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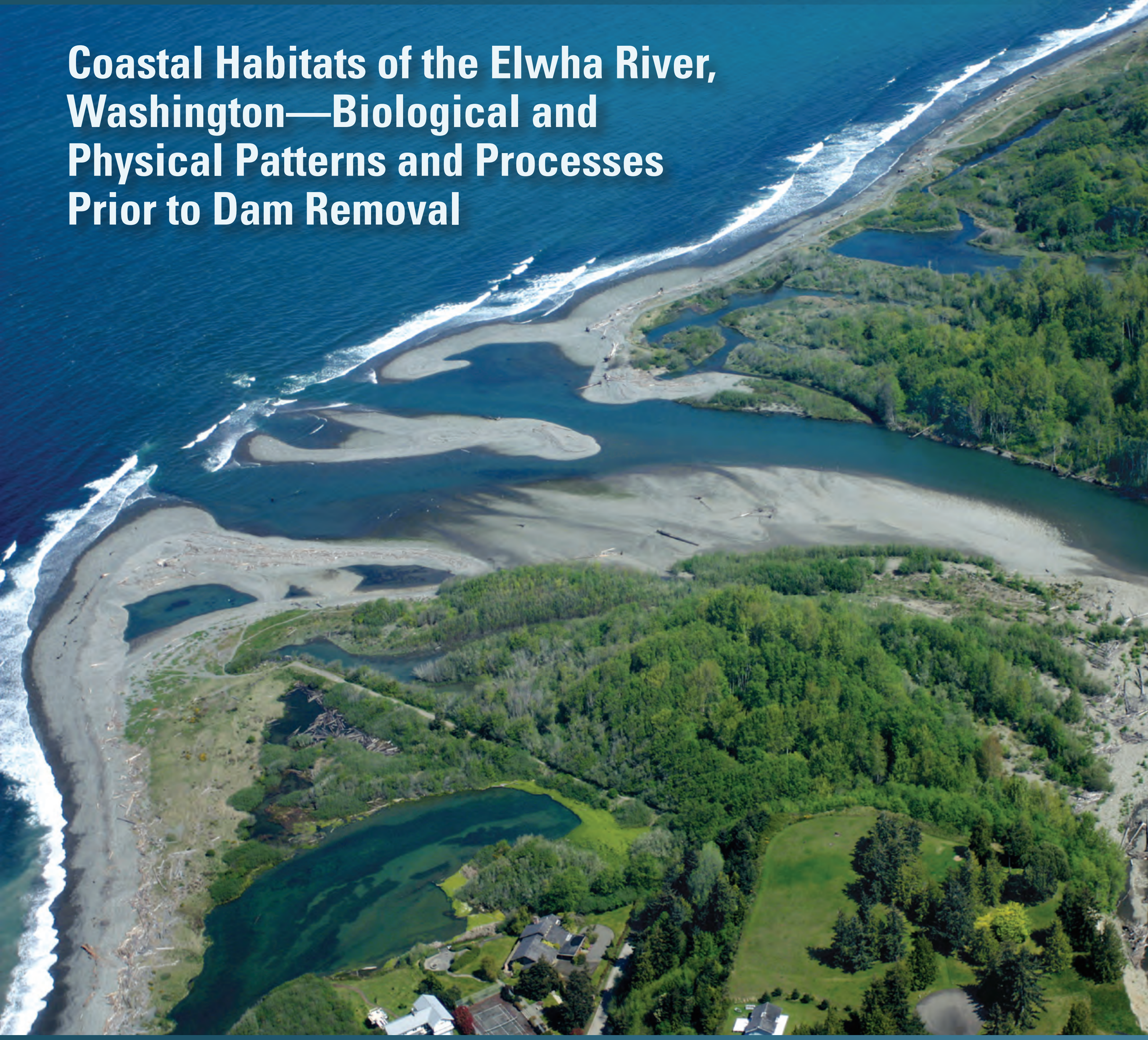
Coastal Habitats of the Elwha River, Washington—Biological and Physical Patterns and Processes Prior to Dam Removal



Scientific Investigations Report 2011–5120

U.S. Department of the Interior
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Front cover: Photograph showing the Elwha River entering the Strait of Juan de Fuca and the estuary of interconnected water bodies east and west of the river. (Photograph taken by John Gussman, Doubleclick Productions, Sequim, Washington, April 22, 2010.)

Inside front cover: Aerial view of Elwha Dam and powerhouse. (Photograph taken by Jet Lowe, National Park Service, Historic American Engineering Record [Library of Congress], 1995.)

Inside back cover: Glines Canyon Dam face. (Photograph taken by Jet Lowe, National Park Service, Historic American Engineering Record [Library of Congress], 1995.)

Back cover (left to right):

Row 1: Sea urchin, clam/starfish (photographs taken by Nancy Elder, U.S. Geological Survey); rockfish (photograph taken by Ian M. Miller, University of California, Santa Cruz, Ocean Sciences Department).

Row 2: Diver/anemone, silverspotted sculpin, cushion star (photographs taken by Nancy Elder).

Row 3: Bull kelp/school of fish (photograph taken by Ian M. Miller); octopus (photograph taken by Reginald R. Reisenbichler, U.S. Geological Survey); mossy chiton (photograph taken by Nancy Elder).

Row 4: Hermit crab (photograph taken by Reginald R. Reisenbichler); diver in kelp (photograph taken by Ian M. Miller); fish eating anemone (photograph taken by Nancy Elder).

Coastal Habitats of the Elwha River, Washington—Biological and Physical Patterns and Processes Prior to Dam Removal

Edited by Jeffrey J. Duda, Jonathan A. Warrick, and Christopher S. Magirl

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U.S. Department of the Interior
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U.S. Geological Survey, Reston, Virginia: 2011

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Suggested reference:

Entire publication:
Duda, J.J., Warrick, J.A., and Magirl, C.S., eds., 2011, Coastal habitats of the Elwha River, Washington—Biological and physical patterns and processes prior to dam removal: U.S. Geological Survey Scientific Investigations Report 2011–5120, 264 p.

Example for chapter within the publication:
Warrick, J.A., Draut, A.E., McHenry, M.L., Miller, I.M., Magirl, C.S., Beirne, M.M., Stevens, A.W., and Logan, J.B., 2011, Geomorphology of the Elwha River and its delta, chap. 3 *of* Duda, J.J., Warrick, J.A., and Magirl, C.S., eds., Coastal habitats of the Elwha River, Washington—Biological and physical patterns and processes prior to dam removal: U.S. Geological Survey Scientific Investigations Report 2011-5120, p. 47-74.

Preface

This report includes chapters that summarize the results of multidisciplinary studies to quantify and characterize the current (2011) status and baseline conditions of the lower Elwha River, its estuary, and the adjacent nearshore ecosystems prior to the historic removal of two long-standing dams that have strongly influenced river, estuary, and nearshore conditions. The studies were conducted as part of the U.S. Geological Survey Multi-disciplinary Coastal Habitats in Puget Sound (MD-CHIPS) project. Chapter 1 is the introductory chapter that provides background and a historical context for the Elwha River dam removal and ecosystem restoration project. In chapter 2, the volume and timing of sediment delivery to the estuary and nearshore are discussed, providing an overview of the sediment stored in the two reservoirs and the expected erosion mechanics of the reservoir sediment deposits after removal of the dams. Chapter 3 describes the geological background of the Olympic Peninsula and the geomorphology of the Elwha River and nearshore. Chapter 4 details a series of hydrological data collected by the MD-CHIPS Elwha project. These include groundwater monitoring, surface water-groundwater interactions in the estuary, an estimated surface-water budget to the estuary, and a series of temperature and salinity measurements. Chapter 5 details the work that has been completed in the nearshore, including the measurement of waves, tides, and currents; the development of a numerical hydrodynamic model; and a description of the freshwater plume entering the Strait of Juan de Fuca. Chapter 6 includes a characterization of the nearshore benthic substrate developed using sonar, which formed a habitat template used to design scuba surveys of the benthic biological communities. Chapter 7 describes the ecological studies conducted in the lower river and estuary and includes characterization of juvenile salmon diets and seasonal estuary utilization patterns using otolith analysis to determine habitat specific and hatchery compared with wild patterns in juvenile Chinook salmon, assessment of benthic and terrestrial macroinvertebrate communities, and seasonal patterns of water nutrients. In Chapter 8, the vegetation communities of the eastern estuary are characterized by mapped vegetation cover types and samples collected for vegetation composition and diversity. Chapter 9 summarizes the existing conditions of the study area as detailed in this report and describes some of the possible outcomes of river restoration on the coastal ecosystems of the Elwha River.

Together, these different scientific perspectives form a basis for understanding the Elwha River ecosystem, an environment that has and will undergo substantial change. A century of change began with the start of dam construction in 1910; additional major change will result from dam removal scheduled to begin in September 2011. This report provides a scientific snapshot of the lower Elwha River, its estuary, and adjacent nearshore ecosystems prior to dam removal that can be used to evaluate the responses and dynamics of various system components following dam removal.

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Conversion Factors, Datums, and Abbreviations and Acronyms

Conversion Factors

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)
kilometer (km)	0.6214	mile (mi)
meter (m)	3.281	foot (ft)
millimeter (mm)	0.03937	inch (in.)
	Area	
hectare (ha)	2.471	acre
square kilometer (km^2)	0.3861	square mile (mi^2)
square meter (m^2)	10.76	square foot (ft^2)
	Volume	
cubic meter (m^3)	0.0008107	acre-foot (acre-ft)
	Flow rate	
centimeter per second (cm/s)	0.394	inch per second (in/s)
cubic meter per second (m^3/s)	35.31	cubic foot per second (ft^3/s)
cubic meter per day (m^3/d)	35.31	cubic foot per day (ft^3/d)
cubic meter per year (m^3/yr)	0.000811	acre-foot per year (acre-ft/yr)
meter per second (m/s)	3.281	foot per second (ft/s)
meter per day (m/d)	3.281	foot per day (ft/d)
meter per year (m/yr)	3.281	foot per year (ft/yr)
	Mass	
gram (g)	0.03527	ounce, avoirdupois (oz)
kilogram per year (kg/yr)	2.205	pound, avoirdupois (lb/yr)
	Density	
kilogram per cubic meter (kg/m^3)	0.06242	pound per cubic foot (lb/ft^3)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$°F=(1.8\times°C)+32.$$

Concentrations of chemical constituents in water are given in micrograms per liter (µg/L).

Datums

Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88) or the Mean Lower Low Water (MLLW) datum of the NOAA tidal station number 9444090 at Port Angeles, Washington during the 1983–2001 epoch. The MLLW datum lies 0.129 meters below the NAVD 88 datum for this tidal station.

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Elevation, as used in this report, refers to distance above the vertical datum.

Abbreviations and Acronyms

ADCP	acoustic Doppler current profiler
AEP	annual exceedance probability
ANOSIM	analysis of similarities
ANOVA	analysis of variance
BA	before and after
BACI	before-after control-impact
BP	before present
CHIPS	Coastal Habitats in Puget Sound
CPUE	catch per unit effort
CRT	Coastal range terrane
CT	conductivity (salinity) and temperature sensor
CTD	conductivity (salinity), temperature, and depth sensor
CWT	coded-wire tag
DTS	distributed temperature sensing
ELJ	engineered log jams
ESU	evolutionarity significant unit
HCl	hydrogen chloride
kcal g^{-1}	kilocalorie per gram
lidar	Light Detection and Ranging
LISST	laser in-situ scattering and transmissometry
LOESS	locally weighted scatterplot smoothing
MD-CHIPS	multi-disciplinary Coastal Habitats in Puget Sound
MHW	mean high water
MHHW	mean higher high water
MLW	mean low water
MLLW	mean lower low water
NAIP	National Agriculture Imagery Program
NH_4+	ammonia
NO_3-	nitrate
NOAA	National Oceanic and Atmospheric Administration
ONP	Olympic National Park
OSC	Olympic Subduction Complex
OTIS	one-dimensional transport with inflow and storage
PSU	practical salinity units
PO_4^{-3}	phosphate
PVC	polyvinylchloride
RFID	radio frequency identifier

Abbreviations and Acronyms—Continued

RhWT	rhodamine WT dye tracer
RKm	river kilometer
RTK-DGPS	real-time kinematic differential global positioning system
RTK-GPS	real-time kinematic global positioning system
SD	standard deviation
SFR	Standard Forage Ratio
TN	total nitrogen
TP	total phosphorus
UPC	uniform point contact
USDA	U.S. Department of Agriculture
USGS	U.S. Geological Survey
WDFW	Washington Department of Fish and Wildlife

Chapter 1

Coastal and Lower Elwha River, Washington, Prior to Dam Removal—History, Status, and Defining Characteristics

By Jeffrey J. Duda, Jonathan A. Warrick, and Christopher S. Magirl

Abstract

Characterizing the physical and biological characteristics of the lower Elwha River, its estuary, and adjacent nearshore habitats prior to dam removal is essential to monitor changes to these areas during and following the historic dam-removal project set to begin in September 2011. Based on the size of the two hydroelectric projects and the amount of sediment that will be released, the Elwha River in Washington State will be home to the largest river restoration through dam removal attempted in the United States. Built in 1912 and 1927, respectively, the Elwha and Glines Canyon Dams have altered key physical and biological characteristics of the Elwha River. Once abundant salmon populations, consisting of all five species of Pacific salmon, are restricted to the lower 7.8 river kilometers downstream of Elwha Dam and are currently in low numbers. Dam removal will reopen access to more than 140 km of mainstem, flood plain, and tributary habitat, most of which is protected within Olympic National Park. The high capture rate of river-borne sediments by the two reservoirs has changed the geomorphology of the riverbed downstream of the dams. Mobilization and downstream transport of these accumulated reservoir sediments during and following dam removal will significantly change downstream river reaches, the estuary complex, and the nearshore environment. To introduce the more detailed studies that follow in this report, we summarize many of the key aspects of the Elwha River ecosystem including a regional and historical context for this unprecedented project.

Introduction

Coastal environments are among the most important ecological components of Puget Sound, a large fjord-estuary in northwestern Washington State. The more than 3,000 km of Puget Sound shoreline is classified into a diverse array of forms according to geologic, oceanographic, and anthropogenic features (Shipman, 2008). Common forms include rocky coasts, beaches, bluffs, embayments, and deltas. Puget Sound and the Georgia Basin are part of the Salish Sea (fig. 1.1), which is fed by rivers in the Cascade Range and Olympic Mountains. These rivers carry melt water from glaciers, snowmelt, and local rainfall. This inland marine water body is connected to the Pacific Ocean by the Strait of Juan de Fuca to the west and the Strait of Georgia to the north. Characterized by a relatively young tectonic- and glacier-influenced geology, spatially variable oceanography, and unevenly distributed levels of urban and anthropogenic impacts, Puget Sound is home to a diverse array of biological communities and charismatic species (Buchanan, 2006; Dethier, 2006; Kriete, 2007; Mumford, 2007; Penttila, 2007; Strickland, 1983). The regions where rivers flowing into Puget Sound meet salt water (sub-estuaries within the greater Puget Sound fjord-estuary) are an especially important habitat type, particularly for salmon, an iconic natural, cultural, and economic symbol for the region (Simenstad and others, 1982).

As an interface between fresh and salt waters, river-mouth estuaries and their surrounding habitats are among the most productive and biologically rich ecosystems on Earth (Goldman and Horne, 1983; Keddy, 2000). The river-mouth estuaries of Puget sound are both dynamic and complex because they are influenced by physical forcings from the Pacific Ocean (such as wave action, currents, and upwelling of nutrients), seasonally and annually varying levels of river inputs (such as sediment, nutrients, and freshwater), and local and large-scale climatological forcings (Moore and others, 2008). The diverse biological communities present in these areas also have complex responses, interactions, and feedbacks with these physical factors. Added to this natural physical and biological variability are human influences that have dramatically altered the amount, quality, and distribution of estuarine habitats throughout the region (Collins and Sheikh, 2005; Todd and others, 2006). Reconstructions of the distribution and size of historical coastal habitats, compared with their current condition, show a dramatic reduction in the amount of river-mouth estuarine and wetland habitat, approaching more than 95 percent loss in some areas. A suite of human drivers, which include industrial and non-point source pollution, resource extraction, recreation, and development affect the remaining estuarine wetlands. Nevertheless, the intact estuarine habitats play an important role in the life history of many species, especially juvenile salmonids, because these young fish reside in river-mouth estuaries during their migration from freshwater rearing areas to the sea (Beamer and others, 2003; Fresh, 2006). Estuarine habitats are believed to offer more protection from predation and higher growth potential compared to alternative nearshore habitats. Exposure of the juvenile salmonids to estuarine salinity gradients also facilitates their physiological transition from freshwater to salt water.

Recently, there has been increased focus on restoring Puget Sound ecosystems and these efforts have highlighted the estuarine ecosystems' capacity to support biodiversity, commerce, and recreational opportunities, despite various impacts and threats to overall system health (Puget Sound Partnership, 2009). The U.S. Geological Survey (USGS) multi-disciplinary Coastal Habitats in Puget Sound (MD-CHIPS) initiative was developed to promote interdisciplinary collaboration among scientists studying Puget Sound ecosystems, with the goal of developing research efforts focused on priorities identified by stakeholders, management agencies, and the public for the restoration and preservation of Puget Sound (Gelfenbaum and others, 2006). To date, MD-CHIPS science has focused on three areas: (1) the effects of urbanization on nearshore ecosystems, (2) restoration of large river deltas, and (3) the recovery of nearshore ecosystems (U.S. Geological Survey, 2006). This report summarizes the research and monitoring activities of MD-CHIPS scientists and partners relating to the Elwha River restoration project.

The Elwha River flows northward from the heart of the Olympic Mountains to the Strait of Juan de Fuca west of Port Angeles, Washington. The Elwha River restoration involves the largest dam removal project to date in the United States and provides the unprecedented opportunity to study the ecological effects of dams and the removal of these dams as an ecosystem restoration technique (Hart and others, 2002; Duda and others, 2008). Although characterized as a restoration project, many of the ecosystem responses and trajectories following dam removal may be novel and precisely predicting ecological outcomes is challenging. Thus, it is imperative that this restoration project is monitored and studied closely, so that ecosystem responses can be characterized and predictive techniques can be advanced. The MD-CHIPS project was developed to characterize the physical and biological characteristics of the Lower Elwha River, its estuary, and nearshore habitats prior to dam removal with the understanding that these characteristics would provide important baseline conditions for comparisons following dam removal. To introduce the more detailed studies that follow in this report, many of the key aspects of the Elwha River ecosystem are summarized here including a regional and historical context for this unprecedented project.

Figure 1.1. The Salish Sea, which includes the Strait of Juan de Fuca, Strait of Georgia, Puget Sound, and the Olympic Peninsula, Washington and British Columbia, Canada. (Used with permission [Freelan, 2009])

Elwha River Dam Removal and River Restoration Project

The Elwha River, on the Olympic Peninsula of Washington State, offers a unique opportunity for restoration because 83 percent of the watershed lies within Olympic National Park (ONP), a World Heritage site (accessed March 12, 2011, at http://whc.unesco.org/en/list) and International Biosphere Reserve (United Nations Educational, Scientific and Cultural Organization, 2010; fig. 1.2). Restoration of the Elwha River was congressionally mandated by The Elwha River Ecosystem and Fisheries Restoration Act (Public Law 102-495) and represents the single largest river-restoration project currently (2011) planned for the greater Puget Sound region. Moreover, the Elwha River restoration project will be the largest dam-decommissioning project in the history of the United States in terms of the projected release of sediment and the size of the existing hydroelectric projects. Decommissioning will involve simultaneous removal of the Elwha Dam (32 m high, constructed from 1910 to 1913 at river kilometer [RKm] 7.9; fig. 1.3) and Glines Canyon Dam (64 m high, completed in 1927 at RKm 21.6; fig. 1.4). These actions offer the unique opportunity to assess the effectiveness of a large dam removal in restoring the watershed and recovering and salmon populations. The upper Glines Canyon Dam and its reservoir Lake Mills are in the National Park, whereas the lower Elwha Dam and its reservoir Lake Aldwell are in an area of mixed ownership and land use (fig. 1.5).

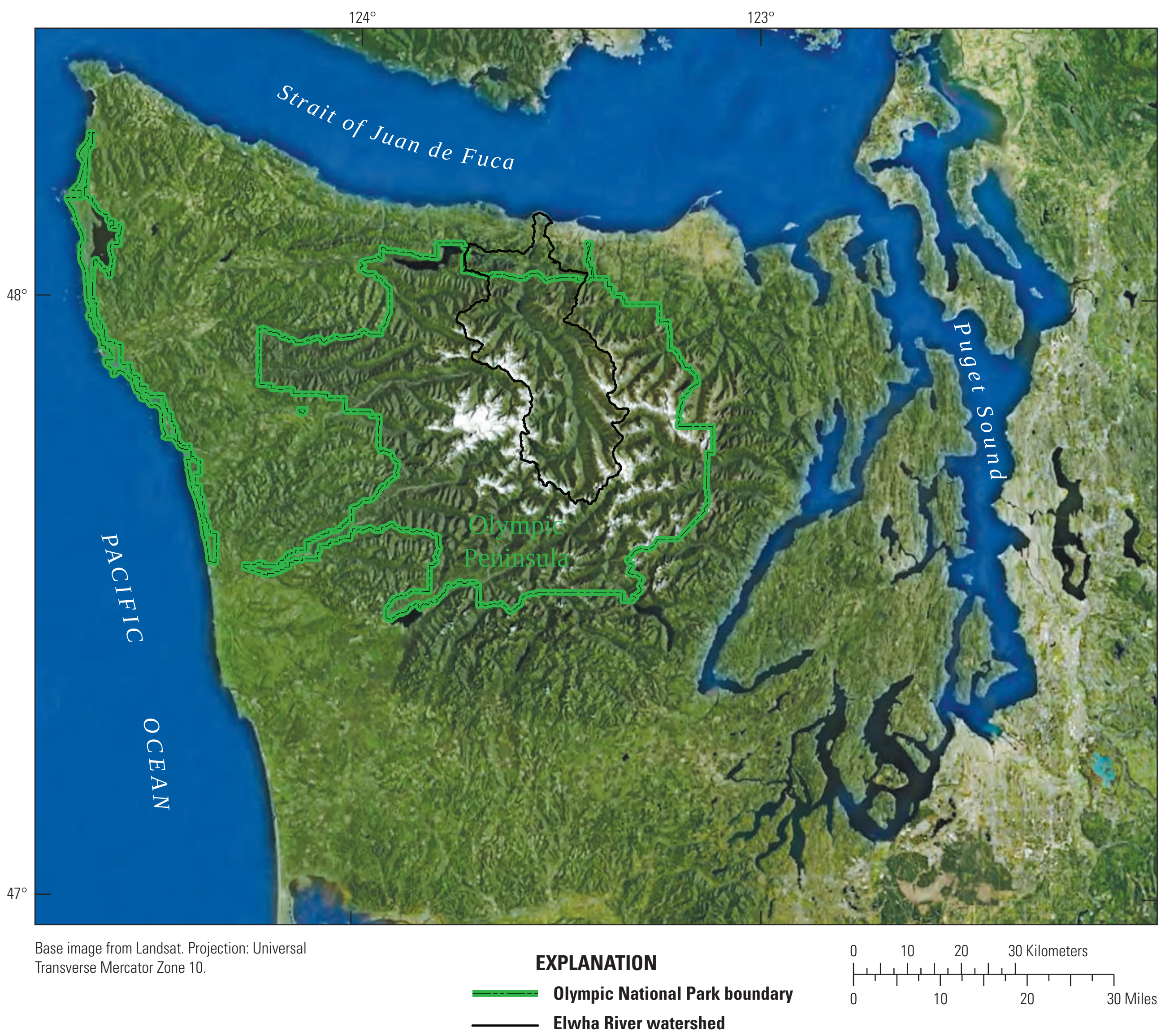


Figure 1.2. Aerial photograph showing view of the Olympic Peninsula and Puget Sound, Washington, showing the Elwha River watershed and the boundaries of Olympic National Park.

