento-San Joaquin Delta system (Bay-Delta). Native to Atlantic and Gulf of Mexico coastal waters striped bass quickly became established in the Bay-Delta after their introduction in the late 1800s. The Bay-Delta population remains the only major one on the Pacific Coast and supports one of the region's most intensive sport fisheries. Recent reductions in the striped bass population, however, have prompted interest in the environmental requirements of the species. Because of the considerable amount of dredging and dredge spoil disposal activity within the Bay-Delta system, interest has been expressed by local resource and regulatory agencies and the public in the potential effects that dredging activity may have on the Bay-Delta striped bass population.

To assess the potential effects that dredging activity may have on the early life stages of striped bass, it is important to consider: (1) the life history characteristics of striped bass, (2) the characteristics of the specific dredging and spoil disposal activities, (3) the potential biological responses of each life history stage exposed to increased levels of suspended sediments associated with dredging activity, and (4) the relationship between the seasonal and geographic distribution of striped bass life stages and dredging activity. In this paper we focus on the information available on potential pathways and mechanisms through which increased suspended sediment concentrations may affect striped bass eggs and larvae including:

- hatching success
- larval survival
- larval foraging success
- adult migration and spawning.

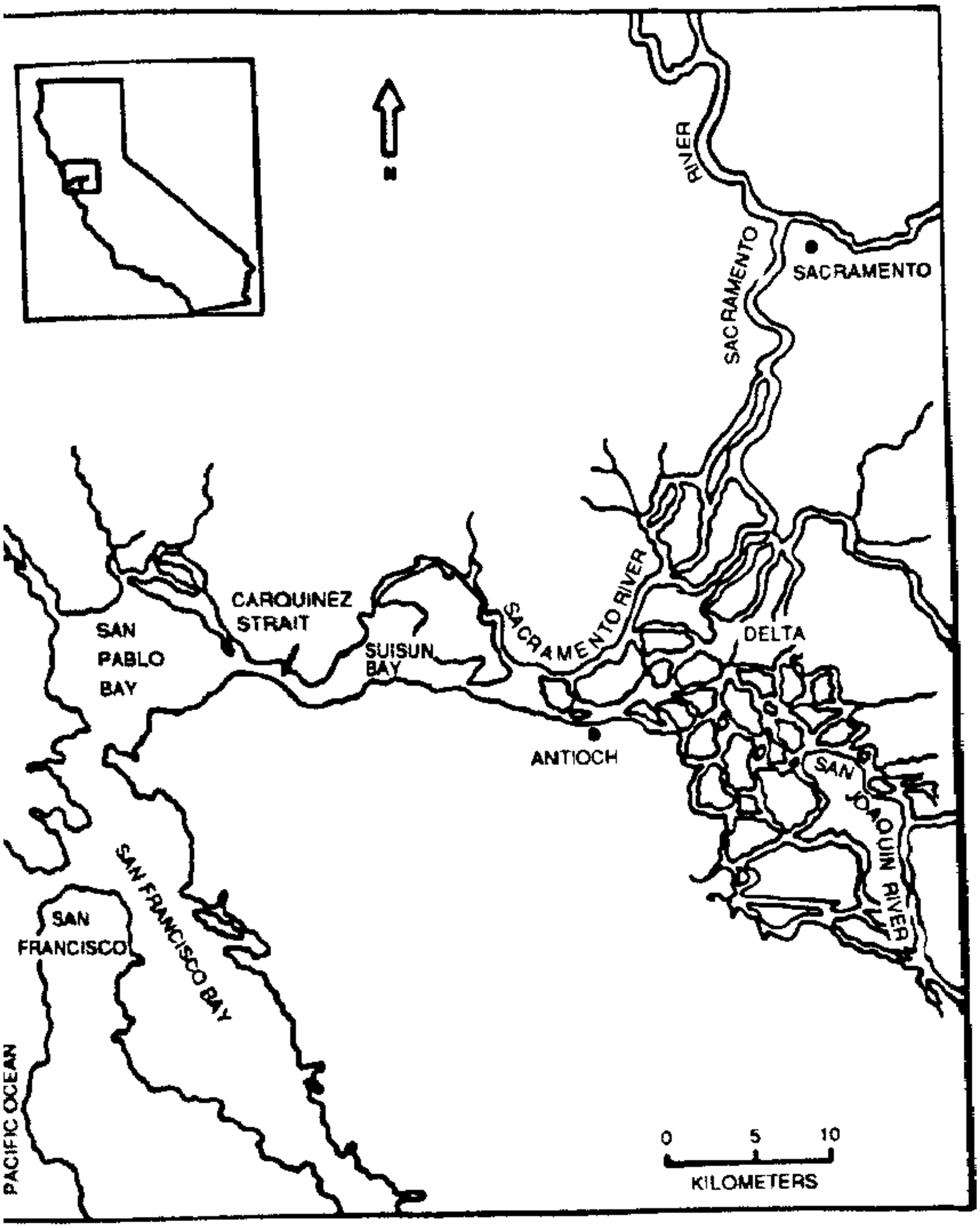
Potential effects associated with exposure of striped bass to toxic materials (e.g., increased heavy metal concentrations, etc.), although important, were not a topic of this workshop and are not addressed in this paper.

LIFE HISTORY OF THE SAN FRANCISCO BAY STRIPED BASS POPULATION

Adult striped bass use San Francisco Bay and coastal waters for feeding during the summer months. In fall they move upstream, concentrating in San Pablo and Suisun Bays (Fig. 1). In spring, mature bass move upstream to spawn in the Sacramento River upstream of Sacramento and in the San Joaquin River upstream of Antioch. Spawning generally occurs from April through early June.

Males mature between age one and three and females usually between age four and six. Both sexes commonly live and reproduce until they are 12-15 years old (females up to 30 years old). Spawning is triggered by water temperatures and occurs generally above 65°F. San Joaquin River spawning generally precedes Sacramento River spawning because of warmer water temperatures in the San Joaquin River.

Striped bass females are prolific; large females have been reported to contain up to 4.5 million eggs. Striped bass fecundity (the number of eggs produced) increases as the bass grow older. Eggs are planktonic and non-adhesive. Suspension and initial exposure to freshwater are both essential for high egg survival (Talbot 1966; Setzler et al. 1980). After fertilization, exposure to low salinity water improves survival of striped bass eggs and larvae. Eggs hatch 29 to 80 hours after fertilization depending on water temperatures (Setzler et al. 1980) and tend to be concentrated near the bottom in midchannel.



igure 1. San Francisco Bay and the Sacramento-San Joaquin river delta.

Larvae (<6mm) carried out of the upper Sacramento and San Joaquin Rivers become distributed throughout the western Delta and Suisun Bay above the entrapment zone, the location of which is determined by freshwater outflow. Larger larvae and juveniles are distributed farther downstream into the entrapment zone (Hassler 1988). Small larvae are distributed fairly evenly across the main channels of the western Delta while larger larvae and juveniles tend to concentrate near the shoreline (Hanson, unpubl. data). Young striped bass remain in and above the entrapment zone during their first summer, but migrate into San Pablo Bay and the central Bay in fall. Some young striped bass remain in the Delta during the winter, and these tend to concentrate in the lower San Joaquin River.

Striped bass feed on a variety of organisms. Larval striped bass initially prey on small zooplankton such as copepods and cladocerans including *Cyclops* spp. and *Eurytemora affinis* (Hassler 1988). Juvenile bass feed primarily on mysid shrimp (*Neomysis mercedis*), but also feed on other invertebrates such as amphipods and small fish. Subadult and adult striped bass are opportunistic predators (Hassler 1988); they are piscivores, and the species eaten depends on availability. During spring and summer, northern anchovy is the most important prey for subadult and adult bass in the south and central Bay and in San Pablo Bay. Pacific herring is an important food item during the winter months when they enter San Francisco Bay to spawn. During the spring, salmon smolts are in the diet of striped bass. Bay shrimp (*Crangon* spp.) and mysids are also important food items for yearling and older bass. Species of fish that prey on young bass in the Delta include catfish, largemouth bass, yellowfin goby, and older bass. Some invertebrates, such as mysids and shrimp (*Palaemon macrodactylus*) may feed on larval striped bass.

Striped bass were initially the object of an intensive commercial fishery in San Francisco Bay waters. Commercial fishing for striped bass in San Francisco Bay was banned in 1935, in part because of conflicts between sport and commercial fisheries. The striped bass sport fishery is currently one of the most important in San Francisco Bay and the Delta. However, the number of striped bass caught by recreational anglers and the catch-per-unit-effort have declined over the past several decades. In recent years, angler harvest has ranged from approximately 100,000 to 400,000 striped bass per year (White 1986).

Additional information on the biology and environmental requirements of striped bass is available in reviews by Talbot (1966), Setzler et al. (1980), and Hassler (1988).

DREDGING AND SPOIL DISPOSAL ACTIVITY IN SAN FRANCISCO BAY

Dredging activity occurs at locations throughout San Francisco Bay, the Delta, and portions of both the Sacramento and San Joaquin rivers. Dredging is primarily undertaken to maintain water depths in navigable waters and to enlarge and deepen channels and port facilities to accommodate larger vessels such as petroleum tankers and containerized cargo vessels. Deepwater channels and port facilities are maintained in San Francisco, Oakland, San Pablo Bay and Suisun Bay and up the Sacramento and San Joaquin rivers.

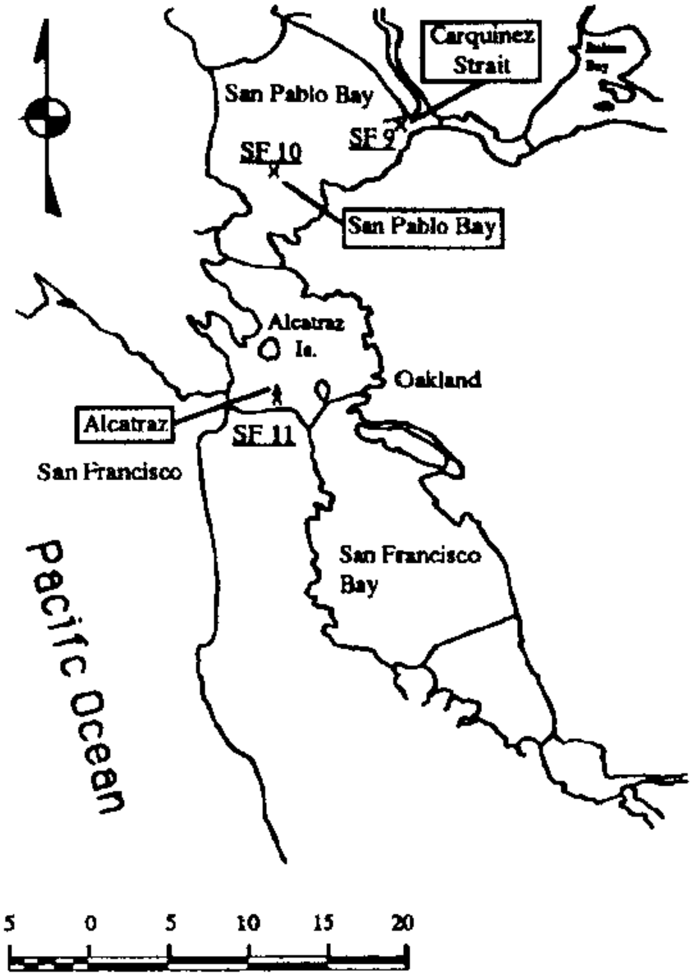
Deposition of silt and sediment associated with freshwater runoff from the Sierra and Central Valley drainage area poses a significant maintenance dredging problem within the San Francisco Bay area. During 1984-85, for example, approximately 8.6 million cubic yards of material were dredged from San Francisco Bay (Dowgiallo et al. 1986), with substantial volumes of additional sediment removed from Sacramento and San Joaquin River channels. Dredging within the San Francisco Bay area includes the use of cutterhead, clamshell bucket, and suction-hopper dredges. Sediment resuspension characteristics of each of these dredges have been investigated in field studies by McLellan et al. (1989) and others (See LaSalle, this volume).

Data available on the increase in suspended sediment concentrations associated with dredging activity in San Francisco Bay and elsewhere (LaSalle, this volume; McLellan et al. 1989) show that substantially elevated sediment concentrations are localized in the immediate area downstream of the dredging operation. Suspended sediment concentrations reported by Sustar et al. (1976; cited in LaSalle) for operation of a Hopper Dredge in a silty-clay area within San Francisco Bay are shown in the following table:

	Suspended Sediment (mg L^{-1})	
Water Column Depth	Average Range	Maximum
Surface	65-210	210
Mid-depth (5 m)	33-64	110
Bottom (10 m)	58-743	1,110

In another set of field measurements made downstream of a bucket dredge operating in San Francisco Bay, suspended sediments 50 m downstream were generally less than 200 mg L^{-1} (see LaSalle). The suspended sediment levels associated with this dredging activity averaged 30-90 mg L^{-1} above a background level of 40 mg L^{1}.

Dredge spoil disposal has been limited to three designated sites within San Francisco Bay (Fig. 2), although the Alcatraz site has been the primary disposal location. Dredged sediments, primarily from Delta and San Joaquin River channels, have also been deposited on Delta levees and islands. In part as a result of concern over the potential impacts of dredge spoil disposal on fisheries populations and water quality within San Francisco Bay, alternative spoil disposal locations (both on land and in coastal waters) are currently being considered.



Description of Designated Sites

SF 11 ALCATRAZ ISLAND 37°49' 17"N, 122°25' 23"W
Distance: About 0.3 nautical miles S of Alcatraz Island.
Depth: 95-160 ft., average 130 ft.
Size: Radius of 1,000 ft.

SF 10 SAN PABLO BAY 38°00' 28"N, 122°24' 55"W
Distance: 2.6 nautical miles NE of Pt. San Pedro at black and white buoy.
Depth: 38-40 ft., average 39 ft.
Size: Rectangle 1,500 x 3,000 ft., long. axis bearing 50° true.

SF 9 CARQUINEZ STRAIT 38 03' 50"N, 122 15' 55"W
Distance: 0.8 nautical miles from Mare Island Strait entrance.
Depth: 28-56 ft., average 42 ft.
Size: Rectangle 1,000 x 2,000 ft., long. axis bearing 80° true.

Figure 2. Designated dredge spoil disposal sites within San Francisco Bay.

Effects of Suspended Sediments on Early Life Stages of Striped Bass

The following section presents a brief overview of available information on the potential effects of dredging and disposal activity on striped bass eggs, larvae, and juveniles. All of the available laboratory and experimental data on the relationship between exposure to suspended sediments and hatching success of eggs, larval survival, and foraging success of striped bass larvae have originated from research conducted using Atlantic striped bass stocks. An extensive field monitoring database does exist on larval and juvenile striped bass within the San Francisco Bay and Delta system; however, no specific studies or tests have been conducted to evaluate the potential adverse effects of dredging activity on the San Francisco Bay striped bass population.

Striped Bass Hatching Success and Development

Schubel and Wang (1973) conducted an extensive series of controlled laboratory studies to investigate the potential effects of suspended sediment exposure on the hatching success of striped bass eggs. Striped bass eggs were obtained from the following sources: (1) naturally fertilized eggs collected with plankton nets from the Chesapeake and Delaware (C and D) Canal, (2) eggs obtained from naturally ripe females and artificially fertilized, and (3) eggs taken from females ripened by hormone injection and artificially fertilized. All adult striped bass used in these tests were collected from the C and D Canal. In addition to control samples, striped bass eggs were exposed to concentrations of naturally occurring Chesapeake Bay sediments at target concentrations of 25, 50, 100, and 500 mg L^{-1}. Actual suspended sediment concentrations varied substantially from the intended concentrations during the two to three-day experimental observation period (for example, measured concentrations ranged from 69 to 1,067 mg L^{-1} within tests having a target concentration of 500 mg L^{-1}). Sediments used in these tests were collected from Chesapeake Bay and separated in the laboratory to select those particles with settling velocities less than 5×10^{-3} cm/sec. Sediments used in these tests comprised primarily three clay minerals (illite, chlorite, and kaolinite), quartz, and feldspar. Test sediments were mixed with Chesapeake Bay water collected from the striped bass spawning grounds for use in each experiment. The suspended sediment concentration was monitored in each test by removing a 200 ml water sample approximately every 4 to 8 hours. The sample was then filtered through pre-weighed 0.5-µm membrane filters, desiccated, and re-weighed. Two replicate tests (with 100 to 200 eggs per test) were performed for each target suspended sediment concentration. Differences in relative and absolute hatching success for striped bass eggs exposed to various suspended sediment concentrations at different developmental stages (e.g., gastrula, early blastula) were tested for statistical significance by analysis of variance.

Schubel and Wang (1973) exposed striped bass eggs to suspended sediment concentrations ranging from an average of 0 to 678 mg L^{-1} during the blastula and gastrula stage. These tests were characterized by a high variance in relative hatching success between replicate tests at each suspended sediment concentration. The absolute hatching rate, expressed as a percentage of the eggs in each test which successfully hatched, of striped bass eggs exposed to various suspended sediment concentrations during the gastrula stage (Fig. 3), showed no evidence of a relationship between hatching rate and exposure to suspended sediment concentrations within the range tested. These results showed that suspensions of natural, fine-grained sediments in concentrations of up to approximately 700 mg L^{-1} had no statistically significant effect on hatching success of striped bass eggs. Thus, Schubel and Wang concluded from these studies that exposure of striped bass eggs to concentrations of suspended sediments within the range tested did not significantly affect hatching success. Similar conclusions were drawn by Schubel and Wang for yellow perch, white perch, and alewife included in their laboratory investigation.

Schubel et al. (1974) and Auld and Schubel (1974) also reported results of experiments in which striped bass eggs were exposed to elevated suspended sediment concentrations. Essentially, the experimental design of these tests was similar to Schubel and Wang (1973) but several modifications were made to reduce variability in suspended sediment concentrations. Using this modified system, sediment suspensions with concentrations up to 1,000 mg L^{-1} were maintained within approximately ±10%. Sediments collected from Chesapeake Bay having a settling velocity of less than 5×10^{-3} cm/sec and particle size ranging from approximately 1 to 4 µm were used in these tests. In addition to controls, striped bass eggs (100-300 per test) were incubated at various stages of

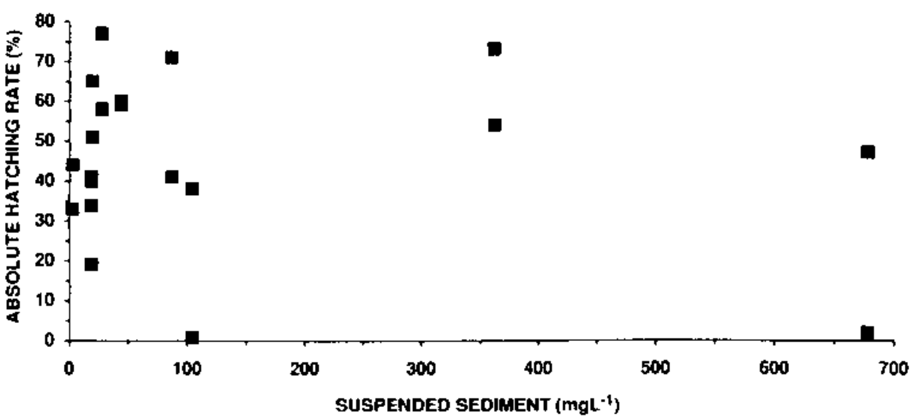


Figure 3. Hatching success of striped bass eggs exposed during the gastrula stage to suspended sediment (Source: Schubel and Wang 1973).

embryonic development in Chesapeake Bay waters. Schubel et al. (1974) and Auld and Schubel (1974) failed to detect any significant difference between hatching success for controls and those eggs exposed to suspended sediment concentrations of 50, 100, or 500 mg L^{-1}. However, hatching success of striped bass eggs exposed to a suspended sediment concentration of 1,000 mg L^{-1} was found to be significantly ($p < 0.005$) less than corresponding controls.

Morgan et al. (1983) conducted a series of laboratory studies exposing Atlantic Coast striped bass eggs to suspended sediment concentrations ranging from 20 to 2,300 mg L^{-1}. The suspended sediments, collected from Chesapeake Bay, were characterized by particle sizes less than 0.0625 mm (98% by weight). The experimental design and testing equipment were similar to those of Auld and Schubel (1974). Up to 50 eggs were used in each test. In addition to hatching success, the rate of egg development was also monitored. Mortality and rate of development in each test were expressed as a percentage of the rates observed in the controls. Data were tested using analysis of variance. Results of these experiments are shown in the following table:

Suspended sediment mg L^{-1}	Hatch (% of controls)	Development rate (% of controls)
95	82	98
265	80	98
595	84	89
890	65	60*
1,100	56	58*
1,600	62	52*
1,900	68	50*
2,300	62	52*

Note: *denotes significant difference $p < 0.05$.

No statistically significant difference was detected between the hatching success of striped bass eggs incubated under control conditions and those exposed to elevated suspended sediment concentrations. The development rate of eggs exposed to suspended sediment concentrations above 890 mg L^{-1} was found to be significantly slower than the development rate for controls.

Larval Survival

Auld and Schubel (1974) also conducted a series of eleven experiments to determine the potential relationship between larval survival and exposure to elevated suspended sediment concentrations. Approximately 100 to 200 larvae, 4 to 8 hours old, were exposed to suspended sediment concentrations of 50, 100, 500, and 1,000 mg L^{-1} and corresponding controls for periods of either 48 or 72 hours. Percent survival at the end of each test was used in an analysis of variance to test statistically for differences in survival between each suspended sediment treatment and controls. Results of these tests showed that larval survival was significantly reduced at suspended sediment concentrations of 500 mg L^{-1} ($p < 0.0025$) and 1,000 mg L^{-1} ($p < 0.005$). No significant differences in survival were detected for suspended sediment concentrations of either 50 or 100 mg L^{-1}.

Morgan et al. (1983) conducted a series of laboratory bioassays to determine the response of larval striped bass to elevated suspended sediment concentrations. The experimental design and testing apparatus were similar to those described by Auld and Schubel (1974). Naturally occurring sediment collected from Chesapeake Bay and the C and D Canal were used in these tests. The suspended sediments ranged in size from 0.0625-2.0 mm (2% by weight), 0.00391-0.0625 mm (38%), and less than 0.00391 mm (60%). Approximately 25-50 striped bass larvae were exposed in each test to suspended sediment concentrations ranging from 1,557 to 5,210 mg L^{-1} for periods of 24 or 48 hours. Larval mortality was determined at the completion of each test. Results of these tests are presented below:

Suspended sediment mg L^{-1}	Mortality %	
	24 hours	48 hours
1,557	20	25
2,450	21	29
3,260	21	37
5,210	31	57

No statistically significant difference was detected between the percent mortality of larval striped bass in these tests and the corresponding controls. The 24-hour LC_{50} was estimated to be 20,417 mg L^{-1} and the 48-hour LC_{50} was 6,292 mg L^{-1}. Morgan et al. (1973) reported a 48-hour LD50 value of 3,400 mg L^{-1} for striped bass larvae exposed to naturally occurring fine-grained Chesapeake Bay sediments in laboratory tests.

Larval Foraging Success

Environmental factors, such as increased turbidity and suspended sediment concentrations associated with dredging activities, could potentially interfere with the ability of larval striped bass to locate and capture prey and could therefore contribute to reduced growth rates and increased larval mortality. Following absorption of their yolk sac, larval striped bass (approximately 6 mm in length and larger) forage on small zooplankton, including calanoid and cyclopoid copepods and cladocerans.

Breitburg (1988) conducted a series of laboratory experiments to examine the effects of suspended sediment on the numbers and sizes of prey (zooplankton) consumed by larval striped bass. Striped bass larvae (10-22 mm TL) from Atlantic coast stocks were exposed to prey organisms composed of either the cladoceran *Daphnia pulex* or an assemblage of naturally occurring zooplankton (approximately 80% of the assemblage was the calanoid copepod *Eurytemora affinis*) at sediment concentrations of 0, 75, 200, and 500 mg L^{-1} (kaolin clay suspensions). Three striped bass larvae were used in each 25-minute feeding test. Between eight and ten replicate tests were conducted at each suspended sediment concentration for natural copepod assemblages and between eight and nineteen replicate tests were performed with *Daphnia pulex*. Prey densities of approximately 100-125 zooplankton L^{-1} were used in each test. Analysis of variance was used to determine whether suspended sediment concentration significantly affected the sizes and number of prey eaten. Prey consumption by larval striped bass feeding on natural copepod assemblages (Fig. 4a) was significantly lower (approximately 40% less) for those fish tested at sediment concentrations of 200 and 500 mg L^{-1} compared to larvae tested at sediment concentrations of 0 and 75

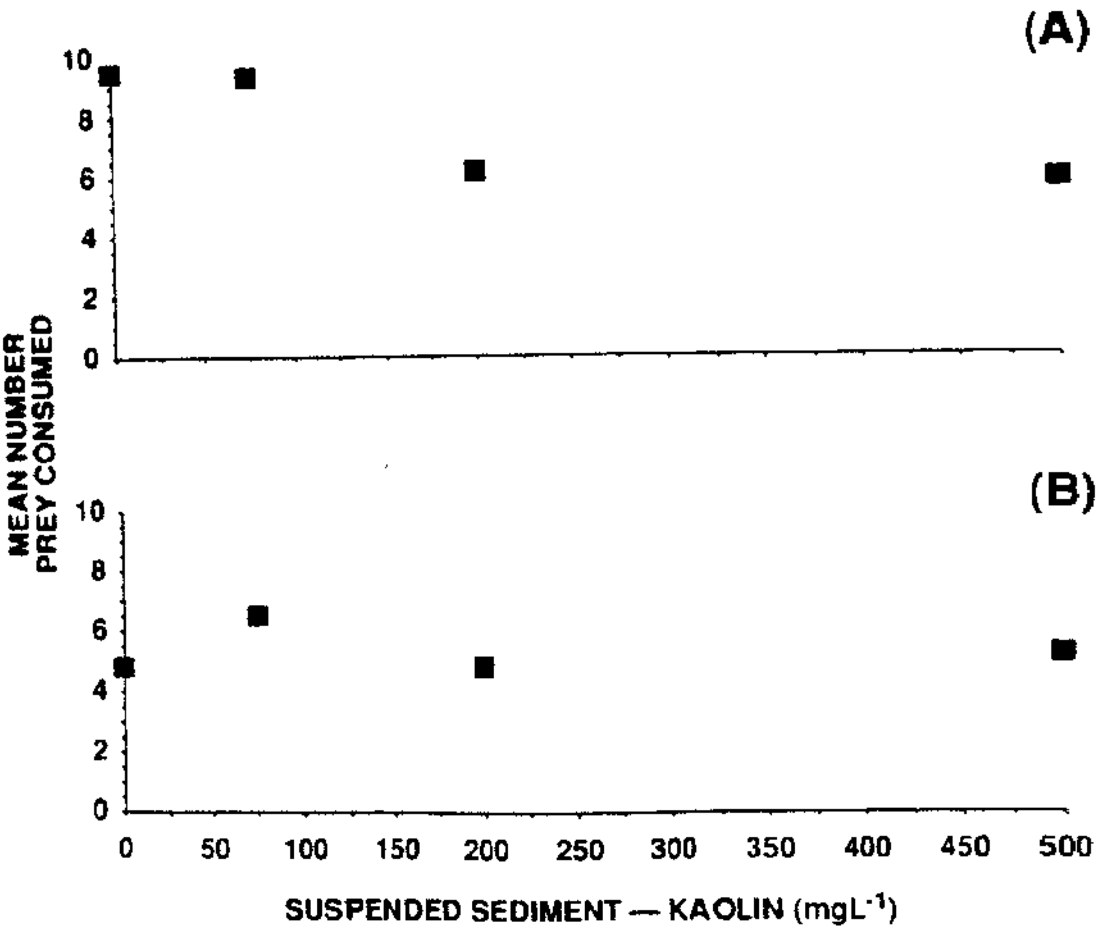


Figure 4. Consumption of (A) a natural assemblage of zooplankton and (B) *Daphnia pulex* by larval striped bass exposed to various suspended sediment concentrations (Source: Breitburg 1988).

mg L^{-1}. In contrast to the results observed for the natural zooplankton assemblage, no statistically significant difference in the number of *Daphnia pulex* consumed by striped bass larvae was observed over the range of sediment concentrations tested (Fig. 4b). Results of these studies, although not conclusive, suggest the possibility of reduced larval foraging success associated with increased suspended sediment levels.

Adult Migration and Spawning Activity

Adult striped bass migrate from the San Francisco Bay region upstream into the Sacramento and San Joaquin rivers for spawning. During their upstream migration adults may encounter and behaviorally respond to elevated suspended sediment levels associated with dredging activity. No mark-recapture tagging or radiotelemetry studies have been conducted on migrating striped bass in the Sacramento-San Joaquin Bay-Delta system which could indicate the potential behavioral response of adults encountering increased suspended sediment concentrations associated with dredging activities. Furthermore, the potential influence of increased suspended sediment levels on spawning activity by striped bass in the Sacramento River and in the Delta has not been investigated.

Flemer et al. (1968) reported no significant differences in the densities or distribution of striped bass eggs and larvae in areas of increased turbidities associated with dredging activity and spoil disposal in upper Chesapeake Bay.

DISCUSSION OF THE RELATIONSHIP OF DREDGING ACTIVITY AND EARLY LIFE STAGES OF STRIPED BASS

Although there has been considerable debate and discussion related to the environmental conditions and factors which contribute to the population dynamics of striped bass in the San Francisco Bay area, there is very little documented, quantitative information available on the potential effects of exposure to increased suspended sediment concentrations associated with dredging activity and dredge spoil disposal practices. The limited information that is available from laboratory studies on Atlantic Coast stocks and indirect field observations from the San Francisco Bay system suggests that striped bass would not be adversely affected by these dredging activities. Striped bass spawning occurs upstream of the designated within-bay spoil disposal sites (Fig. 2), and striped bass eggs do not generally occur in the higher salinity marine waters where dredge disposal takes place. Striped bass eggs could be exposed, however, to elevated suspended sediment concentrations associated with dredging activity in the Sacramento and San Joaquin rivers, Delta channels, and Suisun Bay (Fig. 1).

Striped bass are extremely prolific, broadcast spawners which inhabit estuarine and river systems which are naturally characterized by high turbidity and suspended sediment concentrations (Talbot 1966). Since striped bass eggs are non-adhesive, there is little tendency for suspended sediments to adhere to or accumulate on the eggs. Clean, viable striped bass eggs have been collected from the C and D Canal (Schubel and Wang 1973) and from the Sacramento-San

Joaquin Delta (J. Wang, pers. comm.) during periods of high naturally occurring suspended sediment levels. Furthermore, since striped bass eggs are planktonic, exposure of eggs to localized elevated suspended sediment concentrations will be of short duration.

Laboratory studies such as those conducted by Schubel and Wang (1973), Schubel et al. (1974), and Auld and Schubel (1974), provide experimental evidence that suspended sediment concentrations within the range tested (up to approximately 700 mg L^{-1}) do not adversely affect striped bass egg development and hatching success, although there is evidence that the rate of egg development may be delayed at higher suspended sediment concentrations. Data reported by LaSalle (this volume) show that suspended sediment concentrations in the downstream plume of a dredging operation are at levels less than 1,000 mg L^{-1}. For example, suspended sediment concentrations 50 m downstream of a bucket dredge in San Francisco Bay were generally less than 200 mg L^{-1}, although higher peak suspended sediment concentrations were reported in the immediate downstream vicinity of a hopper dredge in silty clay substrate in San Francisco Bay. These field data show that exposure of planktonic striped bass eggs to high (>500 mg L^{-1}) suspended sediment concentrations will be localized within small areas in the immediate downstream plume of a dredging operation and that the duration of exposure will be short (minutes or hours, depending on stream velocities). The laboratory studies in which reduced egg hatching success was detected after prolonged (days) constant exposure to a 1000 mg L^{-1} suspended sediment concentration are not representative of field conditions in the Sacramento River, Delta, or Suisun Bay, where striped bass eggs may be exposed to dredging activity.

Striped bass larvae are also able to tolerate exposure to elevated suspended sediment concentrations. Results of acute bioassays reported by Auld and Schubel (1974), Morgan et al. (1973), and Morgan et al. (1983) are consistent in showing that larval striped bass are tolerant of suspended sediment exposure at concentrations and durations of exposure associated with dredging activity (LaSalle, this volume). Striped bass larvae are planktonic, and therefore their exposure to elevated suspended sediment concentrations greater than 500 mg L^{-1} resulting directly from dredging activity would be localized to the immediate area in the downstream sediment plume (LaSalle, this volume) for a short period of exposure (minutes or hours). Therefore, the effects on striped bass larvae from exposure to elevated suspended sediment concentrations resulting from dredging are expected to be minimal.