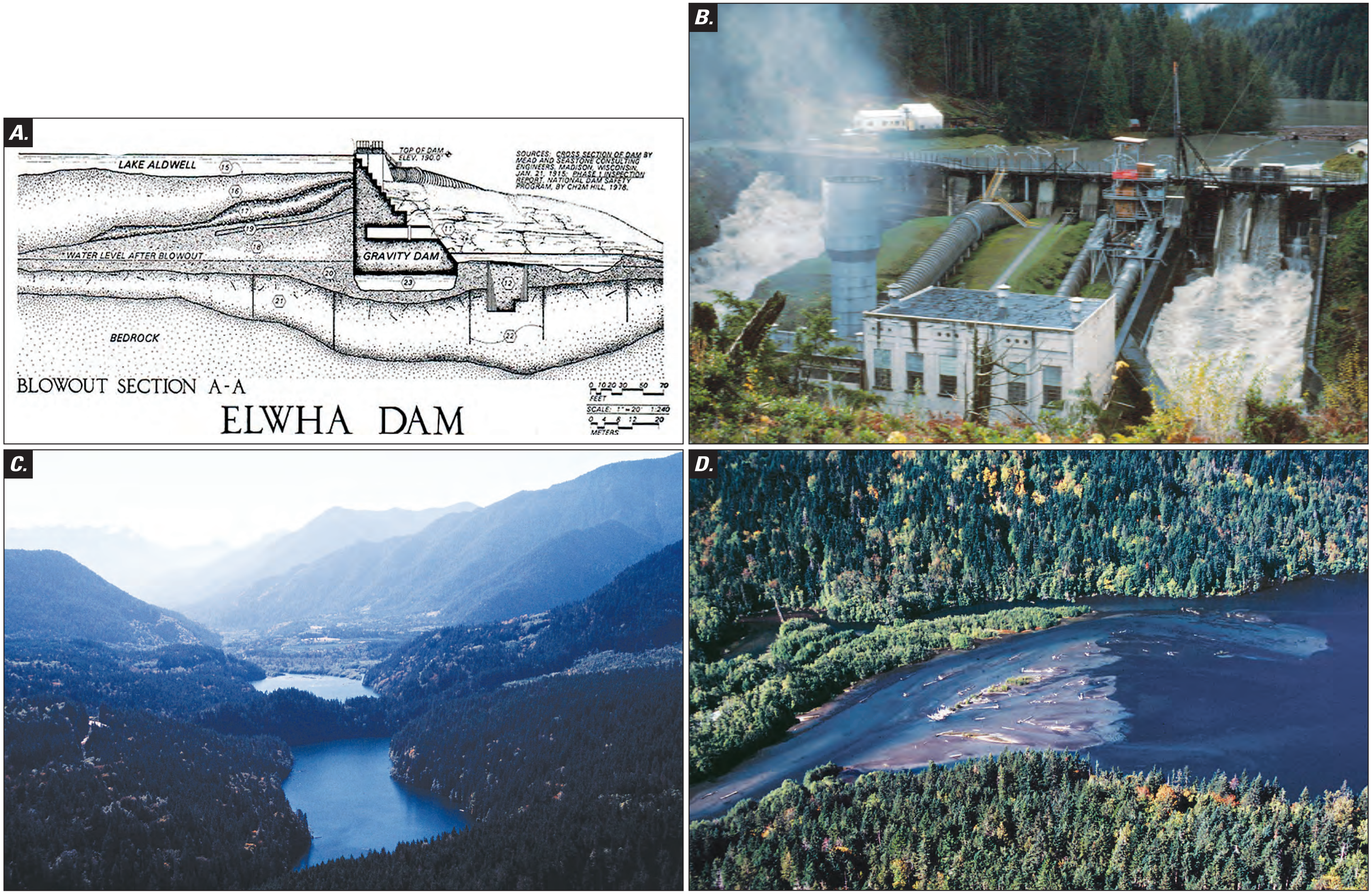


Figure 1.3. Diagram and photographs showing the Elwha Dam, Washington, completed in 1913. (*A*) Elwha Dam is a concrete gravity dam with adjacent buttress-type intake sections flanked by spillways. (*B*) The dam measures 32 meters high and is 137 meters wide at its crest. (*C*) The dam creates the Lake Aldwell reservoir. (*D*) A large delta of sediment deposits has formed at the head of the reservoir.

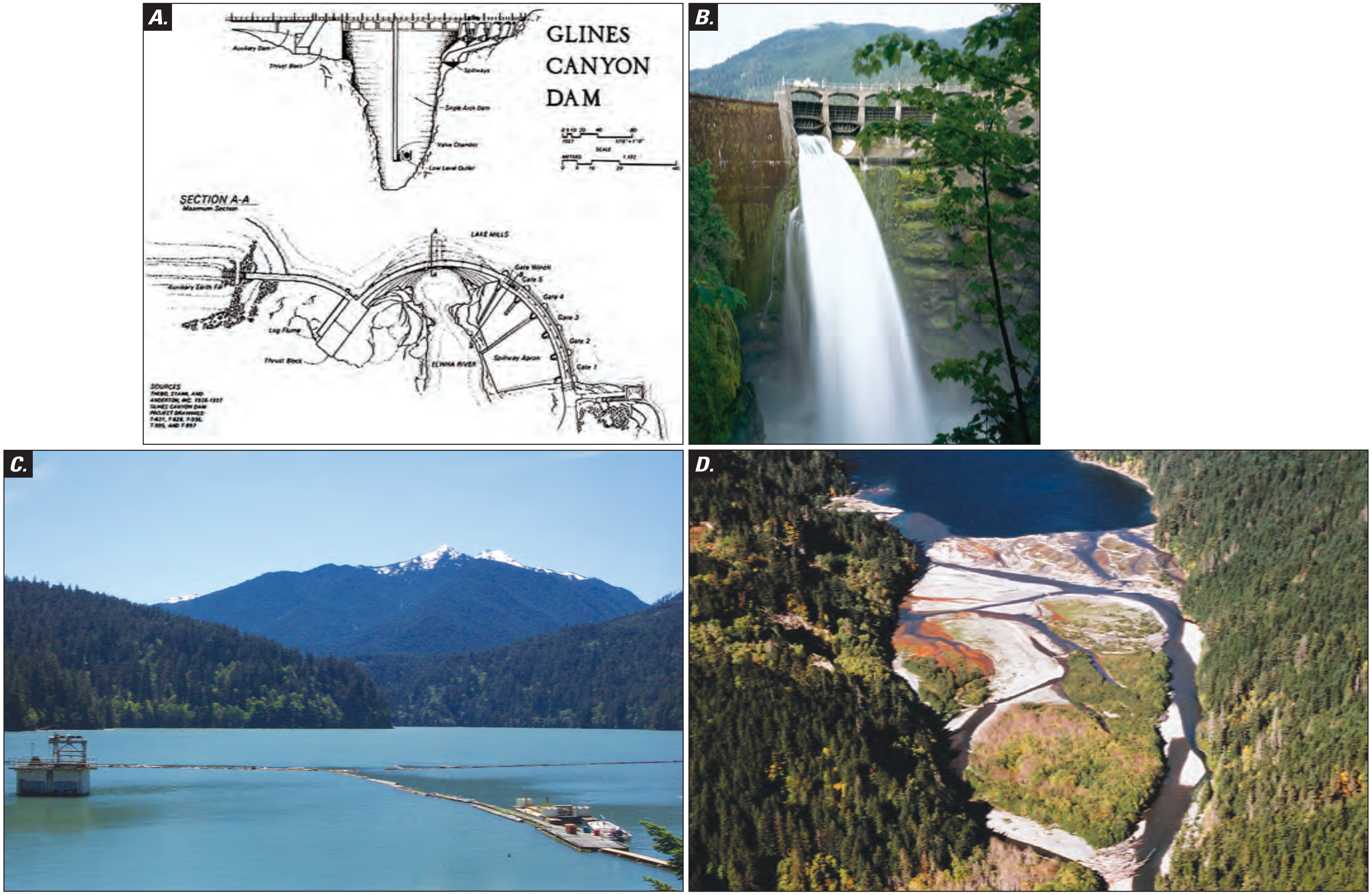


Figure 1.4. Diagram and photographs showing the Glines Canyon Dam, Washington, completed in 1927. (*A*) Glines Canyon Dam is a varied radius, single arch concrete dam. (*B*) The dam measures 64 meters high and width varies from 15.8 meters at its base to 82.3 meters at its crest. The dam is currently (2011) operated as run-of-the-river (equivalent flows entering the reservoir from the Elwha River are released from spillways). (*C*) The dam creates the Lake Mills reservoir. (*D*) A large delta of sediment deposits has formed at the head of the reservoir. (Photograph *B* by Scott Church, private citizen, used with permission, date unknown.)

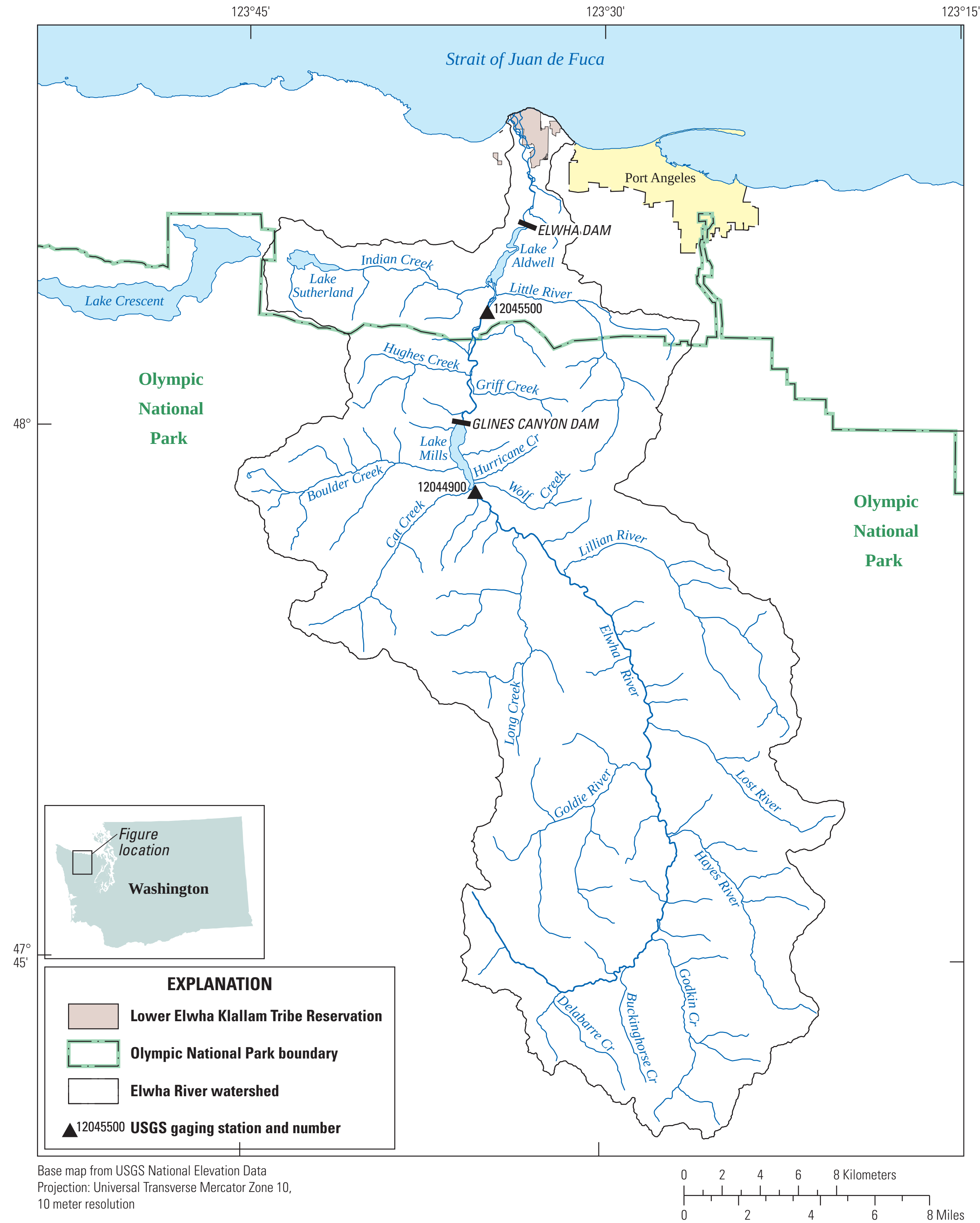


Figure 1.5. Elwha River watershed, northern boundary of Olympic National Park, major tributaries, Lower Elwha Klallam Tribe Reservation, Elwha Dam (Lake Aldwell), Glines Canyon Dam (Lake Mills), and City of Port Angeles, Washington.

Constructed without fish-passage structures, the dams have altered the size and composition of salmon populations in a river that once produced 10 distinct runs, comprised of steelhead trout (*Oncorhynchus mykiss*) and all 5 species of Pacific salmon—Chinook (*O. tshawytscha*), coho (*O. kitsutch*), chum (*O. keta*), pink (*O. gorbuscha*), and sockeye (*O. nerka*). Also affected were sea-run cutthroat trout (*O. clarki*), anadromous bull trout (*Salvelinus confluentus*), eulachon (*Thaleichthys pacificus*), and lamprey (*Lampetra tridentata*), which have life cycles that also require migration to or from freshwater. Salmon populations downstream of Elwha Dam currently are estimated at about 1 percent of their pre-dam numbers (U.S. Department of the Interior, 1995a); the amount of spawning habitat has steadily decreased during the decades following dam construction (Pess and others, 2008). This degradation of spawning habitat has been caused by a massive reduction in the supplies of sand and gravel to the lower river from the sediment trapping effects of the two reservoirs, which in turn has resulted in a coarsening of the riverbed with time. Two fish hatcheries operated by the State of Washington Department of Fish and Wildlife (Chinook salmon) and the Lower Elwha Klallam Tribe (coho, steelhead, and chum) supplement salmon populations. Four fish species (Puget Sound Chinook, steelhead, eulachon, and bull trout) in the Elwha are federally listed as threatened under the U.S. Endangered Species Act.

Construction of the Elwha and Glines Canyon Dams fundamentally changed the Elwha River ecosystem, including its estuarine and nearshore components. Similar changes due to dam construction have occured in other rivers around the world (Baxter, 1977; Petts, 1984). Because of a high sediment trapping capacity (Curran and others, 2009), most of the sediment load transported by the upper Elwha River has accumulated in the two reservoirs. Surveys conducted in 1994–95 determined that approximately 11 million m^3 of sediment had accumulated in Lake Mills (upper dam) and had accumulated behind the lower dam in Lake Aldwell (Randle and others, 1996). The latest estimate (2010) of combined sediment volume in both reservoirs is about 19 million m^3 (Bountry and others, 2010; Czuba and others, 2011, chapter 2, this report). Other changes caused by the dams include increased armoring and channelization in parts of the river downstream of each dam; increased in median particle size (Pohl, 2004; Morley and others, 2008, Draut and others, 2011); increased average age of riparian flood plain forests (Kloehn and others, 2008); and some decreased flood plain complexity and lateral migration of the main channel (Draut and others, 2008; 2011). Geomorphic changes have affected instream benthic invertebrate assemblages and patterns of periphyton standing crop (Morley and others, 2008), as well as salmon populations (Wunderlich and others, 1994; Brenkman and others, 2008; Connolly and Brenkman, 2008; Pess and others, 2008). Another component of functioning flood plain river systems, large woody debris (Abbe and Montgomery, 1996; Latterell and others, 2006), also has been intercepted by the reservoirs (although dam operators regularly pass rafted logs through the dams), further changing flood plain dynamics and fish habitat downstream of each reservoir. However, efforts by the Lower Elwha Klallam Tribe to place engineered log jams in the lower Elwha River has helped rehabilitate some areas (Coe and others, 2006, 2009; see also side bar in Warrick and others, 2011a, chapter 3, this report).

Dam Removal and Release of Sediment Stored in Reservoirs

Mobilization and downstream transport of sediments currently (2011) accumulated in the reservoirs (figs. 1.3*D*, 1.4*D*) during and following dam removal is expected to significantly change downstream river reaches, the estuary complex, and the nearshore environment to the west (Freshwater Bay) and east (Ediz Hook) of the river mouth (fig. 1.6). Studies of the reservoir sediment composition (U.S. Department of the Interior, 1995b; Randle and others, 1996; Childers and others, 2000) indicated that 85 percent and 95 percent (in Lake Mills and Lake Aldwell, respectively) was sand, silt, and clay. A portion of this fine sediment will be readily transported during and immediately after dam removal (Randle and Bountry, 2008). Based on numerical model studies (Randle and others, 1996) and a 1995 reservoir draw down experiment in Lake Mills (Childers and others, 2000), extremely high suspended-sediment concentrations could occur in the Elwha River following dam removal. A simulation model by Konrad (2009) suggests that during the 2- to 3-year deconstruction period, the suspended-sediment concentration in the river could exceed 10,000 mg/L for several weeks each year, with periodically high concentrations for as much as 3–5 years following dam removal, depending upon hydrological conditions (Randle and others, 1996). Anticipation of high-suspended sediment concentrations led to development of unprecedented mitigation measures for the restoration project, including construction of two new water-treatment facilities, the planned suspension of reservoir

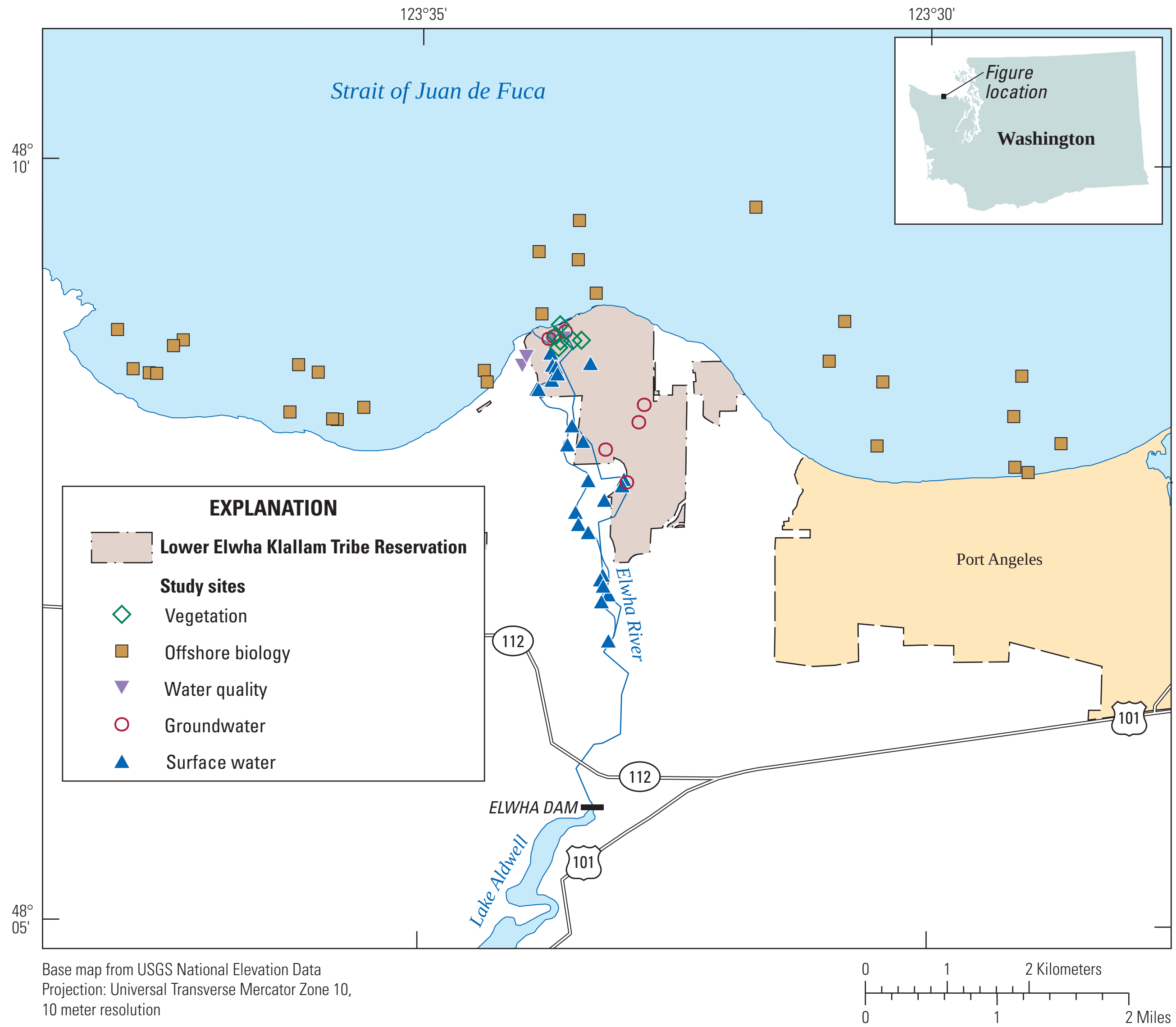


Figure 1.6. Estuary and nearshore study sites, lower Elwha River, Washington. Samples were collected by the U.S. Geological Survey multi-disciplinary Coastal Habitats in Puget Sound team and the Lower Elwha Klallam Tribe.

drawdown when salmon would be affected ("fish windows"), and operation of conservation hatcheries to protect native fish stocks. Over the long-term, the Elwha River bed downstream of the dams is expected to aggrade by as much as 1 m in some areas (U.S. Department of the Interior, 1996; Konrad, 2009), requiring additional mitigation measures such as raising existing flood-protection levees, construction of new levees, and transitioning sewage treatment on the Elwha Klallam Tribal reservation from septic to municipal (City of Port Angeles). Additional details of sediment delivery are provided by Czuba and others (2011), in chapter 2 of this report.

Upon entering the Strait of Juan de Fuca, sediment will be dispersed by waves and tidal currents and deposited on the sediment-starved beaches and seafloor of the Elwha delta (see Warrick and others, 2011, chapter 5, this report). After decades of sediment reduction due to the dams, the nearshore seafloor has coarsened (Warrick and others, 2008) and appears to have developed benthic communities characteristic of coarse sediment and hard bottom substrate (Rubin and others, 2011, chapter 6, this report). Release of massive amounts of fine sediment during and following dam removal will have unknown affects on bottom communities in the nearshore and deep water habitats off the river mouth. The ultimate fate of these sediments depends on complex interactions among waves, currents, and the freshwater plume of the Elwha River (Gelfenbaum and others, 2009; Warrick and Stevens, 2011). Once sediment supply is restored and a long-term equilibrium is reached, the coastal environment will still be affected by other anthropogenic features. For example, coastal bluff armoring that results in a reduced coastal sediment supply (Shaffer and others, 2008) still will be present following dam removal.

The development of key monitoring needs, hypotheses, and the appropriate spatial and temporal scales for data collection and analysis benefited from a series of scientific workshops (Clallam County Marine Resources Committee, 2004; Stolnack and Naiman, 2005; Stolnack and others, 2005; Randle and others, 2006; see also summary in Woodward and others, [2008]). Among the many study designs available for evaluating restoration actions (Roni and others, 2005), the most common study design for the various Elwha River related projects is an intensive "before and after" (BA) approach, although others are using a, "before-after control-impact" (BACI) approach where appropriate control sites exist. The former approach relies on replicating data collection through time from multiple sites over multiple years before dam removal, subsequently returning to these sites for multiple years following removal to measure responses. Strictly speaking, an ideal experimental design would require replication of the treatment (dam removal) in multiple locations. Practically speaking, the Elwha River dam removal is the only such project of its kind in the greater Puget Sound region. Thus, replication is available only within the treatment (termed pseudoreplication by Hurlburt [1984]) and results must be assessed through multiple lines of evidence.

The other approach, a BACI design, incorporates experimental and control treatments (sites). This allows the researcher to account for natural variability in a parameter of interest that is occurring at both sites, which allows a differentiation of these effects from treatment effects. Because there are numerous parameters of interest in the Elwha River restoration project, ranging from terrestrial to aquatic and biological to physical (each of these operating at different spatial scales), there is no single effective "control" site for the Elwha River. For some studies, the Quinault River has been used (Kloehn and others, 2008; Morley and others, 2008) due to its similar watershed area, slope, and discharge (McHenry and Pess, 2008). Others have used the Elwha River upstream of Lake Mills as a reference section (Draut and others, 2008, 2011; Kloehn and others, 2008) for physical processes. Overall, these design constraints necessarily limit the statistical power available to assess the effects of dam removal. However, as Roni and others (2005) point out, valuable knowledge can result from unreplicated BA and BACI designs couched in the framework of a case study, combining multiple lines of inquiry.

Comparison of Regional Estuaries

Estuaries are semi-enclosed water bodies with a connection to the ocean, where freshwater from a river mixes with seawater. In western North America, these estuaries can be highly variable in their size, origin, and functions. Although Puget Sound is the second largest estuary in the United States, most estuaries on the U.S. Pacific coast are relatively small (less than 100 km^2 in area), especially those that drain the steep mountainous terrain in coastal ranges (Emmett and others, 2000). On the Olympic Peninsula, river mouth estuaries and their wetland complexes typically are less than 1–2 km^2 (Todd and others, 2006).

Comparing the river-mouth estuaries of the coastal Olympic Peninsula, the Strait of Juan de Fuca, and the Puget Sound (fig. 1.7) is instructive to highlight the differences in sizes, geomorphologies, and hydrologic conditions. These differences influence the biological conditions among these sites, which can vary considerably. For visual comparisons of these estuaries, we used 2009 aerial imagery from the U.S. Department of Agriculture National Agriculture Imagery Program (NAIP), displayed at 1:24,000 or 1:85,000 scales. The side-by-side comparisons of the river mouth estuaries on the Olympic Peninsula (Hoh, Queets, Quinault, Elwha, and Dungeness Rivers) show that they each cover a relatively small area, with limited off-channel aquatic habitat and virtually non-existent tidal flats (fig. 1.8). The widths of river mouths along the Pacific coast are generally larger than those within the Strait of Juan de Fuca and differences in wave energy are also apparent from the size and extent of breaking waves in the photographs (fig. 1.8; see also Warrick and others, 2011b, chapter 5, this report). The Puget Sound river mouth estuaries, however, are somewhat larger than their coastal counterparts. The Skagit River estuary (fig. 1.9) is the largest of the eight presented in fig. 1.7. In addition to its larger size, a higher diversity of estuarine habitat types is apparent, with complex assemblages of tidal flats, tidal marshes, and tidally influenced distributary channels. Likewise, the Snohomish and Nisqually river mouth estuaries (fig. 1.10) are larger and more diverse than the estuaries of the Olympic Peninsula, although both are smaller than the Skagit.

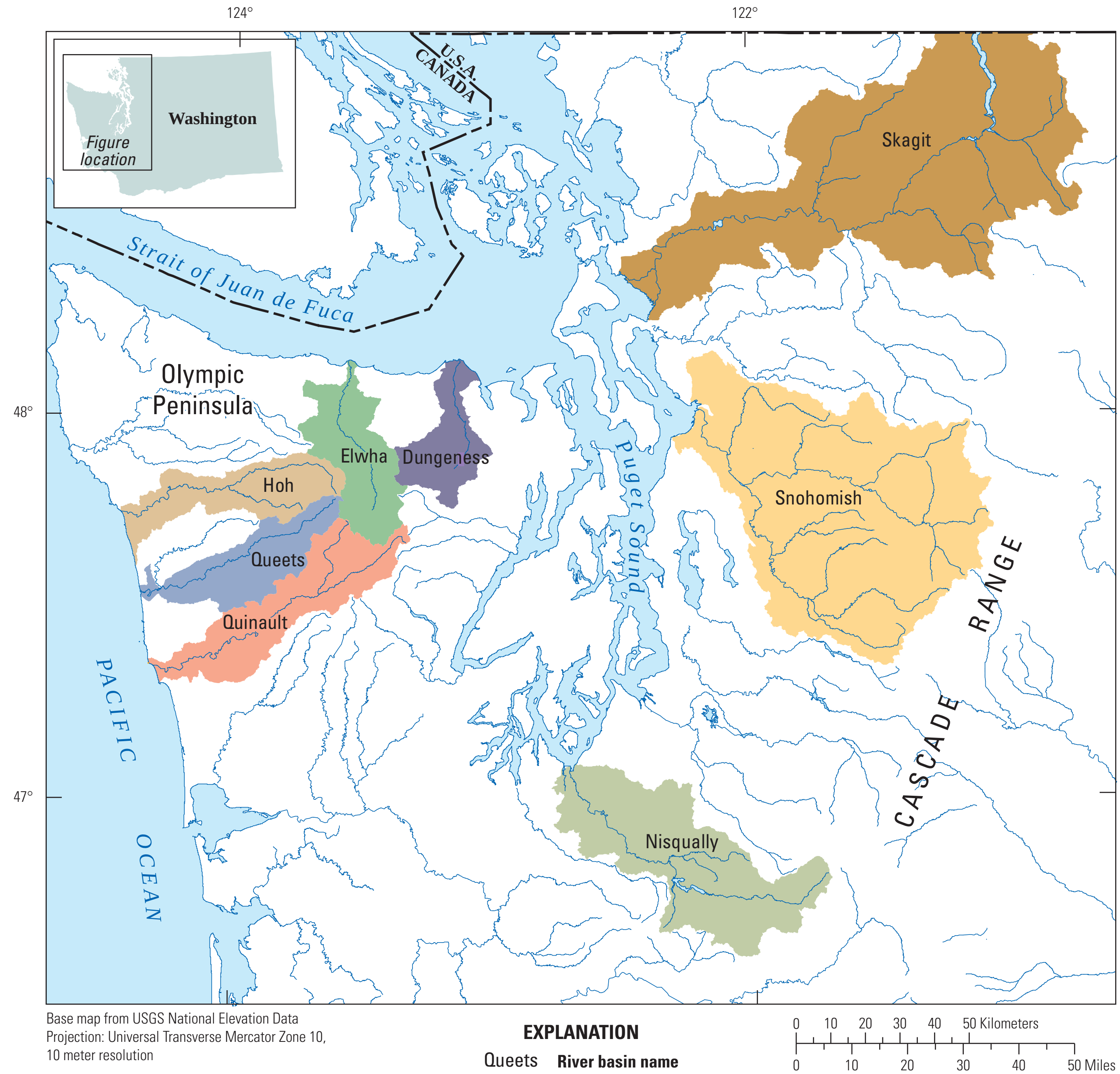


Figure 1.7. Location of eight river watersheds used to compare the size of river mouth estuaries in the Puget Sound and Olympic Peninsula, Washington.

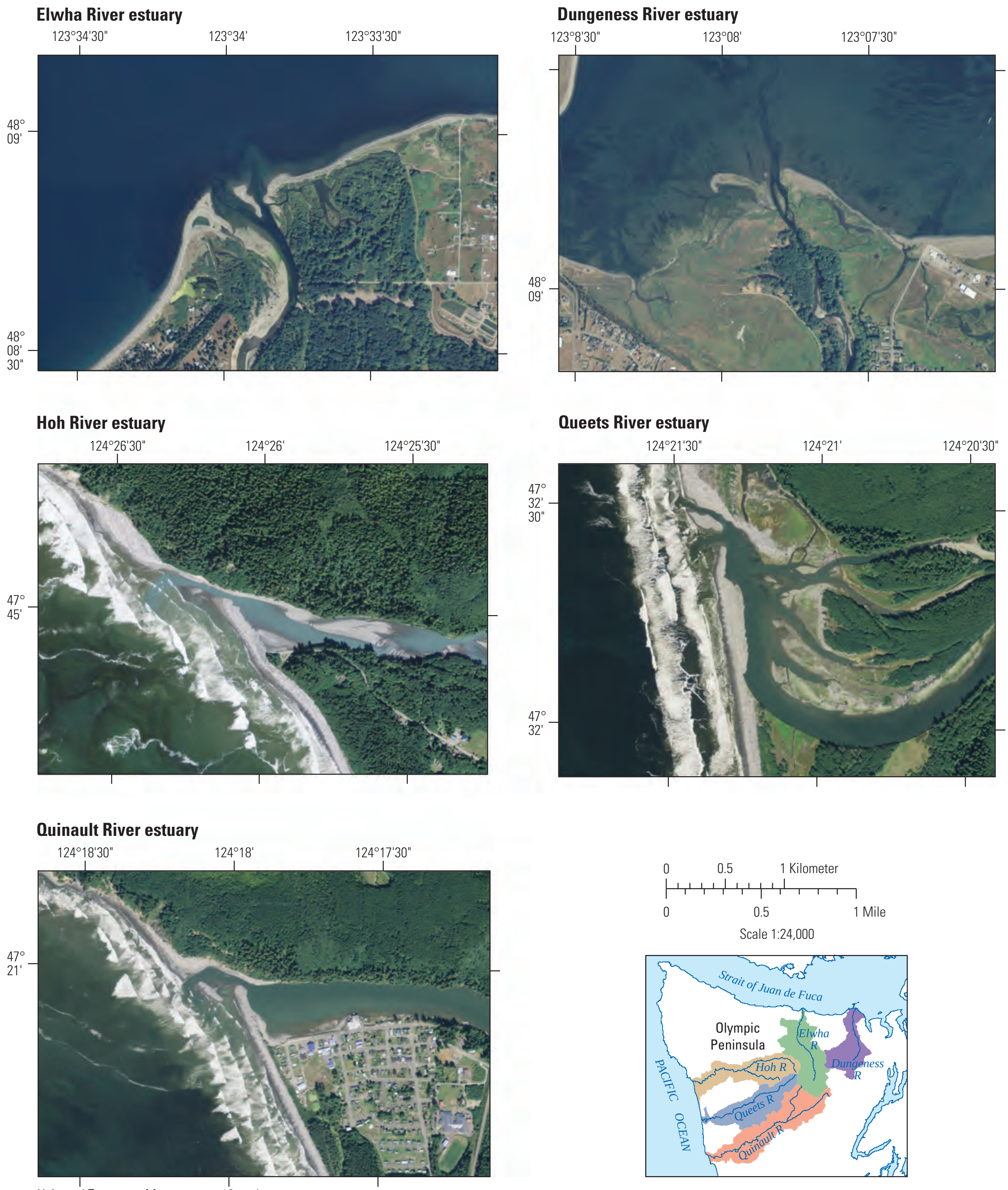


Figure 1.8. Aerial images showing five river mouth estuaries of the Olympic Peninsula, Washington.

Figure 1.9. Aerial images comparing the size of the Skagit River and the Elwha River estuaries, Washington.

Figure 1.10. Aerial images of the Snohomish River and Nisqually River estuaries, Washington.

The Elwha River Estuary: History and Definitions

The mouth of the Elwha River has undergone substantial change over the historical record as sediment sources were curtailed by the dams. Further change resulted from human modification and natural river processes, such as channel meandering and avulsion. The earliest written accounts of the lower Elwha River are from the late 19th and early 20th centuries. Todd and others (2006) suggested that the Elwha River had previously discharged into the Strait of Juan de Fuca through several distributary channels. For example, the General Land Office survey of 1874 noted at least three distributary channels, and an 1891 U.S. Army map and the 1908 U.S. Coast and Geodetic Survey topographic maps showed two channels (Todd and others, 2006).

The number of distributary channels of the Elwha River mouth was reduced to a single channel for most of the 20th century (Todd and others, 2006; Draut and others, 2008). Humans are largely responsible for these changes, due to active modification of the flood plain and channel between the 1940s and 1980s (Draut and others, 2008), as well as from the reduced sediment supply from damming. Human modifications included physical movement of the channel and levee building on both sides of the flood plain. The effects of these modifications can be seen, as an example, in the formation and persistence of Dudley Pond on the west side of the river mouth (fig. 1.11), which was created through the construction of a flood-control levee in 1964 through the middle of the active river mouth (Draut and others, 2008).

A. Western Elwha River estuary

B. Dudley Pond

Figure 1.11. (*A*) West Elwha River estuary and (*B*) the northern shore of Dudley Pond showing bloom of green algae common for this location during summer, near Port Angeles, Washington. ([*A*] Aerial photograph taken by Ian M. Miller, University of California, Santa Cruz, January 14, 2009; [*B*] photograph taken by Jeffrey J. Duda, U.S. Geological Survey October 10, 2007.)

Throughout the 20th century, the Elwha River channel continued to change its position in response to lateral meander migration and episodic avulsion. These natural processes resulted in annualized rates of channel movement of approximately 2–10 m/yr. Some of the highest rates of channel movement were along the lowest 3 km of the river, including the river mouth, an area characterized by low stream gradient and a broad flood plain (Draut and others, 2008; 2011). These historical changes to the lower Elwha River (discussed in detail in Warrick and others, 2011a, chapter 3, this report) are central to the formation and evolution of the river mouth estuary, the coastal wetlands, and the habitats and ecosystem functions they support. For example, most of the coastal lakes along the Elwha River delta, which serve as important rearing habitat for salmon, are former river channels cut off from the current channel by avulsions. The dynamic history of the Elwha River and its flood plain plays an important role in the formation and evolution of the lower river and estuary ecosystem.

The total wetland complex at the mouth of the Elwha River is approximately 0.35 km^2 (table 1.1). The western area of this complex (0.10 km^2) is comprised of two main wetland components (referred to herein as the west estuary). The 0.028 km^2 remnant of the historical estuary—known locally as Dudley Pond (fig. 1.11B)—is contained by the 275 m levee discussed above. The other major feature of the west estuary is a 0.068 km^2 off-channel habitat on the eastern side of the levee that is influenced by river stage and tides. Although groundwater exchange and tidally influenced seepage maintain some degree of connectivity between this habitat and the river channel, the impounded Dudley Pond appears to have limited hydrological connectivity to the Elwha River or the ocean.

Table 1-1. Study areas and descriptions for coastal areas and lower Elwha River, Washington.

Study area name	Description
Elwha River	
Lower Elwha River	The tidally influenced part of the Elwha River near its mouth.
East estuary	
ES1	A part of the eastern estuary complex nearest to the river's mouth. This area was a larger, lentic water body, prior to winter storms in November, 2006 that eroded much of the habitat.
ES2	A large lentic waterbody east of ES1. This is a former river mouth channel that is tidally influenced standing water.
Intraestuarine Channel	A channel that connects ES1 and ES2 that is tidally influenced and flows to the east on a rising tide and to the west on an ebbing tide.
Bosco Creek	A spring-fed perennial water source flowing into the eastern side of ES2. This water source is connected to the Lower Elwha Klallam Tribal Hatchery and receives outflow from rearing ponds.
Western Estuary	
Dudley Pond	A disconnected water body west of a privately constructed dike on the western part of the estuary.
West Estuary Channel (WESC)	A slough (secondary distributary channel) bordered to the west by the dike that is confining Dudley Pond
Nearshore	
Freshwater Bay	The nearshore west of the Elwha River mouth
Ediz Hook	The eastern boundary of the Elwha River nearshore east of the river mouth
West Beach	The beach west of the Elwha River mouth
East Beach	The beach east of the Elwha River mouth

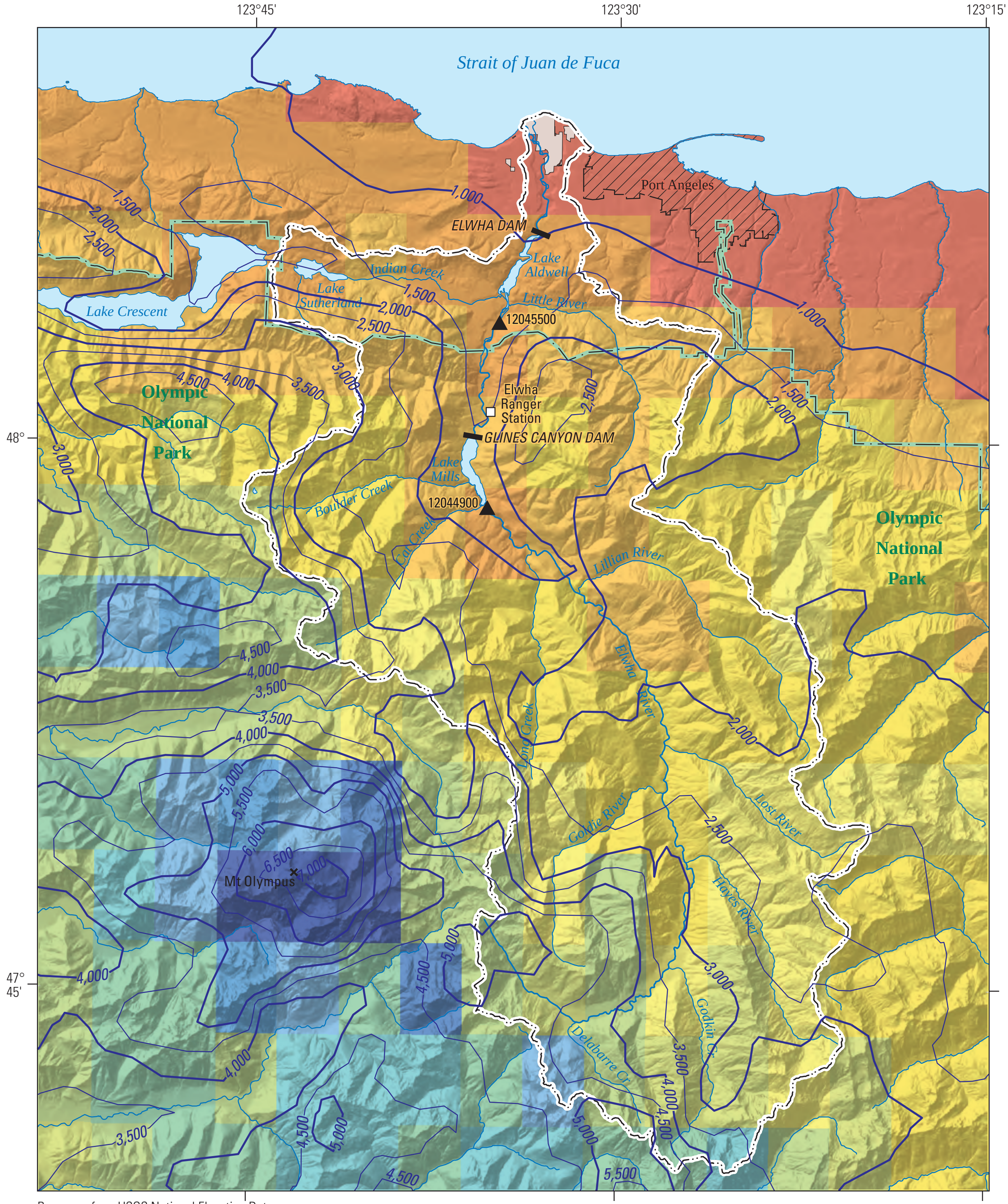
The levee eliminates access to the pond by migrating fish. Resident fish consist largely of a single species, the threespine stickleback (*Gasterosteus aculeatus*) (Shaffer and others, 2009). The east estuary is 0.25 km^2 and is composed of interconnected lagoons and interestuarine channels (fig. 1.12). An important source of freshwater to the east estuary is from Bosco Creek, a spring-fed freshwater source that also serves as the supply and outflow of the Lower Elwha Klallam Tribe's fish hatchery. A mosaic of vegetation south of the beach includes mixed hardwood forests and willow thickets. Vegetation communities of the Elwha River estuary are described by Shafroth and others (2011), chapter 8 of this report.

Figure 1.12. (*A*) East Elwha River estuary and (*B*) beach berm looking south. ([*A*] Aerial photograph taken by Ian M. Miller, University of California, Santa Cruz, March 28, 2008; [*B*] photograph taken by Jeffrey J. Duda, U.S. Geological Survey, October 20, 2007.)

Hydrology of the Elwha River and its Estuary

The climate of the Elwha River basin is characterized by warm, dry summers and cool, wet winters. Most precipitation at upper elevations falls as snow with rain predominating below elevations of about 1,200 m. Because the Elwha River watershed spans the rain shadow created by Mount Olympus and the Bailey Range, the drainage contains the steepest precipitation gradient on the Olympic Peninsula (fig. 1.13). Mean annual precipitation is more than 6,000 mm on Mount Olympus near the headwaters of the Elwha River and about 1,000 mm at the river mouth. Long-term weather records (1948–2005) from the Elwha Ranger Station (approximately RKm 18.2, 110 m elevation) indicate an average annual precipitation of 1,430 mm (Western Regional Climate Center, 2007); most precipitation falls from October through March (fig. 1.14).

Discharge of the Elwha River has been monitored continuously by the USGS since 1918 at McDonald Bridge (streamflow-gaging station 12045500, Elwha River at McDonald Bridge near Port Angeles, WA); the site also was monitored from 1897 to 1901. The gaging site, 7.9 km downstream of Glines Canyon Dam and 13.8 km upstream of the river mouth, has a total drainage area of 697 km^2, 84 percent of the total Elwha River drainage area. Prior to 1975, the discharge at McDonald Bridge was strongly influenced by, "...frequent, rapid, and dramatic stream flow fluctuations for power generation purposes" (U.S. Department of the Interior and others, 1994, p. 8). Between 1975 and 2000, as part of a settlement agreement with the Washington Department of Fisheries, Glines Canyon Dam and Elwha Dam were operated largely as run-of-river by managing discharge at the dams to equal river flow entering the reservoir. Since March 2000, the Bureau of Reclamation, with National Park Service oversight, has operated both projects as run-of-river, except during September and October when lower river flows have been augmented by drafting Lake Mills at the request of the Washington Department of Fish and Wildlife and Lower Elwha Klallam Tribe (Brian Winter, Olympic National Park, written commun., 2010).



Base map from USGS National Elevation Data
Projection: Universal Transverse Mercator Zone 10,
10 meter resolution

0 2 4 6 8 Kilometers
0 2 4 6 8 Miles

EXPLANATION

Lower Elwha Klallam Tribe Reservation

Olympic National Park boundary

Elwha River watershed

12045500 **USGS gaging station and number**

1,000 **Precipitation contour**—Index contour at 1,000 feet, interval is 500 feet

Annual precipitation, in millimeters

- 250 to 1,000
- 1,000 to 1,500
- 1,500 to 2,000
- 2,000 to 2,500
- 2,500 to 3,000
- 3,000 to 3,500
- 3,500 to 4,000
- 4,000 to 4,500
- 4,500 to 5,000
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- 6,000 to 6,500
- 6,500 to 7,000
- 7,000 to 7,500

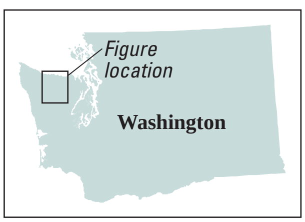
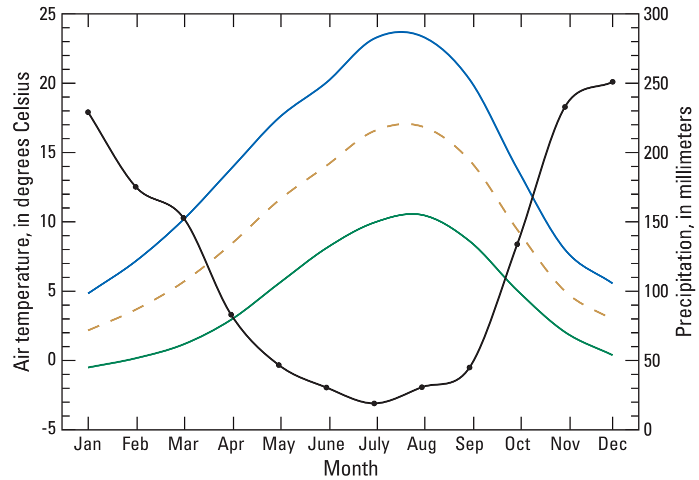


Figure 1.13. Map showing isohyetal contours of annual precipitation across the Elwha River watershed, Washington. Estimated precipitation data from PRISM Climate Group (2010).



EXPLANATION

Air temperature

Precipitation

- Maximum
- Average
- Minimum

Figure 1.14. Graph showing annual trends in air temperature and precipitation in the Elwha River watershed, Washington, June 1948–December 2005. Long-term monthly averages, maximum and minimum air temperatures, and precipitation recorded at the Elwha Ranger Station (RKm 18.2) at an elevation of 110 m. Modified from Duda and others (2008).

The mean annual discharge for the river at the McDonald Bridge gaging station (12045500) is 43 m^3/s (1,520 ft^3/s) based on 95 years of discharge data. Elwha River discharge is bimodal with a peak discharge in the wet winter months due to rainfall and a peak discharge in the late spring and early summer due to snowpack melt (fig. 1.15). The two months with the largest mean monthly discharge are June with 63 m^3/s (2,230 ft^3/s) and December with 60 m^3/s (2,120 ft^3/s). Typically, the smallest flow is in September, with a mean monthly discharge of 11 m^3/s (388 ft^3/s). Extreme low flows often are calculated using a log Pearson Type III approach (Hann, 1977) to determine statistical limits of minimum discharge. For the Elwha River, the 10-year recurrence-interval 7-day low discharge is 7 m^3/s (247 ft^3/s).

The maximum recorded discharge of 1,180 m^3/s (41,600 ft^3/s) occurred November 18, 1897, and seven annual peak discharges in the record were greater than 800 m^3/s (28,300 ft^3/s) (fig. 1.16). The second largest peak of 1,020 m^3/s (35,900 ft^3/s) occurred December 3, 2007, affording researchers the opportunity to observe the Elwha River response to a significant discharge event. All the recorded annual peak discharges have occurred between October and April, and 78 percent of the peak discharges occurred between November and January, consistent with climatology of rivers throughout western Washington (Mass, 2008).

Annual exceedance probabilities (AEP) of peak discharge (also known as flood recurrence intervals) were calculated using the annual peak-discharge record of the Elwha River at McDonald Bridge weighted according to variance (Tim Cohn, U.S. Geological Survey, written commun., 2010) using a regional regression equation for the Elwha River (Sumioka and others, 1998) (table 1.2). The size of the 0.50 AEP event (2-year flood) is 400 m^3/s (14,100 ft^3/s), the size of the 0.04 AEP event (25-year flood) is 948 m^3/s (33,500 ft^3/s), and the size of the 0.01 AEP event (100-year flood) is 1,240 m^3/s (43,700 ft^3/s) (fig. 1.17). These flood-frequency calculations included the peak discharges in 1897 and 2007 as well as the influence of retention capacity of Lake Mills. After the Glines Canyon Dam is removed and the capacity to attenuate the flood hydrograph is lost, peak flows are expected to be slightly larger. Using the watershed regression estimates of Sumioka and others (1998), the 0.01 AEP event (100-year flood) for the Elwha River at McDonald Bridge without the presence of the dam would be about 1,400 m^3/s (49,100 ft^3/s), 10–15 percent greater than peak discharges moderated by the reservoir capacity.

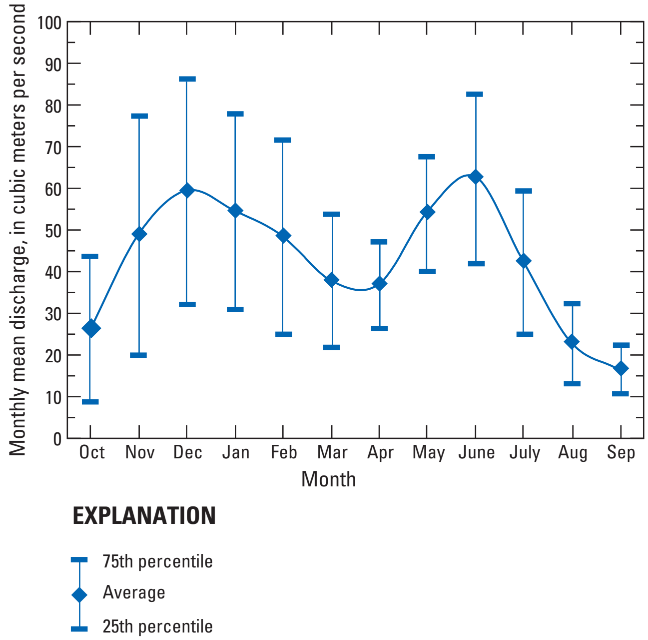


Figure 1.15. The 25th percentile (lower bars), median (50th percentile, diamonds), and 75th percentile (upper bars) of monthly mean discharge as measured at U.S. Geological Survey streamflow-gaging station 12045500, Elwha River at McDonald Bridge near Port Angeles, Washington.

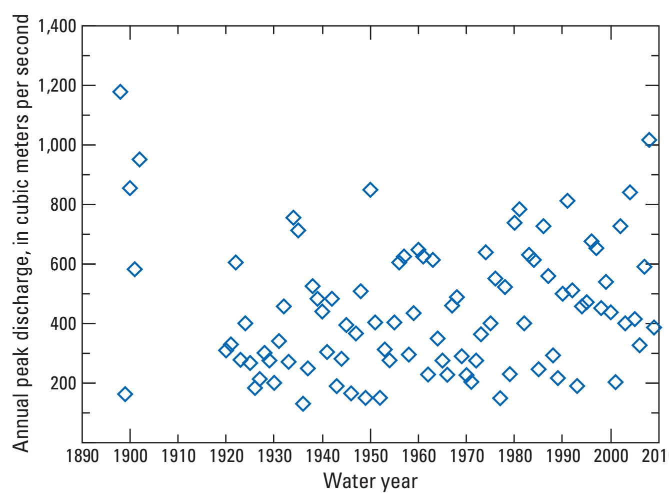


Figure 1.16. Annual peak discharges measured at U.S. Geological Survey streamflow-gaging station 12045500, Elwha River at McDonald Bridge near Port Angeles, Washington.

Table 1.2. Flood-frequency discharge magnitudes calculated for the Elwha River at USGS streamflow-gaging station 12045500, Elwha River at McDonald Bridge near Port Angeles, Washington.