Laboratory tests of the ability of larval striped bass to detect and capture zooplankton suggest that prey consumption is reduced at higher turbidity levels (>200 mg L^{-1}) for naturally occurring assemblages of zooplankton (Breitburg 1988). No relationship, however, was detected between prey consumption and suspended sediment concentrations in tests conducted with *Daphnia pulex*. Chesney (1989) reported results of a series of laboratory growth studies in which larval striped bass were held 25 days in water having suspended sediment (kaolin) concentrations of 0, 50, 100, and 150 mg L^{-1}. Fish were fed the copepod *Eury-*

temora affinis at a density of 100 L^{-1}. The growth rate of larval bass stocked at a rate of 4 L^{-1} was then monitored in each treatment. Results of these tests showed that suspended sediment concentrations within the range tested did not significantly affect striped bass growth. Although the duration of these growth studies (25 days) is not representative of the relatively short period of exposure for a larval striped bass encountering a suspended sediment plume associated with dredging activity, the results are consistent with the conclusion that striped bass larvae are well adapted for foraging on zooplankton under conditions of relatively high natural suspended sediment concentrations (and low light levels associated with high turbidity) in areas such as Suisun Bay and the Delta (Fig. 1) where striped bass larvae are abundant during the spring and summer. Available data are inconclusive on the relationship between larval and juvenile striped bass foraging success and exposure to increased turbidity levels associated with dredging or spoil disposal activities.

To the extent that striped bass locate and capture zooplankton using visual cues, the ability of larval and juvenile fish to successfully locate prey may be decreased by a reduced visual field and reduced light intensities associated with increased suspended sediment levels. The potential significance of increased suspended sediment concentrations on foraging success of larval striped bass is dependent to a large extent on the densities of zooplankton occurring in the areas where larval striped bass are concentrated. Breitburg (1988) hypothesized that potential effects of increased turbidity on larval striped bass prey capture would increase as prey densities decline. This would imply that the average distance between predator and prey increases and the fish become increasingly dependent on their ability to detect and capture distant prey in order to meet their growth and nutritional requirements. The dynamics of larval striped bass predator-prey relationships are, however, poorly understood, and no field studies have been conducted to investigate the potential effect of increased turbidity on feeding success of larval striped bass in the San Francisco Bay-Delta system. Those portions of the Bay-Delta system where foraging larval and juvenile striped bass are located, principally the southern Delta, Suisun Bay, and San Pablo Bay (Fig. 1), are characterized by relatively high naturally occurring suspended sediment and turbidity levels. Although dredging activity does occur periodically in these areas larval and juvenile striped bass exposure would be of limited duration. The principal areas of the Bay where increased suspended sediment concentrations exist as a result of spoil disposal activity are located outside of the principal geographic region where larval and juvenile striped bass foraging occurs.

Subadult and adult striped bass inhabit San Francisco Bay and the Sacramento-San Joaquin Delta and nearshore coastal waters, where they forage on macroinvertebrates and a variety of bait fish, including northern anchovy, Pacific herring, and threadfin shad. The geographic distribution of adult and subadult striped bass within San Francisco Bay and the Delta coincides with areas of dredging and spoil disposal, and therefore elevated suspended sediment concentrations resulting from these activities may have an effect on the behavior and foraging success of striped bass. However, no quantitative information has been compiled on the ability of adult striped bass to locate and successfully capture prey when exposed to conditions of increased turbidity and suspended solids concentrations. Furthermore, no detailed information exists on the behavioral response of forage fish such as Pacific herring and northern anchovy to increased turbidity conditions. Qualitative observations made by recreational fisherman and commercial bait fisherman in San Francisco Bay have led to the speculation that schools of forage fish may disperse and avoid areas of increased turbidity. Commercial bait fisherman have reported a marked reduction in the densities of northern anchovy in those areas in which dredging or disposal activities have resulted in increased turbidity and suspended sediments (S. Cressey, pers. comm.). Although it has been speculated that exposure of forage fish to a localized turbidity plume may disrupt schooling behavior and elicit an avoidance response, thereby making prey detection and capture more difficult for adult striped bass, this potential effect has not been documented by other than indirect qualitative observations. Definitive studies on the behavioral responses of subadult and adult striped bass and their forage species to elevated suspended sediment levels have not been conducted in San Francisco Bay. Also, neither the geographic distribution of adult striped bass nor recreational angler success as it relates to localized dredging activity or spoil disposal practices has been investigated in San Francisco Bay.

More specifically, there is no accurate information on the behavioral re-
sponse of juvenile and adult striped bass and their prey species encountering
localized turbidity plumes such as those occurring downstream of a dredging
operation. No direct measurements are available which document dredging-
induced mortality resulting from the entrainment of larval or juvenile striped
bass caused by suction dredge operations. Furthermore, there exists little quali-
tative information on the potential indirect effects on striped bass distribution
patterns, growth, or survival associated with channelization and the associated
changes in localized current and hydrologic flow patterns.

CONCLUSION

San Francisco Bay and the Sacramento-San Joaquin Delta system are
characterized by high, naturally occurring suspended sediment concentrations and
high turbidity conditions associated with wind-driven turbulence, wave action,
and freshwater outflows from the Sacramento and San Joaquin rivers. A large
proportion of the estuarine environment is shallow, with extensive bottom areas
of accumulated fine-grained silts and clays which are easily resuspended by water
currents, wind, and wave-generated turbulence. Striped bass have been able to
establish high population abundances at these naturally high turbidity levels.
On the basis of their life history strategies and the characteristics of the
environments within which striped bass reside, and the limited available infor-
mation from laboratory and field studies, it is reasonable to conclude that the
early life stages of striped bass are minimally susceptible to effects associated
with increased turbidity and suspended solid concentrations resulting from
dredging and spoil disposal activities.

Current Status of Dredging and Disposal in San Francisco Bay

Since the 1988 workshop on the effects of dredging on anadromous fish, a number of public agencies and private groups have expressed concern over the potential impacts of dredging and dredged sediment disposal on water quality and fisheries resources, including striped bass population in San Francisco Bay. As a result of these concerns, the California Regional Water Quality Control Board-San Francisco Bay Region circulated a staff report in September 1988 entitled "A Review of Issues and Policies Related to Dredge Spoil Disposal in San Francisco Bay." Subsequently the Regional Board held public hearings and received comments on the staff report. After responding to agency and public comments, the Regional Board adopted a policy in July 1989 regarding dredging and sediment disposal in San Francisco Bay (source: Regional Board Summary Report, June 21, 1989), a portion of which is presented below:

- All aquatic disposal of dredged sediment from new work shall be prohibited in San Francisco Bay. This prohibition shall take effect on December 31, 1991. For the purposes of this policy, new work shall include any modification that expands the character, scope, or size of the existing, authorized project.

  Maintenance work refers to periodic dredging which is necessary for the continued use of existing waterways, ports, harbors, or marinas. For example, new work would include the following activities: excavation below current design depth; excavation to accommodate new construction; or the widening of channels or berths to accommodate larger vessels.

  The designation of an ocean disposal site by the EPA and Corps, under Section 102 of the Marine Research, Sanctuaries, and Protection Act, is scheduled for completion by December 31, 1991.

- The Regional Board recognizes that the continued disposal of maintenance work will require a demonstration that there are no significant or irreversible impacts occurring from the disposal of maintenance dredged material in San Francisco Bay. The Regional Board recognizes the Corps expertise in this area and encourages the Corps to coordinate the Demonstration Program. The Regional Board will participate in the development and review of the Demonstration Program prior to adopting further restrictions on dredged material disposal.

- Dredged material disposal in San Francisco Bay shall be restricted to bay sediment.

- The following volume targets shall be utilized each calendar year (i.e., January to December) at each aquatic disposal site (Fig. 2):

Alcatraz Island	(SF-11)	4.0 million cubic yards
San Pablo Bay	(SF-10)	0.5 million cubic yards
Carquinez Straits	(SF-09)	2.0 million cubic yards (NY)
		3.0 million cubic yards (WY)

An examination of disposal patterns at all aquatic disposal sites in San Francisco Bay revealed that the Carquinez Straits area may be influenced by wet weather events. The volume targets for the Carquinez Straits disposal site are 3.0 million cubic yards for wet and above normal years (WY) and 2.0 million cubic yards for all other year classifications (NY).

- The following volume targets shall be utilized on a monthly basis at each aquatic disposal site:

Alcatraz Island	(SF-11)	
October-April		1.0 million cubic yards
May-September		0.3 million cubic yards
San Pablo Bay	(SF-10)	
Any month		0.5 million cubic yards
Carquinez Straits	(SF-09)	
Any month		1.0 million cubic yards

- The Regional Board will restrict dredging or dredge disposal activities during certain periods in order to protect the beneficial uses of San Francisco Bay. These beneficial uses include water contact recreation, ocean commercial and sport fishing, marine habitat, fish migration, fish spawning, shellfish harvesting, and estuarine habitat. These restrictions may include but are not limited to:
  a. Dredging activities from December through February in selected sites along the waterfront where Pacific herring are known to spawn; and
  b. Disposal activities at the Carquinez Straits site (SF-09, Fig. 2) during spring and fall in order to protect striped bass and salmon migrations.

The Regional Board shall continue to encourage land and ocean disposal alternatives whenever practical. The Regional Board will implement these measures through its issuance of Waste Discharge Requirements, Water Quality Certification under Section 401 of the Clean Water Act, or other orders. The Regional Board also encourages EPA and the Corps to expedite the designation of ocean disposal sites in an effort to reduce or eliminate sediment disposal within San Francisco Bay.

The California State Water Resources Control Board recently reviewed and adopted the Regional Board dredging and sediment disposal plan for San Francisco Bay with the exception of the schedule for prohibiting disposal within the bay and designation of an ocean disposal site which was extended beyond the December 31, 1991 date contained in the original Regional Board recommendation.

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Sublethal Effects of Dredged Sediments on Juvenile Salmon

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Abstract: *Histological, immunological, physiological and behavioral responses of juvenile salmonids to suspended sediments were reviewed. Of particular interest were results reported for Fraser River sediments since these naturally occurring sediments are routinely dredged. Histopathology was the least sensitive of the methods for detecting responses. Reports of depressed immunological response owing to suspended sediment exposure were limited. This was considered an area for further research. Biochemical stress indicators were highly sensitive to suspended sediment exposure but appeared to be better suited for detecting effects caused by chronic rather than acute exposures. Feeding and avoidance responses were also highly sensitive. In particular, surfacing response to avoid suspended sediments had implications for short-term exposures since this behavior would increase the potential for predation by birds.*

INTRODUCTION

The concentration, duration and extent of suspended sediment plumes created by dredging have been the subject of various studies reported at this workshop. There is potential for impact when juvenile salmon are present during dredging in Pacific Coast estuaries. This paper examines research which will help managers judge the potential hazard to juvenile salmon caused by re-suspension of sediments associated with dredging.

Of particular interest are experiments conducted with juvenile salmon using sediments collected from the Fraser River, B.C., Canada, where these sediments are dredged to maintain navigation channels. Sediments in the lower Fraser River are composed largely of quartz, feldspar, chlorite, mica and amphibole; montmorillonoid is an important component of the clay fractions and kaolin is present in the coarse clay fractions of some samples (Mackintosh and Gardner 1966). Fine sediments were collected from sandbars in the Fraser River estuary at low river flow (Servizi and Martens 1987). Samples were air-dried and sized. More than 90% of the fine sediment was 5 µm while there was a trace at 55 µm (geometric mean diameter). Particles were described as angular to subangular. These fine sediments were used in a series of tests to measure survival, histopathological, biochemical, respiratory and behavioral responses of juvenile sockeye (*Oncorhynchus nerka*), pink (*O. gorbuscha*), coho (*O. kisutch*) and chinook (*O. tshawytscha*).

HISTOPATHOLOGY

Juvenile sockeye survived a 96 h exposure to 3,100 mg L^{-1} (18% of 96 h LC50) Fraser River suspended sediment but experienced hypertrophy (swelling) and necrosis (cell death) of gill tissue (Servizi and Martens 1987). These authors reported sediment particles lodged between gill lamellae but no evidence of excess mucus. Noggle (1978) reported variable gill damage among juvenile coho which died during exposure to soil derived suspended sediments. He concluded that gill tissue is the primary site of injury for acute exposure to suspended sediments. Redding and Schreck (1987) exposed yearling coho and rainbow trout (steelhead now *O. mykiss*, formerly *Salmo gairdneri*) to high and low concentrations of topsoil (2-3 g L^{-1} and 0.4-0.6 g L^{-1}) for 2 or 7-8 d. Steelhead were also exposed for 2 d to high and low concentrations of suspended clay and volcanic ash. In all cases and both species, gill tissues from treated fish were similar to that of control fish. McLeay et al. (1987) examined reports of gill tissue injury owing to suspended sediments derived from placer mining and concluded that response was highly variable and may be due to differing sediment characteristics, and perhaps biological and experimental factors.

Injury to gill tissue can provide an entry for infectious organisms. Redding and Schreck (1987) reported reduced tolerance among yearling steelhead to the bacterial pathogen *Vibrio anguillarum* following exposure to 2.5 g L^{-1} topsoil for 2 d. Infection occurred even though there was no microscopic evidence of gill injury. Reduced tolerance to infection is consistent with reduced leucocrit (number of white blood cells) reported for Arctic grayling (*Thymallus arcticus*) exposed to suspended sediments (McLeay et al. 1987).

Photomicrographs were used to detect phagocytosed (intercellular) sediment particles in gills and spleens of juvenile sockeye, coho, chinook and pink salmon exposed to Fraser River suspended sediments (D. Martens, Cultus Lake Laboratory, pers. comm.). Particle mineralogy determined by X-ray diffraction analysis confirmed their origin as typical of Fraser River suspended sediment. However, there was no clear correlation between numbers of intercellular particles and exposure concentration. Goldes et al. (1986) reported phagocytosis of suspended clay kaolin by gills of rainbow trout. Phagocytosis of suspended sediments has implications for dredging since this is a mode of entry for radionuclides, toxic sediments and toxic substances adsorbed to sediments.

Because of migration patterns, riverine hydraulics and the nature of dredging operations, juvenile salmon may be exposed to suspended sediments for less than 24 h. However, there were no reports of examinations of gill tissue for exposures of less than 48 h. To help clarify the actual hazard potential, histopathology of gill tissues should be made for brief exposures using representative sediments. Such studies should include disease challenges applied during suspended sediment exposure or immediately thereafter.

SALINITY TOLERANCE

Dredging is usually associated with estuarine habitat where juvenile salmon are adapting to sea water. Since gills play an important role in osmoregulation, injury to gill tissue may interfere with sea water adaptation. Servizi and Martens (1987) exposed feral sockeye smolts to Fraser River suspended sediments for 96 h prior to transfer to sea water or sediment-free fresh water for an additional 48 h. Compared to a control group, suspended sediment levels of 14,400 mg L^{-1} caused a significant decrease ($p<0.01$) in body moisture. On the other hand, plasma chloride levels were not significantly different than controls. An increase in standard deviation of plasma chloride levels suggested that osmoregulatory capacity was reduced slightly after exposure to suspended sediments. However, in no case did osmoregulatory capacity decrease to a level where plasma chlorides reached the acute stress level of 170 mEq L^{-1} (Clarke and Blackburn 1977).

Pink salmon fry move directly to estuarine areas after emerging from incubation areas. The effect of Fraser River suspended sediments on seawater tolerance of this species was measured at Cultus Lake Salmon Research Laboratory using successive groups of feral pink fry from nearby Sweltzer Creek. Pink fry were exposed to suspended sediments for 96 h and then transferred to hyper-salinity bioassays to measure seawater tolerance. A group of control fry was treated in parallel. For small fish, hyper-salinity bioassays are an alternative to plasma chloride measurements (C. Clarke, Pacific Biological Station, Nanaimo, B.C., Canada, pers. comm.).

Pink fry exposed to 11,400 mg L^{-1} suspended sediment experienced 28% mortality and salinity tolerance was 36‰ compared to 42.7‰ for the parallel control group (Table 1). No mortalities occurred at 5,800 mg L^{-1} or less and salinity tolerances of experimental and control fish were equal for these cases.

Redding and Schreck (1987) used plasma sodium content to evaluate osmoregulatory performance of yearling coho and steelhead transferred to 26‰ sea water following exposures of 7-8 d to 2-3 g L^{-1} suspended sediment. No mortality occurred upon transfer to seawater and plasma sodium levels were statistically similar in treatment and control fish.

Judging from these tests using sockeye, pink, coho and steelhead, osmoregulatory capacity would not be impaired owing to short-term exposure (<4 d) to sublethal concentrations of suspended sediments in freshwater.

BIOCHEMICAL STRESS RESPONSE

Elevated levels of plasma glucose are considered to be a secondary response to stressors (Wedemeyer and McLeay 1981). Servizi and Martens (1987)

Table 1. Salinity tolerance of pink salmon fry following exposure to suspended sediments.

Sediment exposure		Salinity Tolerance (96 h LC50)	
SS mg L^{-1}	Mortality percent	Post-sediment* ‰	Post control* ‰
11,400	28	36.0	42.7
7,600	3	41.5	43.1
5,800	0	42.6	43.7
3,200	0	44.2	44.2
1,600	0	44.6	44.6

*Based on 60 specimens except 50 for the first test.

reported plasma glucose elevated 39 and 150% among adult sockeye exposed to 500 and 1,500 mg L^{-1}, respectively, of Fraser River suspended sediments. When tests were extended to include underyearling coho, plasma glucose was similarly elevated (D. Martens, Cultus Lake Laboratory, pers. comm.).

McLeay et al. (1987) reported increased plasma glucose values for Arctic grayling exposed to inorganic suspended sediment at concentrations equal to 500 mg L^{-1} and more. When organic sediments were tested, plasma glucose was elevated at ≥50 mg L^{-1} suspended sediments. Noggle (1978) reported elevated plasma glucose levels for coho exposed to sublethal concentrations of suspended sediments.

Redding and Schreck (1987) used elevated plasma cortisol levels to detect stress in yearling steelhead and coho exposed to low (0.3-0.6 g L^{-1}) and high concentrations (2-3 g L^{-1}) of suspended topsoil. Cortisol levels were also elevated when yearling steelhead were exposed to clay and volcanic ash suspensions. Cortisol levels peaked within 9 h and either stayed elevated or had decreased by 48 h.

Judging from the foregoing reports, it can be presumed that salmon which encounter suspended sediment plumes caused by dredging will respond with elevated glucose and cortisol levels. However, these responses are reversible and recovery occurs when the stressor is removed or the fish escapes the plume. If the stress is chronic, a metabolic cost may be incurred.

RESPIRATORY RESPONSE

Berg and Northcote (1985) observed a significant rise in gill-flaring among juvenile coho exposed to ≥20 NTU (Nephelometric Turbidity Units) of turbidity. Feeding success and social behavior were disrupted by turbidities of 30 NTU and more. The particles were described as very angular and were mainly 0.02 to 0.06 mm. No measurements of suspended sediments were reported.

Gill flaring is a means by which fish can create sudden changes in buccal cavity pressure. This action appears to be a "cough" and may be a reaction to gill irritation caused by suspended sediment. McLeay et al. (1987) observed increased coughing among arctic grayling exposed to 50 and 100 mg L^{-1} suspended sediments.

Coughing can be measured by connecting the buccal cavity to a transducer/ recorder apparatus via fine surgical tubing. Tests at Cultus Lake laboratory which exposed juvenile coho to Fraser River suspended sediments found coughing frequency increased at concentrations of 190 mg L^{-1} and more (D. Martens, Cultus Lake Laboratory, pers. comm.). For Fraser River suspended sediments this value was equivalent to 20 NTU. Coughing rate returned to normal when test fish were transferred to sediment-free water.

BEHAVIOR

Lloyd et al. (1987) reviewed the effects of turbidity on behavior of fish, especially Arctic grayling, Pacific salmon and trout. These species showed a preference for clear water. Turbid water was reported to interfere with feeding and migration in some cases. Juvenile coho avoided 70 NTU derived from road-side sediments when acclimated to clear water (Bisson and Bilby, 1982). The avoidance threshold was 106 NTU when specimens were previously acclimated to 2-15 NTU. On the other hand, there were occasions when fish acclimated to turbid-water exhibited a fright response and preferred turbid water over clear water. Berg and Northcote (1985) observed breakdown in dominance hierarchy and feeding success among juvenile coho subjected to pulses of turbid water (30 NTU). These authors suggested that the fitness of juvenile fish frequently subjected to suspended sediment pulses may be impaired. Sigler et al. (1984) found that coho and steelhead (30-65 mm) emigrated from or died in experimental channels dosed with clay-derived turbidities in the range 100-300 NTU. Turbidities as low as 25 NTU were associated with reduced fish growth.

Servizi and Martens (1987) reported underyearling sockeye were frequently observed at the surface during bioassays with high suspended sediment concentrations. McLeay et al. (1983) made similar observations of arctic grayling and suggested that the fish may have been seeking lower suspended sediment levels associated with surficial water.

The tendency of juvenile coho to swim near the surface when exposed to suspended sediments was quantified in tests at Cultus Lake Laboratory. Surfacing frequency increased with concentration of suspended sediment over the range 140 to 1,350 mg L^{-1} (D. Martens, Cultus Lake Laboratory, pers. comm.). Surfacing frequency was marginally elevated over the range of control values at 200 mg L^{-1}. The latter value is equivalent to 22 NTU for Fraser River sediments.

Behavioral responses to suspended sediments have been reported by several investigators and are initiated in the range 22 to 100 NTU. The cause of these responses has not been identified but since behavioral responses and avoidance occur in similar ranges, avoidance may be prompted by irritation of gill tissues by suspended sediments.

SUMMARY

Juvenile salmon exhibited histological, immunological, physiological, and behavioral responses to suspended sediments. Histopathology occurred in gill tissues at sublethal concentrations but these impacts were less sensitive than other responses. Owing to phagocytosis, suspended sediments composed of or transporting toxic substances may pose a hazard to juvenile salmon. Exposure to suspended sediments reduced tolerance to bacterial infection but further work is needed to define minimum suspended solids exposure levels which cause this effect. Salinity tolerance of juvenile salmon was unaffected by prior exposure to sublethal concentrations of suspended sediments.

Two biochemical indicators, plasma glucose and plasma cortisol, were sensitive indicators of stress caused by suspended sediments. Cough reflex was similarly sensitive to suspended sediment exposure. Both types of responses are reversible when fish re-enter clean water. Potential impact on juvenile salmon owing to exposure to suspended sediment plumes would depend on duration. Chronic exposure (4 d or more) may incur a metabolic cost.

Avoidance and feeding actions of juvenile salmonids were sensitive to suspended sediments. Although juvenile salmonids avoided suspended sediment in laboratory tests, this response may be modified by environmental factors such as salinity, ambient turbidity, temperature gradients and currents in an estuary. However, the tendency of juvenile salmon in laboratory tests to swim near the surface in laboratory tests to avoid higher levels of suspended sediments at depth has implications for the estuarine environment. If juvenile salmon move to the surface to avoid plumes of suspended sediment they will be more vulnerable to bird predation. In such cases, even short-term exposures to dredge plumes could result in significant loss of juvenile salmonids. This subject deserves special consideration in planning dredging operations and should be a consideration for further study.

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Effects of Turbidity on Benthic Foraging and Predation Risk in Juvenile Chinook Salmon

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Abstract: *The foraging behavior of juvenile chinook salmon (*Oncorhynchus tshawytscha*) in conditions of elevated turbidity was investigated in a series of laboratory experiments. These experiments determined the reaction distance to invertebrate prey, the perceived risk to a model predator, and the foraging rate of chinook on benthic* Tubifex *worms, in turbidity conditions ranging from 0 to 800 mg L^{-1}. Both reaction distance and perceived risk declined inversely with turbidity. Foraging rates on* Tubifex *were highest at intermediate levels (50-200 mg L^{-1}) and lowest at 0 mg L^{-1} (control) and 800 mg L^{-1}. The results suggested a tradeoff between the effects of reduced reaction distance and perceived risk to predation. The pertinence of these findings to the impact of dredging-induced increases in suspended sediment level on seaward migrating juvenile salmonids was discussed.*

INTRODUCTION

Estuaries are regions of high primary and secondary productivity and are of great importance to larval and juvenile fishes. Estuaries function both as nurseries and as physiological transition zones. During their seaward migration, juvenile salmonids often inhabit estuarine tidal marsh and slough habitats for periods of up to several months (Levy and Northcote 1982, Simenstad et al. 1982).

Acting as reservoirs for sediment derived from their drainages, estuaries generally possess elevated turbidity levels. Suspended sediment levels also vary with the ebb and flow of the tide. In periods of high flow, especially during spring runoff, estuarine waters can become highly turbid (>100 mg L^{-1}), due to both the increased sediment load from upstream sources and the resuspension of existing sediment within the estuary itself. Larval and juvenile fish are usually present in the estuary during the spring freshet and the subsequent peak in turbidity level (Birtwell et al. 1987, Dunford 1975, Levy and Northcote 1982).

Dredging in estuarine navigation channels is necessary because of the high degree of sediment deposition from upstream and estuarine sources. When dredging occurs, the subsequent increases in turbidity level largely depend upon the water flow, the type of equipment and techniques used, as well as the distance from the dredge operation (LaSalle, this volume). Levels can be as high as 1000 mg L^{-1} above ambient conditions, but levels <200 mg L^{-1} are more common within 250 m of the dredge (Kuo et al. 1985, Lunz and LaSalle 1986). The impact of these changes in turbidity level on migrating salmonid and permanent fish populations is of considerable concern.

Although high turbidity levels can be physiologically detrimental to salmonids (McLeay et al. 1987, Noggle 1978, Servizi, this volume, Servizi and Martens 1987) and many other species (see Bruton 1985 for a review), sublethal effects and modified behavior patterns of fish are of more concern. Levels high enough to cause lethal effects generally are not attained in the natural estuarine environment (Cordone and Kelley 1961) or during most dredging operations (LaSalle, this volume). Reduced growth (Crouse et al. 1981, Sigler et al. 1984) and feeding abilities (Berg and Northcote 1985) have been documented at sublethal levels for several stream resident salmonids. The effect of turbidity on the visual foraging abilities of fish has generated considerable interest (Gardner 1981, Guthrie 1986, Janssen 1983, Miller 1979, Vinyard and O'Brien 1976). The experimental work of Gardner (1981) and Vinyard and O'Brien (1976) demonstrated declines in foraging rates and reaction distances toward planktonic prey with increasing turbidity. Therefore, visual constraints have the potential to severely affect growth and survival in estuarine fishes. Boehlert and Morgan (1985), however, show feeding of Pacific herring larvae may be enhanced by suspended sediment concentrations as high as 1000 mg L^{-1}, compared with clear water controls. The impact of turbidity on the foraging behavior of salmonids remains to be completely elucidated.

Although suspended sediment can adversely affect the visual abilities of estuarine residents, it should similarly affect a reduction in their vulnerability to predation (Gradall and Swenson 1982, Guthrie 1986, Ritchie 1972). However, this effect has yet to be demonstrated. It has been shown that juvenile fishes can be more abundant in turbid than in less turbid coastal areas (Blaber and Blaber 1980, Cyrus and Blaber 1987a). Turbidity gradient experiments have demonstrated some species may actively choose turbid (10-80 NTU) over clear areas (Cyrus and Blaber 1987b). Gradall and Swenson (1982) noted increased activity and less reliance on overhead cover in brook trout and creek chub in slightly turbid water (<10 FTU), while Berg and Northcote (1985) have noted decreases in intraspecific aggression in stream resident coho in turbid conditions (<60 NTU). In light of these studies, it is reasonable to suggest advantages of turbid waters may exist for some fish.

In this paper, I will assess the cost of foraging in turbid water to juvenile chinook salmon (*Oncorhynchus tshawytscha*), in terms of the reduction in reaction distance. Reaction distance here is defined as the distance at which the fish made a deliberate motion toward the prey prior to a strike. With the use of a predatory gull model, I will also examine the reduction in perception of risk in chinook with increases in suspended sediment. These findings will be discussed with respect to the results of a feeding experiment conducted over a range of turbidity conditions. The implications of these experiments will be discussed as they pertain to the impacts of dredging operations within estuaries.

METHODS

Sediment for all laboratory procedures in this study was obtained from a tidal marsh in the south arm of the Fraser River Estuary, at Ladner, British Columbia. Approximately 10 L of this raw sediment was sieved through a 0.40 mm sieve, to remove larger detritus, then suspended in 125 L of freshwater in a plastic bucket and allowed to settle for a period of 2 h. After this time, the supernatant (75 L) was transferred to another bucket and allowed to settle for a further 48 h. The excess water was then poured off, and the remaining sediment slurry was autoclaved for 30 min. This procedure was repeated until the required quantity of slurry was obtained.

In order to determine their reaction distance to prey in turbid water, chinook juveniles were conditioned for two weeks to strike at *Artemia* prey immobilized in 4.0 mm diameter, transparent, glass tubes. A fish striking at the prey in the tube facilitated the release of the latter into the 200 x 30 x 25 cm experimental arena, where it was ingested by the test subject. A video camera was used to record the observations made through the bottom of the tank. Observations were conducted at turbidity levels of 0, 12.5, 25, 50, 100 mg L^{-1} (with one fish also at 200 and 400 mg L^{-1}). Fish were acclimated to the turbidity test condition for one hour prior to observation.

A predatory bird model (gull) was used to investigate the mitigating effects of turbid waters on risk of predation as perceived by chinook juveniles. A 1000 L (200 x 50 x 120 cm) experimental arena was employed, which was fitted with a bottom which sloped to the surface. The arena possessed one plexiglass window (200 x 50 cm) marked off into six regions (two deep and four slope regions), through which observations were made. Fish were restricted, by netting, to the portion of the tank within 40 cm of the window. The gull model was drawn the length of the tank by a drawstring, at a speed of ca. 1 m s^{-1} to represent the slow foraging soar of an avian predator. Twenty-two chinook (65-70 mm FL) were introduced to this apparatus in each of two replicates. The chinook were fed twice daily on live *Drosophila*, *Artemia*, and *Tubifex* introduced to the surface, water column, and bottom, respectively. The fish were acclimated to the arena for one week before observations commenced. The experimental conditions (clear or 25 mg L^{-1} turbidity, predator or no predator) were manipulated randomly on a daily basis in a factorial design. On each day before the lights came on in the laboratory, the water supply to the tank was turned off and the required sediment slurry was added (on appropriate days) to bring the turbidity level up to 25 mg L^{-1}. Observations consisted of counts of fish in each of the six regions during an individual observation set. Three observation sets were performed each day. At the beginning of each appropriate observation set, the gull model would be drawn over the tank. After 30 seconds, counts were made of fish in each of the marked regions of the tank. The change in spatial distribution of fish in the arena was used to assess the perceived predation risk by the juveniles. This change was expressed as the ratio of the number of observed fish in the deepest region of the arena to the number expected from a random distribution.

Foraging rate on *Tubifex* was assessed in an array of seven 70-L aquaria at turbidity levels of 0, 25, 50, 100, 200, 400, and 800 mg L^{-1}. All treatments were run simultaneously. Lighting was provided by double-fixtured, fluorescent lights running the length of the array, 50 cm above water level. Ten fish which had been fed *Drosophila*, *Artemia*, and *Tubifex* for three days before the experiment, then starved for 18 hours, were placed in 4-L, meshed holding chambers within each experimental tank and allowed to acclimate for one hour. Five minutes before the fish were released, 2.4 g (wet weight) of *Tubifex* were introduced to each tank and allowed to settle on the bottom; fish were then released. Aquaria were fitted with false bottoms with concentrated sediment slurry visible to the fish through the glass bottom; 2.0 mm clear glass beads provided a substrate in which the *Tubifex* could burrow. All experimental tanks were surrounded by plywood painted flat white, segregating them from outside disturbances and from each other. After 5 minutes of uninterrupted feeding, tanks were drained (3 minutes), the fish were netted, killed in MS222, and preserved in 5% formalin. Gut contents were dissected, blotted dry, and weighed to the nearest milligram.

RESULTS AND DISCUSSION

My investigations with juvenile chinook salmon support the well established working principle that turbidity reduces the reaction distance of visually foraging fish toward their prey (Gardner 1981, Vinyard and O'Brien 1976). Reaction distances were demonstrated to decrease exponentially with increases in suspended sediment concentration (Fig. 1). Resident fishes may offset the effects of reduced visibility through behavioral changes or the utilization of some optical feature of turbid water. Both processes have been proposed. Smaller fish species and immature stages may profit from either reduced risk to piscivores (Blaber and Blaber 1980, Gradall and Swenson 1982, Guthrie 1986, Ritchie 1972) or enhanced visual contrast within the reduced reactive field (Boehlert and Morgan 1985, Godin and Gregory in prep.).