[**Abbreviations:** m^3/s, cubic meter per second; ft^3/s, cubic foot per second]

Annual exceedance probability	Recurrence Interval (years)	Discharge (m^3/s)	Discharge (ft^3/s)
0.5	2	400	14,100
0.1	10	752	26,600
0.04	25	948	33,500
0.02	50	1,090	38,500
0.01	100	1,240	43,700

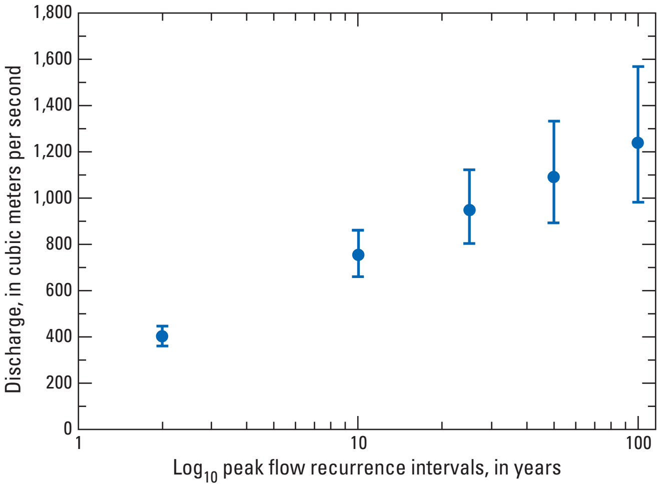


Figure 1.17. Weighted recurrence-interval peak flows calculated for the Elwha River at U.S. Geological Survey streamflow-gaging station 12045500, Elwha River at McDonald Bridge near Port Angeles, Washington, with the upper and lower 95 percent confidence intervals.

Summary

Over the past century, the Elwha River ecosystem has undergone substantial changes due to the presence of the Elwha and Glines Canyon Dams. Dam removal will cause the Elwha River and its coastal environments to change once again as sediments are released from the reservoirs following dam removal and salmon recolonize the watershed. Recent scientific studies conducted in anticipation of dam removal have described the current condition of the river and coastal ecosystems. This report provides a scientific snapshot of the lower Elwha River, its estuary, and adjacent nearshore ecosystems prior to dam removal, serving as a document that can be used to measure and evaluate the responses and dynamics of various ecosystem components following dam removal.

Acknowledgments

Support for the MD-CHIPS Elwha project was provided by the U.S. Geological Survey. Cindi Barton, Susan Benjamin, Leslie Dierauf, Guy Gelfenbaum, John Haines, Frank Shipley, Michael Shulters, and Lyman Thorsteinson provided guidance and support for the MD-CHIPS program. We thank our partners, including the Lower Elwha Klallam Tribe, National Park Service, U.S. Fish and Wildlife Service, Washington Department of Fish and Wildlife, NOAA National Marine Fisheries Service, and Peninsula College for their support and collaboration. Mark Mastin calculated statistical values of flood frequency and low-flow discharge. Special thanks to Theresa Olsen for making the NAIP derived figures and finalizing all GIS-based maps for this Scientific Investigations Report. Our work benefitted from discussions with many colleagues and we would like to thank Guy Gelfenbaum, Eric Grossman, Frank Shipley, Scott Smith, and Tim Randle for their thoughtful discussions.

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Chapter 2

Anticipated Sediment Delivery to the Lower Elwha River During and Following Dam Removal

By Christiana R. Czuba, Timothy J. Randle, Jennifer A. Bountry, Christopher S. Magirl, Jonathan A. Czuba, Christopher A. Curran, and Christopher P. Konrad

Abstract

During and after the planned incremental removal of two large, century-old concrete dams between 2011 and 2014, the sediment-transport regime in the lower Elwha River of western Washington will initially spike above background levels and then return to pre-dam conditions some years after complete dam removal. Measurements indicate the upper reaches of the steep-gradient Elwha River, draining the northeast section of the Olympic Mountains, carries between an estimated 120,000 and 290,000 cubic meters of sediment annually. This large load has deposited an estimated 19 million cubic meters of sediment within the two reservoirs formed by the Elwha and Glines Canyon Dams. It is anticipated that from 7 to 8 million cubic meters of this trapped sediment will mobilize and transport downstream during and after dam decommissioning, restoring the downstream sections of the sediment-starved river and nearshore marine environments. Downstream transport of sediment from the dam sites will have significant effects on channel morphology, water quality, and aquatic habitat during and after dam removal. Sediment concentrations are expected to be between 200 and 1,000 milligrams per liter during and just after dam removal and could rise to as much as 50,000 milligrams per liter during high flows. Downstream sedimentation in the river channel and flood plain will be potentially large, particularly in the lower Elwha River, an alluvial reach with a wide flood plain. Overall aggradation could be as much as one to several meters. Not all reservoir sediment, however, will be released to the river. Some material will remain on hill slopes and flood plains within the drained reservoirs in quantities that will depend on the hydrology, precipitation, and mechanics of the incising channel. Eventually, vegetation will stabilize this remaining reservoir sediment, and the overall sediment load in the restored river will return to pre-dam levels.

Introduction

Sediment transport in the Elwha River has been drastically modified by the construction of Elwha and Glines Canyon Dams (fig. 2.1) nearly a century ago. An estimated 19 million m^3 of sediment has been deposited in the two reservoirs formed behind the dams (Bountry and others, 2010). Bed coarsening and armoring has been identified in the lower Elwha River (Pohl, 2004; Kloehn and others, 2008; Draut and others, 2011) consistent with river response to the reduction in sediment supply downstream of large dams (Williams and Wolman, 1984; Collier and others, 1996; Knighton, 1998; Pizzuto, 2002; Grant and others, 2003). A steady trend of delta erosion at the river mouth is due in part to sediment starvation caused by the upstream dams (Downing, 1983; Schwartz and Johannessen, unpub. data, 1997; Woodward and others, 2008; Warrick and others, 2009). The decreased sediment supply to the lower river has reduced spawning habitat (Pess and others, 2008) and led to the coarsening of the Elwha nearshore zone (Shaffer and others, 2008; Warrick and others, 2008). The planned incremental removal of both dams between 2011 and 2014 will have large short-term effects on the sediment-transport regime in the lower Elwha River, with conditions eventually returning to pre-dam levels some years after complete dam removal.

Although dam decommissioning of smaller structures has become common throughout the United States (Hart and others, 2002; O'Connor and others, 2008), the dam-removal project on the Elwha River will be the largest decommissioning in North America on record, with expected erosion of part of the 19 million m^3 of total reservoir sediment resulting in the largest controlled release of sediment to a river system to date. By comparison, other dam-removal projects in the State of Washington have been small in terms of total volume of trapped sediment and complexity of managing the released sediment (Magirl and others, 2010). The largest dam removal in the United States, in terms of trapped sediment, was the 2008 Milltown Dam removal on the Clark Fork River, Montana, where 5 million m^3 of sediment contaminated from upstream mining activity was trapped behind the dam. However, nearly half of the sediment volume (about 2.2 million m^3) was manually removed and disposed of as part of the dam-removal process (Wilcox, 2010). In the Pacific Northwest, the largest dam-removal project to date has been the Marmot Dam removal on the Sandy River in northwest Oregon in 2007. The total volume of trapped sediment was about 730,000 m^3, consisting of predominantly coarse-grained sediment (sand and gravel), released to the river after a dam breaching and drawdown that lasted just hours (Major and others, 2008). The Marmot Dam removal, because of the volume and characteristics of the trapped sediment and the hydraulics of the river, provides an insightful analogy into the potential geomorphic response of the Elwha River after dam removal. Additionally, the Marmot Dam project has been, to date, the most completely studied and monitored project of its kind in terms of sediment transport.

Sediment eroded from Lake Aldwell and Lake Mills will be transported and deposited downstream of the two dam sites, causing significant effects on channel morphology and ecosystems during and after dam removal. The goals of this chapter are to summarize the current knowledge on the sediment-transport regime of the Elwha River, the nature of sediment currently stored in each reservoir, the expected erosion and downstream transport of the reservoir sediments, and the anticipated effects to the lower Elwha River during and after dam removal.

Base map from USGS National Elevation Data
Projection: Universal Transverse Mercator Zone 10,
10 meter resolution

Figure 2.1. Glines Canyon Dam and Elwha Dam on the Elwha River in northwest Washington. The dams have trapped most of the sediment load from the river during the 20th century that would have otherwise flowed into the Strait of Juan de Fuca. The lower dam, Elwha Dam, forms Lake Aldwell and the upper dam, Glines Canyon Dam, forms Lake Mills.

Sidebar 2.1 Geomorphic Response to Removal of Marmot Dam, Sandy River, Oregon, USA – What Have We Learned?

G.E. Grant, J.J. Major, J.E. O'Connor, C.J. Podolak, M.K. Keith, K.R. Spicer, S. Pittman, H.M. Bragg, J.R. Wallick, D.Q. Tanner, A. Rhode, and P.R. Wilcock

Dam removal has emerged as an important river restoration strategy in the United States. Removals offer an excellent setting for validating analytical models of sediment transport and morphologic change, and testing the capacity of the models to simulate channel evolution in response to changing water and sediment-transport regimes. The largest instantaneous and uncontrolled release of sediment accompanying a dam removal to date (2011) occurred with the breaching of the Marmot Dam coffer dam on the gravel-bed Sandy River in Oregon in October 2007.

Marmot Dam was a concrete diversion dam built in 1913 that stood 14.3 m high by 50 m wide. At the time of removal, the upstream reservoir was completely filled with approximately 750,000 m^3 of sand and gravel. The decision to remove the dam was motivated by a combination of increasing maintenance costs and unfavorable economics. To remove the concrete structure, a temporary coffer dam was constructed out of river bed and reservoir sediment approximately 70 m upstream. The main concrete structure was dynamited in July 2007, but the river was allowed to naturally breach the coffer dam and erode the remaining impounded sediment (730,000 m^3) during the first autumn storms in October 2007, setting off a dramatic sequence of geomorphic events, upstream and downstream.

Sediment transport and morphologic change were monitored in the days, weeks, months, and years following removal through event-based measurements of suspended sediment and bedload, repeat surveys of channel cross-section and planform change, reservoir incision, and repeat lidar surveys. Measurements of sediment transport immediately downstream of the dam following breaching showed a rapid initial increase in fine suspended sediment within minutes, followed by high rates of suspended-sediment load and bedload transport of sand and, later, gravel. The elevated sediment load was derived from eroded reservoir sediment, which mobilized when a meters-tall knickpoint migrated about 200 m upstream in the first hour after breaching. Over the following days and months, the knickpoint migrated upstream more slowly; within 7 months the knickpoint had migrated 2 km upstream. Knickpoint migration, vertical incision, and lateral erosion evacuated about 15 percent of the initial reservoir volume (125,000 m^3) within 60 hours following breaching, and by the end of high flows in May 2008, about 50 percent of the reservoir sediment volume had been evacuated.

About 25 percent of the total volume of sediment eroded from the reservoir was redeposited in a sediment wedge extending about 2 km downstream of the former dam site. The balance of material has been distributed along, and partly fills, pools within the Sandy River gorge, a narrow bedrock canyon extending 2–9 km downstream of the former dam site, but most sediment likely is broadly dispersed along the channel farther downstream. A two-fraction sediment budget indicates that most of the gravel eroded from the reservoir reach was deposited along the channel immediately downstream of the dam site, whereas sand input to and eroded from the reservoir reach largely passed through the gorge and was broadly dispersed farther downstream, chiefly along an 8-km-long channel reach downstream of the river gorge.

Sediment monitoring during dam removal provided insight into the: (1) speed with which unconsolidated reservoir sediment can be eroded even by modest flows; (2) importance of bedrock and longitudinal profile controls on the redistribution of sediment; (3) distinctly different fate of coarse and fine fractions of sediment; (4) importance of the sequence of flows in determining rates of erosion but not final state; and (5) value of coupling upstream and downstream sediment dynamics in models and field campaigns to understand dam removal.

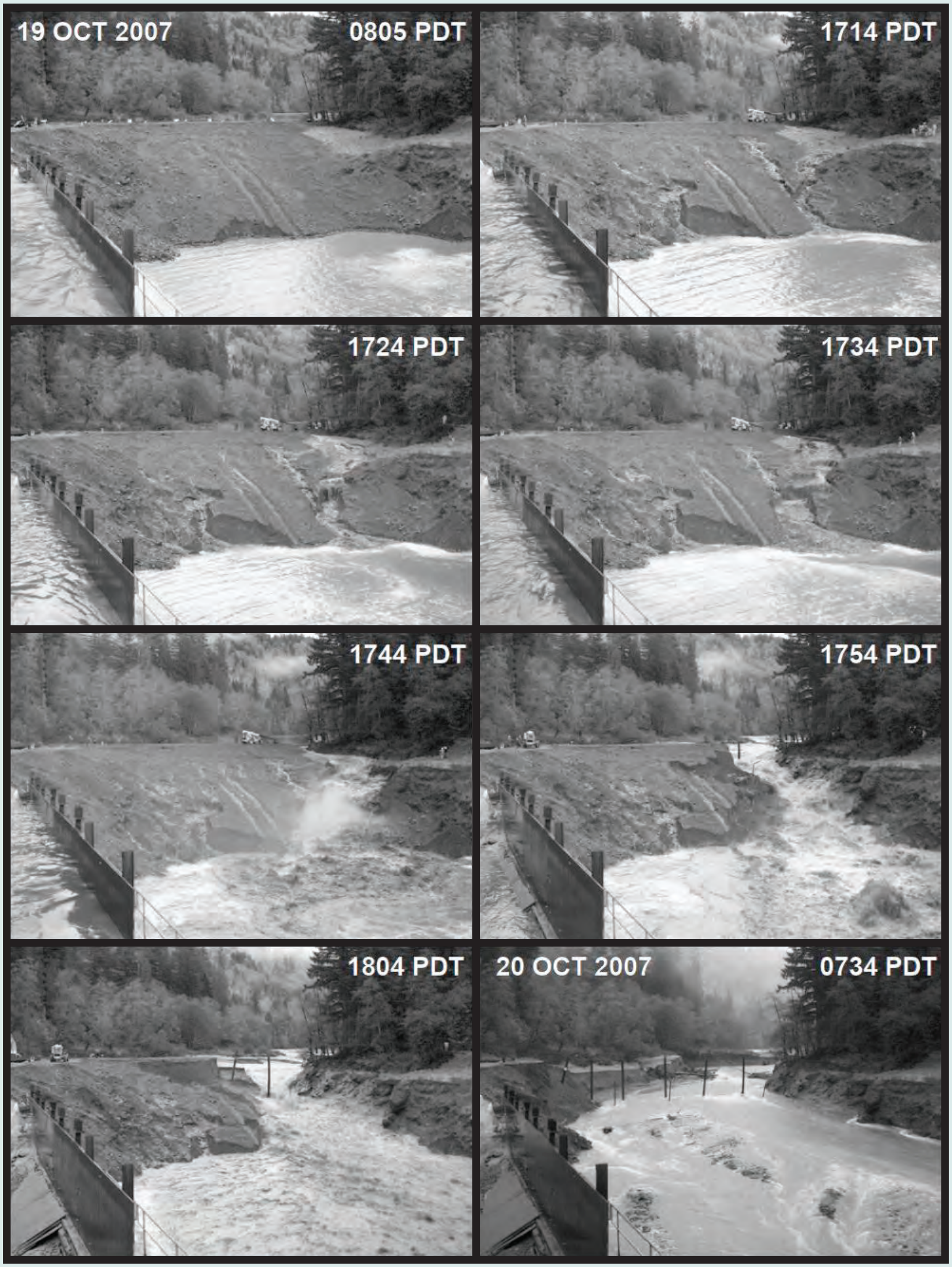

Figure S2.1. Breaching of the temporary earthen coffer dam on October 19–20, 2007, at the site of the manually removed Marmot Dam in northwest Oregon. Note the seepage and mass failures on the downstream face of the dam by 1714 Pacific daylight time (PDT). Incision downward into the coffer dam was nearly complete by 1804 PDT, and nearly all the coffer dam structure had been removed by 0734 PDT the next morning

Sediment Supply

Elwha Dam is a 32-m high concrete gravity dam constructed in 1913 at 7.9 km upstream of the river mouth (fig. 2.1), forming Lake Aldwell with an original storage capacity of 10 million m^3. Glines Canyon Dam is a 64-m high concrete arch dam constructed in 1927 at 21.7 km upstream of the river mouth, forming Lake Mills with an original storage capacity of 50 million m^3. A mapping and sampling effort conducted in 1994–95 on Lake Mills and Lake Aldwell provided much of the current knowledge of sediment deposited in the two reservoirs (Gilbert and Link, 1995), and additional reservoir surveys conducted in July 2010 updated the estimate of total sediment volume accumulated in the two reservoirs combined to 19 million m^3 since construction of the dams **(**Bountry and others, 2010). The upstream sediment supply to Lake Mills was estimated from the volume of accumulated sediment (Gilbert and Link, 1995; Bountry and others, 2010), as well as from field measurements of suspended sediment and bedload transport (Curran and others, 2009).

The Lake Mills delta extends upstream into Rica Canyon, has a total length of about 3,000 m, and is as much as 33-m thick (Bountry and others, 2010). The volume of sediment stored in Lake Mills has been measured by combined topographic and bathymetric surveys resulting in estimates of 10.6 million m^3 of sediment in 1994 (Gilbert and Link, 1995) and 15.6 million m^3 (± 2.7 million m^3) of sediment in July of 2010 (Bountry and others, 2010). In 1994, about 51 percent of this material was stored in the delta at the reservoir entrance, 38 percent was stored on the reservoir floor and sides, and the remaining 11 percent was stored in Rica Canyon and on the alluvial fans of Cat Creek and Boulder Creek (Gilbert and Link, 1995; fig. 2.2).

Sediment in the Lake Mills delta fines in the downstream direction and from upper deposits into deeper, older deposits (Gilbert and Link, 1995). Sediment stored in Lake Mills was composed of 48 percent silt and clay, 37 percent sand, and 15 percent gravel, cobbles, and boulders (Gilbert and Link, 1995; fig. 2.2). The delta consists of gravel, cobbles, and boulders at the upstream end transitioning to sand, gravels, and cobbles in the central area. Post-dam deposits also form a gently sloping wedge of fine-grained sediment on the reservoir floor away from the delta that extends to the dam face (Gilbert and Link, 1995). In the years following extensive study of delta dynamics during the 1990s (Childers and others, 2000), the delta in Lake Mills has prograded about 400 m to the mouth of Boulder Creek, and trees and other vegetation have grown extensively over the delta surface (fig. 2.3).

The Lake Aldwell delta held an estimated 3.0 million m^3 of sediment in 1994 (Gilbert and Link, 1995). Because no record of a pre-dam survey of the Lake Aldwell topography exists, the sediment volume estimate is based on drill holes and surface exploration. Based on additional reservoir surveys in July 2010, the volume of sediment in Lake Aldwell was estimated at 3.0 million m^3 (± 0.8 million m^3) (Bountry and others, 2010), suggesting that there was no measurable increase in reservoir sediment volume between 1994 and 2010. About 46 percent of the sediment in Lake Aldwell is stored in the delta and 54 percent is stored on the reservoir floor and sides; the sediment is composed of 67 percent silt and clay, 28 percent sand, and 5 percent gravel, cobbles, and boulders (fig. 2.4; Gilbert and Link, 1995). The Lake Aldwell delta is about 750-m long and as much as 5–7-m thick and is nested in older river-terrace deposits (Gilbert and Link, 1995). The upstream area of the delta consists primarily of gravel, cobbles, and boulders, and the deltaic deposits transition to predominantly silt and sand at the distal extent. Similar to deposition patterns in Lake Mills, the post-dam, fine-grained sediment forms a gently sloping wedge on the Lake Aldwell reservoir floor that extends to the dam face (Gilbert and Link, 1995). In contrast to the dynamic changes observed on the Lake Mills delta, the Lake Aldwell delta has remained relatively unchanged between 1994 and 2010 (fig. 2.5).

Estimates of the annual sediment load into Lake Mills can be used to approximate the sediment discharge rates in the Elwha River as well as the long-term anticipated sediment load into the Strait of Juan de Fuca after the initial sediment pulse following dam removal subsides. Based on sediment accumulation rates in both reservoirs, the long-term sediment loads were estimated to be 130,000 m^3/yr for coarse sediment and 120,000 m^3/yr for fine sediment, thus totaling 250,000 m^3/yr (Gilbert and Link, 1995; Bountry and others, 2010). These volumetric estimates can be compared to sediment discharge estimates by Curran and others (2009). Using field measurements of suspended sediment (1995–98, 2006–07) and bedload (1994–97) at the Elwha River above Lake Mills (U.S. Geological Survey streamflow-gaging station 12044900), regression equations were developed to estimate both suspended-sediment and bedload discharge as a function of streamflow (Curran and others, 2009). Applying regression equations for suspended-sediment and bedload

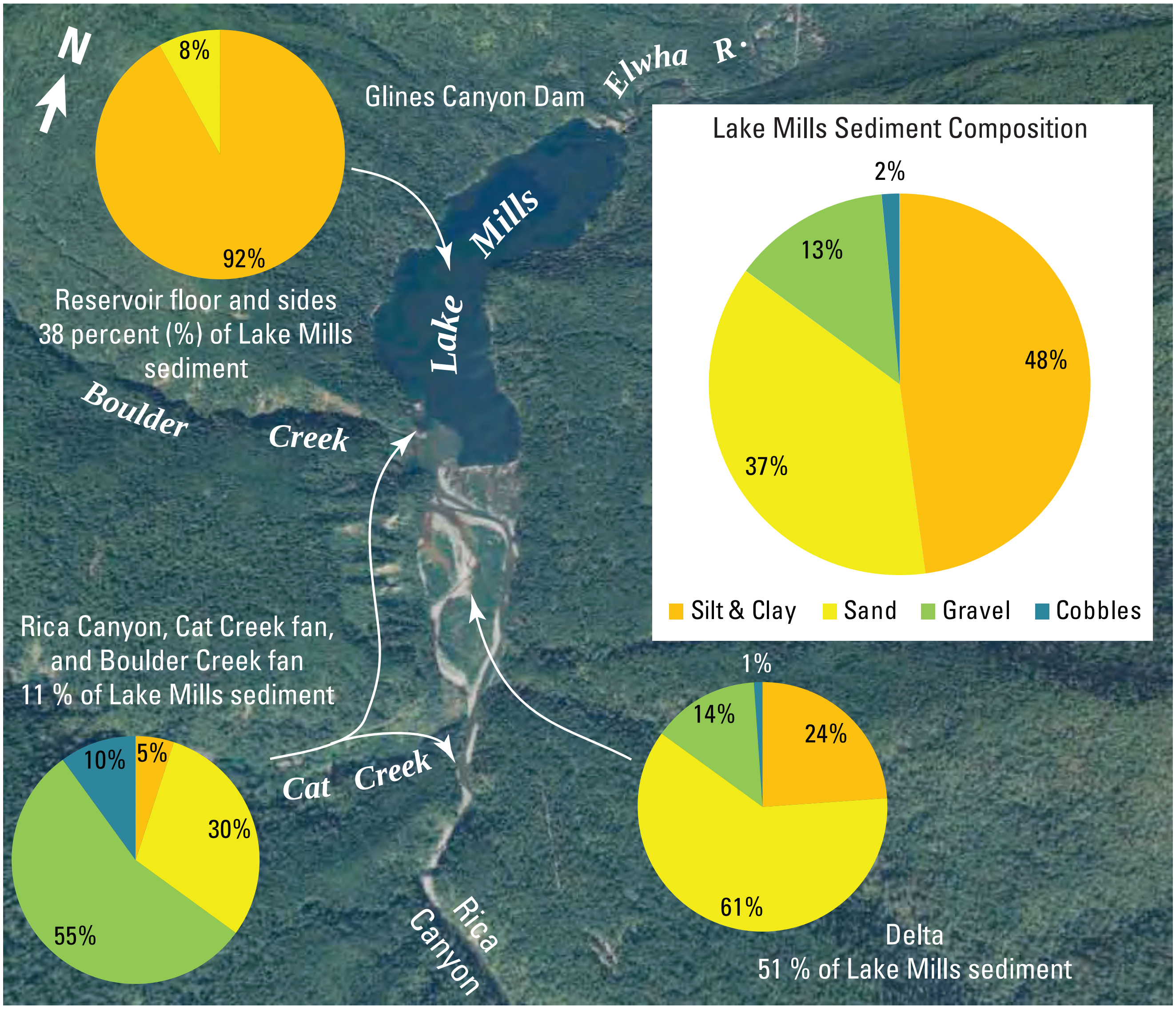

Figure 2.2. Delta and size distribution of stored sediment over an oblique view of Lake Mills, Washington. Although most sediment is sand sized or finer, a considerable volume of sediment is coarse-grained gravel and cobbles. (Aerial photograph from a 2009 National Agriculture Imagery Program image.)

discharge as a function of streamflow during a study period of water years 2006 and 2007, Curran and others (2009) estimated that total sediment load to Lake Mills was between 217 million and 513 million kg/yr. Of that total, 77 percent was transported as suspended sediment and 23 percent was transported as bedload. Based on a comparison of suspended-sediment measurements made upstream and downstream of the reservoir, Curran and others (2009) also estimated the trap efficiency was 0.86 for Lake Mills. Assuming a mean bulk density of 1,760 kg/m^3 for sediment in the Lake Mills delta (Childers and others, 2000), the total sediment load transported to Lake Mills estimated by Curran and others (2009) is equivalent to 120,000–290,000 m^3/yr, which is consistent with total sediment load estimates from reservoir sedimentation (250,000 m^3/yr; Gilbert and Link, 1995; Bountry and others, 2010).