It is generally assumed that predators are visually affected by suspended sediment in a manner similar to the prey fish; risk of juvenile fish to predators is likely to decline with an increase in turbidity (Fig. 2). Experimental studies demonstrating such benefits to fish are lacking. The change in spatial distribution by chinook in response to a predator model was much lower in turbid water than in clear water (Fig. 3). Although the gull model elicited significant responses in both treatments ($p < 0.05$, Chi-square test), the reaction it elicited in the turbid condition was much less pronounced. Also, significant differences in spatial distribution were exhibited in the turbid water as compared to the clear water treatment, in the absence of the predator model. Although the degree to which these responses were cognitive was not apparent, the data suggest that chinook

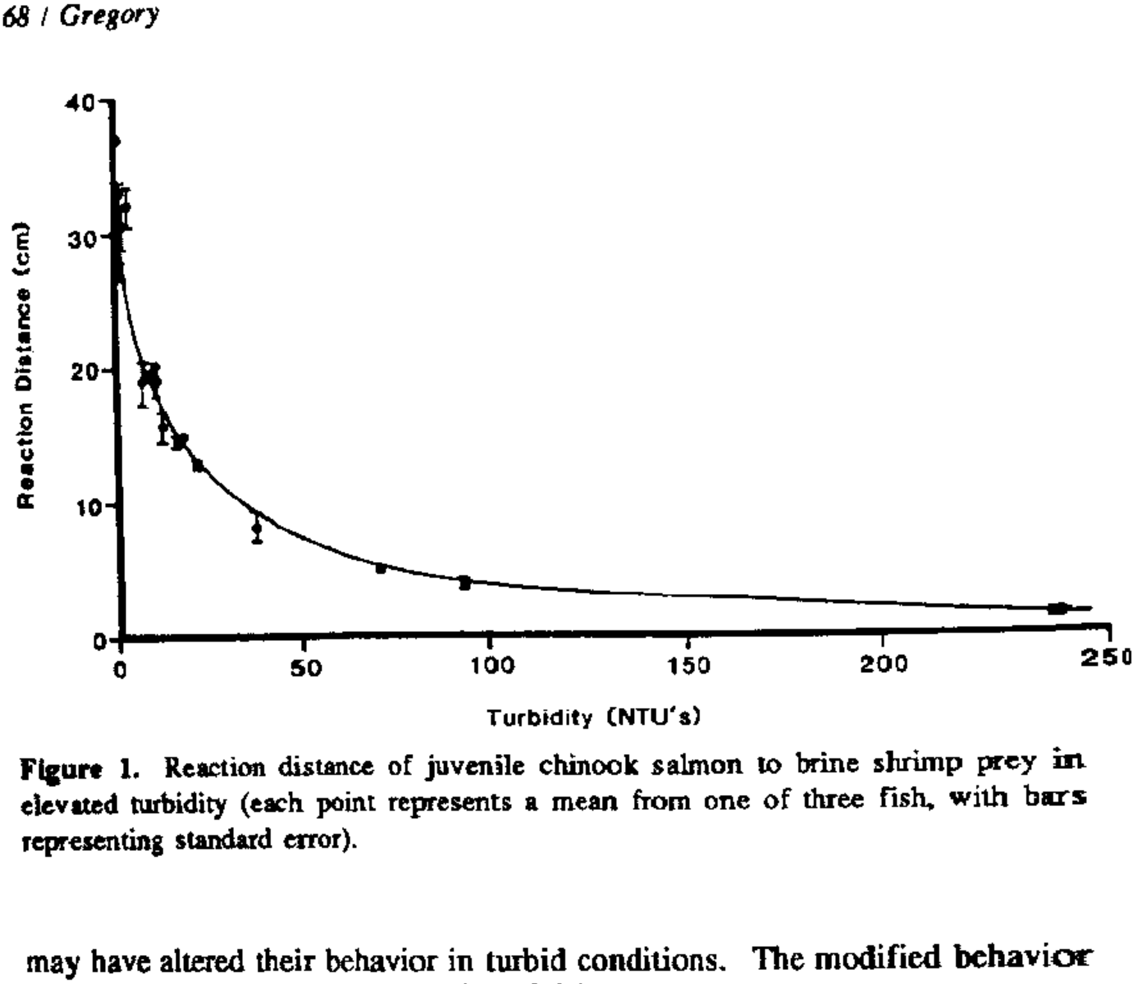

Figure 1. Reaction distance of juvenile chinook salmon to brine shrimp prey in elevated turbidity (each point represents a mean from one of three fish, with bars representing standard error).

may have altered their behavior in turbid conditions. The modified behavior possibly reflected a reduced perception of risk.

The reduction in both reaction distance (Fig. 1) and perceived risk to predation (Figs. 2 and 3) represent conflicting goals to a forager; a tradeoff exists. In clear water, feeding opportunities might be numerous, but any sacrifice in vigilance required to forage effectively may increase the susceptibility to predation. At the opposite extreme, while lowering risk substantially, highly turbid water can reduce the reaction distance to such an extent that visual feeding is not possible. It can be predicted from such a tradeoff that an intermediate condition will provide the forager with acceptable levels of feeding and risk. Therefore, peak foraging rates should occur in intermediate turbidities.

Benthic foraging rates by juvenile chinook salmon on *Tubifex* in the laboratory were highest at an intermediate turbidity level of 100 mg L^{-1} (Fig. 4). Very low foraging rates occurred in both clear water and the highest treatment level (800 mg L^{-1}). High perceived risk at low turbidity levels may have inhibited the foraging behavior of the chinook while the reduced visibility limited feeding at high suspended sediment levels. These data fit the predictions of the tradeoff hypothesis.

Boehlert and Morgan (1985) suggested that similar results for Pacific herring larvae could be explained in terms of prey contrast enhancement within a

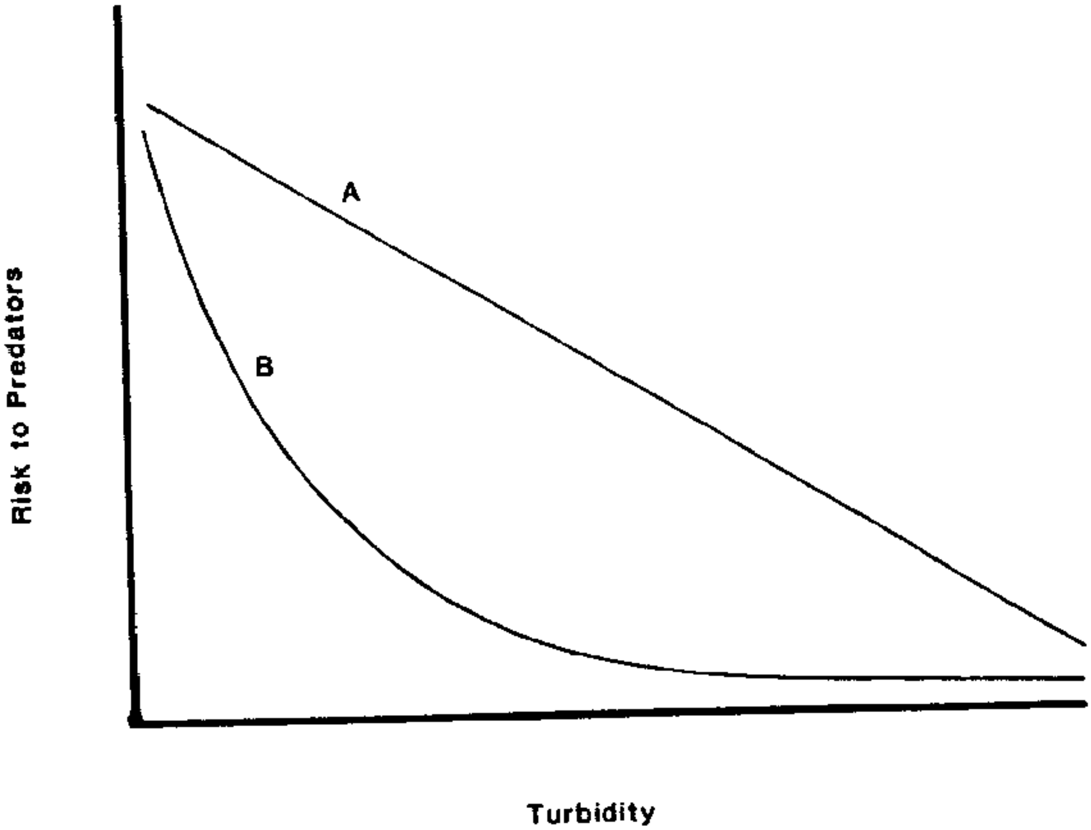

Figure 2. Hypothetical relationship for the risk of fishes to predators in turbid conditions. (A. Linear decline in risk with turbidity increase; B. exponential decline.)

reduced visual field. Prey contrast enhancement is less likely for benthic prey than for planktonic prey, because of the more constant background characteristics of the benthos in different turbidity regimes. I propose a more behaviorally plastic, anti-predation hypothesis. Clearly, compensatory mechanisms mitigating reduced visual range in turbid waters may exist for estuarine fishes.

CONCLUSIONS

This paper presents data which supports a hypothesized tradeoff between reaction distance and perceived risk to predation under conditions of low to moderate turbidity. In light of these data, moderate increases in turbidity level, whether natural or mechanically induced, are not necessarily detrimental to the survival of young salmonids. That is not to say that highly turbid water is desirable. Clearly, elevated turbidity levels of >200 mg L^{-1} have a pronounced negative effect on juvenile chinook foraging rate in the laboratory.

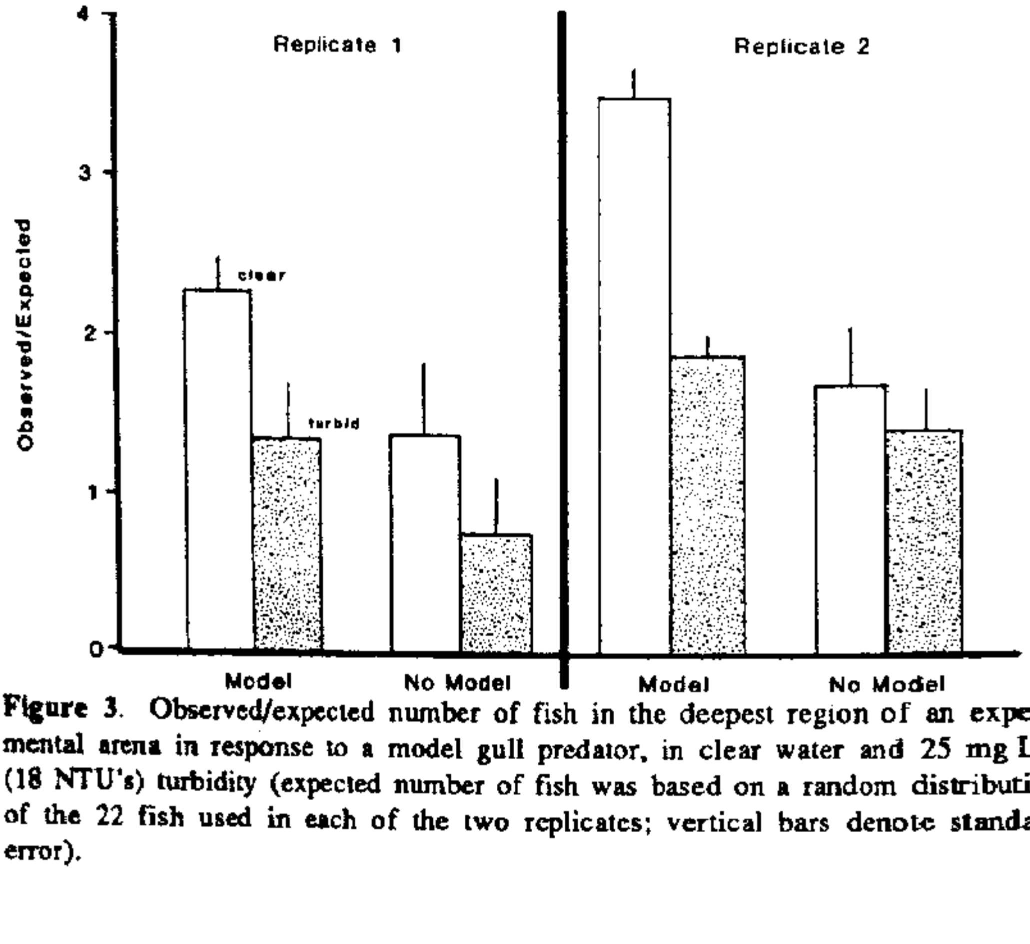

Figure 3. Observed/expected number of fish in the deepest region of an experimental arena in response to a model gull predator, in clear water and 25 mg L^{-1} (18 NTU's) turbidity (expected number of fish was based on a random distribution of the 22 fish used in each of the two replicates; vertical bars denote standard error).

We must look at many other factors related to dredging to determine if a particular operation may be harmful to salmonids or other fish. The effect of elevated suspended sediment loads on the availability of prey to migrating juveniles remains a key question. The secondary productivity of many estuaries is driven by the dynamics of the detrital food web (Healey 1979, Kistritz 1978, Levy et al. 1979, Levy and Northcote 1982). Benthic habitat disturbances over large portions of a given estuary could have severe ramifications for the detritus-based food supply of the fish population. At low prey densities, a relatively small increase in turbidity level can cause reductions in prey encounter rates by a foraging fish (Ware 1973), which may lead to effects on growth rate and survival. Prey availability must be maintained above levels where estuarine fish growth and survival rates are adversely affected. The tradeoff described in this paper is not necessarily to the foraging fish's benefit if prey abundance is greatly limiting.

ACKNOWLEDGMENTS

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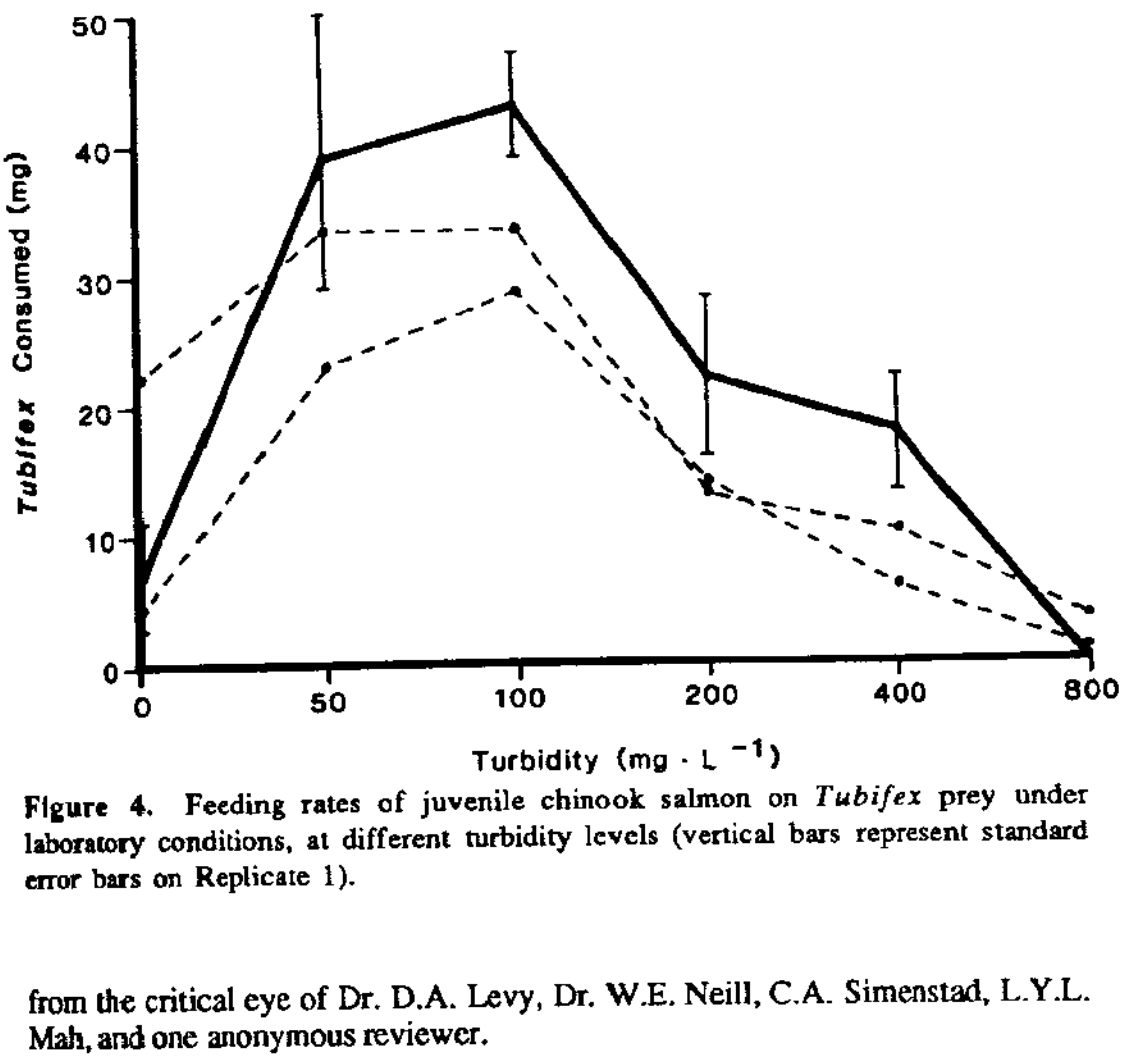

Figure 4. Feeding rates of juvenile chinook salmon on *Tubifex* prey under laboratory conditions, at different turbidity levels (vertical bars represent standard error bars on Replicate 1).

from the critical eye of Dr. D.A. Levy, Dr. W.E. Neill, C.A. Simenstad, L.Y.L. Mah, and one anonymous reviewer.

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Effects of the 1980 Mount St. Helens Eruption on Columbia River Estuarine Fishes: Implications for Dredging in Northwest Estuaries

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Abstract: *The 18 May 1980 eruption of Mount St. Helens markedly increased turbidity in the Columbia River estuary. After the eruption, major changes in abundances and feeding habits of selected fishes were observed. For example, subyearling chinook salmon (*Oncorhynchus tshawytscha*) were less abundant in open-water areas of the upper estuary, more abundant in open-water areas of the central and lower estuary, and consumed fewer* Corophium *spp. Also, amphipod densities at selected locations in the estuary were significantly lower after the eruption. Since dredging often produces high turbidities, the eruption can be viewed as a natural, albeit extreme, example of how high turbidities can alter abundances and feeding habits of estuarine fishes.*

INTRODUCTION

Navigation channels and mooring basins in estuaries of the Pacific Northwest are routinely dredged to maintain depths required by commercial and recreational vessels. In many estuaries, dredging is limited to the spring and summer months when weather and oceanic conditions allow dredges to operate in the sometimes open and turbulent waters (e.g., near the mouth of the Columbia River). This is also the time when large numbers of juvenile salmonids may temporarily reside in or migrate through many of these estuaries. Unfortunately, much of the research on the biological effects of dredging and dredge-disposal in estuaries (e.g., Mortensen et al. 1976, Morton 1977, Hirsch et al. 1978, Peddicord and McFarland 1978, Allen and Hardy 1980) was conducted before estuaries were identified as critical habitats for juvenile salmonids (Levy and Levings 1978, Levy et al. 1979, Healey 1980, Myers 1980, Healey 1982, Kjelson et al. 1982, Levy and Northcote 1982, Myers and Horton 1982, Pearce et al. 1982, Simenstad et al. 1982, McCabe et al. 1983, Dawley et al. 1985, McCabe et al. 1986). Hence, few researchers have considered the potential impacts of dredging on salmonids.

As a result of the 18 May 1980 eruption of Mount St. Helens (Washington state), large quantities of ash and sediment were suddenly washed into the

Columbia River and its estuary. Water quality, particularly turbidity, changed dramatically. Since increased turbidity is one of the major water quality impacts of dredging and dredge-disposal, studying the effects of this dramatic event offered an opportunity to gain insight into how dredging and dredge-disposal could influence estuarine usage by juvenile salmonids and other estuarine fishes.

Although there has yet to be a long-term physical, chemical, or biological monitoring program in the Columbia River estuary, the eruption of Mount St. Helens fortuitously occurred during the sampling phase of the Columbia River Estuary Data Development Program (CREDDP). While not destined to be a long-term study, CREDDP included the most comprehensive, broad-scale biological sampling of the estuary conducted to date. Accordingly, results from CREDDP, as well as those of selected previous studies, could be used as a baseline to compare results of post-eruption monitoring. This report summarizes the results of these comparisons, emphasizing the potential effects of elevated turbidity on estuarine use by juvenile salmonids.

METHODS

Turbidity

Turbidity was measured once a day at five stations in the Columbia River estuary from river kilometer (Rkm) 8.0 to 29.3 on 19, 21, 22, and 26 May 1980 (Fig. 1) using a Hach[1] turbidimeter. Turbidities were measured about 4 m below the surface and recorded as Jackson Turbidity Units (JTU).

Fishes

From February 1980 to July 1981, fishes were sampled in the Columbia River estuary for CREDDP. Sampling consisted of monthly collections of fishes with three types of fishing gear at the stations shown in Figure 1. An 8-m semiballoon shrimp trawl was used to sample demersal habitats. Trawl mesh size was 38.1 mm, with a knotless 12.7-mm mesh liner inserted in the cod end (all mesh sizes are stretched measures). Trawling was done for 5 minutes, generally upstream during a flood tide. A 200 x 9.8-m variable mesh purse seine was used to sample open-water areas; mesh sizes were 19.0 mm and 12.7 mm. The purse seine was set upstream for 5 minutes during selected tidal stages. Beach seines were used to sample intertidal areas (and often adjacent subtidal regions). Two variable mesh (19.0, 12.7, and 9.5 mm) beach seines were used; one was 4.0 m deep, and the other 3.4 m deep. Beach seining was done during selected tidal stages using methods similar to those of Sims and Johnsen (1974). Purse and beach seines had knotless web in their bunts to reduce descaling of fish.

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[1]Reference to trade names does not imply endorsement by the National Marine Fisheries Service, NOAA.

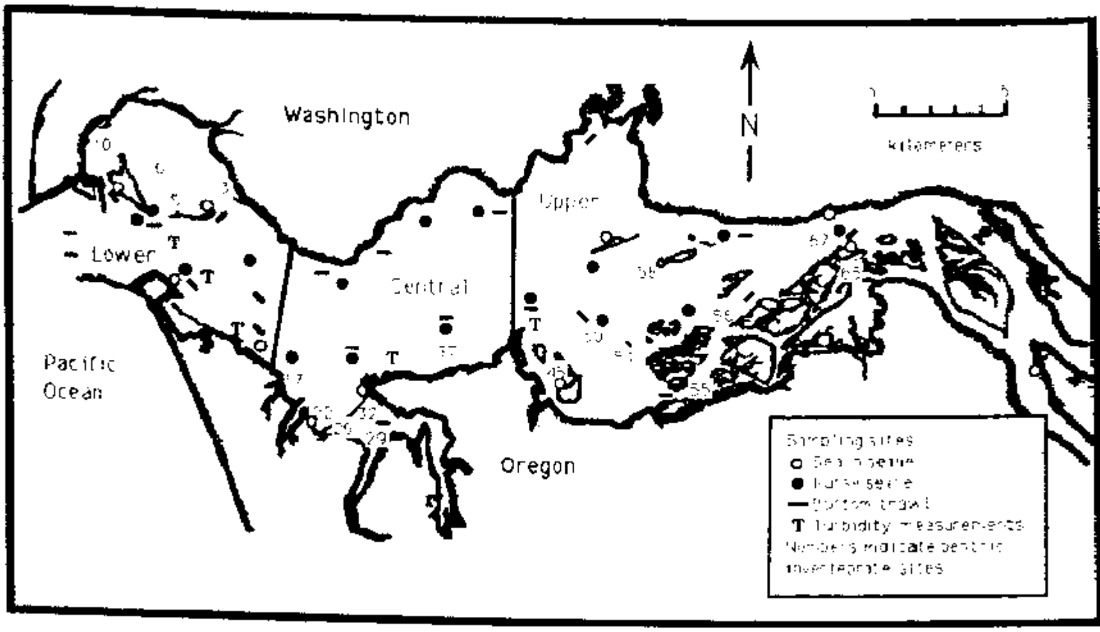

Figure 1. Sampling locations for fishes and benthic invertebrates in the Columbia River estuary, 1977-1981.

Fish collected were identified and enumerated, and a random subsample of up to 50 fish of each species or age group of chinook salmon (*Oncorhynchus tshawytscha*) was measured to the nearest millimeter (total length) and weighed to the nearest gram. When more than 50 fish were collected, the remainder was counted and weighed as a group. Subyearling and yearling chinook salmon were separated using length-frequency histograms.

To analyze fish catches, we divided the estuary into three zones based roughly on salinity (Fig. 1). The lower zone is essentially a marine area; the central zone a mixing area; and the upper zone a freshwater region. Although tides and river flow altered salinity regimes within the zones, these divisions provided a general framework within which to identify any large changes in fish distributions. Since all May 1980 sampling for CREDDP was completed prior to the 18 May eruption, the May collections were used as a baseline to assess how increased turbidities resulting from the eruption altered fish distributions.

Fish Food Habits

To evaluate changes in fish feeding habits, a subsample of five individuals of each species was selected from each sampling effort. These individuals were injected with a buffered 8% formaldehyde solution to preserve stomach contents. Injected fishes were weighed and measured in the laboratory, and their stomachs were removed and placed in vials containing 70% alcohol. Fish stomach contents were identified to the lowest practical taxon, counted, allowed to dry for 10 minutes, and then each taxon weighed.

An Index of Relative Importance (IRI), modified from Pinkas et al. (1971), was used to assess the importance of specific prey items in the diets of selected fish species:

$$IRI = (N + W)F$$

where N = percent number of a prey item in a fish's diet,
W = percent weight of a prey item in a fish's diet, and
F = percent frequency of occurrence of a prey item in a fish's diet.

IRI values were then converted to percentages by dividing by the total IRI and multiplying by 100.

An Index of Feeding (IF) was used to measure fish feeding intensity:

$$IF = \frac{Ws}{Wf} \times 100$$

where Ws = weight of stomach contents of a fish and
Wf = weight of a fish.

Analysis of variance (ANOVA) was used to identify significant differences among IF values. Data were arc sine transformed (Sokal and Rohlf 1969) prior to analysis, and a critical value of $P = 0.05$ was considered statistically significant.

Benthic Invertebrates

In June 1980, 18 estuarine sites that had been sampled during benthic invertebrate surveys in 1977 and 1978 (Durkin and Emmett 1980, Durkin et al. 1983) were reoccupied (Fig. 1). Benthic invertebrate samples were collected with a 0.05-m^2 Ponar dredge (Word 1976), sieved through a 0.6-mm (No. 30) screen, and preserved in a buffered 4% formaldehyde solution containing rose bengal, a protein stain. In the laboratory, formaldehyde was washed from the samples, and invertebrates were separated from the debris, identified, counted, and stored in 70% alcohol. Amphipods, particularly *Corophium* spp., are important prey of many fish species. ANOVA was used to compare differences in mean amphipod densities before and after the eruption. Data were normalized by a $\log_{10}(x+1)$ transformation (Sokal and Rohlf 1969) before analysis.

RESULTS

Turbidity

Turbidities in the Columbia River increased sharply as a result of the 18 May eruption to as high as 1,500 JTU by 21-22 May 1980 (Fig. 2), and then dropped to moderately high levels. Estuarine turbidities appeared to slowly drop

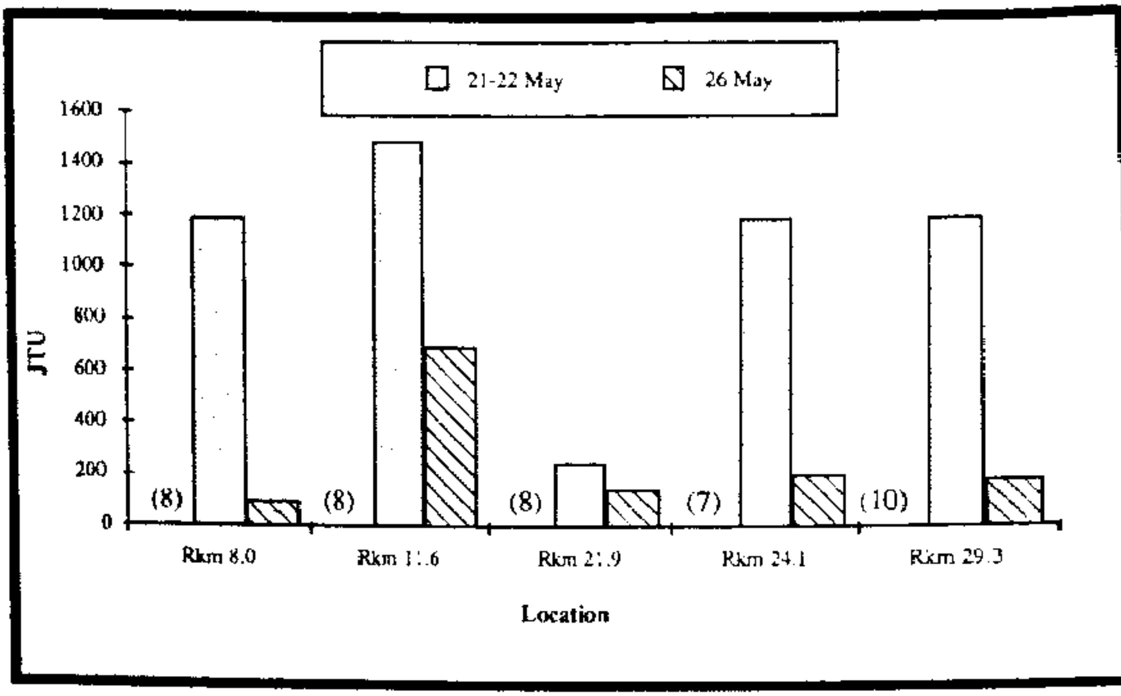

Figure 2. Turbidities in the Columbia River estuary during May 1980. Numbers in parentheses indicate Jackson Turbidity Units (JTU) for 19 May 1980 (prior to effect of eruption).

(to 10-30 JTU) during the following months. Although turbidities were not monitored in 1981, high levels (similar to 1980) were again observed during the spring freshet—May and June (pers. observation). The high turbidities in 1981 were probably also due to Mount St. Helens' ash and sediment.

Fishes

Open-water fishes which showed large changes in distribution pattern after the eruption included juvenile chinook and coho salmon (*Oncorhynchus kisutch*), juvenile steelhead (*O. mykiss*, formerly *Salmo gairdneri*), and Pacific herring (*Clupea harengus pallasi*) (Fig. 3). After the eruption, purse seine catches of juvenile chinook salmon (mostly subyearlings) declined in the upper estuary, but increased in the central and lower estuary. In 1981, a similar decline did not occur during this period (McCabe et al. 1986). Pacific herring were also affected, with reduced catches in the central zone after the eruption. Purse seine catches of juvenile coho salmon and steelhead declined in all three areas after the eruption. However, this was probably related to their natural migration period (Dawley et al. 1985). Other open-water species either showed no changes in their distributions or showed no consistent trends.

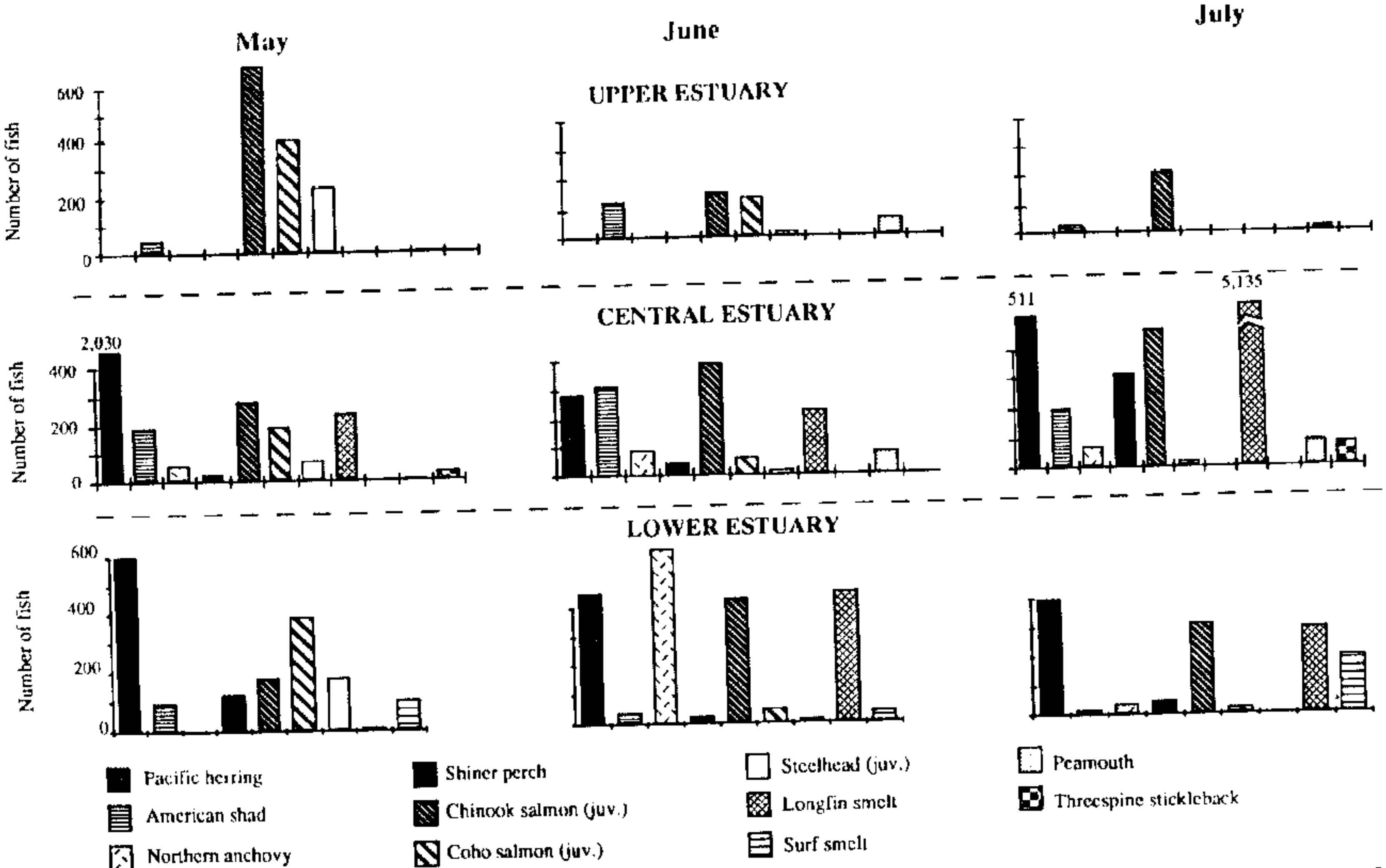

Figure 3. Summary of purse seine catches from three zones in the Columbia River estuary, May to July 1980. See Fig. 1 for zone locations.

The distribution patterns of demersal species also changed after the eruption. For example, catches of starry flounder (*Platichthys stellatus*) in the upper estuary dropped dramatically and did not recover until August. Demersal fish catches overall dropped in both the upper and central estuary during June 1980 (Fig. 4). Catches in the upper and central areas increased in July and reached or exceeded pre-eruption levels by August.

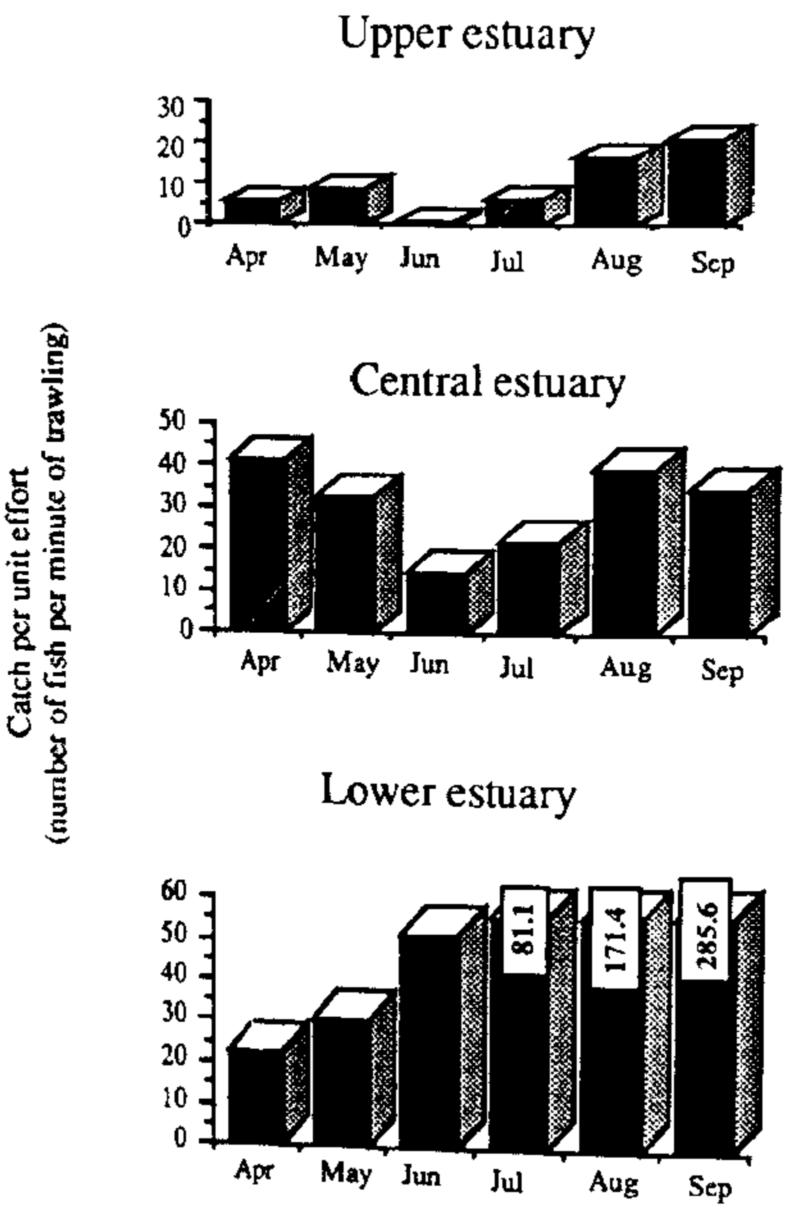

Figure 4. Number of fish caught per trawling minute in three zones in the Columbia River estuary, April to September 1980. See Fig. 1 for zone locations.

Fish Food Habits

In the month prior to the eruption, the gammarid amphipod *Corophium salmonis* was the primary prey item for purse seine-captured subyearling chinook salmon (Fig. 5), yearling chinook salmon, coho salmon, steelhead, and American shad (*Alosa sapidissima*) (Emmett 1982). Secondary prey items were Diptera adults, *Corophium spinicorne*, and Homoptera. In the month after the eruption, the importance of *C. salmonis* declined in the diets of all open-water fishes. Salmon fed primarily on Diptera adults, and American shad fed mainly on copepods. While fewer *C. salmonis* were eaten after the eruption, the numbers of Diptera adults, copepods, and other taxa eaten remained fairly constant, thus their contribution to the total amount of food consumed was higher. One year later (May and June 1981), purse-seine-captured fishes still consumed few *C. salmonis*; however, they consumed more *Daphnia* sp. in May and June 1981 than in the same period in 1980. The comparisons between 1980 and 1981 suggest that either *C. salmonis* were still largely unavailable (see below) or other prey were more abundant.

Beach seine-captured fishes also utilized *C. salmonis* as primary prey during May 1980 (Fig. 6). Unlike purse seine-captured fishes, beach seine-captured fishes showed only a limited reduction in the numbers of *C. salmonis* consumed after the eruption (June 1980). *C. salmonis* was still the primary prey in May and June 1981.

Fish feeding intensity also changed markedly after the eruption (Fig. 7). Purse seine-captured fishes had significantly lower IF values (ANOVA, $P<0.01$) after May 1980. Beach seine-captured subyearling chinook and coho salmon also had significantly lower IF values, but starry flounder did not. All fish species, except American shad, showed an increase in the percentage of empty stomachs after May 1980 (Emmett 1982) .

Amphipod Densities

There was a significant reduction in mean amphipod densities between the pre- and post-eruption benthic surveys (ANOVA, $P<0.05$; Fig. 8). Stations 26-65 were particularly affected. *Corophium* spp. were the dominant amphipods at these stations, representing about 90% of amphipod densities. Stations located in Youngs Bay and Cathlamet Bay appeared to be most affected. For example, Station 26, in Youngs Bay, had one of the highest amphipod densities of all stations before the eruption, but had one of the lowest after the eruption.

DISCUSSION

Although these data suggest that increased turbidity in the Columbia River estuary following the eruption of Mount St. Helens may have altered the distributions and feeding habits of selected estuarine fishes, as well as the densities of important prey species (amphipods of the genus *Corophium)* for many of these fishes, increased turbidity alone may have not been responsible for

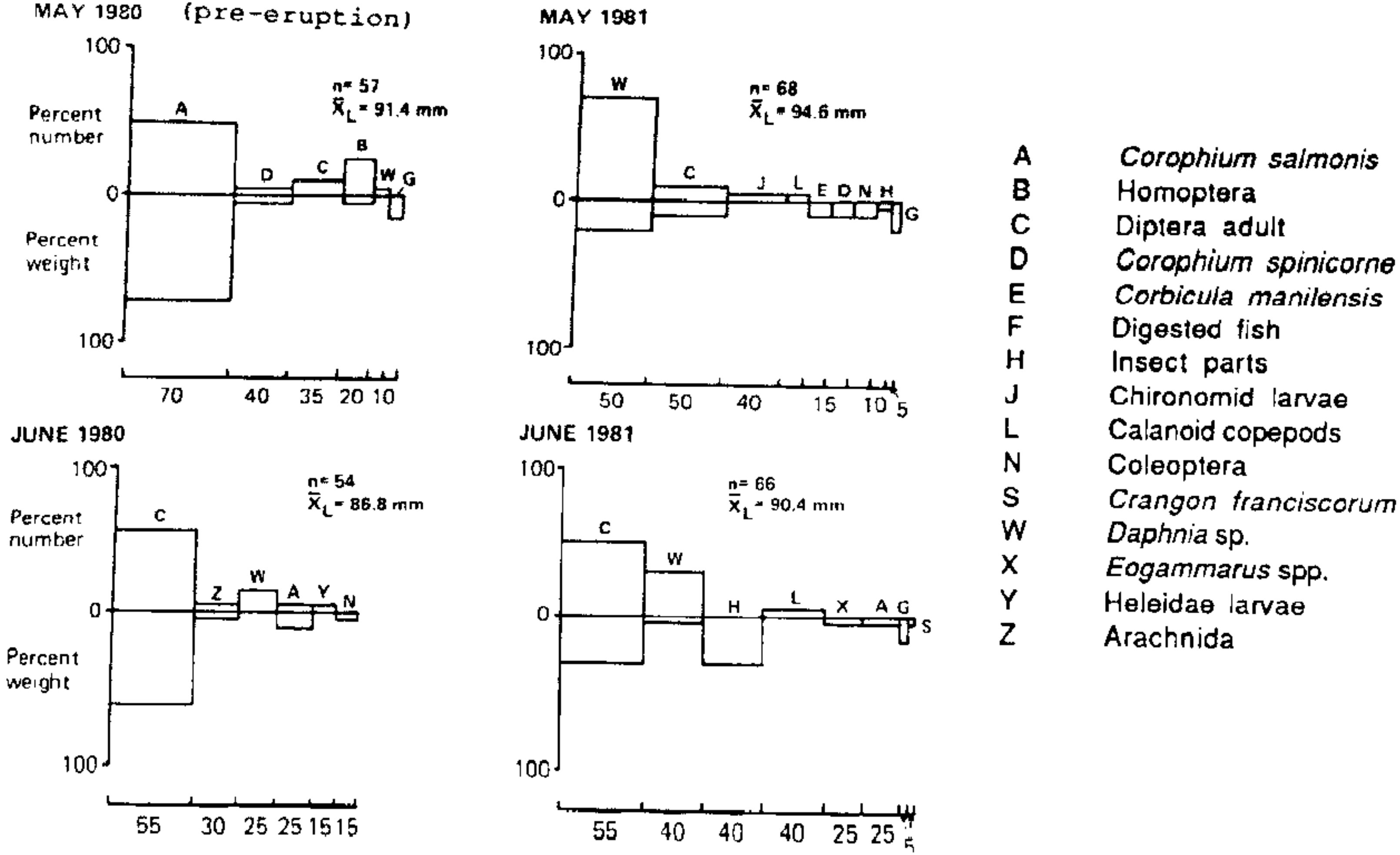

Figure 5. Stomach contents of subyearling chinook salmon captured in purse seines in the Columbia River estuary, May-June 1980 and May-June 1981; n = sample size, $\bar{X}_L$ = mean total length.

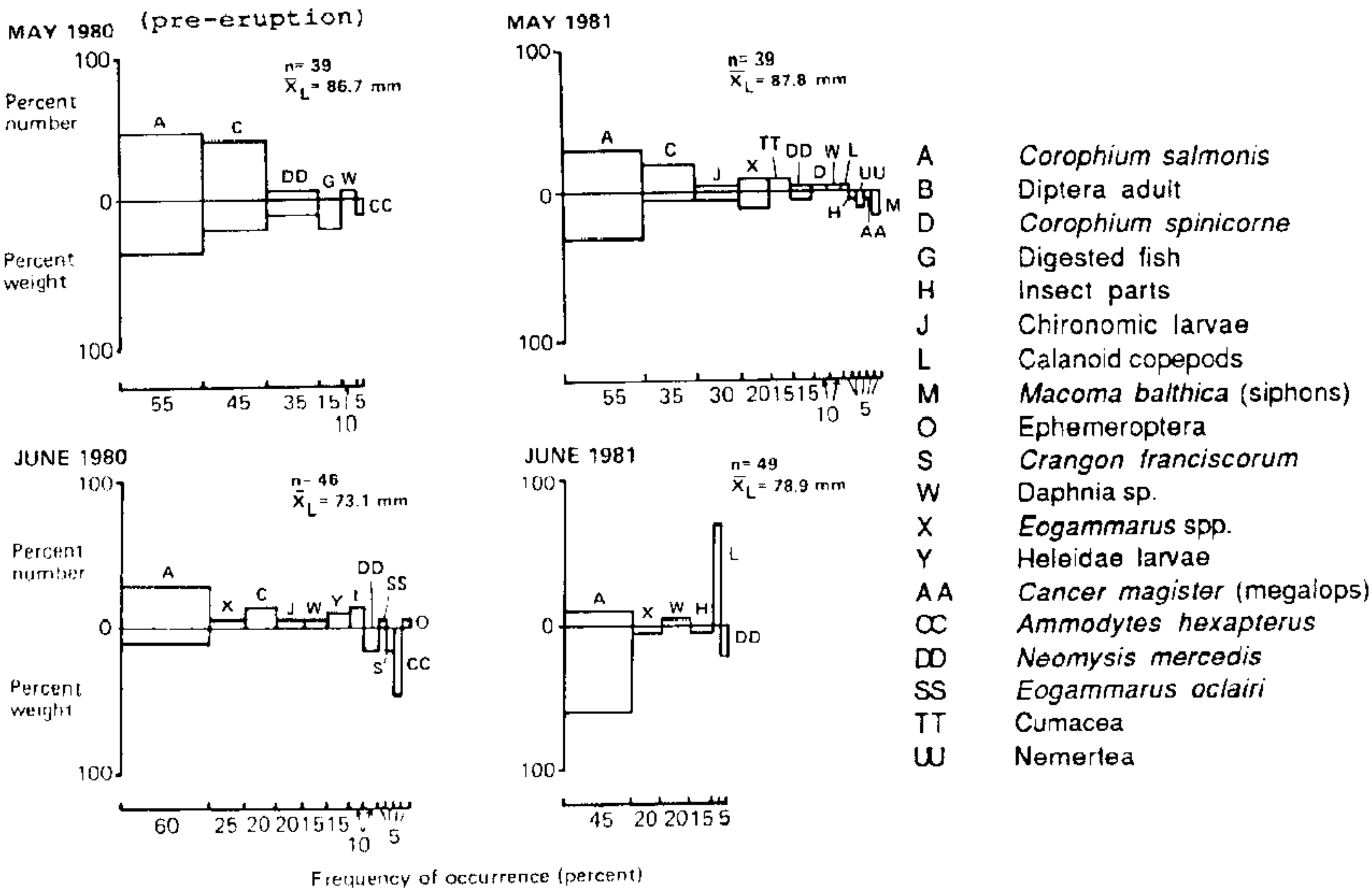

Figure 6. Stomach contents of subyearling chinook salmon captured in beach seines in the Columbia River estuary, May-June 1980 and May-June 1981; n = sample size, X_L = mean total length.

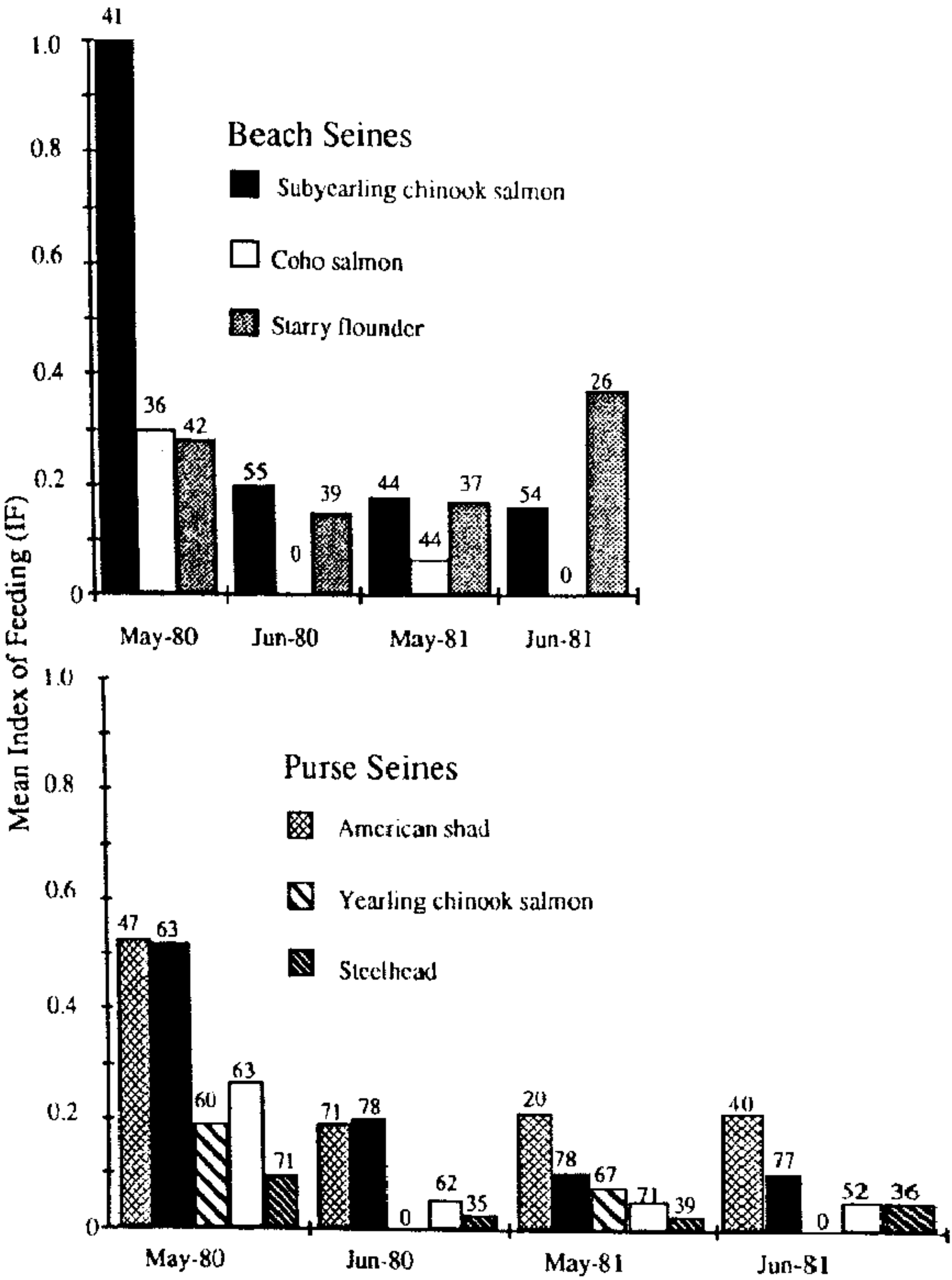

Figure 7. Mean Index of Feeding (IF) values for selected species captured in beach and purse seines in the Columbia River estuary, May-June 1980 and May-June 1981; numbers above columns indicate sample sizes.

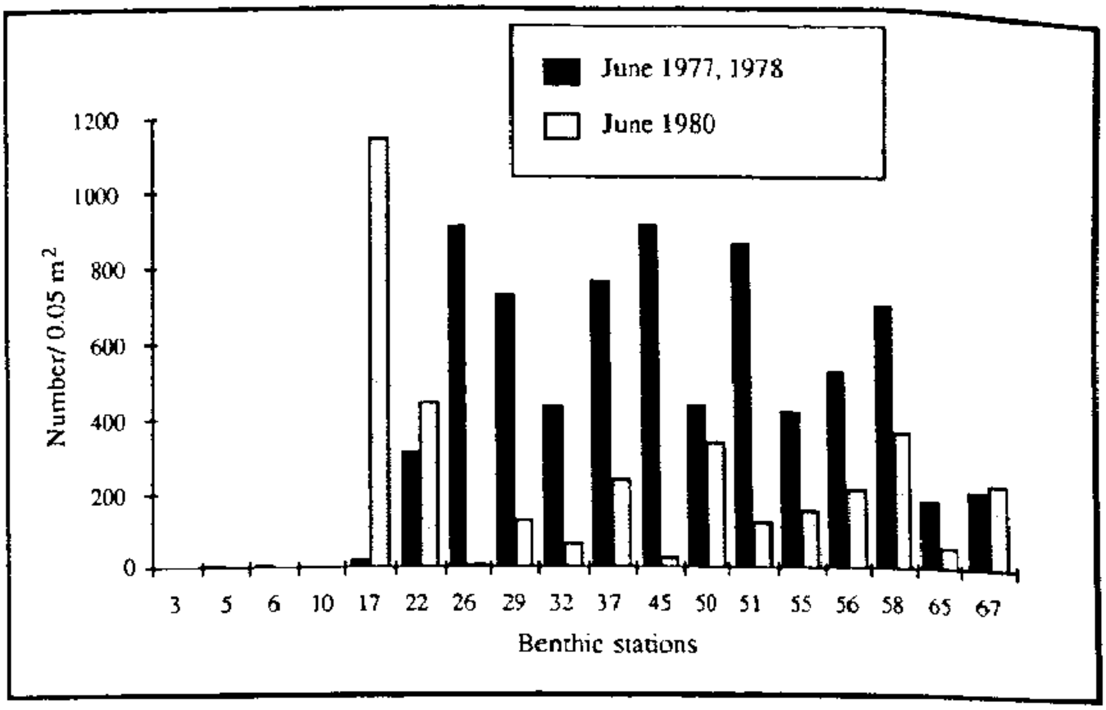

Figure 8. Amphipod (90% *Corophium* spp.) densities at 18 sampling stations in the Columbia River estuary before (June 1977, 1978) and after (June 1980) the 18 May 1980 Mount St. Helens eruption.

all of the observed changes. The lack of a long-term baseline showing seasonal and interannual patterns of abundance, distribution, and feeding habits of these species limits the ability to draw firm conclusions regarding cause-and-effect relationships. Moreover, the abundance of salmonids in the estuary is known to be strongly influenced by the timing of hatchery releases in the Columbia River Basin. Accordingly, the conclusions derived from these data are limited. Nonetheless, these data do provide much needed insight into how increased turbidity may affect estuarine resources. This inability to draw stronger conclusions highlights the importance of long-term monitoring programs as a means of developing the comprehensive databases needed to meaningfully evaluate or predict the impacts of both man-induced and naturally occurring catastrophic events on estuarine ecosystems.

It should also be emphasized that the extremely high turbidities measured in the Columbia River estuary following the eruption were as high as or higher than those which would be expected during typical channel dredging in most estuaries in the Pacific Northwest. In the Columbia River and its estuary, large increases in turbidity seldom occur during main channel dredging projects; most of these sediments are composed of relatively clean sands with minimal silt and clay (Durkin et al. 1979, Blahm and McConnell 1979). Blahm and McConnell (1979) measured turbidities of 2.8 to 30 JTU associated with flow-lane disposal of sediments near Dobelbower Bar in the Columbia River; these levels are generally within the normal range of values measured in this area of the river. However, other studies have reported cases where dredging and disposal of very fine sediments in the Columbia River estuary have caused turbidities near the bottom that were of the magnitude observed following the eruption of Mount St. Helens (Rathburn et al. 1983).

Although most juvenile salmonids (coho and yearling chinook salmon and steelhead) move through the Columbia River estuary quite rapidly (Bottom et al. 1984, Dawley et al. 1985), subyearling chinook salmon often rear in estuaries for considerable time, with this residence being an important component of their life history (Reimers 1973, Nicholas and Hankin 1988). Hence, subyearling chinook salmon would likely be one of the species at greatest risk, and this appeared to be the case. The data on distribution of fish in the estuary before and after the eruption indicated subyearling chinook salmon altered their pattern of estuarine use by moving from the upper to the lower reaches of the estuary where turbidity levels were generally lower due to the inflow of less turbid marine water.

Several factors suggested that the reduced catches of subyearling chinook salmon in the upper estuary following the eruption were a result of altered distribution rather than outright mortality. Prominent among these was the concomitant increase in catches in the central and lower estuary. In addition, there have been several studies that indicated juvenile salmon can tolerate extremely high concentrations of suspended sediments. For example, in a series of bioassays conducted after the eruption, Newcomb and Flagg (1983); found that juvenile salmonids could tolerate aquatic ash loads up to 6,100 mg/L. To put this concentration in perspective, normal suspended sediment levels measured in the Columbia River estuary range from 750 mg/L during high river flows to 260 mg/L during low river flows (Hubbell et al. 1971). In contrast to these reports of high tolerance, at least one report suggested a much lower threshold. Stober et al. (1981); found that juvenile coho and chinook salmon had 96 h-LC_{50}s of 509 and 488 mg/L for suspended sediments, respectively. Also, although high turbidities rarely have been associated with direct mortality, elevated concentrations of suspended sediments have been shown to affect salmonid behavior (Berg 1982, Bisson and Bilby 1982), growth (Crouse et al. 1981, Sigler et al. 1984), feeding (Berg 1982, Berg and Northcote 1985), reproduction (Lloyd et al. 1987), plasma cortisol levels (Redding et al. 1987), and susceptibility to infection (Redding et al. 1987).

Benthic invertebrates are often used to identify the environmental effects of estuarine alterations because they are largely non-motile, are easy to sample, are a critical link in estuarine food webs, and are particularly susceptible to changes in sediment structure. The reductions of *Corophium* spp. in fish diets and benthic samples in our study appear to be related to the large volume of volcanic ash that settled on the bottom of the estuary. Brzezinski and Holton (1983) found that although the ash was not toxic, it adversely affected *Corophium* densities, perhaps by physical suffocation or inhibiting tube building.

Edwards and Schwartz (1981) found the ash acted as a natural insecticide. Turk and Risk (1981) observed that deposition of fine sediments adversely affected *Corophium* populations in an east coast estuary. Sediments with no microflora, such as the ash layer, are unattractive to colonization by *Corophium* (Meadows 1964). How long it took the ash to develop microflora is unknown.

Besides directly affecting benthic invertebrates, high turbidities have also been shown to reduce fish feeding rates (Gardner 1981) and alter the behavior of fishes and invertebrates (Moore 1977). The reduced feeding (lower IF values) observed in estuarine fishes after the eruption may have been related to reduced prey abundances, reduced prey availability, alteration of fish feeding behavior, or a combination of these and unknown factors. Dawley et al. (1981) suggested that high turbidity caused stress, which in turn lowered feeding rates of juvenile salmonids in the upper estuary (Rkm 74) after the eruption. Kirn et al. (1986) showed that subyearling chinook salmon reduced their feeding on *Corophium* for 2 years after the eruption. By 1982, *Corophium* was reestablished in their study area, but its role in salmonid diets was supplanted by mysids.

CONCLUSIONS

The 18 May 1980 eruption of Mount St. Helens dramatically increased turbidities in the Columbia River estuary; the high turbidities appeared to alter the distributions and feeding habits of selected estuarine fishes and lower amphipod densities. However, long-term data are unavailable to determine unequivocally that the observed changes were a direct result of the increased turbidities. Unlike the mud and ash flows from the eruption of Mount St. Helens, dredging usually only affects water quality in areas adjacent to the dredging operations, not entire estuaries. Fish, which have the ability to avoid areas of poor water quality, may not be directly affected by dredging. However, juvenile salmonids and other fishes may be indirectly affected by dredging-caused reductions in food availability and alterations in estuarine habitats. The indirect effects of dredging on fishes, such as reduced feeding and change in estuarine use, may be reduced by protecting known valuable habitat from alteration, scheduling dredging to specific times, and utilizing dredging equipment and techniques that minimize resuspension of sediments.

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Turbidity and Suspended Sediments at the Alcatraz, California, Dumpsite

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Abstract: Dumping of dredged material at the Alcatraz dumpsite located in Central San Francisco Bay, California, contributes substantial quantities of suspended sediments to the water column in this deep, tidally-mixed portion of the estuary. Increased turbidity and/or suspended sediment concentrations from increased dumping in recent years have been hypothesized to be the cause of the concurrent declines of fisheries in the Bay. While dumping at the Alcatraz site contributes significantly to the suspended sediment loading and turbidity of Central San Francisco Bay, the suspended sediment concentrations and turbidity in this area are below those known to adversely affect fish and other organisms if the suspended sediments are uncontaminated. Therefore, increased turbidity and suspended sediment concentrations due to dredged material dumping are unlikely to be the cause of the observed fisheries decline. However, the toxicity of the suspended sediments from some contaminated dredged materials dumped at the Alcatraz site may be a contributing cause to the observed fisheries declines in San Francisco Bay.

INTRODUCTION

The Alcatraz dumpsite is a 610 m (2000 ft) circular area located in Central San Francisco Bay immediately south of Alcatraz Island. Water depths at the site range from about 12 m to 30 m (40-100 ft). The site has been used continuously since 1894. Since 1965, at least 1.5×10^6 m^3 of dredged material have been dumped each year, with approximately 3×10^6 m^3 or more disposed in most years, and 4×10^6 m^3 in 1986 (U.S. Army Corps of Engineers, San Francisco District, Permit files).

The Alcatraz dumpsite is specifically selected because its fast tidal currents maximize dispersion and prevent all but the coarsest or most cohesive dredged material from accumulating on the bottom at the dumpsite. However, during the early 1980s shoaling was observed, attributed primarily to the disposal of cohesive material. Since 1986, there has been a regulatory requirement that, as far as possible, material dumped at the Alcatraz dumpsite should be "slurried" to reduce accumulation of material and shoaling of waters at the dumpsite. One effect of this requirement is to increase the quantities and concentrations of suspended sediments transported from the dumpsite into the suspended sediment regime of Central Bay. The effects of this increased dispersion of suspended solids from the dumpsite have been highly controversial. Recreational fishermen have reported that the frequency of high turbidity conditions in Central Bay has increased and that sport fisheries for striped bass, salmon, halibut, shark, perch, flounder, croaker, and sturgeon in this area have been adversely affected since the slurry requirement was introduced (United Anglers 1987). These observations have been supported by analyses of fish catch and turbidity performed by the Department of Fish and Game, State of California (1987). Both the Department of Fish and Game, State of California (1987) and the National Marine Fisheries Service (1987) have stated that they believe that there is evidence that the slurry requirement may have caused higher turbidity in Central Bay and that this, in turn, may have caused an observed decline in fish stocks. However, the Corps of Engineers (U.S. Army Corps of Engineers 1988) estimates that turbidity resulting from dredged material dumping is of short duration and could not have caused either a general increase in turbidity or deleterious effects on fish.

FATE OF DREDGED MATERIAL DUMPED AT THE ALCATRAZ DUMPSITE

The water column at the Alcatraz dumpsite is less than 30 m deep. It is likely that: (1) dredged material dumped at the site impacts the bottom as a density plume; and (2) fractionation of the solids in the dredged material during the initial fall to the bottom is minimal. When dredged material impacts the bottom sediments, the kinetic energy of the plume is partially dissipated by resuspension of the upper layers of sediments at the impact site, and partially translated into a horizontal plume which disperses near and along the bottom away from the initial impact site. The suspended matter is initially dispersed in the bottom few meters of water over a large area (typically tens to hundreds of thousands of square meters), then settles out at a characteristic velocity that depends primarily on the particle size and density and turbulence at the site (Gordon 1974). Larger mineral grains settle out rapidly, while the lighter fractions remain suspended in the suspended sediments longer. The lighter material is known to be preferentially dispersed and advected away from the disposal site (Basco et al. 1974).