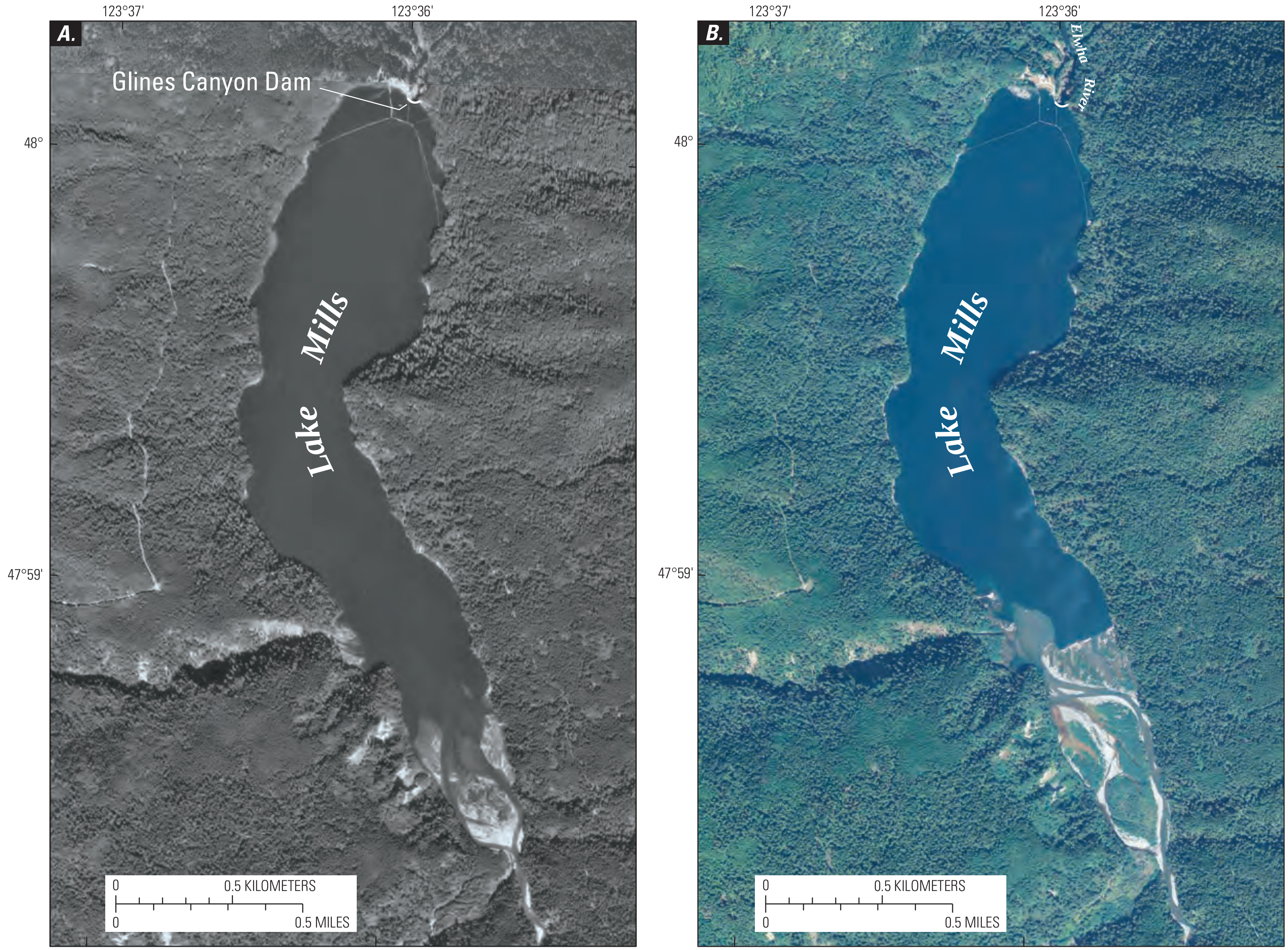

Figure 2.3. Aerial photographs of delta growth from (*A*) 1990 to (*B*) 2009 showing an approximately 400-meter advance of the delta front into the reservoir and significant growth of new vegetation on the deltaic surface at Lake Mills, Washington. (Aerial photographs from a 1990 digital orthophotograph quadrangle and a 2009 National Agriculture Imagery Program image.)

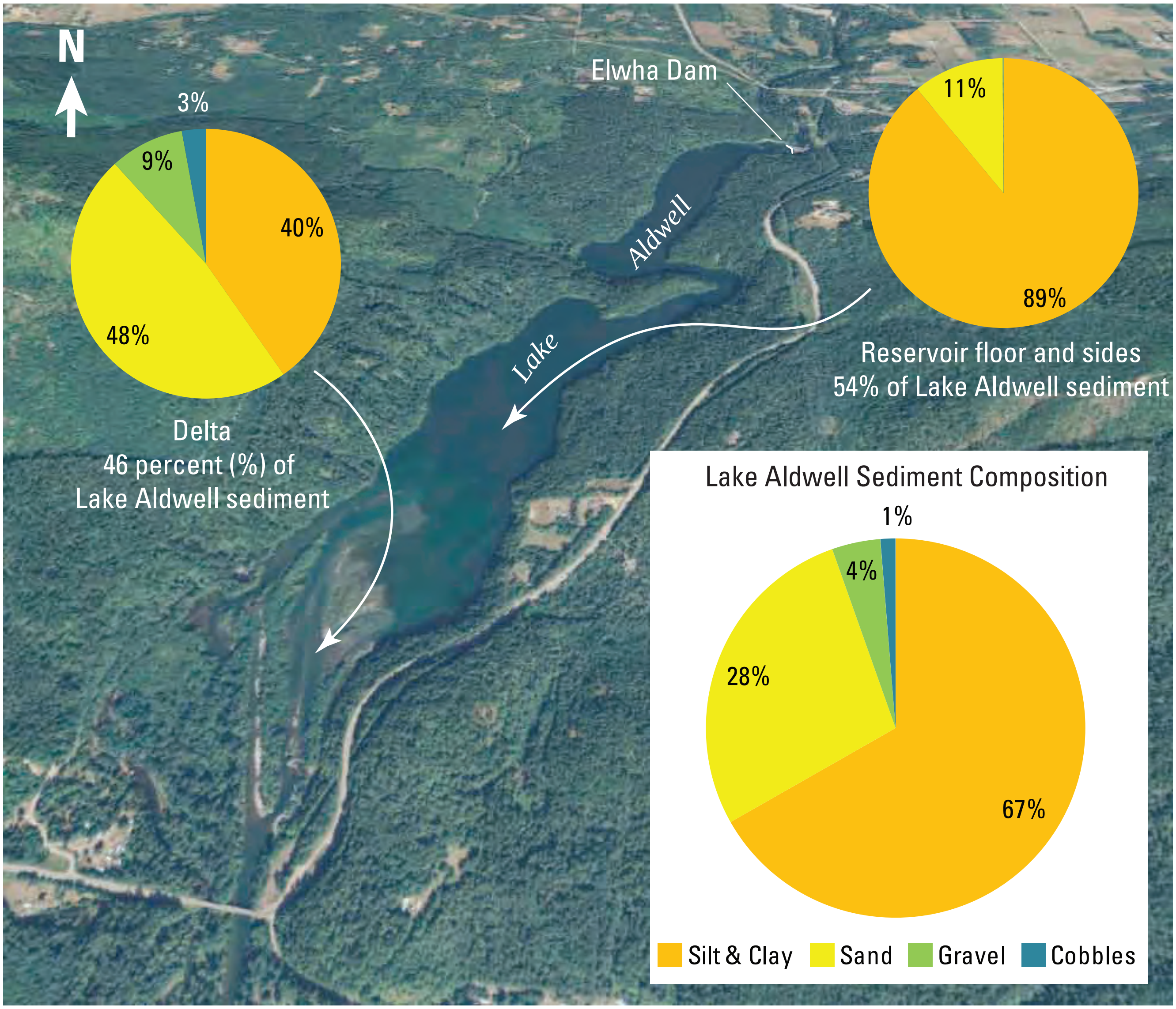

Figure 2.4. Delta and size distribution of stored sediment and an oblique view of Lake Aldwell, Washington. Sand, silt, and clay are the dominant particle-size classifications. (Aerial photograph from a 2009 National Agriculture Imagery Program image.)

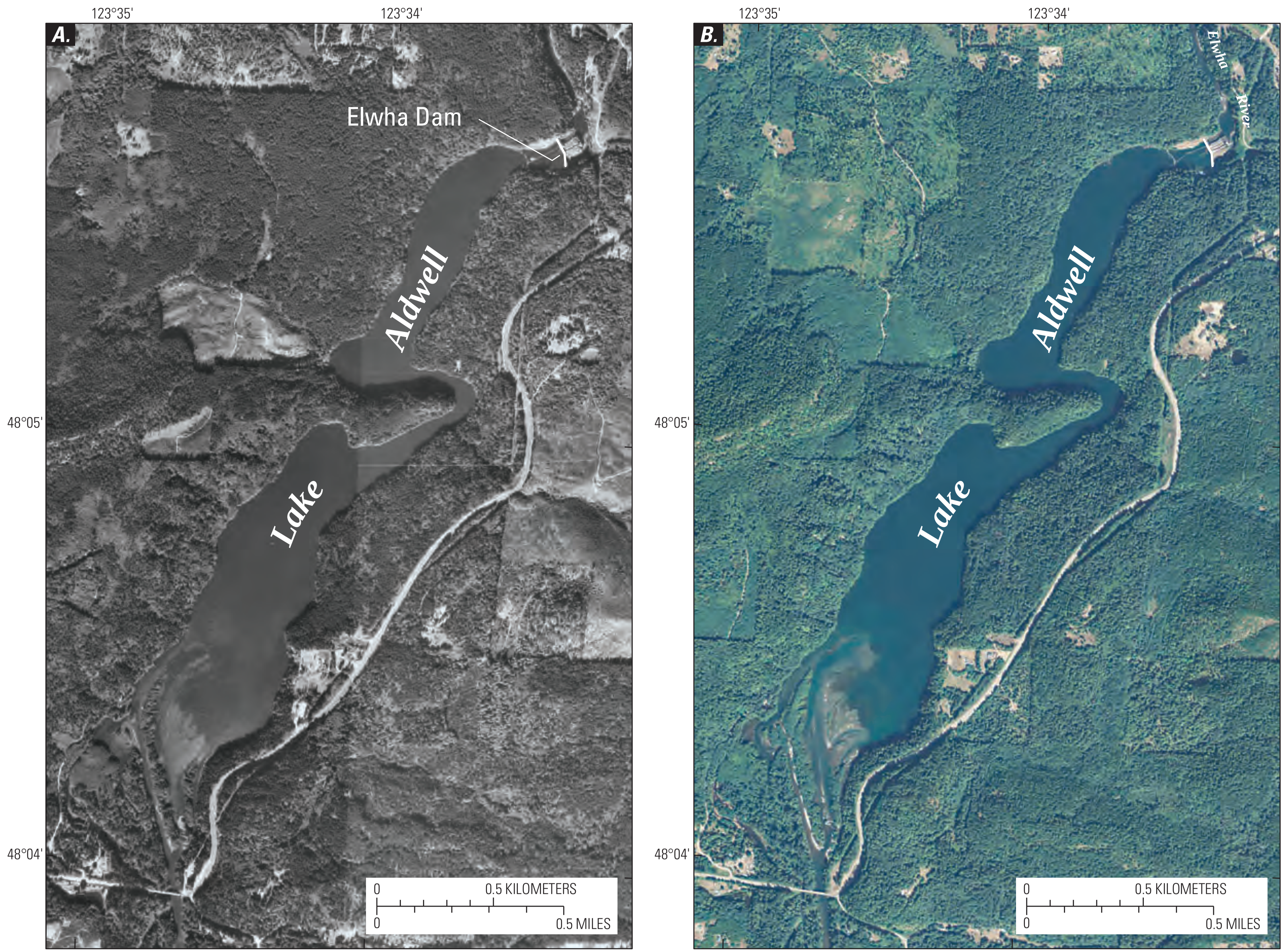

Figure 2.5. Aerial photographs of delta growth from (*A*) 1994 to (*B*) 2009 showing relatively little advancement of the delta at Lake Aldwell, Washington. (Aerial photographs from a 1994 digital orthophotograph quadrangle and a 2009 National Agriculture Imagery Program image.)

Anticipated Sediment Effects from Dam Removal

Elwha and Glines Canyon Dams are slated for decommissioning beginning in September 2011, and one of the primary concerns during and after removal of the dams is the fate of the estimated 19 million m^3 of total stored reservoir sediments. Of the four general categories of sediment-management alternatives originally considered for dam decommissioning (no action, river erosion, mechanical removal, and stabilization; Randle, 2002), the river-erosion alternative; that is, allowing the river to erode part of the stored sediment using natural river-transport processes (Randle and others, 1996), was selected because it was deemed the most cost-effective way to fully restore the river system (U.S. Department of the Interior, 1995, 1996, 2005a, 2005b).

Several key scientific investigations provide insight into the potential erosion and transport of reservoir sediment in response to the removal of Elwha and Glines Canyon Dams. A field experiment was conducted on Lake Mills in April 1994 that used a trial drawdown of the reservoir levels to demonstrate the potential effects of removing Glines Canyon Dam on the erosion and transport of Lake Mills sediment (Childers and others, 2000). Changes in sediment transport and suspended-sediment concentration during and after dam removal were investigated by the Bureau of Reclamation using a mass-balance model to simulate reservoir sediment erosion and a one-dimensional sediment-transport model, HEC-6, to simulate hydraulics and sediment transport of the river downstream of Elwha Dam (Randle and others, 1996). Laboratory physical modeling experiments were conducted by Bromley and others (2011) to investigate the dynamics of sediment erosion from the Lake Mills delta during dam removal. Additionally, Konrad (2009) simulated the dam-removal response in the entire Elwha River using a one-dimensional model developed by Bennett (2001).

Although there is agreement among these various simulations, the responses of the river to dam removal are still not certain. Adaptive management will monitor erosion of the reservoir sediments and effects to the lower river during and after dam removal to determine if alterations to the dam-removal and management strategy are necessary (Randle and Bountry, 2010). Management concerns during dam removal include the release of fine-grained sediment (silt and clay-sized material) affecting downstream water quality and the release of accumulated coarse-grained sediment (sand, gravel, and cobbles) with the potential to affect downstream channel geomorphology, flooding potential, channel-migration rates, habitat productivity, and coastal processes.

Erosion of Reservoir Sediment

Elwha and Glines Canyon Dams will be removed concurrently over a 2–3-year period. Each dam, and associated reservoir water-surface elevation, will be drawn down in controlled increments of 2–6 m to allow erosion and redistribution of the delta sediment in the remaining reservoir. The maximum drawdown rate for each increment will be limited to 0.46 m/d in an effort to prevent initiation of reservoir landslides. As reservoir levels are lowered, headcut erosion will propagate upstream through the reservoir sediment. Periods of constant reservoir elevation between the incremental lowering will allow river processes like meandering and channel migration to induce lateral erosion of the sediment (Randle and others, 1996). Observations during the 1994 drawdown experiment of Lake Mills (Childers and others, 2000) showed that unvegetated deltaic sediment was readily mobilized and transported out of the reservoir (fig. 2.6).

As the two dams are incrementally removed, each delta will prograde, releasing fine sediment into suspension and transporting coarse-grained sediment farther into the shrinking reservoir. In the early stages of drawdown, the large reservoir capacity in the existing pool will allow much of the fine sediment to fall back out of suspension and not pass downstream of the dam site. This response was observed during the 1994 drawdown experiment as the maximum suspended-sediment concentration measured in the draining Lake Mills reservoir was 6,110 mg/L, although the measured suspended-sediment concentration downstream of the dam in Glines Canyon did not exceed 20 mg/L (Childers and others, 2000). As the remaining reservoir water volume decreases from the lowered water surface and infilling from the prograding delta, the capacity of the reservoir to retain suspended sediment will decrease and more suspended sediment will progressively exit the reservoir. Additionally, the volumes of fine-sediment erosion will increase and mix with an ever decreasing reservoir water volume, resulting in increased suspended-sediment concentrations released from the reservoir.

Coarse sediment transported as bedload, however, will remain in the reservoir until the prograding delta reaches the lowered dam structure. After the top one-half to two-thirds of each dam has been removed, the volume of sediment eroded and prograded toward the dam site may be sufficient to completely fill the remaining reservoir volume behind the partially lowered dam (Randle, 2006; Randle and Bountry, 2010). Thereafter, each successive increment of dam removed will draw down the reservoir and release pulses of bedload and suspended-sediment load to the river.

A primary goal during the dam-removal process is to erode as much sediment as possible across the delta flood plain to avoid stranding large wedges of sediment in the former reservoir inundation area. Large wedges of sediment would be susceptible to catastrophic mass wasting even after the dams are completely removed, endangering people and structures downstream and potentially affecting water quality. Physical modeling by Bromley and others (2011) demonstrated that sediment mobilization would be maximized if the incipient knickpoint of the eroding channel was located in the center of the reservoir delta and drawdown steps were relatively small, thus allowing the incising river to meander across the width of the delta. Bromley and others (2011) also determined, however, that if the river were positioned along one of the bedrock canyon walls at the start of drawdown, the river would tend to remain along that bedrock face throughout the dam-removal sequence, leaving a sediment terrace asymmetrically positioned along the opposite canyon (fig. 2.7) that would be susceptible to mass wasting. Having the river channel positioned near one of the bedrock walls also creates the potential that the river would plunge over a bedrock shelf away from the position of the pre-dam river channel, creating a hydraulic barrier to fish passage.

In an effort to lessen the possibility of stranding the incising river channel on one side of the delta, trees and other vegetation were manually removed in September 2010 and a 15-m wide pilot channel was excavated along the upstream 335 m of the Lake Mills delta centerline (fig. 2.8; Bountry and others, 2010). Physical modeling also simulated that much of the trapped sediment will remain in the region of the drained reservoir (Bromley and others, 2011). One salient goal during and after dam removal will be to establish vegetation on relatively stable terraces of sediment remaining in the reservoir areas.

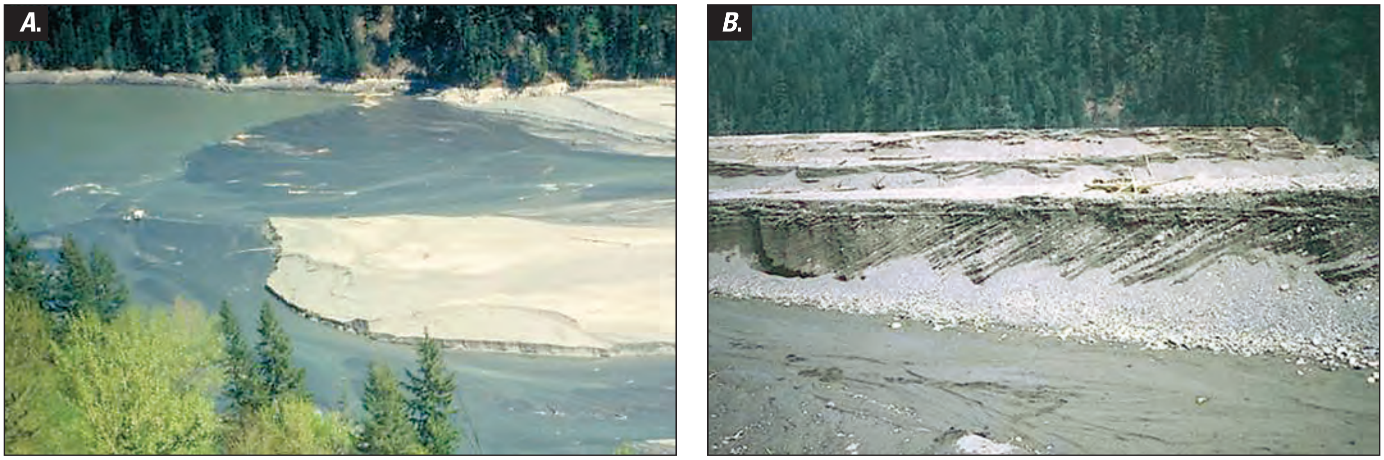

Figure 2.6. Drawdown experiment at the Lake Mills delta, Washington, in 1994. (*A*) Eroded sediment redeposited at the distal end of the prograding delta across the entire width of the receding reservoir. (*B*) The incised active channel showed the underlying stratigraphy of the delta and multiple, inset terraces. Lack of stabilizing vegetation and the coarse, non-cohesive nature of the sediment allowed vigorous incision and downstream transport (adapted from Randle and Bountry, 2008).

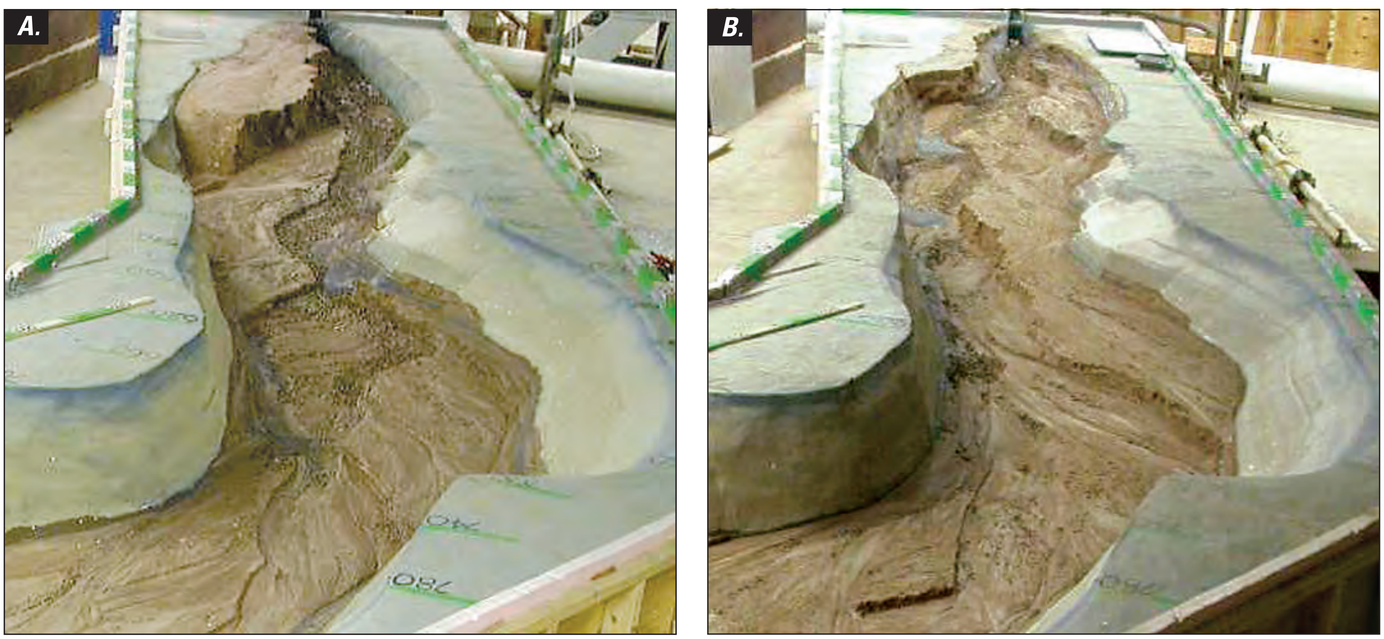

Figure 2.7. Physical model experiments of sediment erosion (*A*) without a pilot channel and (*B*) with a pilot channel showing the extensive and potentially problematic high terrace of alluvial sediment that may remain if the incising channel starts along one of the bedrock walls, Lake Mills, Washington. (Photographs courtesy of Chris Bromley; adapted from Randle and Bountry, 2008.)

Figure 2.8. Lake Mills delta, Washington, (*A*) before construction of a pilot channel and removal of vegetation, September 13, 2010; (*B*) with the newly constructed center pilot channel passing flow after a 5.5-meter reservoir drawdown, November 3, 2010; and (*C*) after a flood that peaked on December 12, 2010, at 632 cubic meters per second (22,300 cubic feet per second), December 15, 2010. ([*A*] Photograph courtesy of Tom Roorda, Northwestern Territories, Inc.; [*B*] photograph courtesy of Dick Bauman, Bureau of Reclamation; and [*C*] photograph courtesy of Dick Bauman, Bureau of Reclamation.)

Sediment Delivery Downstream of Dam Sites

Typical downstream river response to dam removal includes increased turbidity and sediment concentrations (Randle, 2002; Stanley and Doyle, 2002; Major and others, 2008) as well as geomorphic responses to the increased sediment loads, although the magnitude of these effects tends to decrease with time and distance from the dam (Pizzuto, 2002). The modeling efforts by Randle and others (1996) and Konrad (2009), along with observations from the 1994 drawdown (Childers and others, 2000) and the physical modeling experiments by Bromley and others (2011) provided much of the basis for expectations of the sediment delivery to the Elwha River downstream of both dam sites and the response of the river.

Based on results from the Bureau of Reclamation mass-balance modeling, between one-half and two-thirds of silt and clay-sized sediment in the reservoirs are expected to erode and be transported downstream as suspended load, and between one-quarter and one-third of the sand and gravel-sized sediments are expected to erode from the reservoirs and be transported downstream as bedload (Randle and others, 1996; Randle, 2006). Extrapolating these earlier model simulation results (Randle and others, 1996) to the updated estimate of 19 million m^3 of accumulated reservoir sediment since construction of the dams (Bountry and others, 2010), it is estimated that 7–8 million m^3 of sediment will erode from the reservoirs and be released to the river downstream.

Fine-Sediment Transport

Much of the fine sediment released during dam removal will be transported in suspension directly to the Strait of Juan de Fuca, resulting in increased suspended-sediment concentrations and turbidity throughout the lower river (Randle and others, 1996). Simulated suspended-sediment concentrations at Glines Canyon Dam were high during dam removal, exceeding 10,000 mg/L for several weeks each year (Konrad, 2009). Simulated suspended-sediment concentrations at the river mouth did not exceed 500 mg/L nearly as often as downstream of Glines Canyon Dam, generally less than 20 days in a 3 month period. The Konrad (2009) simulations were consistent with earlier model simulation results by the Bureau of Reclamation (Randle and others, 1996). Based on the Bureau of Reclamation model simulation results, suspended-sediment concentrations are expected to be typically between 200 and 1,000 mg/L during the dam-removal period with temporary peaks as high as 30,000–50,000 mg/L (Randle and others, 1996). For comparison, based on the regressions developed with suspended-sediment concentration data collected in the Elwha River just upstream of Lake Mills (Curran and others, 2009), the background suspended-sediment concentration in the Elwha River at a median flow of 34 m^3/s would be 4.6 mg/L and the concentration at 101 m^3/s (a discharge value exceeded only 5 percent of the time) would be 121 mg/L.