The Alcatraz dumpsite is characterized by oscillating tidal currents with maximum speeds of about 150 cm s^{-1} that flow west toward the Golden Gate on the ebb, and east toward Treasure Island on the flood (Dames and Moore 1971; Rubin and McCulloch 1979; Goddard et al. 1985; Winzler and Kelly 1985). Dumping at the Alcatraz site takes place at random times with respect to the stage of the tide. Therefore, suspended solids from an individual dump will be carried away from the site initially in a direction and at a rate determined by the tidal stage when dumping occurred. Dredged material particulates from dumps at slack tide will tend to remain at the dumpsite and accumulate temporarily in the sediments until they are resuspended and/or transported by the increasing currents associated with the next tidal cycle. Dredged material particles entering the suspended sediment regime are carried away from the dumpsite by currents and are undoubtedly subject to subsequent cycles of deposition and resuspension.

These resuspension/deposition cycles will continue in the Bay until the particles deposit in areas of consistently low current velocities, or are carried out to sea.

Laboratory resuspension studies have shown that maximum tidal current velocities at the Alcatraz dumpsite will resuspend most of the dredged material particles initially deposited at slack water if the dumped material has a density of 1.3 g cm^{-3} or less (Teeter 1987). The fine-grained fraction of dredged materials has a density of less than 1.3 g cm^{-3} and will, therefore, be resuspended unless it is incorporated in cohesive clumps of mud with low water content. The requirement that materials be slurried before disposal at the Alcatraz dumpsite is intended to ensure that such clumps of mud are broken up and mixed with water before dumping. Therefore, essentially all of the fine-grained particles dumped in dredged materials at the Alcatraz dumpsite are resuspended and transported by tidal currents away from the dumpsite into the Central Bay suspended sediment regime. This conclusion is supported by recent observations that no measurable accumulation has taken place at the site during the past several years (U.S. Army Corps of Engineers, San Francisco District, personal communication).

A hydraulic model of San Francisco Bay was used by Schultz (1965) and the U.S. Army Corps of Engineers (1988) to investigate the ultimate fate of dredged material dumped at Alcatraz. These model studies provided estimates that 47% of the material is transported to the ocean, 27% deposits in Central Bay, 22% in upper South Bay, and 2% in San Pablo bay. However, these studies are unreliable and of limited value since the hydraulic model does not accurately reproduce critical processes such as: (1) vertical variation in current velocity, particularly the mean estuarine circulation; net flow of surface water seaward and net flow of bottom water shoreward (Denton and Hunt 1986, McCulloch et al. 1970); (2) resuspension of sediments; and (3) vertical movements of suspended particles in response to their characteristic settling velocity and turbulence. In addition, the model studies consider only the ultimate fate and provide no information on the residence time of suspended sediments in Central Bay.

In fact, there is very limited reliable information concerning the fate of suspended dredged material particles in Central Bay. However, certain characteristics of this fate can be inferred from a knowledge of basic processes. First, as a result of the dumping, settling and resuspension described above, suspended sediments will tend to be concentrated and transported in near-bottom water. Therefore, the estuarine circulation that occurs at the Golden Gate (i.e., mean flow out of the Bay at the surface and mean flow into the Bay in bottom waters) will tend to retain the suspended particulates in the Bay, particularly when river flow rates are very low as in the summers of 1986 and 1987. Estimates obtained from the hydraulic model of the Bay suggest that 47% of the dredged material is eventually transported to the ocean. This is probably a large overestimate because the model does not reproduce the vertical structure of currents in this estuarine circulation. Second, the suspended dredged material particles will tend to be transported and dispersed within the deeper western portion of Central Bay (which is distinct from and somewhat isolated from exchange with waters from shallower parts of the Bay). Because there is little or no net deposition in these deep areas (U.S. Army Corps of Engineers 1988) these particles will remain in the active suspended sediment regime until: (1) accumulated in sediments where tidal currents are low, including harbors and pier areas of the San Francisco shoreline and embayments on the Marin County shoreline; (2) transported out to the ocean; or (3) transported south of the Bay Bridge. There is no data available to enable estimation of the mean residence time of suspended sediments in Central Bay. However, the mean residence time of water in Central Bay is estimated to be about 15-20 days at low river flow rates (Denton and Hunt 1986).

In summary, studies of the fate of dredged material suspended sediments in Central Bay are extremely limited. However, available evidence suggests that suspended sediments created by dumping at the Alcatraz site remain in the suspended sediment regime (and thereby in contact with fish, shellfish, and other biota) of western Central San Francisco Bay for an average length of time that almost certainly exceeds several tidal cycles and probably several days or weeks. This is consistent with the recent conclusion that dredged material particulates dumped at the Alcatraz dumpsite "are expected to remain suspended for a considerable time due to the water currents existing within the Bay" (U.S. Army Corps of Engineers 1988).

SUSPENDED SEDIMENT BUDGET OF CENTRAL SAN FRANCISCO BAY

The Corps of Engineers (1988) has argued that: (1) increased turbidity at the Alcatraz dumpsite immediately after a dump disappears rapidly, and (2) suspended sediment loading due to dredged material dumping is very small compared to the Bay's total suspended sediment load. Both of these observations, while correct, are misleading.

First, while the turbidity plume created immediately following a dump does disperse and disappear from the site within an hour or two after dumping, its disappearance is most likely not solely due to dispersion but also due to transport of the plume away from the site by tidal currents. Thus, turbidity may remain high in the plume as it is transported beyond the turbidity monitoring network at the dumpsite. No systematic study of this possibility has been performed but observers have reported it to occur. If dumping takes place during slack tide, the turbid plume may be reduced by temporary sedimentation but a turbid near-bottom plume will be created during subsequent periods of high tidal current velocity. In addition, most observations of the turbid plume at and near the Alcatraz dumpsite are made in surface waters, not in the near-bottom waters where the turbidity concentrations are expected to be highest.

Second, although the suspended solids load due to dredged material dumping is small compared to the Bay's total suspended sediment load, it is probably the dominant source of suspended sediments within west Central Bay during most times of the year (Table 1). Except for dredged material, estimates of each of the inputs to west Central San Francisco Bay listed in Table 1 are uncertain and temporally variable. For example, the quantities of suspended sediment transported by currents into western Central Bay from South Bay, San Pablo Bay, and eastern Central Bay depend upon river flow rate and wind and tide induced mixing. The quantity thus transported into Central Bay throughout most of the year is probably not large, because much of the river borne material is deposited in the upper parts of the estuary, except during the brief periods of high fresh water inflow from the rivers during winter. The quantity of suspended sediment derived from wind/wave resuspension in west Central San Francisco Bay (Table 1) is also likely to be substantially overestimated because this estimate assumes that wind/wave resuspension is uniform throughout the Bay. In practice, wind/wave resuspension is much less effective in the deep basin of west Central Bay. The majority of the Bay's suspended sediment load is created by wind/wave resuspension in the water column of shallow areas of Berkeley Flats, South Bay, San Pablo Bay, and Suisun Bay (Krone 1966). The predominantly deeper, oceanic-influenced west Central Bay surrounding the Alcatraz site should, in the absence of dredged material disposal, have much lower turbidity than other parts of the Bay except during high outflow periods. Thus, the high background turbidity observed in west Central Bay is most likely caused, at least partially, by the dredged material particulates dumped at the Alcatraz dumpsite.

Additional evidence of the importance of dredged material dumping as a source of suspended sediment in west Central San Francisco Bay is provided by two other observations. First, the quantity of suspended sediment that is estimated (U.S. Army Corps of Engineers 1988) to be transported annually from the Bay through the Golden Gate to the ocean (1.8-4 x 10^6 m^3) is lower than the quantity of dredged material dumped annually at the Alcatraz dumpsite (approxi

Table 1. Estimated suspended particulate loading (m^3) to San Francisco Bay waters.

Source of input	Total SFB*	Central SFB dispersal area
Wind/wave suspension	130,000,000	2,500,000
Riverine inflow	8,000,000	<8,000,000
Alcatraz dumping	2,800,000	2,800,000
Surface runoff	2,010,000	<200,000
San Pablo Bay dumpsites	994,000	Negligible
Net erosion/South Bay	443,000	443,000?
Point sources	174,000	<17,000
Aerial 157,000	65,000	

Data, in part, from U.S. Army Corps of Engineers (1988).

*SFB = San Francisco Bay. Total SFB includes Central and South Bay, San Pablo Bay, and Suisun Bay.

mately 5-6 x 10^6 m^3). Second, the suspended sediment load from one day's average dumping is approximately 10^5 kg, or about 1-2% of the estimated total suspended sediment load of the entire water column in the west Central Bay deep basin. If the mean residence time of suspended sediments (counting only the time particles spend in suspension, not time spent at temporary deposition sites) were 10 days (about one half the dry season residence time for water), dumping at Alcatraz would contribute approximately 15% of the total suspended particulate loading of west Central San Francisco Bay.

BIOLOGICAL EFFECTS

While the impacts of dredged material dumping on benthic communities within the dumpsite are relatively well studied and understood, impacts on free-swimming or floating organisms have been studied less and are poorly understood, partly because of the greater technical difficulties associated with studies of such impacts. These difficulties of studying these impacts are related primarily to: (1) mobility of planktonic and nektonic organisms that prohibit sampling of organisms in the field that have a known history of exposure to the dredged material plume; (2) difficulty of performing laboratory studies which can reproduce the variable exposure to the dredged material plume that is experienced by the organism in the natural environment; (3) lack of suitable dumpsites for study where the dispersing dredged material plumes are transported within a known area that is not also impacted by other anthropogenic influences; and (4) subtlety of the anticipated sublethal and/or long-term impacts and the consequent difficulty in observing such effects in the naturally highly variable ecosystems surrounding most dredged material dumpsites such as the Alcatraz dumpsite. Nevertheless, if turbidity is high enough, adverse effects on phytoplankton production and sublethal effects including avoidance reactions can occur in fish and shellfish.

It has been hypothesized that increased dredged material disposal at the Alcatraz site has caused a decline of fisheries in San Francisco Bay through increased turbidity and/or suspended sediment concentrations. This observation was based primarily on the decreased fishing success in Central Bay observed by fishermen during several recent years when slurrying and increased annual dumping rates were thought to have increased the turbidity and suspended sediment load at and near the site. Unfortunately, very few records exist of temporal changes in either turbidity or suspended sediment concentrations at the dumpsite. The fisheries success data are also poor.

The primary turbidity records are visual observations by fishermen of turbid surface water plumes and secchi disk measurement taken at stations several kilometers or more from the dumpsite. Such visual observations and secchi disc readings are both notoriously inaccurate and give information only for turbidity in the upper few meters of the water column. Dredged material dumping will affect surface turbidity less than near-bottom turbidity, and there are no multi-year, temporal trend data for the near-bottom suspended sediment concentrations at and near the site.

The multi-year, temporal trend, data for Central San Francisco Bay fisheries consists primarily of party boat fishing log data and does not include population surveys employing statistically designed scientific sampling. Both fish populations and the success of party boat fishermen vary greatly from year to year with many factors, particularly weather and climate, that are unrelated to dredged material dumping or other anthropogenic influences. Therefore, the available fishery data is sparse, of limited reliability, and affected by natural variability that is difficult or impossible to distinguish from any impacts of dredged material dumping.

Although the existing data demonstrate that fisheries have progressively declined in Central San Francisco Bay over the last decade or more, a period when dredged material dumping at the Alcatraz site has also progressively increased, the data are unsuitable to validate any relationship that may exist between increased dumping, fisheries decline, and increased turbidity. Similarly, the lack of significant correlation does not necessarily preclude the existence of such a relationship. Considerably more evidence would be necessary to unequivocally demonstrate or reject the hypothesized cause-and-effect relationship between dredged material dumping and/or the slurry requirement and reduced fish populations and fishing success. Such a cause-and-effect relationship may exist even if other factors (such as reduced river flow rates) also contributed to the dramatically decreased fishing success in Central Bay during the past decade or during 1986 and 1987.

Finally, the measured suspended sediment concentrations at and near the Alcatraz dumpsite (ranging from 10 to 50 mg l^{-1}, Winzler and Kelly 1985) do not exceed those concentrations of uncontaminated suspended sediments that have been shown to produce adverse effects on a variety of fish species in laboratory studies and studies in other rivers or estuaries (U.S. Army Corps of Engineers 1988). However, it has been hypothesized that the mechanism involved in the possible cause-and-effect relationship between dredged material dumping and fisheries decline may not be related to increased turbidity but to toxics in the suspended sediments released by dumping (Segar 1988, 1989). The lighter dredged material, which remains suspended longer, is known to have higher concentrations of metals and organic contaminants than the coarse-grained material (Chen et al. 1976; U.S. Army Engineer District San Francisco 1975).

SUMMARY

The majority (approximately 80%) of the dredged material dumped at the Alcatraz site is not permanently deposited in the bottom sediments of the site. Strong tidal currents sweep dredged material away from the site as suspended particulates either before the dredged material settles on the bottom or by resuspension of dredged material temporarily deposited in the dumpsite sediments during slack water periods.

Little data exists concerning the fate of the suspended sediments created by dredged material dumping at the Alcatraz site. However, it is apparent that these suspended sediments remain in Central San Francisco Bay for some time and are deposited and resuspended many times before leaving the suspended sediment regime in Central Bay. Because most suspended sediment movement takes place in the few meters above the sediments, bottom dwelling or bottom feeding organisms are potentially the most affected.

Existing evidence suggests that: (1) turbidity and/or the persistence and frequency of turbid plumes has increased in west Central San Francisco Bay in the past several years; (2) fishing success for several species has substantially declined in Central Bay, particularly in the past decade; (3) dredged material disposal is a major, perhaps dominant, contributor to the suspended sediment loads of west Central Bay and may increase the background turbidity and suspended sediment concentrations in this area, especially during the dry season when other source of suspended sediments are at a minimum; (4) the slurry requirement established in 1986 undoubtedly caused an increase in the percentage of dredged materials introduced to the suspended sediments at the Alcatraz dumpsite and, therefore, an increase in the loading rate of suspended sediments in the area, (5) the incremental turbidity and suspended sediment concentrations associated with dredged material plumes are probably too small compared to the normal background range of turbidity in Central Bay to support a conclusion that these adverse effects to planktonic and nektonic organisms are caused by dredged material plume turbidity, and (6) it is possible that the observed fisheries declines are at least partly due to the suspended sediment toxicity of the dredged material.

ACKNOWLEDGMENTS

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Entrainment of Anadromous Fish by Hopper Dredge at the Mouth of the Columbia River

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Abstract: *Studies were conducted at the mouth of the Columbia River to determine the number and types of estuarine organisms entrained by hopper dredging. As part of the study, information was obtained on the number and types of fish species entrained. Fourteen species or species groups of fish were collected during the four year study. Eulachon (*Thaleichthys pacificus*), was the only anadromous species entrained. No juvenile or adult salmonids were collected. Numbers of individuals entrained were low for all species except Pacific sand lance (*Ammodytes hexapterus*), which were collected in moderate numbers throughout the study. None of the species collected showed any seasonality except Pacific sand lance, which were slightly more abundant in the late summer. These results indicated that anadromous species were not entrained in any numbers by hopper dredging at the mouth of the Columbia River.*

INTRODUCTION

Dredging at the mouth of estuaries has raised concern over the number and types of organisms entrained by dredges. In the Pacific Northwest, entrainment of anadromous fish, particularly salmonids, is of concern because they are a major economic resource to the region. In addition, the majority of estuary dredging is done during the peak outmigration of juvenile salmonids, thus making them most susceptible to entrainment.

Previous research on entrainment of fish by hopper dredge has been fairly limited. Dutta and Sookachoff (1975) summarized the earlier work done by Fisheries and Marine Services of Canada on entrainment of juvenile salmonids by hydraulic dredging in the Fraser River. Although the sampling methods required approximations of the volume of material sampled and mortality estimates were based on a number of assumptions, it was felt that a sufficient number of juvenile salmonids were entrained to limit dredging to times outside of the peak juvenile salmon outmigration period. Subsequently, a dredging guide for the lower Fraser River was issued (Boyd 1975). However, later studies (Arseneault 1982) provided evidence which lowered the mortality estimates for fish entrained, and resulted in a lessening of dredging restrictions.

Entrainment studies have also been conducted in Grays Harbor, Washington. Bengston and Brown (1977) made some limited observations of pipeline dredged material as it was being discharged. They observed three adult spiny dogfish (*Squalus acanthias*) which had been entrained by the dredge. Tegelberg and Authur (1977) made observations with regard to fish entrained both by hopper and pipeline dredges. They collected nine species of fish from the hopper dredge and five species from the pipeline discharge, three of which, longfin smelt (*Spirinchus thaleichthys*), American shad (*Alosa sapidissima*), and eulachon are anadromous. Stevens (1981) collected data in Grays Harbor on fish entrained by various dredging operations, including pipeline, hopper and clamshell dredges. Unidentified smelt were the only possible anadromous species collected. Armstrong et al. (1982) evaluated impacts of dredging on fish as part of a Dungeness crab study in Grays Harbor. They collected a total of 15 species of fish of which only one, the chum salmon (*Oncorhynchus keta*), is anadromous. This study was the first to calculated fish entrainment per cubic yard of material dredged. In most cases, entrainment was less than 0.1 fish per cubic yard and was 0.008 per cubic yard for the one chum salmon collected.

Subsequent information on fish entrainment was obtained by the Portland District Corps of Engineers as part of a study begun in 1985 to assess impacts to Dungeness crab (*Cancer magister*) populations due to hopper dredging at the mouth of the Columbia River (MCR). The study was designed to determine the number and types of organisms entrained, and whether entrainment was correlated with any environmental or dredging parameters. This information was then used to determine if entrainment could be reduced by altering the dredging operation or schedule.

The MCR has the largest maintenance dredging program in the Portland District. The navigation channel is 5 miles (8 km) long and 2640 feet (805 m) wide (Fig. 1), of which the outer 2000 feet is authorized to a depth of -55 feet (-17 meters), while the remaining 640 feet is authorized to a depth of -48 feet (-15 meters). Approximately 7 million cubic yards of material, consisting largely of medium to fine grain sand, is dredged annually from the channel between April and October. The channel has also been considered for deepening which could substantially increase the future maintenance dredging requirement.

The Columbia River supports some of the largest anadromous fish populations on the west coast with over 100,000,000 juvenile salmonids migrating out of the estuary each year. Large runs of American shad and eulachon also migrate through the estuary each year.

METHODS

The entrainment studies were conducted aboard the Corps of Engineers' dredge ESSAYONS. The ESSAYONS is a 350-foot-long, medium-class hopper dredge with a 6000 cubic yard capacity hopper which is normally filled to approximately 4000 cubic yards. It has an average dredging speed over the bottom of 2-4 knots. The ESSAYONS has two dragarms, each equipped with

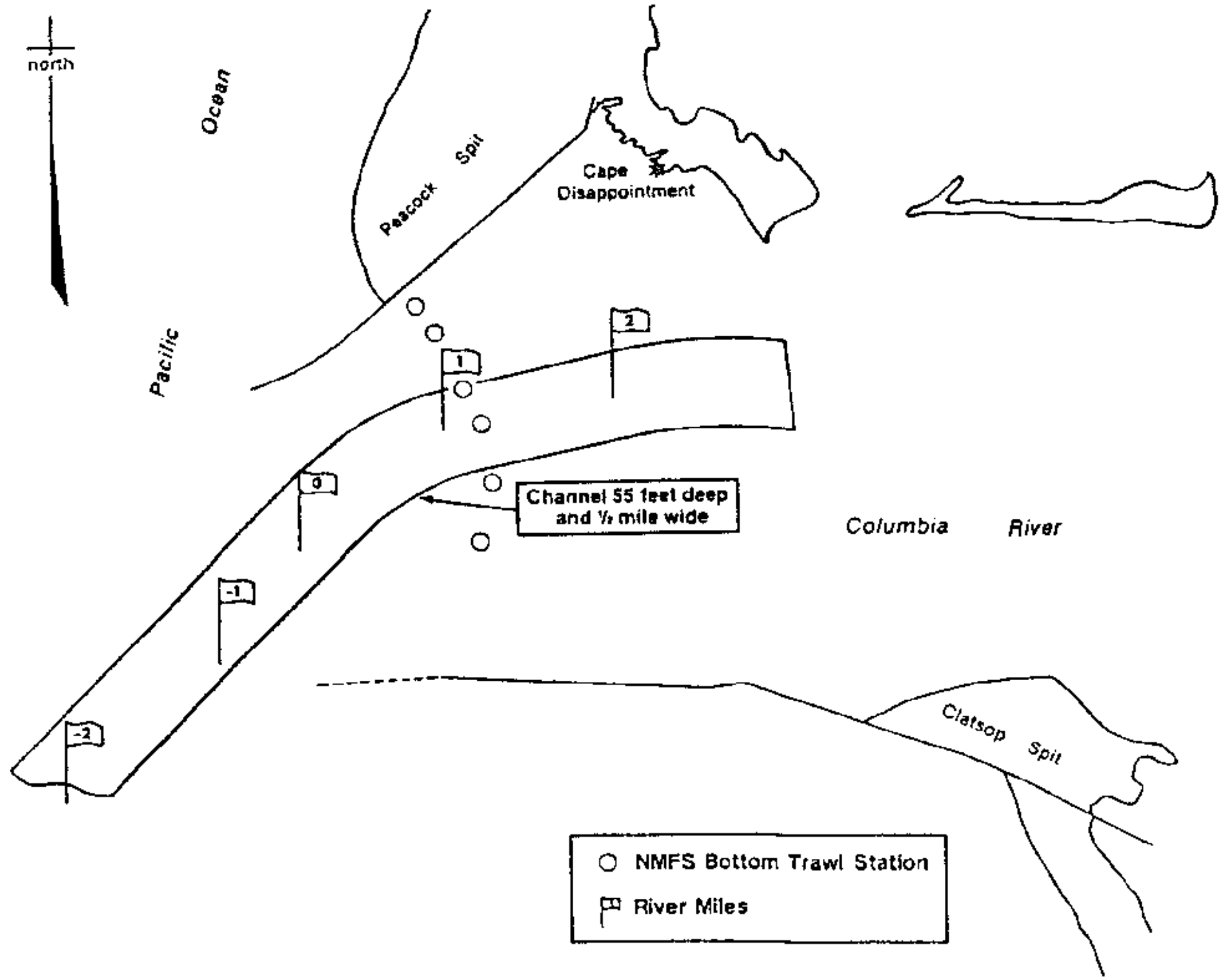

Figure 1. The navigation channel at the mouth of the Columbia River.

two pumps totaling 1500 hp. Dredged material is pumped up two 28 inch diameter pipes which converge at a central manifold, then diverge to allow material to flow to the hopper through any combination of four 26 inch diameter hopper discharge pipes, two aft and two forward. There is a lander at the end of each discharge pipe, designed to disperse the flow of material into the hopper. There are four adjustable 5 foot diameter overflow pipes in the hopper used to skim off water and lighter material and discharge it overboard. The ESSAYONS has a normal operation cycle of 120 minutes; 54 minutes dredging, 20 minutes turning, 40 minutes running time to the dump site, and 6 minutes dumping dredge material. During a normal dredging day of 24 hours, the ESSAYONS would have a total dredging time of approximately 8 hours.

The entrainment sampler was described in detail in Larson et al 1988 and Larson and Patterson 1989. It was designed to sample the dredged material before it was discharged into the hopper, by replacing the lander of the port aft discharge pipe (Fig. 2a). It is similar to the standard landers, except that the bottom is a grate of 0.64 cm x 3.14 cm steel bar set on edge and spaced 0.64 cm apart (Fig. 2b). The grate allows sand and water to pass through to the hopper while crabs, other organisms, and debris are washed into a 0.64 cm mesh collection basket. The collection basket can be raised to deck level so that the sample can be removed without delaying the dredging operation.

Evaluation of the sampler during the 1984 dredging season showed that if all four landers were open and discharging equal amounts the sampler could handle the full discharge of the port aft lander for 30 to 60 seconds. The collection basket usually became clogged with debris if operated for longer than 60 seconds.

Sampling was conducted at the MCR during the May to October dredging seasons 1985-88. The duration of each sampling cruise was generally four days. Most cruises took place during spring tides, though some were conducted during neap tides. A particular effort was made to collect samples during the spring cycles as this is the period of maximum salt water intrusion into the estuary, and is potentially the period when crab abundance is the highest in the navigation channel. Samples were collected at slack and at maximum flood and ebb tides over the four day cruises. Four samples were collected at each tide stage, two with and two against the direction of the tide, or two east and two west at slack tide. Samples were taken over a 24 hour period during 1985, but the hours were limited to 16 each day centering around maximum flood and ebb tides for the years 1986-88. A total of 789 samples were obtained during the four years of the study. Tidal stage, direction of dredging, dredge speed (knots), velocity (feet/second) and density (grams/liter) of dredge material, and dragarm depth (feet) were recorded. The duration of each sample was recorded, then samples were sorted, species identified and individuals counted.

Entrainment was calculated by relating the number of organisms collected to the total number of cubic yards dredged for each load sampled. This relationship is shown in the following formula.

A.

B.

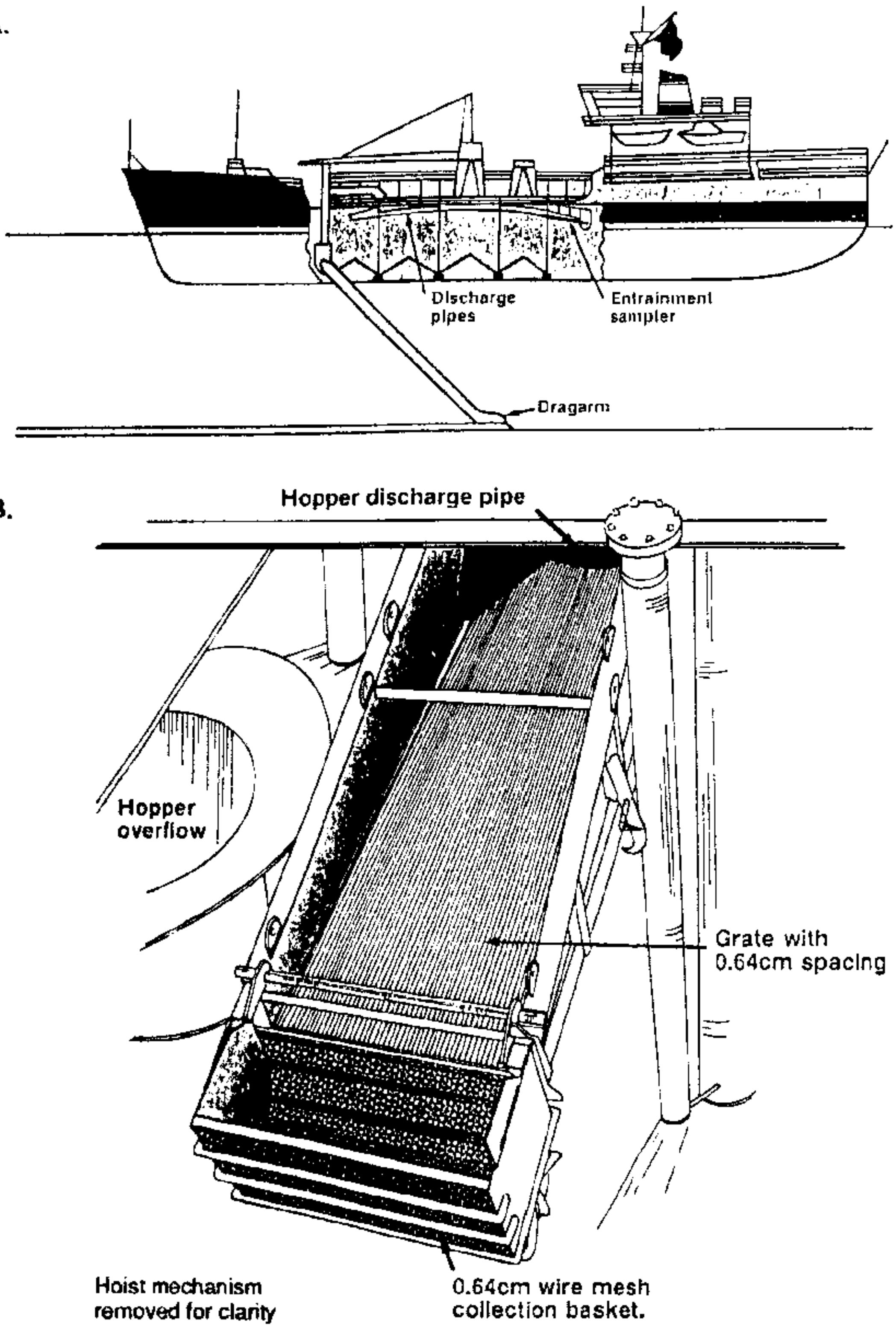

Figure 2. A. Location of entrainment sampler on dredge ESSAYONS. B. Entrainment sampler.

$$E = \frac{\frac{4N}{T}}{\frac{Y}{PT}}$$

where: E = entrainment (number of organisms per cubic yard) 4 = correction factor for 4 landers N = number of organism collected per individual sample
T = time sampled in minutes per individual sample
Y = cubic yards per load
PT = total pumping time in minutes per load

Results and Discussion

A total of 14 species or species groups of fish were encountered during the study (Table 1). Most of the fish collected were demersal species typical of the mobile sand community along the Oregon coast. They included: staghorn sculpins, Pacific sand lance, showy snailfish, poachers and juvenile flatfish. Relatively few pelagic fish species were collected; those encountered include anchovy, herring, eulachon, smelt and surfperch. Eulachon was the only anadromous species collected. No juvenile or adult salmonids were collected.

Pacific sand lance was the dominate fish species collected, accounting for 92% of all individuals entrained. They were collected in moderate numbers throughout the study (X = 0.341 individuals/cy). Sand lance showed some seasonality, with the average number entrained per month increasing between May and August, then dropping again in September and October (Fig. 3). No other relationships were observed between number of fish entrained and any of the environmental or dredging parameters recorded. Mean entrainment, standard deviation and range were calculated for the most commonly entrained species for all samples taken (Table 1). Most species were collected in small numbers with some encountered only once, i.e., Pacific sandfish, cabezon and spiny dogfish. All individuals collected were adults except the flatfish, some of the Pacific sand lance, and the spiny dogfish.

Species entrained were similar to those collected by trawl (stations shown on Fig. 1) during an earlier study (McCabe et al 1988) at the MCR (Table 2). They are also similar to species collected during the Grays Harbor study (McGraw and Armstrong 1989). Results of entrainment studies from MCR and Grays Harbor differed from the Fraser River studies. At the Fraser River, juvenile salmonids and Eulachon were the dominate species entrained, while at the mouth of the Columbia River and in Grays Harbor, estuarine and marine demersal species were the most commonly entrained species. The difference between the studies is probably due to the location of the dredge. In the Fraser River studies, the dredges were located upstream of the estuary at a point where the river was constricted. Because of this constriction, it is likely that juvenile salmon and smelt were more concentrated, making them more

Table 1. Mean entrainment, standard deviation and range of fish species entrained at MCR 1985-88. Total number of samples = 789.

Species	Mean	(S.D.)	Range
Pacific sand lance *Ammodytes hexapterus*	0.341	(1.370)	0 - 18.89
Juvenile flatfish Pleuronectiformes	0.008	(0.048)	0 - 0.90
Pacific staghorn sculpin *Leptocottus armatus*	0.003	(0.021)	0 - 0.21
Poacher Agonidae	0.009	(0.086)	0 - 2.25
Showy snailfish *Liparis pulchellus*	0.002	(0.015)	0 - 0.18
Eulachon *Thaleichthys pacificus*	0.002	(0.021)	0 - 0.32
Herring and anchovy Clupeiformes	0.008	(0.057)	0 - 1.08
Cabezon *Scorpaenichthys marmoratus*	<0.001		
Pacific tomcod *Microgadus proximus*	<0.001		
Spiny Dogfish *Squalus acanthias*	<0.001		
Big skate *Raja binoculata*	<0.001		
Gunnel Pholidae	<0.001		
Pacific sandfish *Trichodon trichodon*	<0.001		
Surfperch Embiotocidae	<0.001		

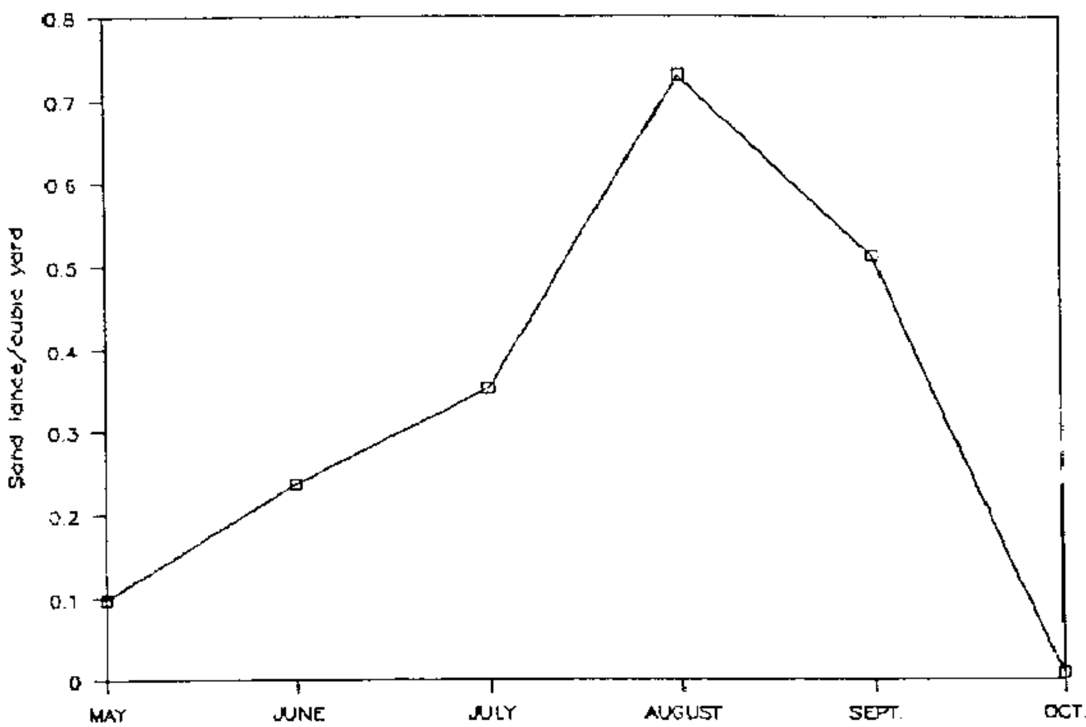

Figure 3. Pacific sand lance mean monthly entrainment 1985-88.

susceptible to entrainment. At the mouth of the Columbia River and in Grays Harbor, the juvenile salmonids and smelt may have had more area through which to migrate, thus avoiding the dredge.

On the basis of the results of the studies in Grays Harbor and at the mouth of the Columbia River, it seems unlikely that anadromous fish are entrained in any significant amounts by hopper dredging in channels through estuaries, or at large river mouths such as at the Columbia River. However, dredging in river channels where the river is constricted, particularly during periods of peak outmigration, may entrain juvenile salmonids or smelt. Additional sampling upstream of the estuary would be necessary to assess the number of fish entrained at the MCR in these situations.

Table 2. Fish species collected at the mouth of the Columbia River by trawl from1980-1981 (McCabe 1988).

Species	Apr	May	Jun	Jul	Aug	Sep	Oct
Spiny dogfish *Squalus acanthias*	x			x		x	x
Big skate *Raja binoculata*			x		x	x	
Pacific herring *Clupea pallasii*			x	x		x	
Northern anchovy *Engraulis mordax*		x	x	x		x	x
Surf smelt *Hypomesus pretiosus*	x						
Eulachon *Thaleichthys pacificus*	x						
Longfin smelt *Spirinchus thaleichthys*	x	x	x	x	x		
Whitebait smelt *Allosmerus elongatus*	x		x		x	x	
Pacific hake *Merluccius productus*			x	x			
Pacific tomcod *Microgadus proximus*	x	x	x	x	x	x	x
Walleye pollock *Theragra chalcogramma*				x			
Shiner perch *Cymatogaster aggregata*	x	x				x	x
Redtail surfperch *Amphistichus rhodoterus*	x	x		x		x	x
Spotfins surfperch *Hyperprosopon anale*	x					x	x
Sand lance *Ammodytes hexapterus*	x	x	x	x	x	x	
Black rockfish *Sebastes melanops*				x			

Table 2—cont.

Species	Apr	May	Jun	Jul	Aug	Sep	Oct
Unidentified rockfish Scorpaenidae				x			
Kelp greenling *Hexagrammos decagrammus*				x			
Pacific staghorn sculpin *Leptocottus armatus*		x	x	x		x	x
Warty poacher *Occella verrucosa*	x			x			
Pricklebreast poacher *Stellerina xyosterna*		x	x		x	x	x
Showy snailfish *Liparis pulchellus*			x		x		
Ringtail snailfish *Liparis rutteri*		x	x				
Slipskin snailfish *Liparis fucensis*	x						
Pacific sanddab *Citharichthys sordidus*					x		
Speckled sanddab *Citharichthys stigmaeus*			x	x		x	
Butter sole *Isopsetta isolepis*	x	x	x	x	x	x	
English sole *Parophrys vetulus*		x	x		x	x	
Starry flounder *Platichthys stellatus*			x			x	
Sand sole *Psettichthys melanostictus*		x	x	x	x	x	

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Fish Entrainment by Dredges in Grays Harbor, Washington

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*Abstract: Fish entrainment data were collected during several different studies conducted on pipeline and hopper dredges from 1978 to 1989 in Grays Harbor, Washington. In general, the highest entrainment rates and numbers of species of fish were observed in hopper dredge samples from the outer harbor. A total of twenty-eight species of fish were identified from entrainment samples, twenty-four of which occurred in outer harbor samples and eight from the inner harbor. Pacific sand lance (*Ammodytes hexapterus*) were entrained at the highest rate, 594/1000 cubic yard (kcy), observed in all the studies, followed by Pacific staghorn sculpin (*Leptocottus armatus*) (92/kcy), and Pacific sanddab (*Citharichthys sordidus*) (76/kcy). Only one chum salmon (*Oncorhynchus keta*) fry was caught in all samples taken, and comparison with trawl samples taken during one study indicated that some fish species actively avoid the dredges.*

INTRODUCTION

The U.S. Army Corps of Engineers (COE) dredges about 230 million cubic yards (cy) of sediment each year to maintain the waterways of the nation and about 70 million cy in new dredging projects. An additional 100-150 million cy of sediments are dredged by others under permits issued by the Corps (Engler et al. 1988). In the last 20 years, there has been increasing public and agency concern regarding the impacts of dredging and disposal on aquatic resources. Most early studies regarding dredging emphasized water quality and sediment contamination (Engler 1980), but more recent ones have focused on biological impacts, including effects of turbidity and suspended solids on fish, and the impacts of direct uptake, or entrainment, by dredges to mobile epibenthic organisms and demersal fish.

Much of the information on fish entrainment comes from studies conducted since the early 1970s on the lower Fraser River in British Columbia, Canada (Braun 1974 a,b; Dutta and Sookachoff 1975 a,b). Environment Canada, Fisheries and Marine Service, began monitoring dredging operations in 1972 by observing spoil outflow from pipeline dredges and using dipnets to sample overflow pipes (Dutta and Sookachoff 1975a). Monitoring continued and techniques were refined in subsequent years to quantify pipeline and hopper dredge entrainment during salmon outmigration and to assess mortality rates. Mortality rates were measured by introducing a known number of fry near the cutter head of a pipeline dredge or into the head of a hopper dredge. The total mortality rate for pipeline dredges was estimated at approximately 99% of those fish entrained (Braun 1974a; Dutta and Sookachoff 1975a). Using dip nets to sample overflow ports on a hopper dredge, Tutty (1976) reported recovery rates of only 1% for juvenile salmon. Histopathologic studies of surviving entrained fish revealed that salmon smolts suffered internal lesions and other conditions, indicating overall mortality due to entrainment by a hopper dredge was probably 100% (Tutty and McBride 1976). As a result of these entrainment studies, the Canadian government established dredging guidelines for the Fraser River (Boyd 1975; Arsenault 1981).

The impact of entrainment by dredges has not been studied intensively in the United States. The first entrainment studies in the U.S. of which we are aware were conducted in Grays Harbor, WA (Fig. 1). The COE presently maintains the navigation channel, removing an average of 1.6 million cy of dredged material annually, mostly with pipeline and hopper dredges. In 1974 the Seattle District COE initiated studies to determine the effects of channel dredging on the environment and biota of Grays Harbor (U.S. Army Corps of Engineers 1976-1977). Bengston and Brown (1976) conducted trawl surveys to obtain information on fish species in the harbor, and observed three spiny dogfish (*Squalus acanthias*) in a pipeline dredge spoil area. They postulated that these large fish may feed on organisms disturbed by the cutter head. Also included in the same series of studies was the first attempt to monitor crab entrainment by a hopper dredge (Tegelberg and Arthur 1977). The latter effort was not intended to assess dredging impacts to fish, but Tegelberg and Arthur reported the following species of fish collected in samples taken aboard dredges: Pacific staghorn sculpin (*Leptocottus armatus*), juvenile English sole (*Pleuronectes* [*Parophrys*] *vetulus*), juvenile American shad (*Alosa sapidissima*), juvenile longfin smelt (*Spirinchus thaleichthys*), threespine stickleback (*Gasterosteus aculeatus*), northern anchovy (*Engraulis mordax*), sanddab (*Citharichthys stigmaeus*), bay pipefish (*Syngnathus leptorhynchus*), and eulachon (*Thaleichthys pacificus*). Three subsequent studies were conducted by Stevens (1981), Armstrong et al. (1982), and McGraw et al.(1988), primarily to assess crab impacts due to dredging; however some data on fish entrainment were also collected (Dinnel et al. 1986, 1987). The following information on fish entrainment is synthesized in this paper because it is either unpublished or contained only in government reports, and has not been widely available previously for those interested in dredging impacts.

METHODS

The entrainment studies by Stevens (1981), Armstrong et al. (1982), McGraw et al. (1988) and Wainwright et al. (1990) were conducted in various portions of the Grays Harbor navigation channel (Fig. 2) aboard several different dredges. However, the sampling methodology was generally the same for all studies, in that discharged sediment was strained through steel baskets or nets, entrained organisms were identified and counted, and some estimates were made of the volume of material sampled. Entrainment rates were usually calculated by

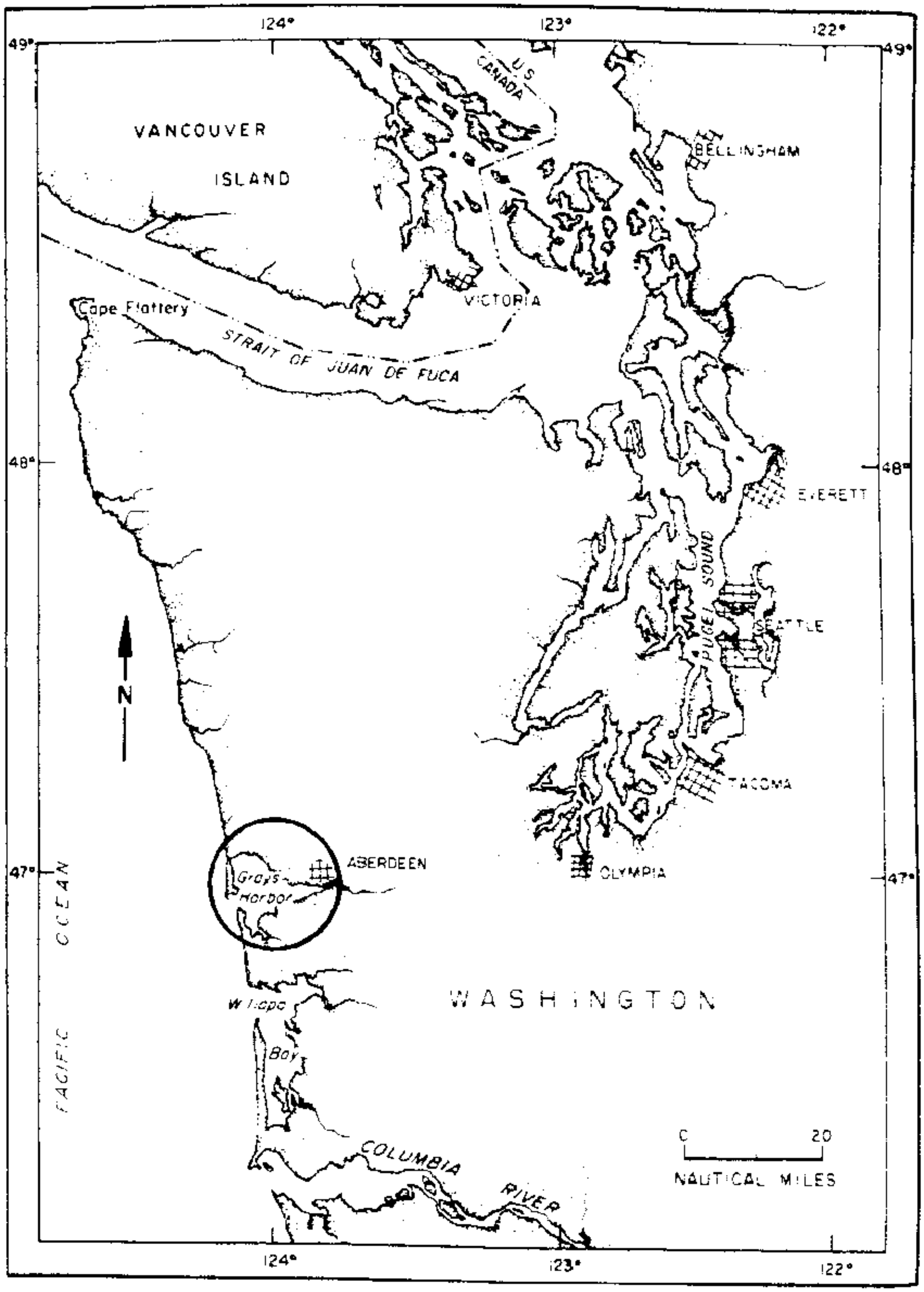

Figure 1. Map of Washington State showing the location of Grays Harbor.

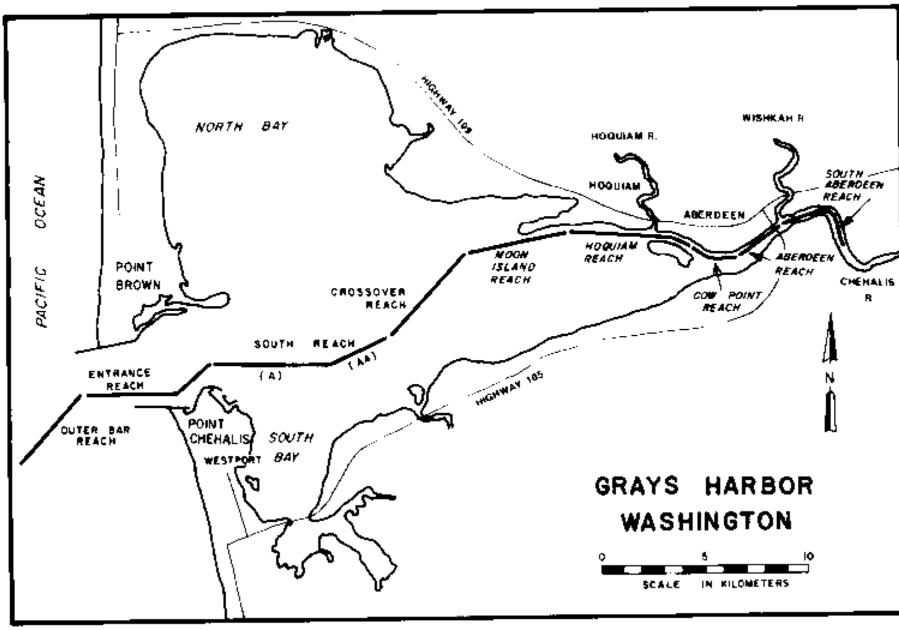

Figure 2. Map of Grays Harbor showing the location of the navigation channel.

multiplying the number of entrained organisms per unit of time by the pumping rate (in cy per unit time), yielding an entrainment rate in number of organisms per cy or other convenient volumes (e.g., # fish/kcy). This is a practical and useful term since dredging is performed and recorded on a volumetric basis. Methodologies for each of the three studies vary somewhat and are briefly summarized below. Details can be found in the respective report or publication for that study.

In his study from October 1978 to December 1979, Stevens (1981) estimated crab and fish entrainment rates for two hopper dredges (PACIFIC and SANDSUCKER) in South Reach, Crossover Reach, and the Westport Marina, and a pipeline dredge (MALAMUTE) in the inner harbor reaches (i.e., Moon Island Reach to Aberdeen), and Terminal Four, adjacent to the Aberdeen Reach (Fig. 2). He tried several different sampling techniques aboard the PACIFIC, which was equipped with a modified sidecaster arm used to take samples by straining material through a steel basket suspended below the sidecaster. The basket was constructed of diamond-shaped mesh, 16 mm x 38 mm; basket dimensions were 76 cm x 76 cm x 61 cm. Other samples were taken with a 2.4 m trawl net with a 2.0 cm mesh. The net was tied around the mouth of the modified discharge pipe and was nearly submerged during sampling. A similar method was employed aboard the SANDSUCKER. Collapsible sampling baskets (35.6 cm x 40.6 cm x 76.2 cm deep) made of expanded steel were positioned below one of eleven discharge ports on a rotating basis. The mesh size was 44 mm (long axis) x 16 mm (short axis). Sampling aboard the pipeline dredge was accomplished by straining material through a large steel basket (91 cm x 76 cm x 122 cm, with a mesh size of 44x16 mm) held with a forklift under the discharge end of the pipeline.

Armstrong et al. (1982) collected entrainment samples aboard two pipeline dredges (MALAMUTE and McCURDY) in the general vicinity of the Aberdeen Reach during two different seasons (summer and winter/spring) and aboard the hopper dredge SANDSUCKER in South Reach, Crossover Reach, and portions of the inner harbor area during May to September 1980. Protocol and gear used to sample the discharge pipe aboard the SANDSUCKER were the same as described by Stevens (1981) except that the mesh size on the metal baskets was reduced to increase sampling efficiency for young-of-the-year Dungeness crabs and outmigrating salmon fry and smolts. The baskets (35.6 cm x 40.6 cm x 76.2 cm depth, with an outside mesh of 44 x 16 mm), were lined with a plastic mesh (12.7 mm) attached to the inside to reduce the overall mesh size to less than 12.7 mm. Armstrong et al. used a different method to take samples for the pipeline dredges. They placed a 15.2 m net at the perimeter of the discharge area through which effluent flowed and estimated the percent of total discharge sampled.

Trawl samples were also collected during the 1980 studies at stations throughout the harbor, from Westport to Aberdeen, to provide estimates of crab and fish populations during the study. Trawls were made biweekly from May through October in 1980 and at 4-5 weeks intervals from November 1980 through June 1981. All organisms caught in trawl samples were identified, measured, and counted.

The Corps hopper dredge YAQUINA was used in October 1985, and August 1986, 1987 and 1989 for further studies (McGraw et al. 1988; Wainwright et al. 1990). Areas sampled included the Grays Harbor Bar, South Reach, and Crossover Reach (Fig. 2). The dredge was equipped with a modified distribution system and collection baskets for each dragarm (Figs. 3 and 4) which permitted sampling discrete amounts of dredged material that could be accurately estimated by instruments on the vessel. During sample collection, all dredged material from each draghead was diverted to collection baskets for a specified period of time. The collection baskets were perforated with 0.6 cm (diameter) holes so that most small organisms could be retained. This sampling method was an improvement over previous studies because of increased accuracy of estimating the volume of material being sampled. In addition, simultaneous trawl samples were conducted in the channel using a 3-m beam trawl to obtain more accurate density estimates of crab and fish. Fish data were obtained from 6 of the 1985 trawl samples, 5 of the 1986 trawl samples (Dinnel et al. 1986, 1987), and 42 of the 1989 samples (Wainwright et al. 1990).

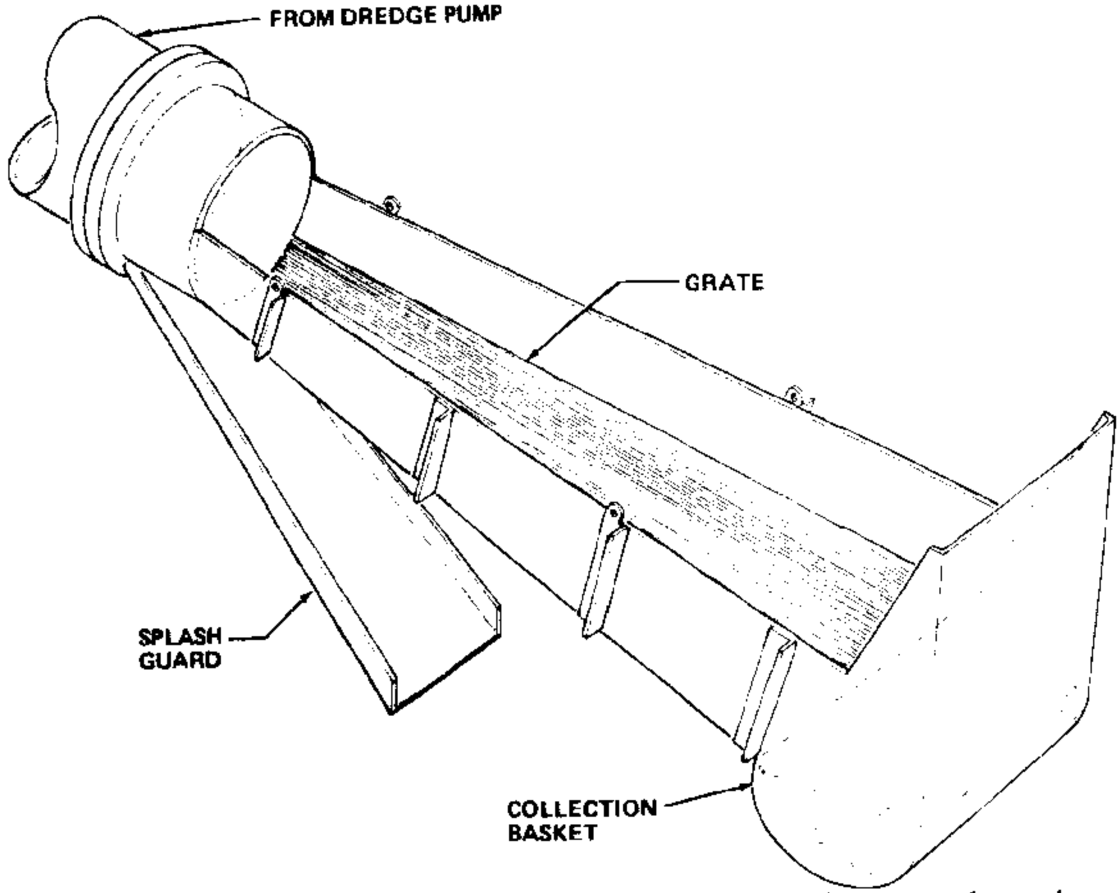

Figure 3. Drawing of the collection basket used in the entrainment study on the YAQUINA.

RESULTS AND DISCUSSION

Although the major entrainment studies in Grays Harbor occurred over a ten-year period (1979-1989), results were similar in some respects. For example, the greatest entrainment rates and number of species of fish occurred in the South Reach portion of the navigation channel (Tables 1 and 2) and decreased with distance toward the inner harbor area (i.e., Moon Island Reach to Aberdeen). One notable exception was the entrainment rate for snake prickleback (135/kcy) in inner harbor samples obtained with the hopper dredge SANDSUCKER during the summer of 1980 (Armstrong et al. 1982). This species was not observed in any pipeline dredge (MALAMUTE) samples during the same season, however (Table 2), indicating that the population is highly variable or, possibly, that hopper dredges may entrain certain types of fish at higher rates than pipeline dredges. All of the fish species identified during entrainment studies in the last decade in Grays Harbor are listed in Tables 1 and 2 and are divided according to location and the type of dredge (i.e., hopper or pipeline dredge) used in the study.

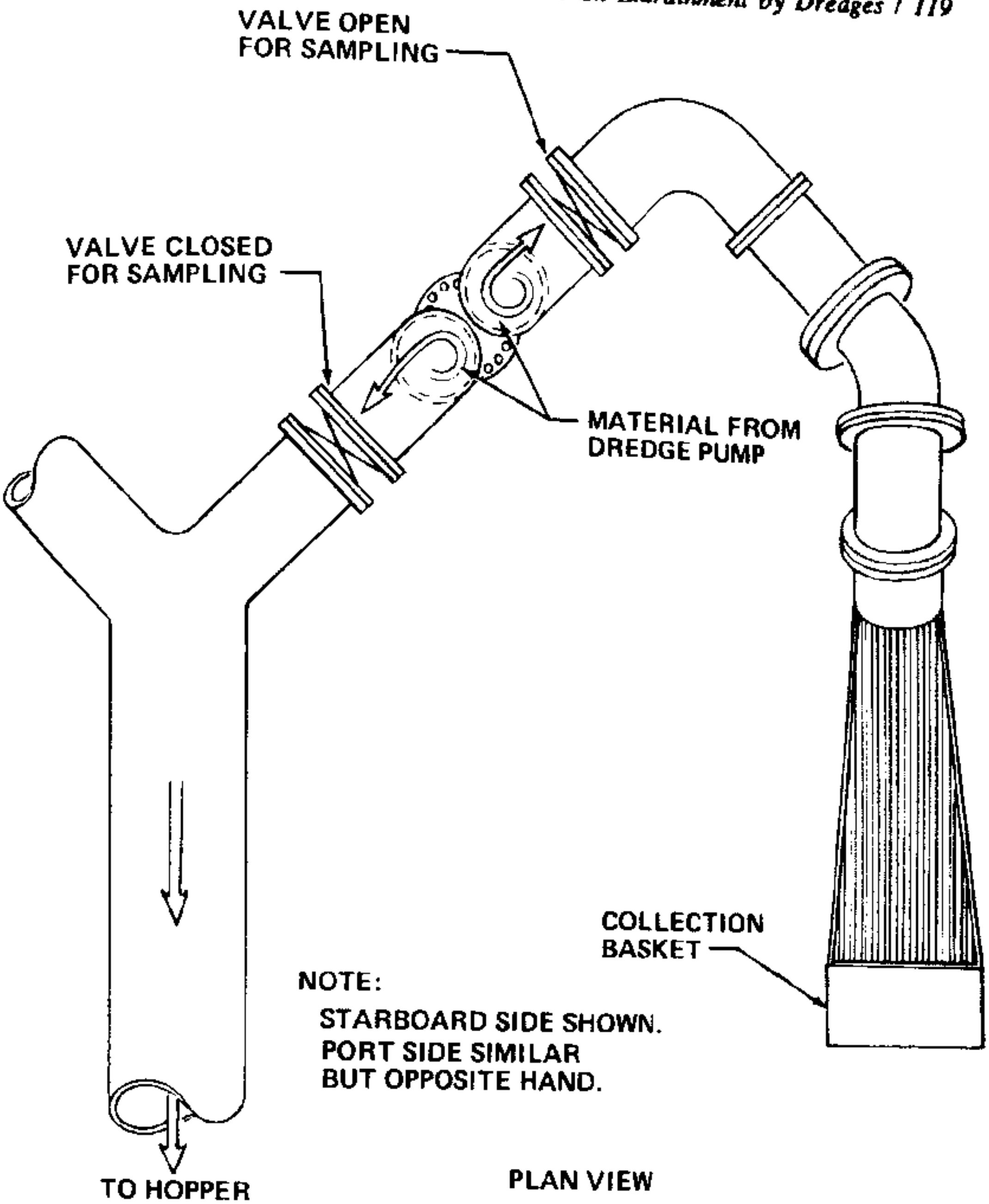

Figure 4. Schematic drawing of the distribution system for the collection baskets aboard the YAQUINA.

Overall, the species with the highest entrainment rates in the three studies were Pacific staghorn sculpin (92/kcy), Pacific sand lance (594/kcy), and Pacific sanddab (76/kcy). However, sand lance were conspicuously absent from the Armstrong et al. 1980 dredge and trawl samples (Tables 1 and 3). This may indicate a very low population of sand lance in the harbor during that time, a very aggregated distribution, or a less intense sampling effort compared to later

Table 1. Mean fish entrainment rates (# fish per 1000 cy) for hopper dredges in various reaches of the Grays Harbor Navigation channel.

Species	Bar Reach*	South Reach**								Crossover Reach***			Inner Harbor****	
	8/87	11/78	12/78	3/79	Sum 1980	10/85	8/86	8/87	8/89	3/79	Sum 1980	8/86	3/79	1980
Anchovy (Engraulididae)									1					
Arrowtooth flounder (*Atheresthes stomias*)						8	22							
Bay pipefish (*Syngnathus leptorhynchus*)		6												
Buffalo sculpin (*Enophrys bison*)		6												
English sole (*Pleuronectes [Parophrys] vetulus*)	21	6			35		25	32	9					
Flatfish (unident.) (Pleuronectiformes)	12						28	11	1			4		
Kelp greenling (*Hexagrammos decagrammus*)							1							
Lingcod (*Ophiodon elongatus*)					2			1						
Longnose skate (*Raja rhina*)	3													
Northern anchovy (*Engraulis mordax*)					18									

Table 1—cont.

Species	Bar Reach*	South Reach**								Crossover Reach***			Inner Harbor****	
	8/87	11/78	12/78	3/79	Sum 1980	10/85	8/86	8/87	8/89	3/79	Sum 1980	8/86	3/79	1980
Pacific sanddab (*Citharichthys sordidus*)	24	19	21	8	76		10	9	9		3	4		
Pacific sandfish (*Trichodon trichodon*)					2									
Pacific sand lance (*Ammodytes hexapterus*)	57			47		36	175	72	594	39				
Pacific staghorn sculpin (*Leptocottus armatus*)	18	81	62	31	92	35	47	12	7		27	8		16
Pacific tomcod (*Microgadus proximus*)				1			2					8		
Pipefish (Syngnathidae)				8										
Prickly sculpin (*Cottus asper*)														20
Saddleback gunnel (*Pholis ornata*)					5		5	1	3			4		
Sand sole (*Psettichthys melanostictus*)	9	6		16		1	5	2	6					
Slender sole (*Lyopsetta exilis*)								1						
Smelt (Osmeridae)													9	

Table 1—cont.

Species	Bar Reach*	South Reach**								Crossover Reach***			Inner Harbor****	
	8/87	11/78	12/78	3/79	Sum 1980	10/85	8/86	8/87	8/89	3/79	Sum 1980	8/86	3/79	1980
Snailfish (Cyclopteridae)				1										
Snake prickleback (*Lumpenus sagitta*)								6	3		8			135
Speckled sanddab (*Citharichthys stigmaeus*)					3									
Starry flounder (*Platichthys stellatus*)				1			2		1					
Surfperch (Embiotocidae)							1							
Warty poacher (*Occella verrucosa*)	9													

*Bar Reach data were collected aboard the Corps dredge YAQUINA (n=84).

**South Reach data were collected as follows: 1) November 1978 (n=24) and December 1978 (n=24), aboard the SANDSUCKER; 2) March 1979 (n=18), aboard the PACIFIC; 3) Summer 1980 (n=10) aboard the SANDSUCKER; 4) October 1985 (n=171) and August 1986 (n=248), 1987 (n=385), and 1989 (n=569) aboard the Corps dredge YAQUINA.

***Crossover Reach data were collected in 1979 aboard the dredge PACIFIC (n=2), in 1980 aboard the SANDSUCKER (n=11), and in 1986 aboard the Corps dredge YAQUINA (n=52).

****Inner Harbor samples were collected in 1979 aboard the PACIFIC (n=3), and in 1980 aboard the SANDSUCKER (n=4).

The 1980 samples include Cow Point Reach samples taken in June and Moon Island samples taken in August 1980.

Table 2. Mean fish entrainment rates (# fish per 1000 cy) for pipeline dredges in various locations in Grays Harbor.

		Inner Harbor Area		
	Westport Marina*	Terminal Four*	Navigation channel	
Species	Nov/Dec 1979	Dec 1979	Summer 1980*	Win/Sp 1981**
Chum salmon (*Oncorhynchus keta*)				8
English sole (*Pleuronectes* [*Parophrys*] *vetulus*)	3			1
Gunnel (Pholidae)	3			
Pacific staghorn sculpin (*Leptocottus armatus*)	37	15		1
Prickly sculpin (*Cottus asper*)				4
Saddleback gunnel (*Pholis ornata*)			23	
Stickleback (Gasterosteidae)		2		
Threespine stickleback (*Gasterosteus aculeatus*)			4	

*Samples were taken aboard the hopper dredge MALAMUTE.

**Samples were taken aboard the hopper dredge MCCURDY.

studies. Flatfish, especially English sole, were a main component of fish entrained by the YAQUINA in South Reach in 1986 and 1987 (25 and 32/kcy, respectively) (Table 1). Armstrong et al. (1982) found a similar rate (35/kcy) for South Reach. Comparisons with other studies show that relatively high entrainment rates for sand lance and staghorn sculpins were also reported by Larson and Moehl (1990) and Tutty and Morrison (1976). The former reported that sand lance was the dominant species collected in their study (approximately 380/kcy) and accounted for 92% of all fish entrained, while staghorn sculpin had the next highest entrainment rate (10/kcy). The highest entrainment rates for sand lance and staghorn sculpin reported by Tutty and Morrison (1976) were 25,000 and 798 per day, respectively, with total entrainment estimates of about 165,000 and 3,000 for the three month dredging period. Sookachoff (1977) reported that the hopper dredge FORT LANGLEY entrained over 216,000 sand lance during a four

Table 3. Comparison of fish entrainment rates (pipeline dredge) and trawl density in Cow Point Reach during May 1980 (adapted from Armstrong et al. 1982).