After dam removal, fine-sediment concentrations are expected to return to natural, background levels within 2–5 years after complete removal (Randle and others, 1996). Similarly, Konrad's (2009) simulations suggested that suspended-sediment transport likely will return to normal levels during low to moderate flows within 4 years after dam removal. However, larger flow events may still generate high suspended-sediment concentrations, as erosion is expected to further widen the channels in the drained reservoir region (Randle and others, 1996). These episodic high discharge values may cause high suspended-sediment concentrations and bed instability that may have deleterious effects on Chinook salmon incubation through infill of spawning gravels and reduction of flow to salmon redds (Konrad, 2009).

Coarse-Sediment Transport

The exact timing of the arrival of the coarse-grained bedload to the lower Elwha River is not certain. Coarse material will be released downstream once the prograding deltas have reached the lowered dam faces. However, the rate of transport downstream will depend on the rate of dam removal and the driving hydrology in the years during and following dam removal (Randle and others, 1996; Konrad, 2009). If the winter seasons are wet, the resulting high river flows will be able to transport larger quantities and sizes of sediment downstream at a faster rate. In contrast, if dry conditions persist, the river will have less capacity to transport this coarse material downstream and more time will elapse before this coarse sediment arrives in the lower river.

Although 95 percent of the sediment from the Lake Aldwell delta consists of sand, silt, and clay and probably will have a minimal geomorphic affect on the lower Elwha River, much of the coarser material from the Lake Aldwell delta may start to arrive in deposits in the lower river during the second winter (2012–13), accreting some existing gravel bars and probably transporting downstream relatively quickly. Farther upstream, no coarse bedload will exit Lake Mills until the prograding delta impinges against the lowered Glines Canyon Dam structure. Coarse sediment release may begin during the second winter season (2012–13), although large quantities of bedload probably will not pass the Glines Canyon Dam site until the winter of the final year of dam removal (2013–14). This enhanced bedload will then transport down the middle reach of the river for another 2–5 years after complete removal of Glines Canyon Dam (Randle and others, 1996).

Although the first appreciable transport of coarse reservoir material to the lower river may start to be observed in the winter of the third year of dam removal (2013–14), whole-scale geomorphic changes and concomitant increases in bedload and sediment caliber associated with downstream translation of reservoir sediment will affect the lower river 4–7 years after the start of dam removal. Bedload transport rates are then expected to return to background conditions 7–10 years after the start of dam removal. However, there is considerable uncertainty in the bedload transport and system response.

Geomorphic Effects due to Released Sediment

The lower river downstream of Elwha Dam will undergo geomorphic changes due to the increased sediment supply during and after dam removal (Draut and others, 2008; Konrad, 2009). Initially, as only fine material is released from the reservoirs, bimodal grain-size distributions will result downstream due to the newly transported sand combined with the armored cobble and boulder bed (Kloehn and others, 2008). As dam removal progresses and fine and coarse sediment are released downstream, the river is expected to become transport-limited: sediment will deposit in areas where the transport capacity of the river is less than the sediment supply (that is, pools, side channels, recirculation zones, and flood plain edges along the channel). Most aggradation is expected to occur in river pools during periods of low river discharge. Aggradation on riffles is not as likely to occur, but is possible depending on the magnitude and frequency of peak flows during and after dam removal.

Simulations by Konrad (2009) showed that within 4 years after completion of dam removal the lower Elwha River will aggrade as much as about 2.5 m in the upper part of the lower river and relatively little near the river mouth. Similarly, Bureau of Reclamation simulations suggest that areas of the lower river will aggrade between 0–3 m in the short term and in the decades after dam removal up to an average of 1.4 m throughout the lower river (Randle and others, 1996; Randle, 2006). Overall, river reaches with the greatest potential for morphological change include depositional reaches with lower slopes (for example, lower Elwha River), valley expansions (for example, upstream of Lake Aldwell), or upstream of constrictions where backwater exists at higher flow (for example, upstream of Glines Canyon) (Konrad, 2009). In the decades after dam removal, channel migration is expected to increase in the lower reach (Kloehn and others, 2008). Due to aggradation and fining of material in the lower river, the channel could shift from the current anabranching form (see Warrick and others, 2011, chapter 3, this report) to a braided system (Draut and others, 2008), likely resulting in a straighter channel alignment (steeper slope), which is consistent with a braided channel planform (Randle and Bountry, 2008).

The sediment released from these dam removals will affect downstream flood risk and ecosystems. Simulations from the Bureau of Reclamation suggest that sedimentation would increase the 100-year flood stage along the lower Elwha River by less than 1 m (Randle and others, 1996; Randle, 2006). In response to these simulations, existing levees along the lower Elwha River were raised and new levees were constructed to protect landowners downstream. In the lower river reaches, the initial increase in sediment load will disrupt aquatic and riparian ecology. However, sediment deposition and the incorporation of large woody debris and organic matter will lead to an overall increase in quantity and quality of fish habitat (Woodward and others, 2008).

Sediment Monitoring During and After Dam Removal

Even with the current simulations, there is considerable uncertainty in the timing and magnitude of the system response to the removal of Elwha and Glines Canyon Dams. Observations from the Marmot Dam removal suggested smaller overall aggradation rates downstream and a faster chronology of river responses than numerical simulations (Major and others, 2008). Studies of the Milltown Dam removal determined that (1) reservoir erosion occurred at greater rates than simulated by numerical modeling due to greater-than-anticipated lateral erosion (Johnsen, 2011), and (2) a sediment pulse that traveled by both translation and dispersion produced downstream pool filling, fine sediment infiltration (Evans and Wilcox, 2010), and sedimentation of side channels, but minimal overall aggradation (Wilcox, 2010).

Monitoring of river reaches downstream of the Elwha and Glines Canyon Dam sites will provide important insight into these geomorphic processes as the Elwha River adjusts to and recovers from the dam-removal process. During initial reservoir drawdown, monitoring activities will be focused on reservoir surveys, reservoir elevation, turbidity, discharge, and investigations to characterize the areas of erosion and the advancement of the delta fronts (Randle and Bountry, 2010). Turbidity will be measured continuously upstream of Lake Mills and at the new surface-water diversion structure at river kilometer (RKm) 5.3 to monitor water quality in the river where water is diverted into the Elwha Water Treatment Plant (Randle and Bountry, 2010). If suspended sediment degrades water quality beyond an acceptable threshold, the pace of dam removal will be adjusted within the adaptive management framework to reduce effects for downstream water users. Additionally, an acoustic Doppler current profiler (ADCP) and a laser in-situ scattering and transmissometry (LISST) system will be installed at the water diversion structure to enable the quantification of suspended-sediment load in the lower river at least through 2013. Physical samples will be collected through at least the end of 2013 to quantify the suspended-sediment concentration carried by the river under different hydrologic conditions. These samples, in turn, will be used with turbidity data, acoustic-backscatter data, and LISST data to estimate the continuous suspended-sediment load carried within the lower Elwha River.

Monitoring of riverbed aggradation and potential increase in flood stage downstream of the dams will begin once sand and gravel have begun to transport downstream of the reservoir sites, using a combination of stage recorders, channel cross-section surveys, and longitudinal profile surveys (Randle and Bountry, 2010). A series of geophone bedload sensors were installed along the crest of the diversion weir at the new Elwha Water Treatment Plant (RKm 5.3) to continuously measure acoustic energy generated by bedload transport during and after dam removal (Marr and others, 2009; Randle and others, 2009). To fully quantify flux of bedload, however, physical measurements of bedload are needed to correlate acoustic energy measured with the geophones to gravel transport. There are currently no plans to measure bedload in the lower river during the dam-removal project. Over the long term, river-based monitoring will focus on the restoration of vegetation, habitat, sediment dynamics, the function of large woody debris, and morphological processes in the lower river, estuary, and nearshore zone (Woodward and others, 2008).

Although some monitoring of sediment load and geomorphic response of the river corridor during and after dam removal will occur, the amount of basic geomorphic research currently planned during the project is relatively modest. No measurements of bedload transport are currently planned. Suspended-sediment load will be measured in the lower river through 2013, but monitoring beyond 2014 is unfunded and no suspended-sediment measurements are planned upstream of Lake Mills or between the two reservoirs. These additional monitoring sites would provide spatial resolution of the rate of sediment mobilization and downstream transport and enable the construction of a quantifiable sediment budget throughout the river. Finally, there are no plans to collect remotely-sensed geomorphic data sets of repeat aerial photography or lidar during the project. Such public data sets would enable a wealth of follow-on analyses that could provide important insight well after completion of the dam removals.

The restoration of the Elwha River represents a true “best case” for what a full-on watershed restoration scheme might hope to accomplish. The removal of Elwha and Glines Canyon Dams is unique, in part, because the project represents the largest controlled release of sediment in the history of North America. This particular project is also unique because it encompasses an entire watershed including the estuarine and marine habitats of a predominantly pristine wilderness historically home to all runs of Pacific Northwest anadromous salmonids. It is imperative that the response and recovery of the Elwha River during and after dam decommissioning are studied if reasonable expectations for other watershed-based restoration efforts are to be established. A sound technical narrative of this unprecedented project is needed to describe the fate of the sediment in the years to decades after dam removal.

Summary

The presence of Elwha Dam and Glines Canyon Dam have significantly altered the sediment transport and geomorphic processes of the Elwha River and the nearshore environment in the Strait of Juan de Fuca. The planned removal of the dams from 2011 to 2014 will release an anticipated 7–8 million cubic meters of the estimated 19 million cubic meters of stored reservoir sediment to erosion and transport downstream. During dam decommissioning, fine sediment is expected to be more readily mobilized than coarse sediment.

Studies indicate that suspended-sediment concentration during low to moderate flows will return to pre-dam levels within a few years after the dams are completely removed. Storm events will continue to generate higher concentrations of suspended sediment until the reservoir sediment is no longer significantly eroding over the long term. Generally, recovery rates of sediment-transport regimes relative to background conditions are expected to take 4–7 years following complete dam removal. The magnitude, timing, and frequency of storm events during and soon after dam removal are the mechanisms causing the overall transport of released sediment to the river downstream and the time until long-term stabilization of the sediment has been obtained.

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Chapter 3

Geomorphology of the Elwha River and its Delta

By Jonathan A. Warrick, Amy E. Draut, Michael L. McHenry, Ian M. Miller, Christopher S. Magirl, Matthew M. Beirne, Andrew W. Stevens, and Joshua B. Logan

Abstract

The removal of two dams on the Elwha River will introduce massive volumes of sediment to the river, and this increase in sediment supply in the river will likely modify the shapes and forms of the river and coastal landscape downstream of the dams. This chapter provides the geologic and geomorphologic background of the Olympic Peninsula and the Elwha River with emphasis on the present river and shoreline. The Elwha River watershed was formed through the uplift of the Olympic Mountains, erosion and movement of sediment throughout the watershed from glaciers, and downslope movement of sediment from gravitational and hydrologic forces. Recent alterations to the river morphology and sediment movement through the river include the two large dams slated to be removed in 2011, but also include repeated bulldozing of channel boundaries, construction and maintenance of flood plain levees, a weir and diversion channel for water supply purposes, and engineered log jams to help enhance river habitat for salmon. The shoreline of the Elwha River delta has changed in location by several kilometers during the past 14,000 years, in response to variations in the local sea-level of approximately 150 meters. Erosion of the shoreline has accelerated during the past 80 years, resulting in landward movement of the beach by more than 200 meters near the river mouth, net reduction in the area of coastal wetlands, and the development of an armored low-tide terrace of the beach consisting primarily of cobble. Changes to the river and coastal morphology during and following dam removal may be substantial, and consistent, long-term monitoring of these systems will be needed to characterize the effects of the dam removal project.

Introduction

The removal of the Elwha and Glines Canyon Dams will introduce massive volumes of sediment into the Elwha River, some of which will be transported to the Strait of Juan de Fuca coastal waters. This sediment perturbation will likely modify the shapes and forms, that is, the "geomorphology," of the landscape downstream of the dams. One important step to characterizing these future changes is a thorough description of the present geomorphology of the Elwha River and its coast. If these changes are not measured and described, future changes to the river, flood plain, and coastal landforms cannot be tracked, and the effects of the dam removal will remain unknown.

The present geomorphology of the Elwha River results from an uplifting mountain range, large changes in sea level, former alterations to the landscape by glaciers, and continual modification and movement of materials by winds, waves, rain, snow, river discharge, and the force of gravity. The watershed has also been modified by human activity, most notably from the two large dams that prevent sediment movement down the river. Less obvious, but also important, are the numerous human alterations to the river channel.

This chapter introduces the geomorphic setting of the Elwha River and its coast, explores the recent history of the Elwha River and its delta, and describes the effects that human intervention has had on these systems. This geomorphic framework will help track future changes to this system after the dams are removed.

Geologic Setting

Tectonics

The steep and glaciated Olympic Mountains include some of the highest coastal peaks along Cascadia and form the headwaters of the Elwha River drainage basin. These mountains were formed by the convergence of the oceanic Juan de Fuca plate with the continental North American plate through the Cascadia subduction zone (fig. 3.1). As the Juan de Fuca plate converged with the North American plate at a rate of approximately 43 mm/yr, part of the oceanic rocks and sediments accreted onto the front of the North American plate rather than subducting under it. The Olympic Mountains are the highest part of this "accretionary wedge" of marine rocks that spans the majority of the Cascadia subduction zone (fig. 3.1).

Evidence for the accretion of marine rocks abounds throughout the mountainous terrain of the Olympic Peninsula, where the primary rock types are slightly altered sedimentary (that is, "metasedimentary") rocks that originated underneath the seafloor (fig. 3.2). These uplifted rocks are known as the Olympic subduction complex (OSC; Brandon and others, 1998), which is an exposed part of the accretionary wedge that underlies most of the offshore continental margin (Stewart and Brandon, 2004). The deformed phyllites and schists that make up the OSC were originally deposited on the seafloor in Eocene to Miocene time (55.8 to 5.3 million years before present [BP]) as a result of numerous sediment-laden turbidity currents plunging down the continental slope into deep water (Orange and others, 1993; Brandon and others, 1998; U.S. Geological Survey Geologic Names Committee, 2010).

A distinct boundary exists between the sedimentary rocks of the OSC and the Coast Range terrane (CRT; Brandon and others, 1998), which are sequences of volcanic and sedimentary rocks (fig. 3.2). The CRT also was formed in the ocean, as evidenced by its marine sedimentary rocks and pillow basalts, the latter being the result of underwater volcanism. The structural boundary between the older CRT and the OSC is the extensive Hurricane Ridge Fault and its equivalents (Tabor and Cady, 1978; Gertsel and Lingley, 2003; Polenz and others, 2004). Because the Hurricane Ridge Fault crosses the Elwha River basin near Lake Mills (fig. 3.2), bedrock lithology of the Elwha River basin upstream of Lake Mills differs substantially from the basin downstream of Lake Mills.

Over geologic time, the rocks of the central Olympic Mountains are uplifting at a rate of approximately 0.6 mm/yr (Brandon and others, 1998; Batt and others, 2001). This uplift of the Olympic Mountains maintains steep slopes in the Elwha River watershed and erosional processes such as rockfalls, shallow and deep landslides, and debris flows that generate substantial quantities of sediment for the river (Montgomery and Brandon, 2002; Acker and others, 2008). The rate of tectonic uplift in the Elwha River watershed decreases between the central Olympic Mountains and the coast to rates of less than 0.3 mm/yr (Tabor and Cady, 1978; Brandon and others, 1998; Polenz and others, 2004).

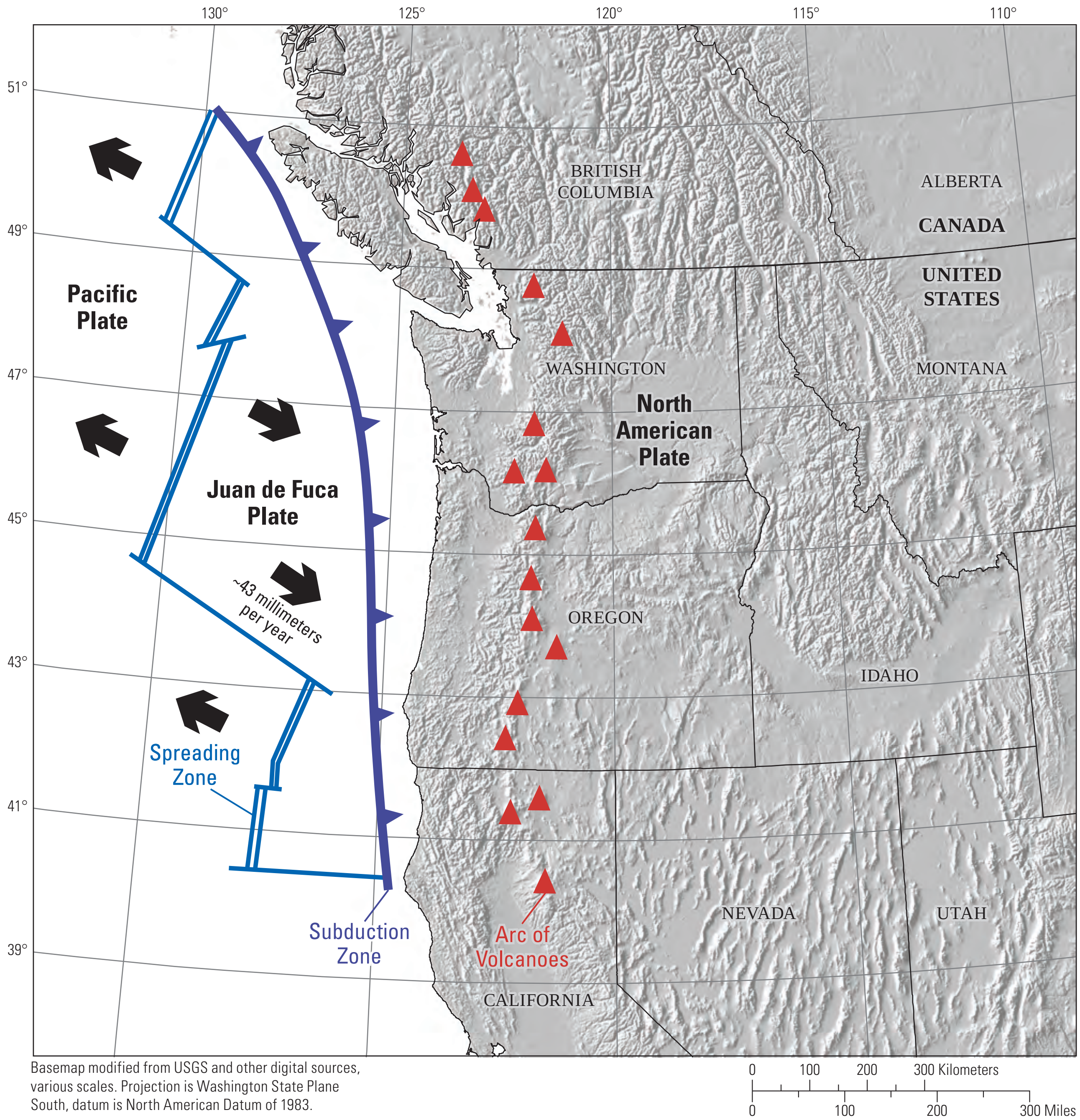


Figure 3.1 Generalized plate tectonics map of the Cascadia subduction zone of North America.

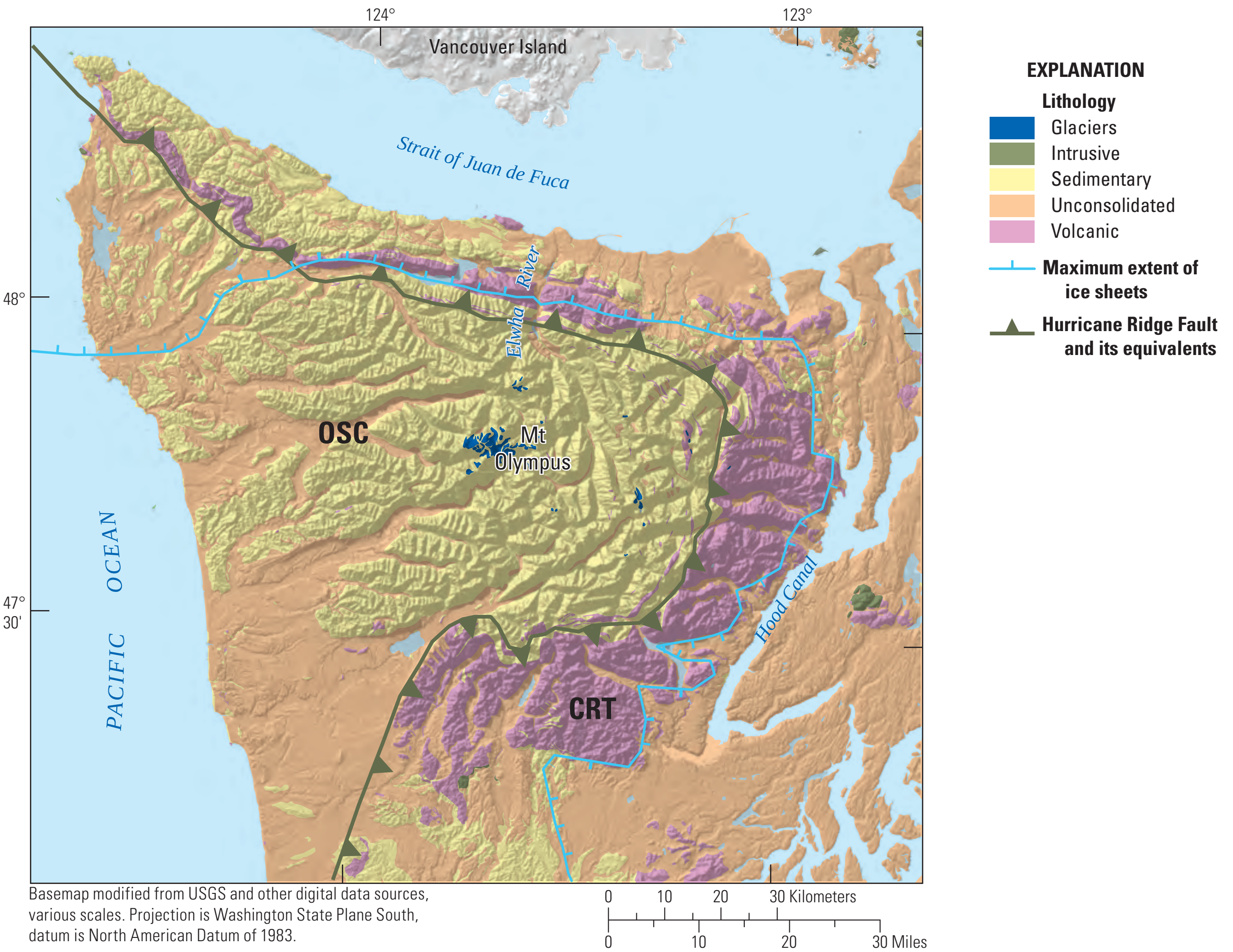


Figure 3.2 Generalized geological map of the Olympic Peninsula, Washington. The fault between the Olympic subduction complex (OSC) and the Coast Range terrane (CRT; Brandon and others, 1998) is shown with a green line. Maximum extent of the most recent Cordilleran ice sheet was to the north and east of the blue line. Map created after Tabor (1987), Brandon and others (1998), and Schuster (2005).

Glacial Processes During the Quaternary Period

Profound new forces at the close of the Pliocene (approximately 2.6 million years ago) shaped the uplifted rocks of the Olympic Mountains. Sweeping down from their northern source in the highlands of British Columbia, massive continental ice sheets buried the Puget Sound and its margins. At the same time, valley glaciers sourced in the hinterlands of the Olympic Mountains carved out deep U-shaped troughs in the valleys (Easterbrook, 1986; Batt and others, 2001; Mosher and Hewitt, 2004). In the most recent glacial episode, these valley glaciers reached their maximum extent at about 19,000 years BP (Easterbrook, 1986), before being overridden at their margins by the ice sheets advancing from the north. At about 14,000 years BP, the continental ice sheet reached its maximum extent, approximately 11 km inland of the present coastline in the region of the lower Elwha River watershed (fig. 3.2). The maximum thickness of the ice sheet has been estimated to be 1,900 m in the central Strait of Juan de Fuca (McNulty, 1996; Porter and Swanson, 1998; Mosher and Hewitt, 2004; Polenz and others, 2004). The weight of the ice sheet depressed the land surface, sinking the ground surface almost 150 m in the region of the Strait (Mathews and others 1970; Waitt and Thorson, 1983; Dethier and others, 1995; Huntley and others, 2001). The land surface rose quickly as the ice retreated northward toward Canada, recovering much of its former elevation after about 2,000 years.

Remnants of the ice age persist today as small alpine glaciers around Mount Olympus (fig. 3.2). Additional evidence of the ice age is scattered widely across the Olympic Mountains, particularly where the Juan de Fuca continental ice lobe repeatedly dammed the waters of the Elwha River watershed, creating a lake in the lower Elwha River watershed. Sediment deposits from this pro-glacial lake, with poorly sorted, poorly stratified glacial outwash alluvium and stratified sediment dominated by silt and clay, are visible today in bluff exposures along the lower Elwha River and may represent the most substantial modern sources of sediment downstream of Elwha Dam (Tabor, 1975, 1987; McNulty, 1996; Polenz and others, 2004).