Species	Entrainment rate (Fish/kcy) (n=4)	Trawl density (Fish/1000 m^2) (n=2)
Buffalo sculpin (*Enophrys bison*)	0	1
English sole (*Pleuronectes [Parophrys] vetulus*)	0	5
Longfin smelt (*Spirinchus thaleichthys*)	0	16
Pacific staghorn sculpin (*Leptocottus armatus*)	0	3
Prickly sculpin (*Cottus asper*)	0	4
Saddleback gunnel (*Pholis ornata*)	23	4
Snake prickleback (*Lumpenus sagitta*)	0	4
Starry flounder (*Platichthys stellatus*)	0	6
Threespine stickleback (*Gasterosteus aculeatus*)	4	1

month dredging period. Unfortunately, the numbers from the last two studies were not reported in # fish/kcy, so comparisons to other data sets based on a volumetric entrainment rates cannot be made.

Eulachon, an anadromous species reported in the Fraser River entrainment studies and by Larson and Moehl (1990), was observed by Tegelberg and Arthur (1977) in their study, but not in any other Grays Harbor entrainment studies. Some possible explanations may be that it is a relatively fast swimming fish and that the Grays Harbor population may be very sparse. No eulachon were caught in trawl samples by Armstrong et al (1982), indicating a possible avoidance of the net or relatively low density.

Over the last few years, some Washington state resource agencies have expressed concern about entrainment of lingcod (*Ophiodon elongatus*) by hopper dredges in Grays Harbor because of the close proximity of the navigation channel to areas used by juvenile lingcod. However, only two lingcod were observed in all study samples, one by Armstrong et al. (1982) in hopper dredge samples in 1980 and one in 1987 YAQUINA samples (Table 1).

Dredging effects on salmon have also been an issue in Grays Harbor. Unlike the Fraser River studies however, entrainment of salmon did not appear to be a problem in areas sampled during the Grays Harbor studies. The only salmon observed was a 37 mm chum salmon fry entrained by a pipeline dredge (Table 2) operating at Cow Point in February 1981 (Armstrong et al. 1982). This contrasts greatly with juvenile salmon entrainment rates reported for the Fraser River (up to 858 pink salmon (*O. gorbuscha*) per day. The probable reasons for the difference in entrainment rates between the Grays Harbor studies and the Fraser River studies were: the time of the dredging in relation to outmigration; the location of dredging in relation to the river mouth; and the populations of outmigrants in each system. Outmigrating salmon fry and smolts are much more available to a dredge in confined upstream portions of a river during their peak downstream migration. This occurs from February through May for chum and coho salmon in the Chehalis River, one of six rivers emptying into Grays Harbor. Most of the Grays Harbor entrainment studies were conducted outside peak salmonid periods and were mostly within the harbor proper. However, much of the dredging in the Fraser River estuary was actually within portions of the Fraser River rather than in the harbor proper (Braun 1974a; Dutta and Sookachoff 1975b; Boyd 1975) and occurred during peak periods of outmigration (March through June). If the dredging in Grays Harbor were to be conducted during February through May farther up the Chehalis River, east of the Aberdeen Reach (Fig. 2), more salmon might be entrained. In addition, salmon outmigration in the Fraser River, which has a drainage area approximately 35 times that of the Grays Harbor system (Dutta 1976; Loehr and Colias 1981) has been estimated to be as high as 411,000,000 chum and chinook fry (Sookachoff, 1977). Although no reliable figures are available for Grays Harbor salmon outmigration, the six rivers which empty into the estuary probably produce only a fraction of the salmon fry observed in the Fraser, reducing the probability of entrainment.

A comparison of trawl data with entrainment data collected by Armstrong et al. (1982) indicates that larger crabs and some fish were avoiding the dredges. For example, only two fish species were found in the May pipeline entrainment samples at Cow Point (inner harbor area), while nine fish species were found in the trawls at that location and time (Table 3). Some of those fish, such as longfin smelt, the most abundant fish in the trawls, were absent from the dredge samples. Data from hopper dredge samples at Cow Point and other portions of the navigation channel also indicated that some fish species were sometimes numerous in trawls but never present in the hopper dredge samples; those included: buffalo sculpin (*Enophrys bison*), starry flounder (*Platichthys stellatus*), and shiner perch (*Eumatogaster aggregata*). However, these three species are not ubiquitous in the navigation channel, as are English sole (Rogers et al. 1988), which are frequently entrained. For the fish species which were entrained, size did not appear to be a factor in avoidance, as the average sizes were similar for both dredge and trawl samples. The largest fish in the 1980/81 hopper dredge entrainment samples was a 234 mm tomcod, indicating that, as noted by Bengston and Brown (1976), larger fish of some species are sometimes entrained.

Trawl data from the 1985, 1986, and 1989 entrainment studies (Dinnel et al. 1986, 1987; Wainwright et al. 1990) showed that the trawl caught several times more species in the navigation channel than were entrained by the dredge (Table 4). The most frequently observed fish in the trawls were English sole, sculpins, and Pacific sanddabs; these fish also occurred in entrainment samples. However, Pacific sand lance, the fish with the highest entrainment rates, was either absent from trawl samples or occurred in very low numbers, indicating an ability to avoid trawls when disturbed. This apparent contradiction is probably explained by the fast swimming ability of sand lance, which allows them to escape a trawl when touched first by a tickler chain, and their burrowing behavior in sandy substrates, which makes them vulnerable to the powerful suction generated by the dragheads of hopper dredges.

CONCLUSIONS

Entrainment data from Grays Harbor do not indicate any substantial impacts to major commercial or sport fish species in Grays Harbor. Some indirect impacts to juvenile salmonids and piscivores may result from entrainment of prey species, such as sand lance; however, the effects are difficult to evaluate at this time. Conversely, the relatively high entrainment rate for staghorn sculpins may reduce predation on some other fish species and Dungeness crab which they consume. It is also difficult to assess the relative impacts to some fish species because seasonal population densities of many are not well known or monitored. For example, dredging impacts may be most direct on juvenile English sole (<150 mm), which occur seasonally in high abundance throughout the estuary from late winter/early spring to late summer (Rogers 1985; Shi 1987; Rogers et al. 1988; Gunderson et al. 1989). Using the entrainment data for English sole in Table 3, (mean=21 English sole/kcy), we can estimate dredging impacts in the South Reach portion of the navigation channel from the planned widening and deepening project. Approximately 1.8 million cy of material will be dredged from the South Reach (U.S. Army Corps of Engineers 1988), which would result in the entrainment of 38 thousand juvenile English sole (i.e., 21 English sole/kcy x 1.8 million cy). Depending on which population estimate is used, this represents from about 0.5% to 2.5% of the total juvenile English sole population in Grays Harbor. However, this method of calculating impacts does not address the density dependent nature of entrainment, habitat preference, size, natural mortality, migration, and other important factors. Unlike the effort to estimate entrainment impacts on crab (Armstrong et al. 1987), no similar impact models have been developed for English sole or any other species of juvenile fish which utilize Grays Harbor as a nursery area.

Table 4. Ratio of fish entrainment rates (in # fish/1000 m^2) for the Corps hopper dredge YAQUINA to trawl densities (in # fish/1000 m^2) for the South Reach in October 1985, August 1986, and August 1989 and the Crossover Reach during August 1986.

Species	South Reach			Crossover Reach
	Oct 1985	Aug 1986	Aug 1989	Aug 1986
Anchovy (Engraulididae)	0:3		0.1:0	0:25
Arrowtooth flounder (*Atheresthes stomias*)	1:0	2.5:0		
Butter sole (*Isopsetta isolepsis*)	0:6		0:1.5	
Curlfin sole (*Pleuronichthys decurrans*)			0:0.04	
Dogfish shark (*Squalus acanthias*)	0:0.2	0:1	0:1	0:4
English sole (*Pleuronectes* [*Parophrys*] *vetulus*)	0:14	3:51	1.7:71	0:50
Flatfish (unident.) (Pleuronectiformes)		3:0	0.1:0	0.6:0
Flathead sole (*Hippoglossoides elassodon*)				0:0.5
Goby/Blenny (Gobiidae/Blenniidae)			0:0.1	
Greenling (Hexagrammidae)	0:0.2		0:0.2	
Gunnels (Pholidae)	0:3	0:0.3	0:0.7	0:11
Herring (Clupeidae)			0:0.7	
Kelp greenling (*Hexagrammos decagrammus*)		0.1:0 (see greenling)		
Pacific herring (*Clupea pallasi*)				0:0.5 (see herring)
Pacific sanddab (*Citharichthys sordidus*)	0:16	1:13	2:0.3	0.6:14
Pacific sand lance (*Ammodytes hexapterus*)	4.2:0	20:1	103:1.5	0.6:0
Pacific staghorn sculpin (*Leptocottus armatus*)	4.2:? (see sculpins)	5:?	1.2:? (see sculpins)	1:?

Table 4—cont.

Species	South Reach Oct 1985	South Reach Aug 1986	South Reach Aug 1989	Crossover Reach Aug 1986
Pacific tomcod (*Microgadus proximus*)	0:12	0.3:14	0:13	0:4
Pipefish (Syngnathidae)	0:3		0:0.2	
Poacher (Agonidae)			0:0.08	
Pricklebacks (Stichaeidae)	0:1	0:2	0:3	0:1
Rockfish (Scorpaenidae)			0:0.2	
Saddleback gunnel (*Pholis ornata*)		0.6:0 (see gunnels)	0.5:?	0.6:0
Salmon (*Oncorhynchus* sp.)			0:0.04	
Sand sole (*Psettichthys melanostictus*)	0.1:1	0.6:0	1:0.3	
Sculpins (Cottidae)	?:21	?:21	?:10.2 (see Pacific staghorn sculpin)	?:38
Skate (Rajiidae)			0:0.04	
Smelt (Osmeridae)	0:0.4	0:1	0:0.8	0:24
Snailfish (Cyclopteridae)			0:0.3	
Snake prickleback (*Lumpenus sagitta*)			0.5:? (see pricklebacks)	
Starry flounder (*Platichthys stellatus*)		0.3:0	0.1:0.2	
Stickleback (Gasterosteidae)			0:0.2	
Surfperch (Embiotocidae)	0:3	0.1:1	0:4.3	0:0.5
Tubesnout (Gasterosteidae)			0:0.07	

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Dec 76 R-4 Vol 4= Appendix G (Fish), H (Avian Fauna), I (Mammals), and J (Ecological Relationships)

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Use of Dredged Material to Enhance Fish Habitat in Lower Granite Reservoir, Idaho-Washington

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***Abstract**: The focus of the workshop was to examine the effects of dredging on anadromous fishes on the Pacific Coast. This paper discusses a field demonstration project involving the use of dredged material to enhance habitat for salmonid fishes in Lower Granite Reservoir, Idaho-Washington. Deposition of sediment at the confluence of the Snake and Clearwater rivers, near Lewiston, Idaho threatens the integrity of a levee system used for flood control. Experimental dredging and disposal of the material about 32 km downstream at a mid-depth disposal (3-8 m) site has been preliminarily evaluated for beneficial uses to fishery resources. Water quality was not affected about 100 m downstream from the dumping site shortly after disposal at the surface and mid-column but increased suspended solids and turbidity were measured on the bottom. A total of 4,100 fish were collected during spring 1988. Sixteen species of fish were collected at the disposal site compared to 16 at a mid-depth reference and 19 at a shallow water reference site. Catches of target fishes at the disposal site appear to fall between the most productive shallow water sites and the mid-depth reference sites. Recolonization of benthos occurred four months after deposition of dredged material, although standing crops of chironomids and oligochaetes were lower than reference sites.*

INTRODUCTION

The focus of this workshop was to examine the effects of dredging on anadromous fishes on the Pacific Coast. This paper discusses a field demonstration project involving the use of dredged material to enhance habitat for salmonid fishes. The U.S. Army Corps of Engineers is evaluating habitat enhancement as part of their long-term alternative to resolving the sediment deposition problem in Lower Granite Reservoir, Idaho-Washington.

Use of dredged material to enhance habitat for fish and wildlife is a fairly widespread practice (Landin and Smith 1987). To date, most uses have been to benefit wildlife resources or coastal fish habitat (McKern and Iadanza 1987). Anderson (1985) reported that islands created in the Campbell River, B.C. provided valuable feeding habitat for wild chinook salmon (*Oncorhynchus tshawytscha*); abundance of food organisms for salmon was as high as in the natural marsh. McConnell et al. (1978) reported that the Miller Sands, Columbia River, Oregon habitat development sites created by dredged disposal were used extensively by juvenile chinook salmon for feeding. However, both the Campbell River and Miller Sands studies dealt with estuarine systems. Estuarine systems naturally receive a high deposition of sediment. As a result, sediment size structure is generally smaller, which produces a different benthic fauna compared to larger substrates more common in "up-river" systems. However, the results of estuarine situations can not necessarily be applied to dredge and disposal in upriver fluvial habitats. Therefore, biotic responses to the deposition of dredged materials may be totally different than in the lower river systems or estuary. The key to successful use of dredged material for habitat enhancement in any system is to identify habitat requirements of the target species and then create habitat attributes that are limiting to the target species.

Sediment deposition in the upstream end of Lower Granite Reservoir, Idaho-Washington, was originally predicted prior to reservoir impoundment in 1975. Since then, the U.S. Army Corps of Engineers have been monitoring sediment deposition and presently estimate that 1,529,200 m^3 (2,000,000 yd^3) of sediment enters the reservoir annually. The majority of this material moves downstream and out of the reservoir but approximately 611,680 m^3 (800,000 yd^3) is deposited in the vicinity of the confluence of the Snake and Clearwater rivers near Lewiston, Idaho and Clarkston, Washington. Deposition of this material threatens the flood control and navigational purposes of the reservoir. The integrity of the levee system constructed as part of the Lower Granite project to protect portions of Lewiston is the greatest concern, followed by navigational access to local ports. A number of possible alternatives to increase freeboard on the levees are being examined. It appears that any long-term solution will involve some level of dredging. In-water disposal appears most feasible because the availability of upland sites is limited as a result of the steep, talus cliffs. Disposal of dredged material may be feasible at least 33 km downstream of the confluence, without affecting the flood control profile at the confluence.

To provide information on the distribution and abundance of fishes and benthos, several studies have been conducted in Lower Granite Reservoir. Initial studies focused on use of deep (>20 m) and shallow (<6.6 m) habitats by fish and benthic communities (Bennett and Shrier 1986). Their findings suggested that the benthic community was dominated by dipteran insect larvae and oligochaetes. Benthos abundance was similar among sites but varied seasonally. Shallow waters were important resting and feeding areas for migrating chinook salmon and rainbow (steelhead) trout (*Oncorhynchus mykiss*) juveniles. Deep water fish communities were dominated primarily by nongame fishes, although white sturgeon (*Acipenser transmontanus*) were present.

Initial dredging and land disposal were monitored in 1986. Results indicated that water quality from dredging and disposal was not significantly altered during the winter dredging (Bennett and Shrier 1987). In 1987, monitoring was expanded to include proposed in-water sediment disposal at mid-depth (>6.6-<19.8 m) sites. Preliminary data collection for fishes and benthos was conducted at mid-depth sites and a more extensive data base on shallow and deep water sites was developed. In 1988, the Corps began the first year of testing in-water disposal. The second year of flood control dredging was conducted and, for the first time, experimental placement of dredged material was made in-water at selected mid-depth and deep sites. Interest in mid-depth disposal was the potential for creation of shallow water habitat. Disposal specifications indicated that water depth at the disposal site would be decreased to about 4 m. Desired characteristics of the disposal were based on recommendations from an Adaptive Environmental Assessment Management (AEAM) workshop held in 1987 to resolve technical issues among resource agencies (Webb et al. 1988). At the workshop, three disposal strategies were recommended: (1) disposal at a mid-depth underwater plateau; (2) creation of an island; and, (3) deep water disposal. This report summarizes findings at the mid-depth disposal site (underwater plateau).

Study Area

Lower Granite Reservoir is a 3602 ha impoundment on the Snake River. The Snake River originates in southwestern Wyoming, flows west through southern Idaho, cuts north to Lewiston and flows west into the Columbia River in Washington. Four lower Snake River dams have created about 210 km of slack water to Lewiston, river kilometer (RKm) 224. Lower Granite is formed by the uppermost dam at RKm 173 on the lower Snake River. The reservoir is approximately 51 km long and averages 643 m in width. Depth averages 17 m but ranges from shallow, sandy shorelines <1 m in depth to a maximum of 42 m. Shallow water habitat (<6.6 m) comprises less than 8% of the total surface area and most of that occurs in the upper end of the reservoir.

As part of the 1988 survey, eight study stations were established in Lower Granite Reservoir from about RKm 179 to RKm 204. Three of the eight stations will be discussed for purposes of this paper (Fig. 1). The site at RKm 204 was the shallow water reference site. A mid-depth reference (RKm 179) site was selected for comparison with the mid-depth disposal site (RKm 193). A thorough description of these sites is given by Bennett et al. (1988).

Methods

We sampled the fish assemblage at all three sites with an 8 m shrimp (bottom) trawl, a 3.3 m surface trawl, and 66x2 m gillnets. A 165x5 m purse seine was also used at the mid-depth and shallow reference sites; the mid-depth disposal site was too shallow for its use. At least three hauls were taken with the bottom trawl on each of five days during May and June. Similarly, at least three to five hauls were taken with the surface trawl on each of 2 days during

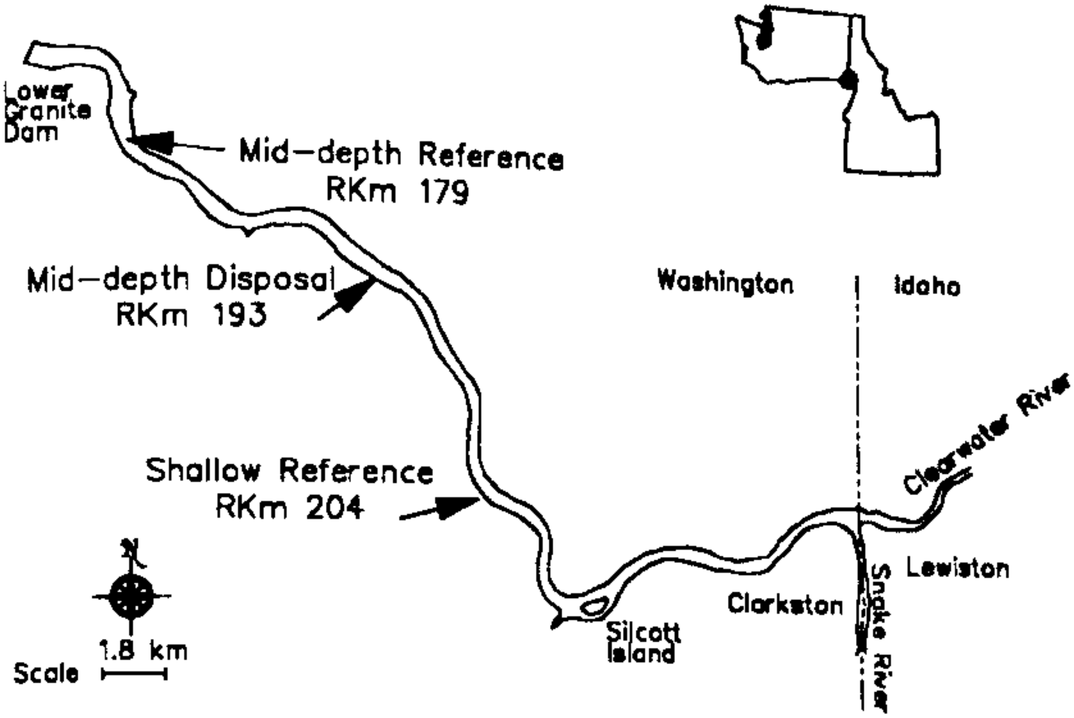

Figure 1. Map of Lower Granite Reservoir, Idaho-Washington showing locations of Mid-depth Reference, Mid-depth Disposal and Shallow Reference sites.

June. Trawling speed varied with wind and water velocity, ranging from 0.6-2.6 m/sec (mean of 1.3 m/sec) for bottom trawling and 0.3-1.1 m/sec (mean of 0.8 m/sec) for surface trawling. Eight multi- and monofilament horizontal gillnets with 3.1, 4.1, and 5.1 bar mesh were fished on the bottom for five to six days at each site. Nets were set three hours before sunset, checked at two hour intervals to minimize mortalities, and fished until three hours after sunset.

Station and year comparisons of fish abundance (CPUE) were compared by a two-way analysis of variance (ANOVA) using a repeated measures design. CPUE was transformed into rank data as described by Conover (1980) to account for possible non-normality. A Fischer's LSD test was used when significant ($P<0.05$) differences were determined.

We sampled benthic assemblages in June 1988 using a Shipek dredge (1072.5 cm^2). A total of twelve dredge samples were systematically collected along four transects (three samples per transect) at each site. Samples were sieved in the field, preserved in 5% formalin, and sorted, identified, and enumerated in the laboratory. Wet weights were taken on groups of preserved organisms, and summed by groups. All biomass data was converted to standing crop (g/m^2).

A two-way ANOVA using a nested repeated measures design with subsampling was used to compare benthic biomass among stations and between years. A Fischer's LSD test was used when significant ($P<0.05$) differences were determined.

RESULTS

Results of in-water disposal monitoring indicate that measurable impacts were limited to the bottom of the water column (Army Corps of Engineers 1988). No significant increases in turbidity or suspended solids were recorded in surface or mid-depth water samples and did not exceed 5 NTU and 5 mg/l above background, respectively. In bottom samples, a peak increase in suspended solids and turbidity occurred from 10 to 30 minutes following disposal, then gradually decreased to near background. Maximum suspended solids on the bottom were about 250 mg/l above ambient but returned to ambient in about 60 minutes. Maximum turbidity on the bottom was about 70 NTU above ambient but returned to near ambient within 60 minutes. Differences in the magnitude of the "peak" was related to the size of sediment being disposed.

Disposal of dredged materials in Lower Granite Reservoir resulted only in a temporary visual increase in turbidity. The overall increase in turbidity was of short duration and restricted to a plume noticeable downstream about 17 km from the air (Glen Mendel, WA Dept. of Game, personal communication).

We collected a total of 4,100 fish during the 1988 spring season at the three stations in Lower Granite Reservoir (Table 1). Highest total catches were for steelhead, largescale suckers (*Catostomus macrocheilus*), bluegill (*Lepomis macrochirus*) and chinook salmon. Sixteen species were collected at the mid-depth disposal site, compared to 19 at the shallow reference site and 16 at the mid-depth reference site. The six species of interest, chinook salmon, steelhead, white sturgeon, northern squawfish (*Ptychocheilus oregonensis*), smallmouth bass (*Micropterus dolomieui*), and channel catfish (*Ictalurus punctatus*) all were collected at the three sites, except white sturgeon which was not collected at the mid-depth reference site.

Comparison of catch rates (CPUE: catch per hour) among gear types provides insight into habitat use. Catches at the reference sites generally were similar between 1987 and 1988. Catch rates of gillnets at the mid-depth disposal site indicate that the abundance of species of interest were not significantly different prior to (1987) and immediately after (1988) deposition of dredged material (Fig. 2). Catches of steelhead at the disposal site were slightly higher in 1988 than in 1987 prior to the disposal. Catch rates of northern squawfish were significantly higher ($P=0.03$) at the shallow reference site than the mid-depth reference and disposal sites. However, catch rates of northern squawfish at the disposal site were similar ($P=0.30$) in both years. The most noticeable difference between pre- and post-disposal fish catches, possibly related to decreased depth, is

Table 1. Fish captured by gear types during spring 1988 sampling in Lower Granite Reservoir, Washington. BT—bottom trawling, ST—surface trawling, and GN—gillnetting. Bottom trawling and surface trawling are expressed as number of hauls, gill netting effort as number of net hours. Species codes: ATR—white sturgeon, ONE—sockeye salmon, OTS—chinook salmon, PWI—mountain whitefish, SGA—rainbow trout, AAL—chiselmouth, CCA—common carp, MCA—peamouth, POR—northern squawfish, RBA—redside shiner, CCO—bridgelip sucker, INE—brown bullhead, IPU—channel catfish, NGY—tadpole madtom, LGI—pumpkinseed sunfish, LMA—bluegill sunfish, MDO—smallmouth bass, PAN—white crappie, PNI—black crappie, PFL—yellow perch. Site abbreviations: MD—middepth disposal, SR—shallow reference, MR—middepth reference.

Gear type	Site	Total spring effort	Total numbers by species																				
			ATR	ONE	OTS	PWI	SGA	AAL	CCA	MCA	POR	RBA	CCO	CMA	INE	IPU	NGY	LGI	LMA	MDO	PAN	PNI	PFL	TOTALS
BT	MD	15	0	0	11	8	6	0	0	0	0	0	0	18	0	0	0	0	0	6	0	0	1	50
	SR	16	0	0	35	12	0	3	6	2	19	0	4	241	5	0	0	66	105	4	40	0	14	556
	MR	17	0	2	2	5	0	2	6	0	6	0	33	2401	1	1	1	0	9	5	19	1	0	2494
Total		48	0	2	48	25	6	5	12	2	25	0	37	2660	6	1	1	66	114	15	59	1	15	3100
ST	MD	7	0	0	6	0	48	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	54
	SR	5	0	0	14	0	163	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	180
	MR	6	0	0	7	0	79	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	86
Total		18	0	0	27	0	290	0	0	0	0	0	1	0	0	0	0	0	0	0	2	0	0	320
GN	MD	306.2	3	0	10	0	44	1	11	1	20	2	4	59	1	7	0	0	0	9	5	0	5	182
	SR	247.0	3	1	6	0	17	9	32	3	34	4	6	188	9	4	0	8	0	10	16	0	38	388
	MR	310.4	0	0	2	0	52	0	20	0	10	0	0	17	0	2	0	0	0	4	3	0	0	110
Total		863.6	6	1	18	0	113	10	63	4	64	6	10	264	10	13	0	8	0	23	24	0	43	680
Grand total			6	3	93	25	409	15	75	6	89	6	48	2924	16	14	1	74	114	38	85	1	58	4100

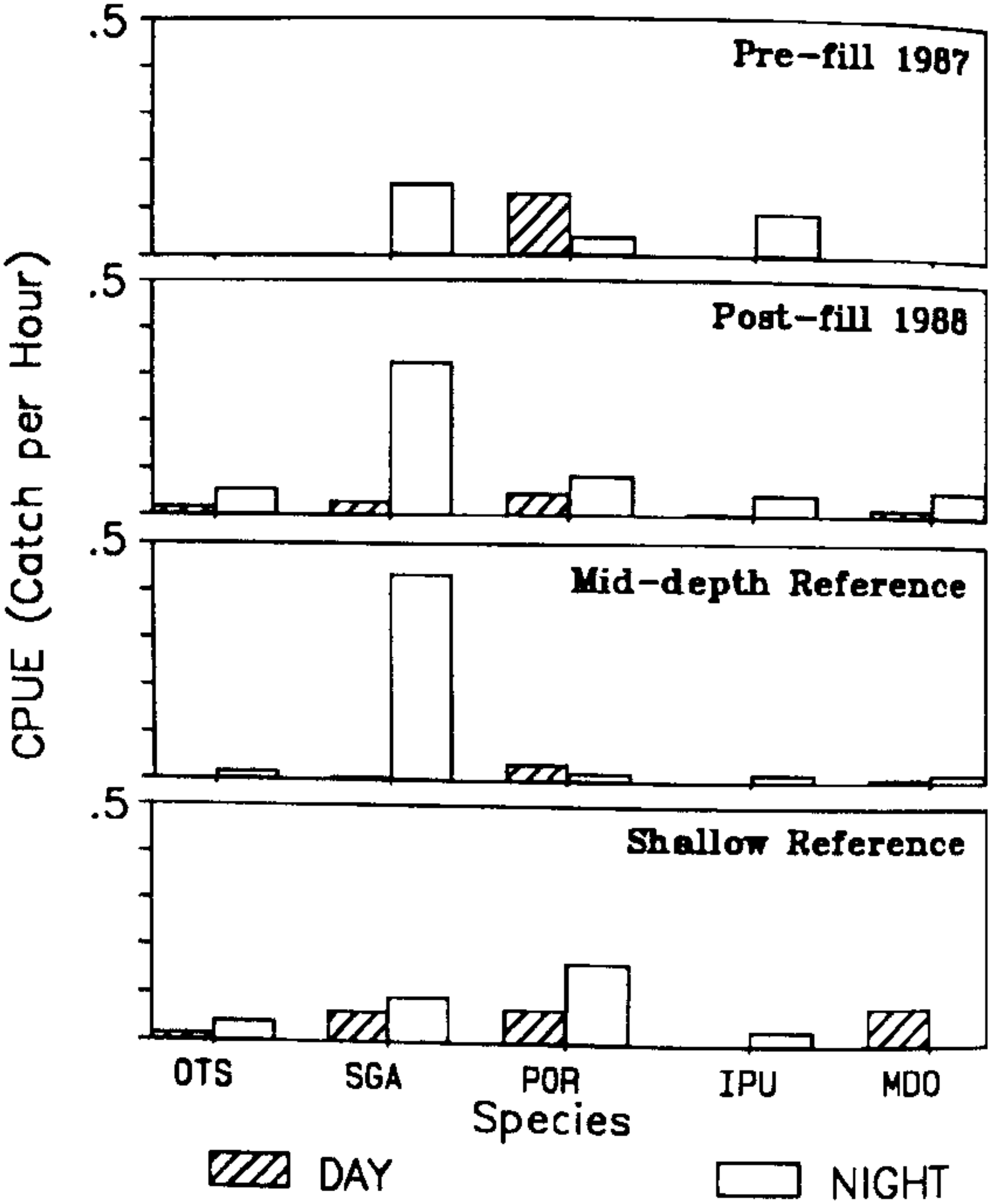

Figure 2. CPUE (Catch per hour) for mono and multifilament gill nets from Lower Granite Reservoir, Idaho-Washington. Species codes are: OTS-chinook salmon; SGA-steelhead; POR-northern squawfish; IPU-channel catfish; and MDO-smallmouth bass.

the appearance of chinook salmon and smallmouth bass in post-disposal catches where none were captured in pre-disposal sampling.

Results from surface and bottom trawling suggest the abundance of chinook salmon and steelhead differs at each of the study sites (Fig. 3). Catch rates from surface trawling were considerably higher for steelhead than chinook salmon although surface trawling was not initiated until June following the peak of the chinook salmon emigration. Highest catch rates were observed at the shallow reference, followed by the mid-depth disposal and then lowest at the mid-depth reference. Overall trends in catch rates from bottom trawling (May-June

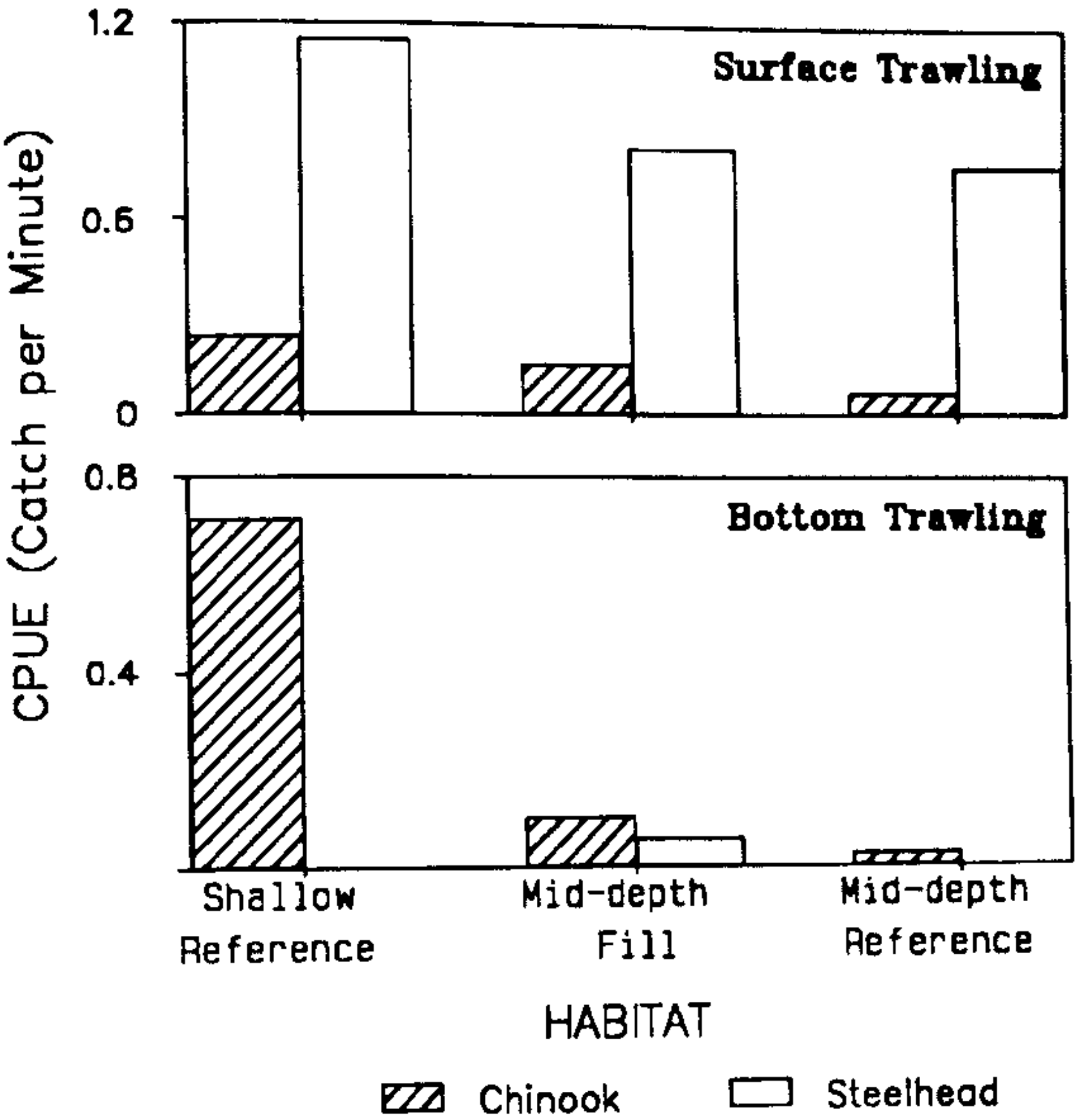

Figure 3. CPUE (catch per minute) for chinook salmon and steelhead by surface and bottom trawling in Lower Granite Reservoir, Idaho-Washington.

sampling) were similar to those from surface trawling, although chinook salmon were more abundant and steelhead were captured only at the mid-depth disposal site. Differences in catches between bottom and surface trawls suggest a pelagic orientation for steelhead compared to a more benthic orientation by chinook salmon.