Profile of the Elwha River

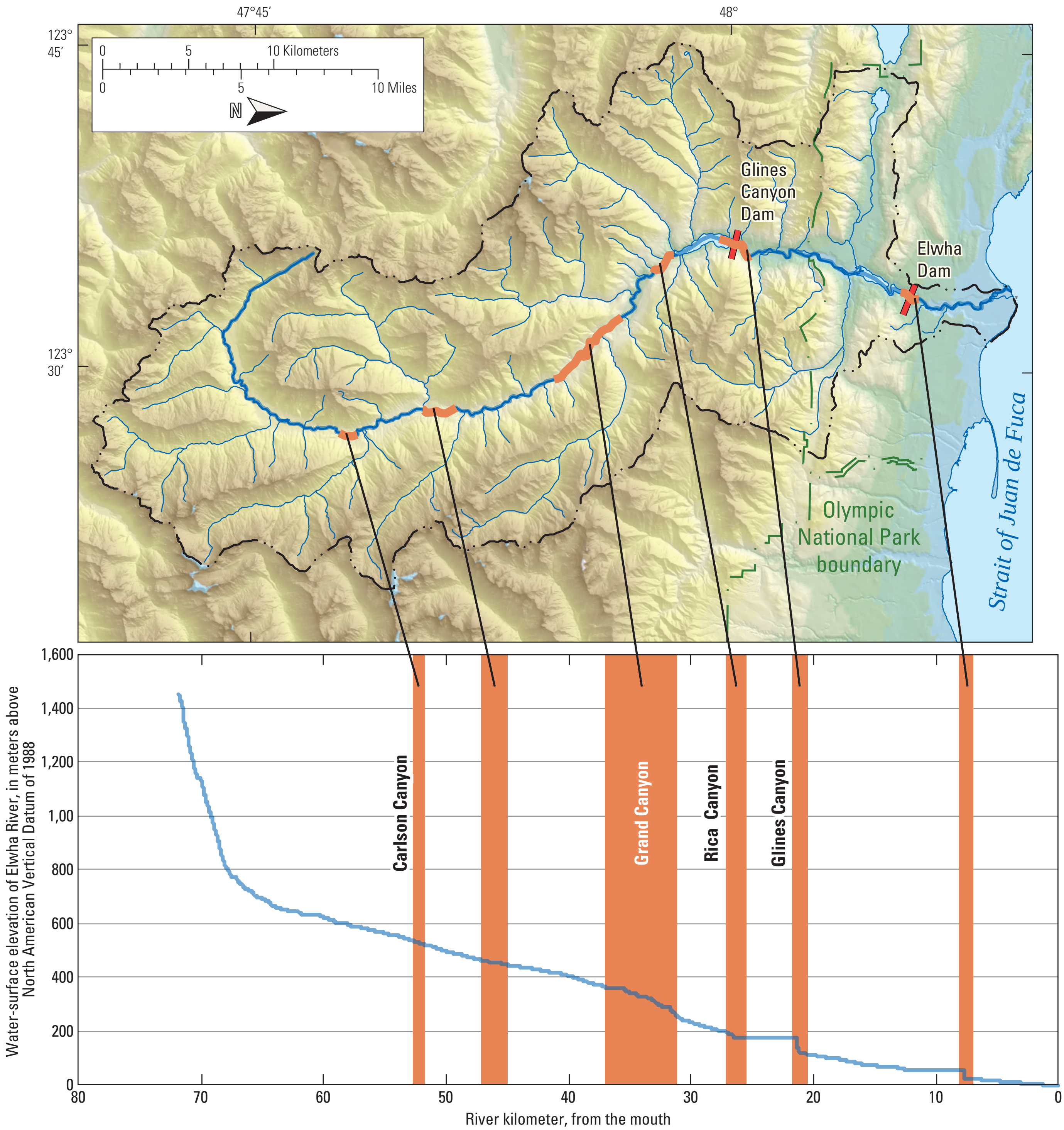
Wherever the waters of the Elwha River cross over the hard, folded rocks of the ancient accretionary wedge, the river is confined within bedrock canyons. Between these canyon segments, wide valleys of alluvium create conditions for active bank erosion and frequent channel migration (Kloehn and others, 2008; Draut and others, 2011). The longitudinal profile of the Elwha River is steepest in the headwaters and flattest near sea level (fig. 3.3). Below approximately 600 m elevation, the slope of the Elwha River is roughly constant at about 0.009 until the Lake Mills delta. There are short, steeper reaches within the Grand Canyon of the Elwha River and Rica Canyon, where the river slope increases to about 0.006. Between Glines Canyon Dam and the coast, the river slope decreases to an average of approximately 0.0013, and the flattest part of the river is immediately upstream of the river mouth (fig. 3.3). The two reservoirs formed by Glines Canyon Dam and Elwha Dam create prominent steps in the longitudinal water surface profile, which will be removed through the lowering of the reservoirs and the erosion of sediment that should follow dam removal.

Lower Elwha River Morphology

The final 7.8 kilometers of the river between the Elwha Dam and the Strait of Juan de Fuca is the flattest reach and is termed the “lower river” (fig. 3.4). This final reach is the most heavily altered reach of the river and will experience the largest effects of renewed sediment supply following dam removal (Draut and others, 2008; Konrad, 2009).

The upper 1.3 km of the lower river lies within a bedrock gorge, and the lowermost 6.5 km is within a broad alluvial flood plain (fig. 3.4). The river channel splits into two to three narrower channels for much of the alluvial section, a planform morphology called “anabranching” (compare to Smith and Smith, 1980; Harwood and Brown, 1993; Knighton and Nanson, 1993). Gravel bars and vegetated islands separate these narrow channels (Pohl, 1999, 2004; Draut and others, 2008; Kloehn and others, 2008). Unconsolidated sediment of these bars and islands contains large proportions of reworked glacial materials that are poorly sorted and have grain sizes ranging from silt and clay to cobbles (Draut and others, 2011).

An anabranching river such as the lower Elwha is understood to be distinctive from single-channel meandering rivers and braided rivers but has elements of both (fig. 3.5). Islands in anabranching rivers can be large relative to the width of the channel and commonly are stable over decades and even centuries. Individual channels meander or may exhibit smaller-scale braiding but eventually rejoin together (Knighton, 1998). Processes that cause anabranching are not fully understood.



Base map modified from USGS and other digital sources, various scales. Projection is Washington State Plane South. Datum is North American Datum of 1983.

Figure 3.3. Longitudinal profile of the water-surface elevation of the main-stem Elwha River calculated from a 10-meter digital elevation model of the Olympic Peninsula, Washington.

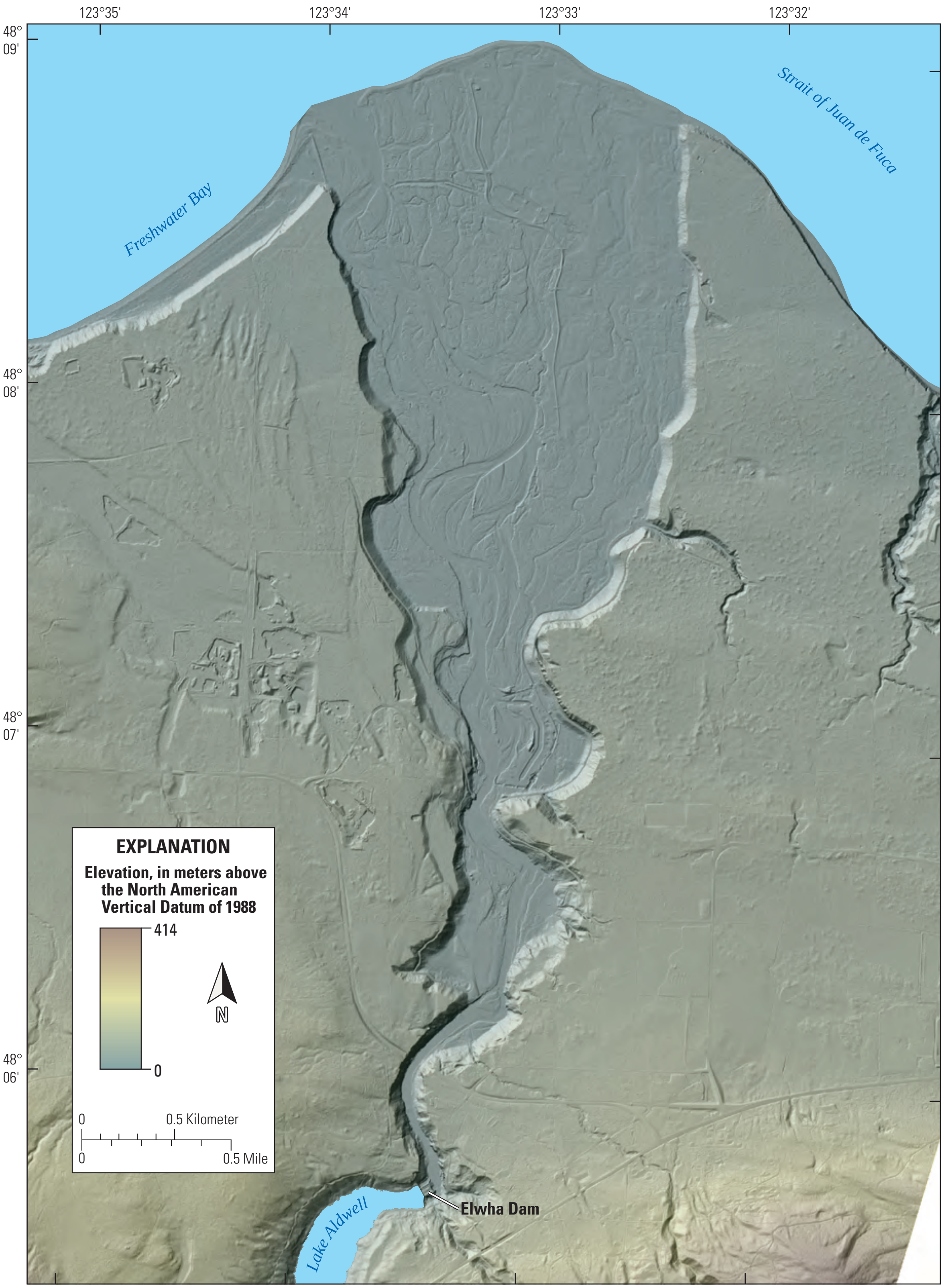


Figure 3.4. Topography of the flood plain and delta of lower Elwha River, Washington. Lidar survey dated between April 4 and 6, 2009, by Terra Remote Sensing, Inc. Horizontal resolution is 1.8 meters.

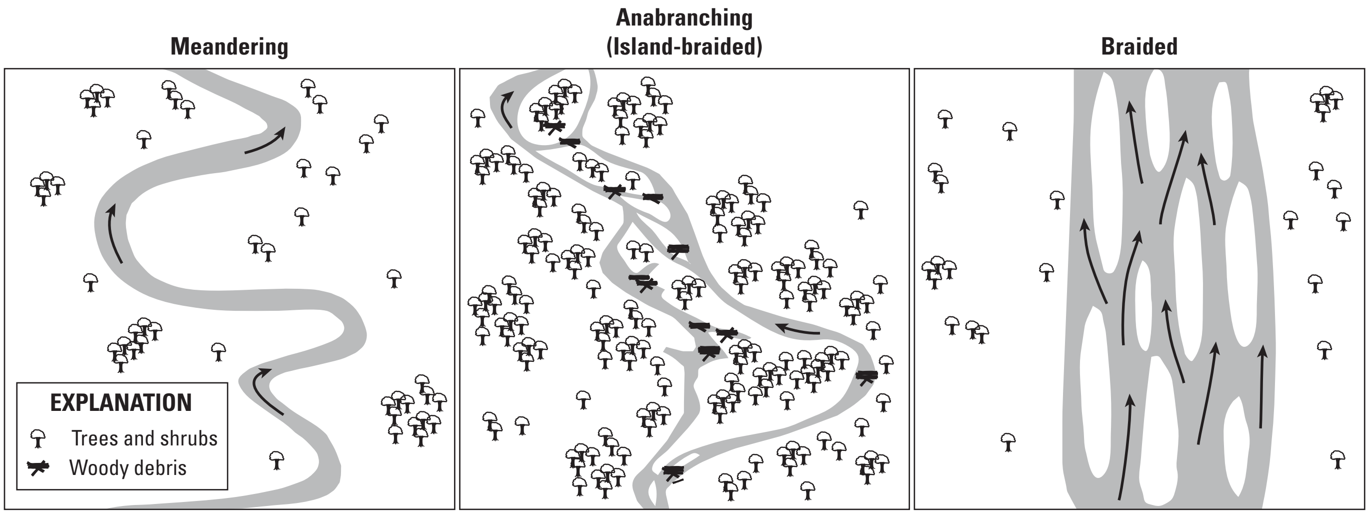


Figure 3.5. Schematic diagrams showing channel planforms for meandering to braided geomorphic types. The lower Elwha River is characterized as anabranching, which has characteristics of meandering and braided systems.

The total number of channels along a cross-section of the lower Elwha River varies between one and three. Large islands between the channels can be many times wider than the individual channels themselves and can persist for decades (Draut and others, 2008, 2011). The lower Elwha River is similar in size, shape, slope, and discharge rates to other anabranching rivers of the western Washington region described by Beechie and others (2006) (fig. 3.6), which suggests that this morphology is not unique for the region. The flood plain and islands of the lower Elwha River are heavily wooded. Initially these landforms are vegetated by alder and cottonwood trees, and older surfaces abound with secondary successional trees including spruce, hemlock, and cedar (for example, Beechie and others, 2006; Kloehn and others, 2008).

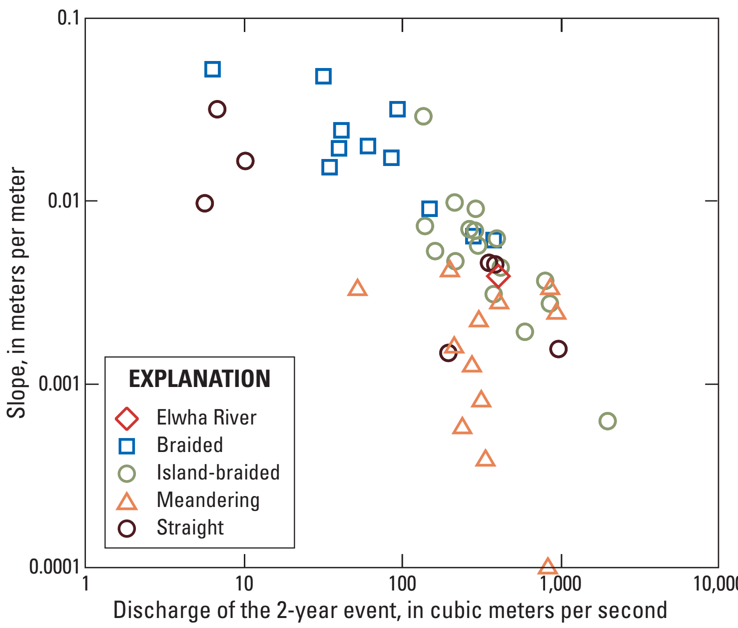


Figure 3.6. Comparison of the channel form of the lower Elwha River to 42 river reaches from other rivers in western Washington. Slope in the average slope of the river section and discharge is calculated as the 2-year peak discharge which represents an approximation of bankfull flow. Adapted from Beechie and others (2006).

Historical Channel Changes in the Lower River

Draut and others (2008, 2011) used 14 sets of sequential aerial photographs obtained between 1939 and 2009 to document historical changes to channel position and morphology in the lower Elwha River. The photographs provided evidence of effects from large floods and human modifications of the flood plain. Changes were greatest along the lowermost 3 km of the river, where absolute changes in the channel position between 1939 and 2009 averaged approximately 160 m and in some places measured as much as 660 m (Draut and others, 2011).

The centerline of the main channel has moved laterally in time due to the erosion of channel banks and the avulsion of new channels (fig. 3.7; Draut and others, 2011). Although new channels can form rapidly in response to high flow events or human modifications, the older "abandoned" channels often appear to be active (unvegetated) for several years after the avulsion, even though they receive little flow.

A few of the channel changes between 1939 and 2009 in the lowest part of the Elwha River are highlighted in figure 3.8. Most notably the two major active channels observed in the 1939 photograph were reduced to only one by 1965 (fig. 3.8). Dike construction in 1950 (Pohl, 1999) led to the abandonment of river flow into the formerly dominant eastern channel, although this eastern channel remained open to coastal waters for many years, as seen in the 1956 photograph (fig. 3.8*B*, see also fig. 3.7*A*). By 1965, the mouth of the abandoned eastern channel was closed (fig. 3.8*C*); this former channel became coastal lakes and wetlands that have served as salmonid rearing habitats or high-flow refugia when hydraulically connected to the river. The 1965 photograph also shows the effects of levee construction on the western side of the river mouth and straightening of the channel sponsored by Clallam County in 1964 (fig. 3.8*C*). This western levee redirected the river toward the north and cut off Dudley Pond from river and tidal flow.

Following construction of the 1964 west levee, the channel began to meander in the lowest 3 km ("C" in fig. 3.7). The rate of lateral channel migration in the final river meander has been measured at several meters per year from topographic surveys and aerial photographs (fig. 3.9). Most channel movement occurs during high flows of winter, and negligible channel change is observed during the summer.

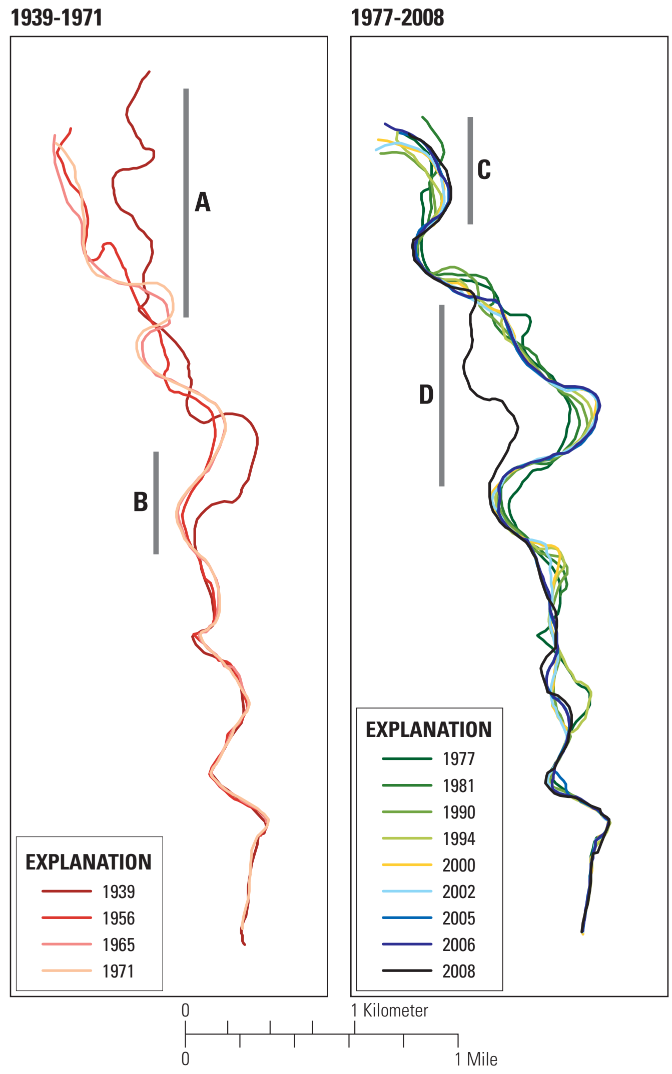


Figure 3.7. Historical changes in the location of the channel midline, Elwha River, Washington, during 1939–1971 and 1977–2008. The dominant channel was mapped in reaches where multiple channels were present (after Draut and others, 2011). Four historical changes to the lower river thalweg are described the text and labeled with *A*–*D*.

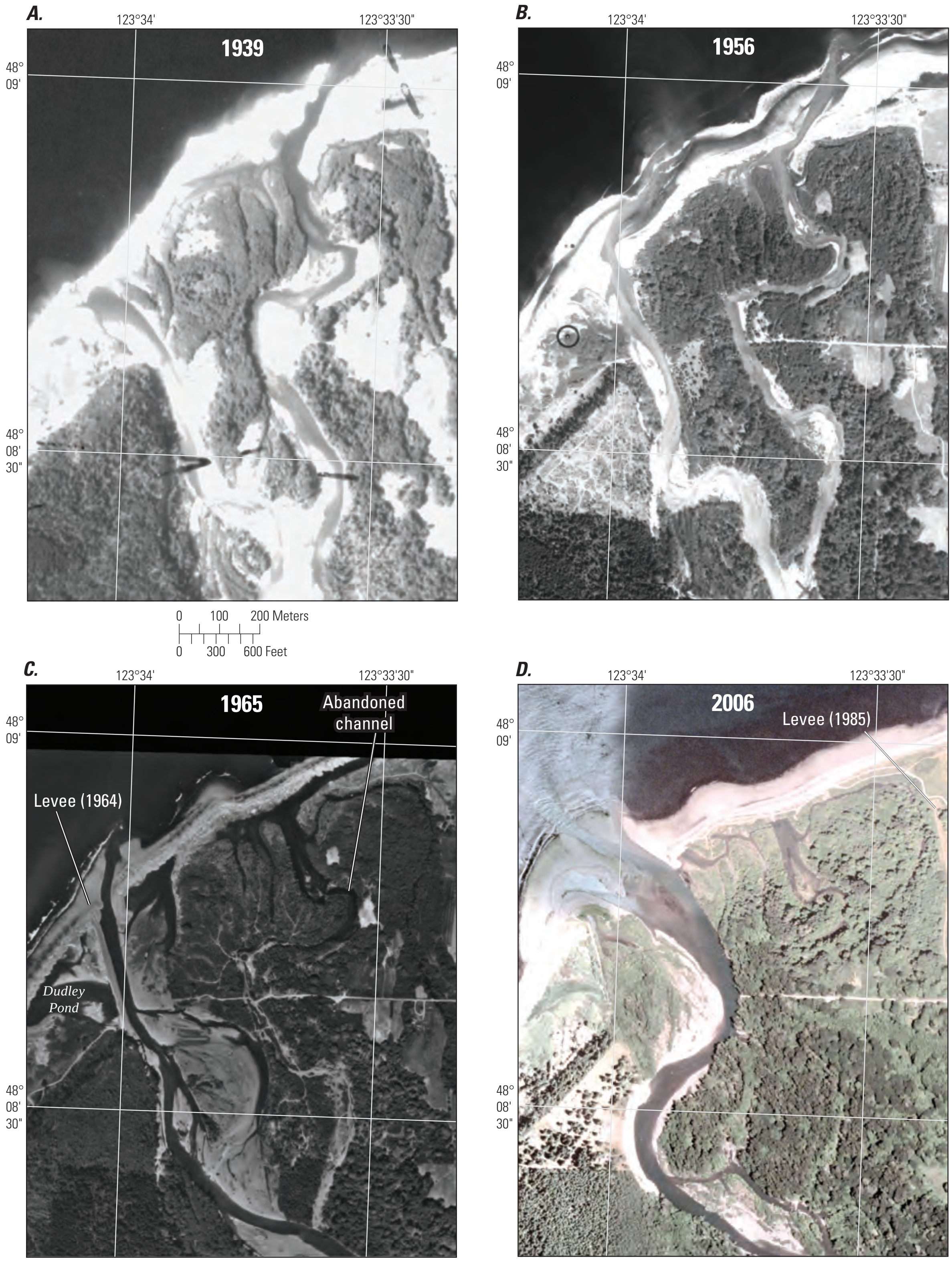


Figure 3.8. Aerial photographs showing effects of human modifications and channel meander and avulsion changes in the mouth of the Elwha River, Washington, between 1939 and 2006. (After Draut and others, 2008.)

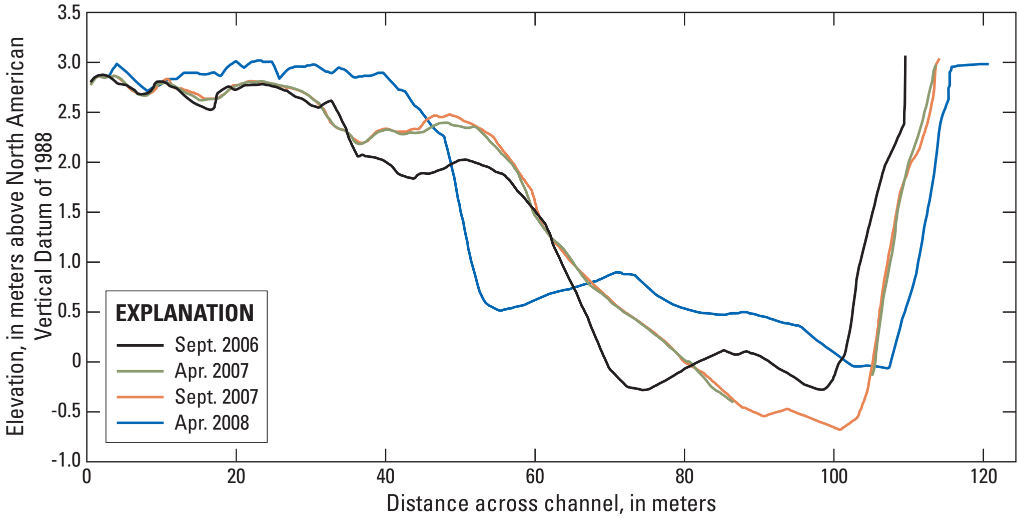


Figure 3.9. Measured channel change on the final meander showing several meters of lateral migration per year of the lower Elwha River, Washington. (After Draut and others, 2010.)

During the winter high flows between 1994 and 2000, an avulsion formed a new westernmost channel in the Elwha River; this new channel has been referred to as the Hunt Road Channel ("*D*" in fig. 3.7). A large flood in December 2007 broadened this new channel, after which most of river discharge flowed down the Hunt Road Channel (fig. 3.10). This 2007 flood also downed hundreds trees that accumulated in woody debris piles throughout this reach (fig. 3.11).

Several other human modifications have been made to the lower Elwha River during the 20th century. Additional modifications include the following:

1. Repeated bulldozing of channel boundaries by private-party landowners between the 1950s and 1980s;

2. Excavation of a north-trending artificial meander cutoff by Clallam County in 1947 to reduce the flood risk east of the natural (eastward migrating) meander ("*B*" in fig. 3.7);

3. Construction of a levee that is set back from the river on its east side by the U.S. Army Corps of Engineers in 1985 (fig. 3.8*D*);

4. Construction of an outfall channel leading from the tribal fish hatchery to the main river channel;

5. Construction of a weir and diversion channels for near the State-run fish hatchery about 4.5 km upstream of the river mouth (Johnson, 1994; Pohl, 1999; Draut and others, 2008); and

6. Restoration of in-stream habitat by the Lower Elwha Klallam Tribe primarily through the placement of large woody debris in channels (see sidebar 3.1).