Benthic assemblages and standing crops were similar among the three sites. Chironomids and oligochaetes dominated the community and accounted for nearly 100% of organisms sampled. Benthic community standing crops ranged from 0.92 g/m^2 (wet weight) at the mid-depth disposal site to 10.4 g/m^2 at the mid-depth reference site (Table 2). Biomass of chironomids four months after deposition of the dredged material was significantly different (P=0.004) among sites. The mid-depth disposal site was significantly lower in chironomid biomass than the shallow reference but similar to the mid-depth reference site. Differences between years within a site were not significant (P=0.93). Biomass of oligochaetes at the mid-depth disposal site was significantly lower than the mid-depth reference although statistically similar to the shallow reference. No differences were observed between years within a site (P=0.63).

Large variances in estimates of benthic standing crops (Table 2) resulted in no significant differences between 1987 and 1988 (P=0.64). Mean standing crops at the shallow water reference site were similar although differences at the mid-depth reference site were about an order of magnitude different between 1987 to 1988 (Table 2). Estimates of benthic standing crops in 1987 at the mid-depth disposal site, prior to disposal were similar to the other sites although variances were dramatically higher.

DISCUSSION

Preliminary results of the use of dredged materials to enhance habitat for salmonid fishes in Lower Granite Reservoir appear promising. While we stress the preliminary nature of our results, they do seem to indicate increased salmonid use of the mid-depth disposal site compared to results from pre-disposal sampling. Salmonids emigrate through Lower Granite Reservoir during the spring (Bennett and Shrier 1986; Bennett et al. 1988), and an estimated 5 million more chinook emigrated in 1988 than in 1987 (unpublished data, Fish Passage Center, Portland, Oregon). Despite the millions of chinook passing through Lower Granite on their out migration, none were collected at the pre-disposal site in 1987. However, only gillnetting was used in 1987. In 1988, however, after site construction, chinook were captured. Whether chinook appearance at the disposal site reflects increased attractiveness of the site, or is merely a reflection of the increased number of the outmigrants is unclear. However, the fact that no chinook salmon were captured prior to disposal strongly suggests lack of use. Interpreting a single season of data requires caution, but initial results suggest a change in salmonid use in response to the disposal of dredged material.

Results of the AEAM workshop indicated that fall chinook salmon were of main interest for habitat improvement although potential existed for

Table 2. Abundance of chironomid and oligochaetes from dredge samples in Lower Granite Reservoir during June 1987 and July 1988. Sample size (n), mean biomass estimates and variances (S^2) are shown based on a per m^2 area. During 1988, abundance estimates were made using a Shipek dredge; in 1987, a Ponar dredge was used. Site abbreviations are: MR—mid-depth reference, MD—mid-depth disposal, SR—shallow reference.

			Chironomid		Oligochaete		Total
Site	Year	n	Biomass	S^2	Biomass	S^2	biomass
MR	1987	6	2.10	3.18	2.64	0.78	4.83
MR	1988	12	2.24	0.43	8.16	24.32	10.40
MD	1987	8	1.63	2.94	4.34	12.13	5.97
MD	1988	12	0.75	0.43	0.17	0.03	0.92
SR	1987	12	3.63	6.40	2.86	3.63	6.49
SR	1988	12	4.75	4.54	3.10	4.40	7.86

other juvenile salmonids for overwintering habitat (Webb et al. 1988). Other anadromous salmonids such as steelhead and spring and summer chinook salmon migrate through the reservoir so quickly that most workshop participants saw little potential for habitat improvement although habitat quality could be improved. In 1985, Bennett and Shrier (1986) reported that some chinook salmon over-wintered in Lower Granite Reservoir. Their presence in Lower Granite coincided with fall releases from upstream hatcheries. In contrast, no chinook salmon were captured during winter sampling in 1987.

We believe that the newly created shallow-water site could be ideal rearing habitat for fall chinook salmon. Bennett et al. (1988) demonstrated that abundance of chinook salmon smaller than 75 mm (fall chinook) in Lower Granite Reservoir habitats was highly correlated with low cover, low gradient shorelines with fine substrate. At this time, we do not know whether the disposal site will be utilized by fall chinook even though the habitat created at the mid-depth disposal site may have similar characteristics to habitat that fall chinook salmon are currently using. Few fall chinook salmon were collected at the disposal site in 1988. However, very cursory sampling in 1988 indicated that fall chinook were not abundant in habitats used in previous years. We believe that low recruitment to the area upstream of Lower Granite Reservoir was probably one reason why no fall chinook were captured at the disposal site during 1988. Several years of additional sampling will be necessary to assess the value of the newly created habitat to fall chinook salmon.

Potential for increased predator use of newly created habitat is another important consideration, especially if salmonid use increases. The two predators of highest concern in Lower Granite Reservoir are northern squawfish and smallmouth bass. Squawfish catches in gillnets during 1988 were higher at the disposal site than at the mid-depth reference site, but lower than at the shallow reference site (Bennett et al. 1988). Squawfish catches in 1987 were similar between the pre-disposal and reference mid-depth sites, both of which were lower than at the shallow reference site. These data suggest a possible increased use of the post-disposal area by squawfish, although the differences are relatively small and may not reflect trends related to habitat.

Preliminary trawling results during the summer have not demonstrated an abundance of young-of-the-year squawfish, suggesting that the disposal site may not be attractive for spawning and/or rearing of these potential predators. Appearance of smallmouth bass after, and their absence before disposal may, however, indicate increased use by this predator. This pattern coincided with that of chinook salmon, although it is premature to suggest that smallmouth bass changed their habitat use in response to chinook salmon distribution, especially since few salmon were consumed by bass sampled in 1987 (Bennett et al. 1988).

Characteristics of the disposal make sampling the mid-depth disposal area difficult. Soundings indicate a highly irregular bottom varying in depth between two to four meters below full pool elevation. This type of surface is very difficult to effectively sample by bottom trawl and, thus, some of our estimates of salmonid abundance at that site represent a minimum level of abundance. We anticipated purse seining the disposal site but were unable to effectively sample it with this gear type because of the shallow depth. Even though sampling is difficult and some of our estimates of use may be minimal, the resulting habitat actually may be more suitable for fishes than a broad, flat under-water plateau which is considered easier to create.

Recolonization by benthos appeared to occur four months after deposition of the dredged material. Chironomids are an important food item to both chinook salmon and steelhead trout in Lower Granite Reservoir (Bennett and Shrier 1986). Oligochaetes although not found to be consumed directly by fishes in Lower Granite are believed to be significant in the trophic dynamics of the system. As of the first sampling, mean chironomid and oligochaete standing crops were lower than those at reference sites. Although highly variable and not statistically different, a substantially lower oligochaete mean biomass at the disposal site than the reference sites suggests a slow rate of colonization of oligochaetes (Table 2). In comparison, the slight difference among chironomid mean biomass suggests more rapid colonization than oligochaetes. Further increases in benthic standing crop will probably increase the attractiveness of the site to fish as a feeding station.

At present, catches of target fish at the disposal site appear to be intermediate between the most productive sites in shallow water and the mid-depth reference site. Sampling four months after disposal indicated that benthic colonization has started, although standing crops were considerably lower than reference sites. Further seasonal sampling is planned to enhance understanding of the use of newly created habitat and the potential for beneficial use.

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Summary and Conclusions from Workshop and Working Group Discussions

INTRODUCTION

While numerous reports, papers and theses have been written on the issue, synthetic treatments of dredging effects on anadromous fishes in the Pacific Northwest have been typically site specific, rare and comparatively outdated (O'Neal and Sceva 1971; Slotta et al. 1973, 1974; Westley et al. 1973; Dutta and Sookachoff 1975; Smith et al. 1976). More comprehensive examinations are available from other regions (e.g., Sherk 1971; May 1973; Sherk et al. 1974; O'Connor and Sherk 1975), but the applicability to Pacific Northwest estuaries and dredging operations is dubious. Finally, few, if any, of these synopses evaluate potential dredging effects in the context of natural estuarine conditions (e.g., suspended sediment concentrations) and the behavior of the fish. This workshop was designed, in part, to synthesize and update dredging impact information for anadromous fishes on the West Coast in the same manner as Manooch (1988) for the East Coast.

The general objective of the working group discussions was to pursue consensus among the invited workshop technical experts as to the actual or potential effects on anadromous fishes of dredging activities in Pacific Northwest estuaries. When we removed the issue of the effects of sediment-associated contaminants from consideration in this workshop, the evidence for impacts of suspended sediments on anadromous fish became less obvious. To better understand these impacts, we segregated our discussions into (1) *near-field effects* and *far-field effects* and (2) *ecosystem-level effects*, which could be considered as direct and indirect effects, respectively. A final goal was to consider recommendations for future research that would address substantiated, but unresolved, effects of dredging.

Near-field effects were considered to be those associated with immediate injury to fish brought into contact with suspended sediment plumes, and the associated water mass, created during a dredging operation. *Far-field effects*, which we still categorized as direct effects on the fish, were considered those that induced modifications in fish behavior (e.g., migration rate, feeding, predator avoidance) that could result in reduced fitness of the fish over the long term. In contrast to the direct effects on fish survival and behavior, *ecosystem effects* were considered those indirect effects on the estuarine ecosystem's ability to provide the basic functions of habitat for reproduction and refuge from predation, (secondary) production of prey resources and (primary) production of the organic matter supporting the estuarine food web upon which fish production ultimately depends.

NEAR-FIELD EFFECTS

At least within the sphere of experience of these workshop technical experts, few instances of documentable near-field dredging effects on anadromous fishes in this region are not associated with contaminated dredge material. Significant depletion of dissolved oxygen is usually not a factor because high sediment biological oxygen demand (BOD) is not common; however, several instances were cited. Correspondingly, everyone agreed that the principal mechanisms of potential near-field injury were through histopathological effects (e.g., hypertrophy and necrosis) on the fishes' gills when they are present in high suspended sediment concentrations. In particular, phagocytosis (intercellular incorporation) of sediment particles in gill tissue was the most recognized histological evidence of effects on the fish. While this is an apparently significant mechanism of direct entry of contaminants adsorbed to sediment particles, it also appears to be a common phenomenon of juvenile salmon migrating in naturally turbid estuaries (e.g., Fraser; see Servizi); moreover, phagocetized particles in the fishes' spleen do not correlate well with the concentrations of suspended sediment (*ibid*). Thus, anadromous fishes migrating in estuaries with a high natural turbidity have probably adapted physiologically to sediment particle impingement on gill tissues. The coughing response (also described by Servizi) may also be a protective response, rather than necessarily a symptom of injury, of fish dealing with an irritant, as indicated by the immediate initiation and cessation of that response when fish move into or out of waters with suspended sediment concentrations >187 mg l^{-1}. In addition, ancillary evidence suggests that repair of affected gill tissue begins immediately (e.g., within 2 d after a 10-d exposure) after removal of the irritant.

Perhaps more relevant issues were judged to be (1) the fishes' option to detect and avoid elevated suspended sediments and (2) the duration of exposure in elevated concentrations. Controlled experiments have yielded *obvious* evidence (e.g., elevated blood plasma glucose) that fish are stressed at *sustained* levels of high (>500 mg l^{-1}) suspended sediment concentration, but what is unknown is the actual extent and duration of exposure in the natural environment. These issues rest in part on two primary factors: the distribution of a plume in the migratory pathway of the fish and the avoidance capability and motivation of the fish to move into and through the plume. We have a moderately precise understanding of the former factor. Most dredging plumes in the Pacific Northwest region (e.g., hydraulic dredging) form a "tear-drop" suspended sediment distribution in the water column, with highest concentrations within 50-100 m of the dredge cutterhead and declining exponentially away from that point. Thus, *the proportion of the migratory pathway in the dredge plume may be the primary determinant of fish exposure to elevated suspended sediments.* Therefore, the most valid assessment of dredging impact would require a "bottleneck" analysis that evaluated the risk of exposure to a dredge plume given a predictable behavior of migrating fish.

However, the natural behavior of fish in estuaries, much less their avoidance of dredging plumes, is poorly known. Probably most of the information relates to their vertical and horizontal position in the water column. In the case of juvenile Pacific salmon, general observations augmented by some systematic sampling would indicate that chum and chinook fry tend to move in shallow waters along the shoreline, juvenile pinks occupy surface waters but may occur further out in channels, and larger fish (e.g., sockeye, coho and chinook salmon, and steelhead trout, smolts) will occur deeper and throughout channels. We could not identify any species-specific migratory behavior to adult salmon in estuaries, and we characterized adult migrations in general to be highly variable or at least stock- and estuary-specific. While no generalizations could be drawn about any of the other anadromous fishes under consideration, a working group consensus was that juvenile migration is more vulnerable to disruption than the adult migration. Delayed timing of adults has been shown to impair reproductive success in some stocks (e.g., Stuart River stocks in Fraser River system), but no evidence exists to indicate that turbidity will induce such a delay. Obviously, the evolutionary motivation behind the adult spawning migration is a strong one.

As illustrated by Quinn, there are alternative scenarios of juvenile fish migration through estuaries that have not been explained. The assumed paradigm until recently is that most juvenile salmon move rapidly (e.g., 2 body lengths s^{-1}) and directly through estuaries; only small chum and chinook "fry," and pink fry to a lesser extent, reside in estuaries for very long (e.g., >days). If we can assume minimal effects of the tag *per se*, Quinn's recent experience with ultrasonic-tagged smolts suggests considerable individual variability in migratory behavior: some fish do swim rapidly down-channel irrespective of tide and habitat, but some "park" in backwater embayments, "idle" in the current or even reverse their migration. That salmon have an extraordinary ability to detect and distinguish turbidity and other water quality gradients is obvious; the variant appears to be the behavioral motivation to sustain an ongoing behavior such as active migration. We also know the fish are not necessarily reluctant to enter high turbidity waters. The unknown factors are the cue(s), and threshold(s), that will induce the fish to deviate from the natural behavior. Therefore, given our ignorance of what environmental cues and stimuli *naturally* shape a fish's behavioral response during its downstream migration, we cannot at present construct a descriptive model of a juvenile salmon's response to a dredging plume.

Near-field effects of dredging entrainment of anadromous fishes do not appear to be a consideration, as the occurrence of these fishes was not found to be significant in either McGraw and Armstrong's or Larson and Moehl's studies of pipeline and hopper dredging operations.

FAR-FIELD EFFECTS

The empirical evidence for far-field effects is, if possible, even more ambiguous. As with the histological responses to elevated suspended sediments, behavioral effects that could impact migrating anadromous fishes, such as reduced foraging success and increased vulnerability to predation, are highly dependent upon the duration of exposure. Gregory's laboratory experiments on the effects of turbidity on the prey reaction distance and predator avoidance of juvenile chinook salmon illustrated that behavioral changes were pronounced when turbidity levels were >200 mg l^{-1}. Servizi substantiated potentially increased vulnerability to predation by the surfacing response of juvenile coho salmon at approximately the same threshold. However, the significance of these responses is less obvious where juvenile salmon are naturally adapted to relatively high turbidity levels; where localized dredging plume concentrations are not appreciably elevated in surface, nearshore habitats, the common foraging habitat of juvenile salmon; or where the fish are not residing in turbid waters very long. As with near-field effects, the primary determinant will probably be the spatial and temporal overlap between the distribution of elevated turbidity and fish. Dredging operations where the turbidity plume encompasses a small proportion of the center of the channel, and highest concentrations are at depth, were considered to pose insignificant effects compared to operations that produced highly turbid plumes from bank to bank and for long distances along the channel.

ECOSYSTEM EFFECTS

Potential ecosystem effects were assumed to include loss or change in critical habitat, reduction of primary and secondary production (food web effects), and changes in hydrology and sedimentology. Permanent changes in channel configuration and bathymetry, rather than effects of the dredging activity *per se*, were not considered because these were assessed in the site designation and other stages of the overall project evaluation. Given the inherently dynamic and non-deterministic nature of estuaries, we doubted that with the current state of the science we can ever discriminate such ecosystem-level effects. Excessive sediment accretion of productive habitats likely poses the greatest potential impact, but defining deleterious sedimentation rates is dubious when dealing with estuarine floral and faunal communities that by-and-large are adapted to rapid sedimentation rates. The working group estimated conservatively that maximum sediment accumulation of 1-cm is usually limited to the area within 500 m of the dredge. Two "red flag" situations were identified, however: (1) dredging within active, cross-current, shallow channels, especially with agitation dredging; and (2) dredging within close proximity to hard substrate (e.g., rocky subtidal) communities. In the latter case, we noted that hard substrate communities in estuaries indicate high current regimes, where significant settlement of sediments is unlikely in most cases.

Despite considerable controversy about the role of dredging in altering fish distributions, most of the technical experts believed that the quantitative and reliable documentation of free-swimming organisms required to make these interpretations just does not exist. Even in San Francisco Bay, undoubtedly one of the most impacted and probably the most studied of the West Coast estuaries, the long-term changes in climatic and oceanographic effects, as well as historic changes in the watershed and estuary, completely confound and obscure the effects of more localized suspended sediments introduced by dredging and dredge disposal (Segur). A considerably more thorough understanding of estuarine hydrodynamics, sedimentology, primary production and trophic energy flow is needed before we can distinguish comparatively short-term, episodic perturbations such as dredging.

RECOMMENDATIONS

If there was a "bottom line" to the workshop and working group discussions, it was that, while there is meager evidence that near-field, direct and ecosystem-level impacts from dredging-associated suspended sediments are significant or common, we do not know enough about fish migrating through estuaries to exclude indirect, far-field effects. Those potential near-field effects of most concern are delay of either juvenile or adult migration, inhibition of feeding and increased vulnerability to predation. These predominantly behavioral responses, or constraints upon natural behavioral repertoires, simply cannot be predicted at this time. This provides no reassurance to either fish or habitat managers, who must be conservative in regulating potentially harmful dredging activity in the absence of definitive scientific knowledge, or to the dredge operators, who have no fish behavior criteria useful for modifying dredging techniques or schedules to avoid impact. We can obtain this knowledge in two ways: (1) conducting dedicated *research* that tests hypotheses about the impact of different dredging conditions on different species under different conditions in different estuaries; or (2) developing standardized protocols for systematic *monitoring* of fish responses and environmental conditions resulting from active dredging projects. The research approach has the potential power of rigorous scientific design and the opportunity to conduct controlled experiments; a primary drawback of this approach is its limitation to just a few dredging situations, and the resulting lack of inference. The monitoring approach has the advantage of building inferential application over a wide variety of dredging situations; the primary drawbacks of this approach are the long time-frame required to build an adequate picture of the scope of effects and the potential for inconsistent application of the monitoring protocol. A further potential problem associated with the research approach is the extensive cost, especially if the dredging activity is to be manipulated for experiments, which could cost \$50,000 d^{-1}! Until the funding and institutional impetus emerges to support the necessary research, our only improvement in this meager understanding of dredging effects will require *in situ* monitoring of permitted dredging projects.

Our final discussions focused on other information needs and research products that would breach some of these gaps in our understanding, as well as approaches to evaluating dredging impact in the far-field.

Scientific Studies to Address Information Needs

1. More predictive, three-dimensional understanding of turbidity plumes under representative estuarine conditions: The scenarios used to identify the suspended sediment exposures that anadromous fish are potentially vulnerable to in estuaries being dredged appears to be quite subjective; more rigorous sampling of dredge-induced turbidity plumes needs to be conducted over a variety of estuarine conditions and dredging operations; the resulting data should be incorporated into a predictive model of the distribution and transport of suspended sediment particles under known conditions.

2. Significance of elevated phagocytosis: Although we have identified phagocytosis of sediment particles to be a histological response to suspended sediments, the effect of this response on the fish's physiology and behavior needs clarification over the spectrum of turbidity levels, durations and particle characteristics that encompass natural to worst-case dredging conditions; it would be particularly worthwhile to examine the differences in this response between natural and contaminated particles.

3. Depth and lateral distribution of migrating juvenile salmon: Despite the plethora of studies on juvenile salmon in the region, no definitive model exists on the distribution of the different species, sizes and stocks in the water column and laterally across estuarine channels; a study design would, in addition, have to address the effect of turbidity, salinity, current speed and direction, temperature and other water characteristics on these distributions.

4. Migratory behavior models: Some descriptive models of juvenile salmon migration behavior through estuaries need to be developed from ultrasonic or other real-time tracking of fish movement; the study designs and models need to evaluate variation of fish movement relative to current directions and velocities, salinity, depth and water quality characteristics, and should specifically test the natural responses of fish migration into water masses with elevated turbidities. The working group also recommended that both laboratory avoidance experiments (e.g., what is the suspended sediment concentration threshold that initiates fish avoidance?) and field measurements be coordinated to describe and explain the mechanisms for the observed behavior.

5. Quantitative definition of sediment characteristics: Presently, extremely subjective indices are being used to describe the physical characteristics of suspended sediment particles, if any are used at all; a system needs to be developed (and evaluated) that describes characteristics of natural sediment particles, including such descriptors as mineralogy, angularity, surface charge, etc.; it is somewhat bewildering when considering that this problem prompted a similarly strong recommendation almost two decades ago (Sherk 1971)!

Approaches to Evaluating Far-Field Dredging Impact

1. Effects of sedimentation on peripheral habitats: In conjunction with developing a more predictive, three-dimensional understanding of turbidity plumes under representative estuarine conditions, we need to evaluate the transport and deposition of dredge-suspended sediments in peripheral habitats. Shallow habitats such as littoral flats and marshes would be degraded if sedimentation rates were sufficient to smother primary producers (e.g., benthic diatoms, marsh plants and eelgrass) or consumers (e.g., benthic infauna and epibenthic organisms) that are important prey resources of anadromous fishes. This issue is particularly germane to the question of far-field impacts resulting from dredging in small, shallow channels.

2. Relationship between fishing success and turbidity: The role of turbidity in influencing the catchability of fish in estuaries is also exceedingly controversial; data sets that have been or may be applied to examine this issue are extremely poor because of all the compounding variables that affect the distribution of fish and fishing effort, and the efficiency of the fishing gear; if this question is to be addressed properly, and potentially resolved, controlled fishing under various natural and dredge-induced turbidity levels will have to be tested relative to more comprehensive assessment of the actual number and distribution of fish available to be captured.

3. Evaluation criteria that assesses habitat function: No evaluation technique presently available provides a quantitative, community (e.g., multi-species) and multi-function approach to assessing the value of existing, restored or created estuarine habitats; this is a particular void when addressing the beneficial use concept of creating or enhancing habitats with dredged material.

In summary, the pervasive need is certainly for a more mechanistic understanding of how suspended sediments impact anadromous fishes in Pacific Northwest estuaries. However, a probably equally or more critical gap is the behavior and response of the fish under natural turbidity levels. The inherent activity patterns of anadromous fishes in estuaries of this region have evolved in most cases under conditions of moderate to high turbidity. Thus, a turbidity threshold needs to be defined under which the fish have adaptive options to avoid or minimize their susceptibility to higher risk. Identifying these thresholds is necessary before we can both understand the significance of dredging activities and develop strategies to prevent impacts when dredging activities exceed these thresholds. The most effective approach to gaining this mechanistic understanding will probably be a combination, optimally overlapping in time and space, of dedicated research and systematic monitoring. The only way this can be effectively accomplished is to allow specific dredging effects experiments and monitoring during periods of anadromous fish migrations. Unequivocal evidence of measurable impact that cannot be avoided by altering dredging operations or procedures (e.g., Huston and Huston 1976; Hayes 1986; Havis 1988) will justify continuance of dredging prohibitions during fish migration periods. Alternatively, properly conducted research will considerably advance our meager knowledge of anadromous fish responses to suspended sediment concentrations and other consequences of dredging activities (e.g., noise), and could facilitate development of dredging criteria and monitoring protocols that would enable dredging during anadromous fish migrations under certain circumstances without undue impact to these vital fisheries resources.

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Appendix A: Agenda

THURSDAY, SEPTEMBER 8, 1988

0800-0815	**Welcome** JACK PULLEN, Chief Coastal Ecology Group U.S. Army Engineer Waterways Experiment Station Vicksburg, MS
0815-0830	**Introduction** CHARLES SIMENSTAD, Marine Biologist Wetlands Ecosystem Team, Fisheries Research Institute University of Washington, Seattle, WA
0830-0900	**Contemporary Issues Involving Impacts of Dredging Activities** DOUG CLARKE Coastal Ecology Group U.S. Army Engineer Waterways Experiment Station Vicksburg, MS
0900-0930	**Changes Induced by Dredging** MARK LASALLE Coastal Ecology Group U.S. Army Engineer Waterways Experiment Station Vicksburg, MS
0930-1000	Break
1000-1030	**New Revelations on the Natural Migratory Behavior of Juvenile Salmon** THOMAS QUINN, Assistant Professor School of Fisheries, University of Washington, Seattle, WA
1030-1100	**Potential Physiological Impacts on Migratory Behavior** WALTER PEARSON, Technical Group Leader Battelle Northwest, Marine Research Laboratory, Sequim, WA

1100-1130 **Effects of Chronic Turbidity on Density and Growth of Steelhead Trout and Coho Salmon**
JOHN SIGLER, Manager-Environmental Science Group
Spectrum Sciences and Software, Logan, UT

1130-1200 **Suspended Sediment Effects on Striped Bass Eggs and Larvae**
CHARLES HANSON, Senior Biologist
TENERA Corporation, Berkeley, CA

1200-1300 Lunch

1300-1330 **Some Sublethal Effects of Suspended Sediments on Juvenile Salmon**
JAMES SERVIZI, Program Head-Contaminant Interactions
Department of the Environment, Cultus Lake, B.C., Canada

1330-1400 **Turbidity Influences on Feeding Behavior and the Implications to Predation Pressure**
ROBERT GREGORY, Research Associate
University of British Columbia, Vancouver, B. C., Canada

1400-1430 **The Effects of Increased Turbidity Resulting from the Mt. St. Helens Eruption on Columbia River Estuarine Fishes**
ROBERT EMMETT, Fishery Biologist
NOAA-National Marine Fisheries Service
Hammond Laboratory, Hammond, OR

1430-1500 **Turbidity and Suspended Sediments at the Alcatraz, CA Dump Site**
DOUGLAS SEGAR, Senior Research Associate
Tiburon Center for Environmental Studies,
San Francisco State University, Tiburon, CA

1500-1530 Break

1530-1600	**Inwater Disposal of Dredge Materials in Freshwater: Potential for Enhancement?** DAVID BENNETT, Professor Department of Fish and Wildlife Resources University of Idaho, Moscow, ID
1600-1630	**Is Entrainment of Fishes a Significant Impact of Dredging?** DAVID ARMSTRONG, Associate Professor School of Fisheries, University of Washington Seattle, WA
1630-1700	**Entrainment of Anadromous Fish by Hopper Dredging at the Mouth of the Columbia River, Oregon and Washington** KIM LARSON, Biologist U.S. Army Corps of Engineers, Portland District, Portland, OR
1700-1730	**Discussion and Summation of Presentations** Convenor: CHARLES SIMENSTAD, Marine
Biologist	Wetlands Ecosystem Team, Fisheries Research
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FRIDAY, SEPTEMBER 9, 1988

0800-0815	**Introduction to Mandate**
0815-0945	**Impacts on Larval and Juvenile Life History Stages**
0945-1000	Coffee Break
1000-1100	**Impacts on Adult Populations**
1100-1130	**Needs for Future Research**
1130-1200	**Conclusions and Recommendations**

Appendix B: List of Speakers and Participants

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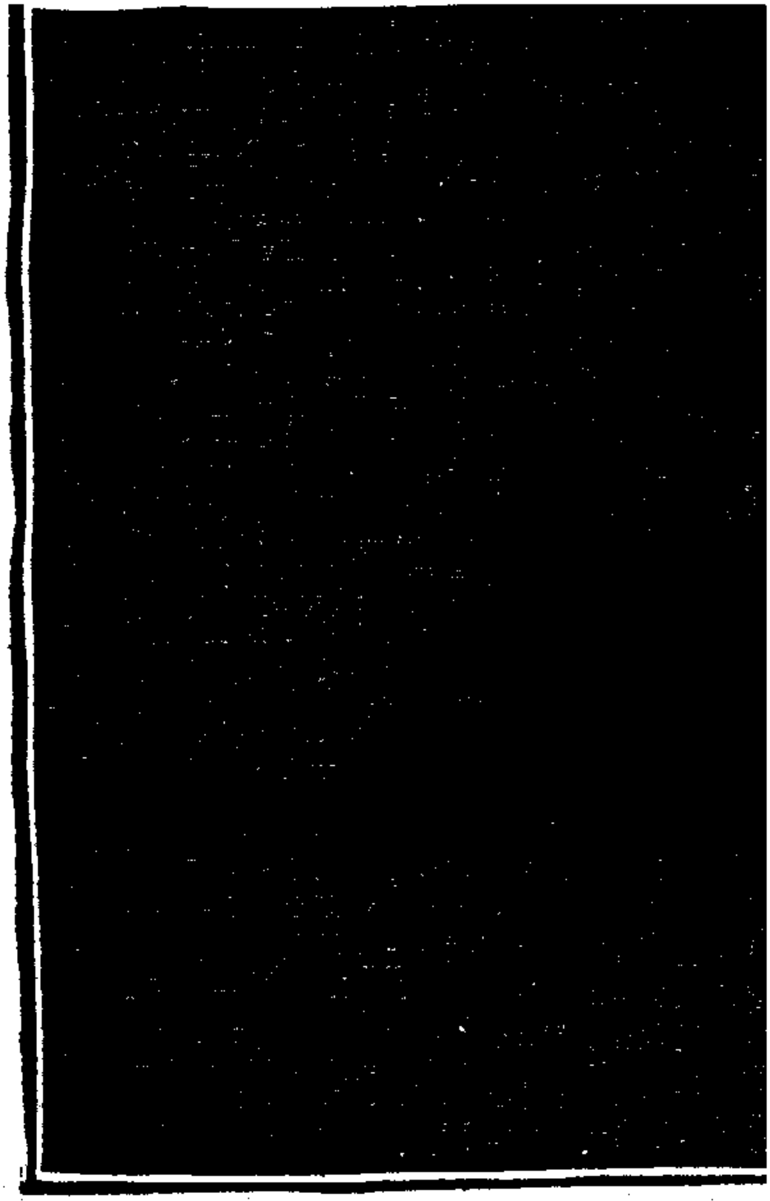

What are the effects of turbidity caused by dredging on the physiology, behavior, and survival of anadromous fishes?

Can anadromous fishes detect and avoid suspended sediment concentrations that may be harmful to them?

Is it possible to use dredged materials for habitat enhancement.?

These and other questions were addressed at a special workshop to assess dredging impacts, to re-evaluate strategie for minimal-impact dredging, and to recommend future approaches to dredging in Pacific Northwest waters.

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