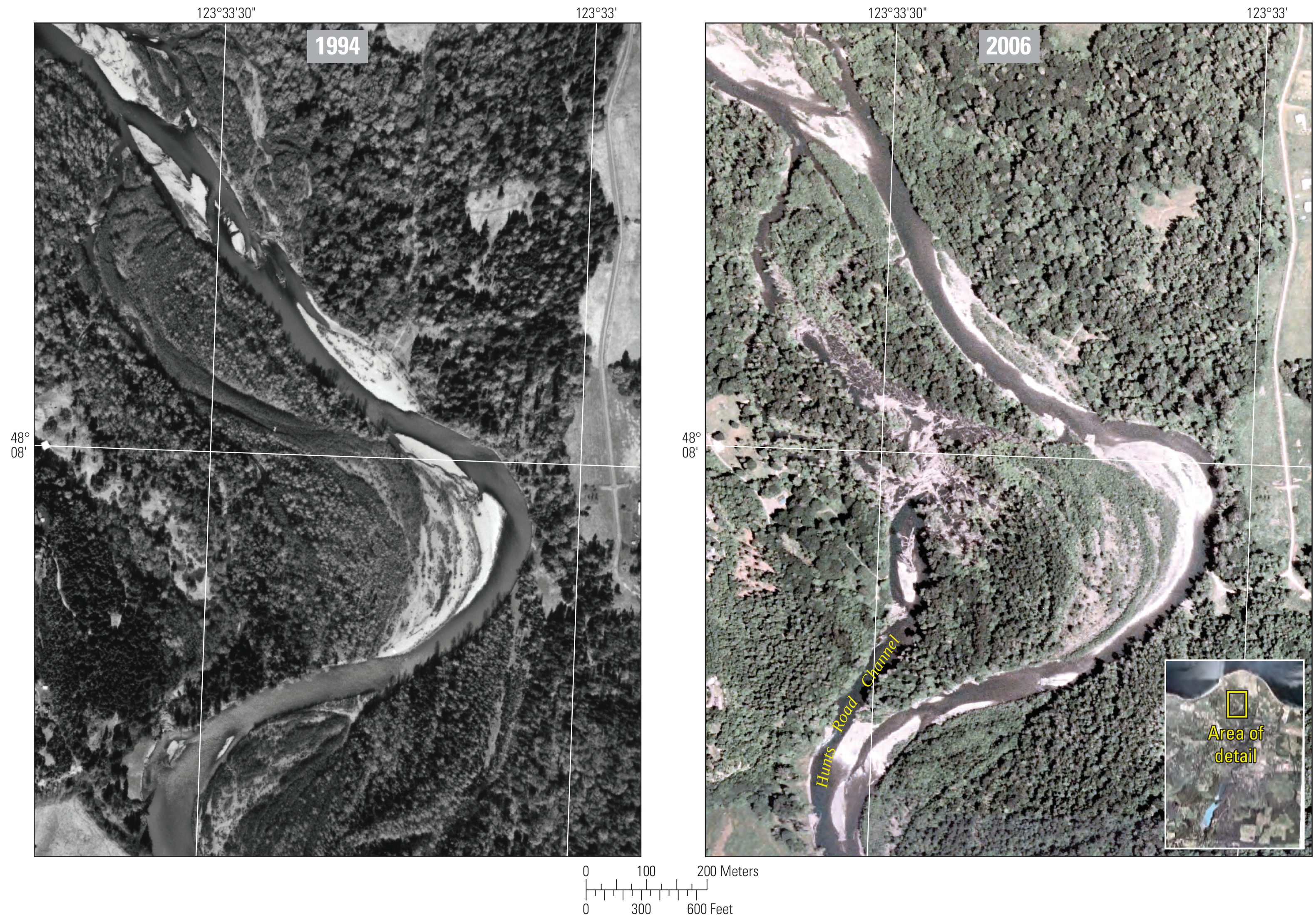


Figure 3.10. Historical aerial photographs showing a major channel avulsion in the lower Elwha River, Washington, 1994 and 2006.

Figure 3.11. Woody debris resulting from channel avulsion in the lower Elwha River, Washington. Flow direction is from the top to the bottom of the photograph. The avulsion point is shown at the top of the photograph. (Photograph taken by Tom Roorda, Northwestern Territories, Inc., April 4, 2007.)

Current Sediment Supply to the Lower River

For almost 100 years, the Elwha River dams have captured all upper-watershed sand and gravel before it could reach the lower river, leading to bed armoring and coarsening in the lower river (Pohl, 2004; Kloehn and others, 2008; Curran and others, 2009; Draut and others, 2011). In the absence of this supply, bank erosion of older fluvial or glacial deposits constitutes the primary supply of coarse sediment to the lower river. We know of no estimate of the amount of this sediment recruited by bank erosion and channel migration, although research on other rivers demonstrates this process can be important for sediment supply (Collins and Dunne, 1989; Martin and Church, 1995; Wallick and others, 2009).

Lateral erosion of the river into a 38-m-high bluff of glacial till immediately upstream of the river mouth may contribute an important input of bed material (fig. 3.12). Using repeat aerial photography (fig. 3.8), Draut and others (2008) showed that this bluff retreated about 70 m between 1965 and 2006. Extrapolated to the full 310 m bluff length, this retreat would have contributed about 820,000 m^3 during these 41 years, which is equivalent to an annual rate of about 21,000 m^3/yr.

The final potential sources of sediment to the lower watershed are the drainage area and the small tributaries that drain directly into the lower river. Most of this lower catchment area is privately owned and used for wood production, small farms, and open space, and little information is available about the rates and patterns of sediment contributions from this lower landscape.

Figure 3.12. Large bluff immediately upstream of the mouth of the Elwha River, Washington. This bluff has eroded substantially during the past several decades, and it may be a primary source of sediment to the lower river and littoral cell. (Photograph taken by Amy E. Draut, U.S. Geological Survey, September 8, 2008.)

Sidebar 3.1 Flood Plain Restoration in the Lower Elwha River

Michael L. McHenry, Habitat Manager, Lower Elwha Klallam Tribe

The construction of the Elwha and Glines Canyon Dams in the Elwha River in the early 20th century prevented fish migration and captured sediment and wood that previously helped form and maintain the ecosystem of the river flood plain. Downstream of the dams, these effects were magnified by flood plain logging, removal of logjams, and channelization, activities that were common practice in western Washington rivers. As a result of the cumulative effects of dam construction and historical flood plain management practices, habitat conditions for anadromous salmon in the lower river are degraded. For instance, the lower river has incised into its river bed, lost suitable-sized spawning gravels, increased in temperature, lost connectivity with parts of its flood plain, and lost habitat complexity.

Although dam removal will assist with correcting many of the ecological problems besetting the Elwha River and its imperiled salmon populations, it will not immediately correct the degraded flood plain habitats of the lower river. In an effort to improve flood plain habitats in the Elwha River prior to dam removals, the Lower Elwha Klallam Tribe has been conducting numerous flood plain restoration actions, which include removing unneeded dikes, installing dozens of engineered log jams throughout the channel and flood plain, correcting fish migration barriers, planting thousands of native trees and shrubs, and controlling the spread of exotic vegetation (figs. S3.1.1 and S3.1.2). The restoration efforts are designed to maximize potential habitat areas that can accommodate all the sediment that will be released by dam removal. For example, dike removals increase flood plain capacity and allow the river to occupy areas currently inaccessible to habitat forming processes.

Engineered log jams (ELJs) are designed to mimic the natural architecture of large, stable logjams that are critical to anastomosing of forested island channel morphology commonly found in large western Washington rivers. ELJs at the river reach level may be used to promote island formation, to activate side channels (in conjunction with dike removals), and reduce flow velocities. ELJs also promote habitats that are used heavily by adult and juvenile salmonids of all species. ELJs typically develop large, complex scour pools that interact with groundwater to provide the cool temperature refugia preferred by salmon. Monitoring has shown that these habitats contain 2–6 times as many juvenile salmon as similar habitats without complex wood structures.

Efforts to revegetate the flood plain have focused on previously disturbed areas in the flood plain that were either open or lightly vegetated. Elevation models and existing tree height data obtained by lidar were used to identify areas for planting a mixture of native coniferous and deciduous trees. More than 20,000 trees and shrubs have been planted in this effort. Concurrently, the tribe has been aggressively controlling non-native exotic vegetation that has invaded the flood plain.

Figure S3.1.1. Newly constructed engineered log jam on the Elwha River, Washington during winter flooding 2009. This structure is one of 50 that the Lower Elwha Klallam Tribe has installed in the lower Elwha River prior to dam removal.

Figure S3.1.2. Newly activated flood plain overflow channel during the first winter following removal of a 1,500 foot flood plain dike on the Elwha River, Washington, 2009.

Coastal Morphology

The Elwha River delta (fig. 3.13) is a wave-dominated deltas within the broader Puget Sound region (Shipman, 2008; Warrick and others, 2009a). The morphology of wave-dominated deltas are controlled largely by littoral redistribution of river sediment. Waves at the Elwha River delta arrive from the west, and these waves induce littoral currents and sediment transport toward the east (toward Ediz Hook), as described in Warrick and others (2011, chapter 5, this report).

Glacial processes have carved irregular shoreline shapes into the entire coastline of Puget Sound and deposited the coarse sediments that compose the beaches (Finlayson, 2006; Shipman, 2008). The Elwha River delta is different from inner Puget Sound beaches and deltas, however, because it is subject to waves from the Pacific Ocean. As shown below, these Pacific-derived waves—in addition to sea level and fluvial histories—are important characteristics that define the Elwha River coastal morphology. The Elwha River delta is lobate in shape and its beaches are composed largely of gravel, with some sand. These beaches have been eroding quickly during the past 80 years, a phenomenon which appears to be related to the loss of sediment supply caused by the Elwha River dams. The rapid erosion of the beaches is causing a distinct erosional morphology on the shoreline of the Elwha River delta.

Figure 3.13. The mouth and delta shoreline of the Elwha River, Washington. (Photograph taken by Ian M. Miller, University of California, Santa Cruz, February 12, 2005, at 11:24 Pacific standard time)

Sea-Level History

The shoreline of the Elwha River delta has moved more than 100 m in elevation over glacial time as a result of the rises and falls in the global (or "eustatic") sea level that have resulted from varying quantities of ice storage, fluctuations in ocean water temperature, and the rising and falling of the Earth's crust as it shifts under the weight of glacial ice sheets. With the retreat of the Juan de Fuca Lobe of the continental ice sheet approximately 14,000 years BP, the Strait of Juan de Fuca filled with seawater. Mosher and Hewitt (2004) detailed the subsequent sea-level history for the Strait of Juan de Fuca, and this record is compared graphically to the global eustatic sea level shown in figure 3.14. At the time of the ice sheet retreat, the eustatic sea level was approximately 70 m below current (2011) levels because of the tremendous amount of water trapped in the world's ice sheets (fig. 3.14). In contrast, the relative sea level in the Strait of Juan de Fuca at this same time was approximately 50–75 m higher than the current levels, as evidenced by high altitude carbon-14 (^{14}C) dated marine sediments (Matthews and others, 1970; Waitt and Thorson, 1983; Dethier and others, 1995; Huntley and others, 2001). The approximately 150 m difference between the

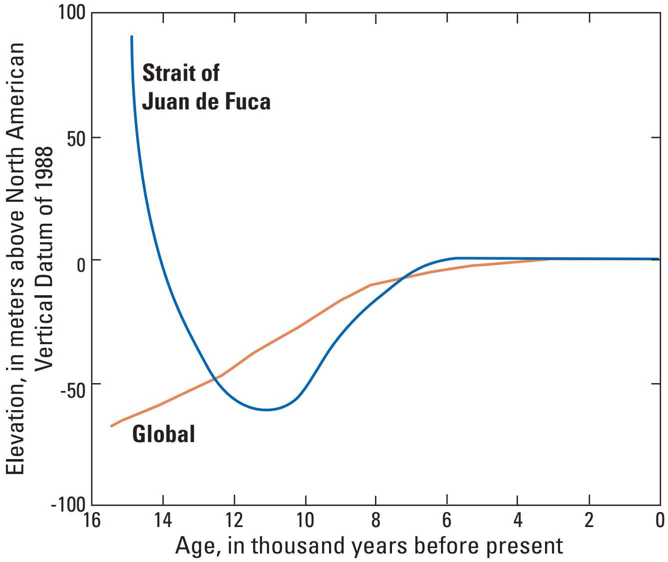


Figure 3.14. History of relative sea level for the Strait of Juan de Fuca off the coast of Washington (after Mosher and Hewitt, 2004). The global eustatic sea level curve (Global) is based on Peltier (2002).

global eustatic and the regional Strait of Juan de Fuca sea levels resulted from the depression of the Earth's crust under the thick ice sheet that covered the region.

The high-stand of relative sea level within the Strait of Juan de Fuca was short-lived, however, because the crust rebounded upward at rates much faster than the eustatic changes in sea level. This rebound induced rapid falling of relative sea level in the Strait, which reached an absolute minimum of approximately -55 m at about 10,500 years BP (Mosher and Hewitt, 2004; fig. 3.14). Following this minimum in sea level, changes in relative sea level have resulted from the combined effects of eustatic and glacial-isostatic changes. The eustatic changes brought relative sea level approximately to its present height at ca. 6,000 years BP, after which relative sea level has been relatively constant compared to the fluctuations between 6,000 and 14,000 years BP (fig. 3.14). Although the eustatic sea level has increased about 1.7 mm/yr over the past century (Intergovernmental Panel on Climate Change, 2007), the relative sea level changes in the region of the Elwha River have been negligible because of the continued uplifting of the landscape. This differs from the inner Puget Sound where relative sea level has been rising at about 2.0 mm/yr, and the outer coast where relative sea level has been falling at about 1.6 mm/yr during the past century (National Oceanic and Atmospheric Administration stations 9447130, Seattle and 9443090, Neah Bay).

Coastal Evolution Since the Pleistocene Time

The dynamic sea level following the most recent ice sheet retreat left clear imprints on the landscape of the Strait of Juan de Fuca. The transgression (or rising) of sea level in the Elwha River delta region started about 10,500 years BP and was followed by relative sea level stability since about 6,000 years BP. This transgression left the most defining physical features on the shoreline and submarine (or underwater) landscape. Coastal bathymetric surveys have identified a broad terrace extending several kilometers offshore of the present day Elwha River delta (fig. 3.15). This feature is an ancient delta that was built during the lowest sea levels, and three drowned littoral spits appear along its eastern edge (I–III in fig. 3.15). Such spits are built out at the termination of littoral sediment cells, and the presence and location of these drowned spits suggest that the shoreline position of the Elwha River delta was 3–5 km offshore of the present shoreline during lower sea levels (Galster and Schwartz, 1990; Mosher and Hewitt, 2004). Furthermore, these drowned spits step up in elevation from -25 m for the farthest offshore spit (I), to -8 and -6 m for spits II and III, respectively (Mosher and Hewitt, 2004). The consistent orientation of the drowned spits and Ediz Hook (spit labeled IV in fig. 3.15) indicates that transport of sediment along this shoreline has consistently been toward the east, which is in the same direction of littoral currents from the waves of the Pacific Ocean (see Warrick and others, 2011, chapter 5, this report). One hypothesis is that as relative sea level rose between approximately 10,500 and 6,000 years BP, the delta shoreline was eroded inland at an average rate of about 0.7 m/yr (Galster and Schwartz, 1990). During at least three episodes of this transgression, the terminal spit was abandoned and a new spit was formed farther upslope.

The present-day Elwha River delta has a lobate planform centered on the river flood plain (fig. 3.15). This geomorphic shape suggests that either the presence of the flood plain, or discharge of sediment from the river, has defined this sinuous morphology of the delta. One leading hypothesis holds that this sinuous form did not result solely from river depositional patterns but rather from a combination of sediment supply from the river that kept the position of the delta's shoreline relatively steady during the past 6,000 years, coupled with more than 1 km of erosion of the adjacent coastal bluffs on both sides of the delta during the past 6,000 years (equivalent to an average erosion rate of 0.2 m/yr; Galster and Schwartz, 1990). The sediment supplied by the combination of erosion of these coastal bluffs and sediment discharge from the river built the 5 km spit, Ediz Hook, in this strongly unidirectional eastward littoral cell (Galster and Schwartz, 1990). The beaches of the Elwha littoral cell are composed of mixed sediment grain sizes (sand to boulder), which is generally consistent with many of the Strait of Juan de Fuca beaches, owing to the abundance of coarse sediment in the glacial landforms of the region (Shipman, 2008).

Historical Shoreline Changes

The 20th century oral histories from Lower Klallam Elwha Tribal members describe hundreds of meters of land lost from coastal erosion of the beach east of the river mouth ("east beach") and a fundamental shift of the low-tide area of the beach from being favorable shellfish habitat to a cobble substrate poorly suited for shellfish habitat (Reavey, 2007; Warrick and others, 2009a). The changes described by the Lower Elwha Klallam Tribe are consistent with aerial-photograph-derived erosion rates (fig. 3.16), which have accelerated significantly from 0.8 to 1.4 m/yr during 1939–90 and 1990–2006, respectively (Warrick and others, 2009a). Further, this oral history is consistent with the current differences between the west beach, which has not changed significantly in position and is mostly sand and gravel, and the east beach, which is eroding and has a cobble low-tide terrace (fig. 3.17).

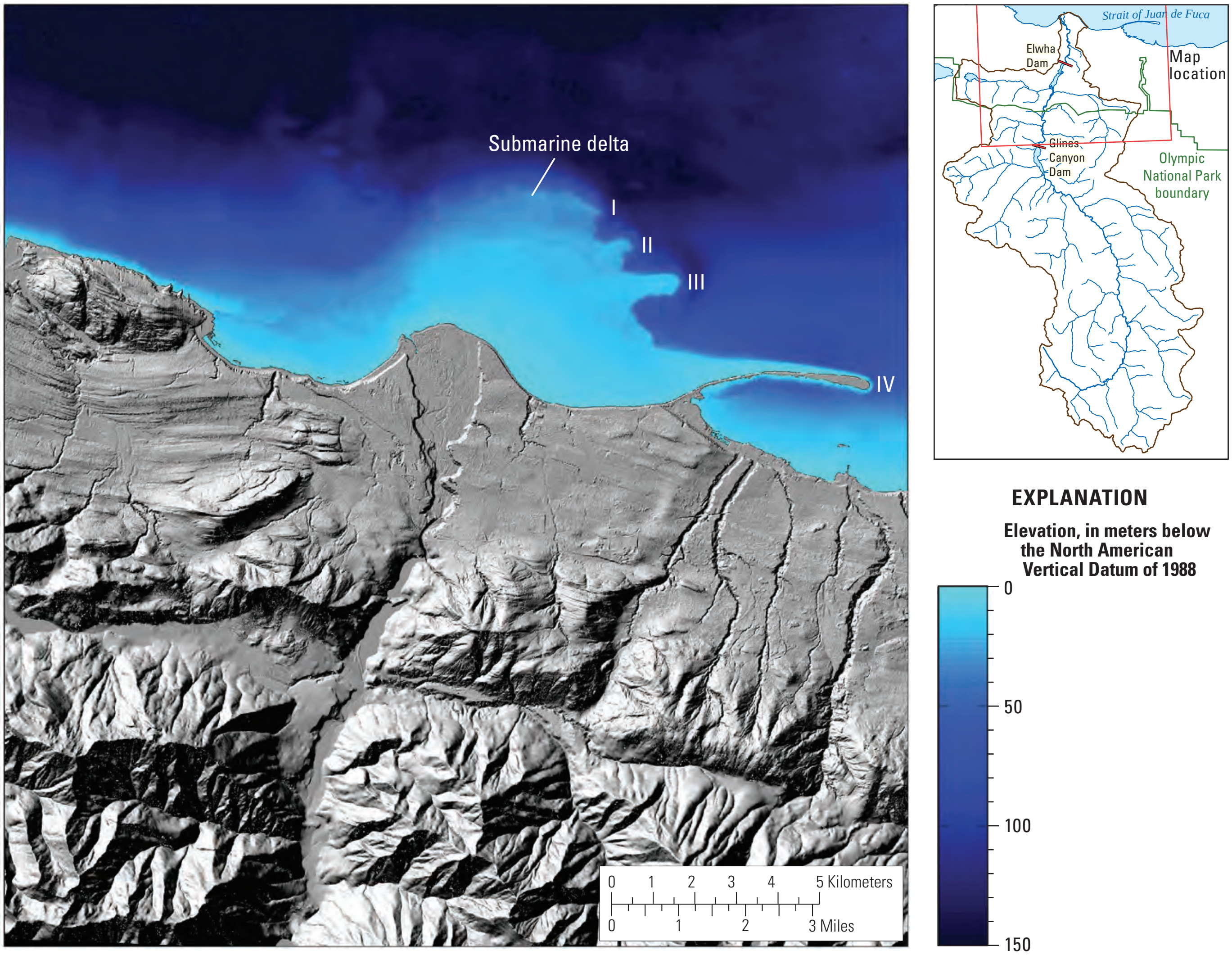
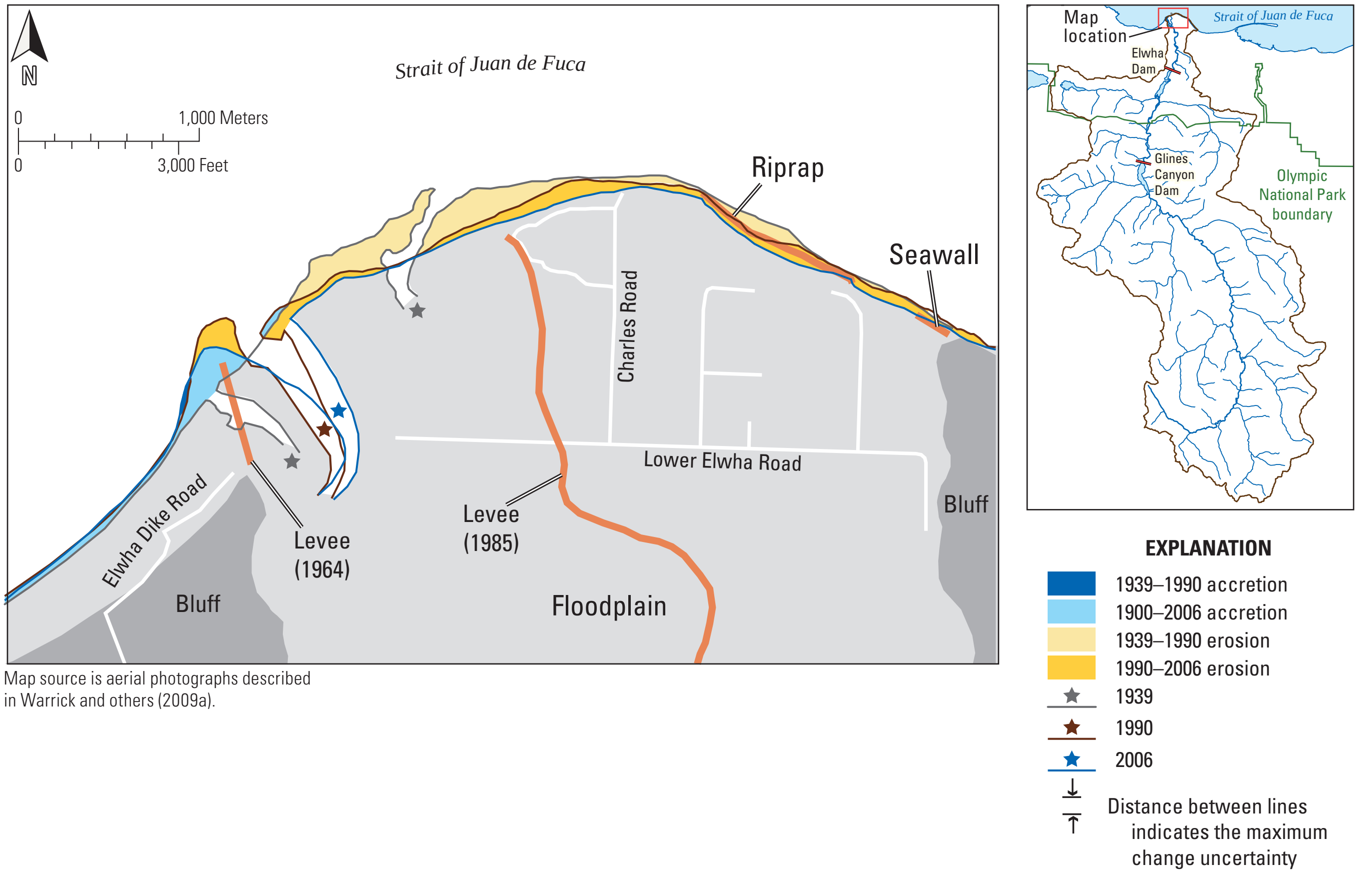


Figure 3.15. Coastal bathymetry of the broad submarine delta plain, three drowned spits (I–III), and the Ediz Hook (IV), which represents the terminus of the current littoral cell of the Elwha River delta, Washington. Source data are a merger of Finlayson (2005), Cochrane and others (2008), and Warrick and others (2010).



Map source is aerial photographs described in Warrick and others (2009a).

Figure 3.16. Shoreline change of the Elwha River delta, Washington, between 1939 and 2006. (After Warrick and others, 2009a.)

Figure 3.17. Differences between the beach on the west and east sides of the current river mouth of the Elwha River delta, Washington.

Historical changes to the position of the river and the shoreline also are largely responsible for the formation and destruction of the coastal wetlands near the river mouth. The series of aerial photographs, shown in figure 3.8, shows how abandoned river channels become coastal lakes and estuarine channels once the river changes course. Erosion of the shoreline commonly pushes some sand and gravel inland over the beach berm, into sediment lobes termed "overwash deposits" (fig. 3.18). These overwash deposits decrease the size of coastal lakes and estuarine channels by filling the water body with sediment (fig. 3.18). Thus, coastal erosion, which has been considerable since the dams have been in place, has resulted in wetland loss around the Elwha River delta.

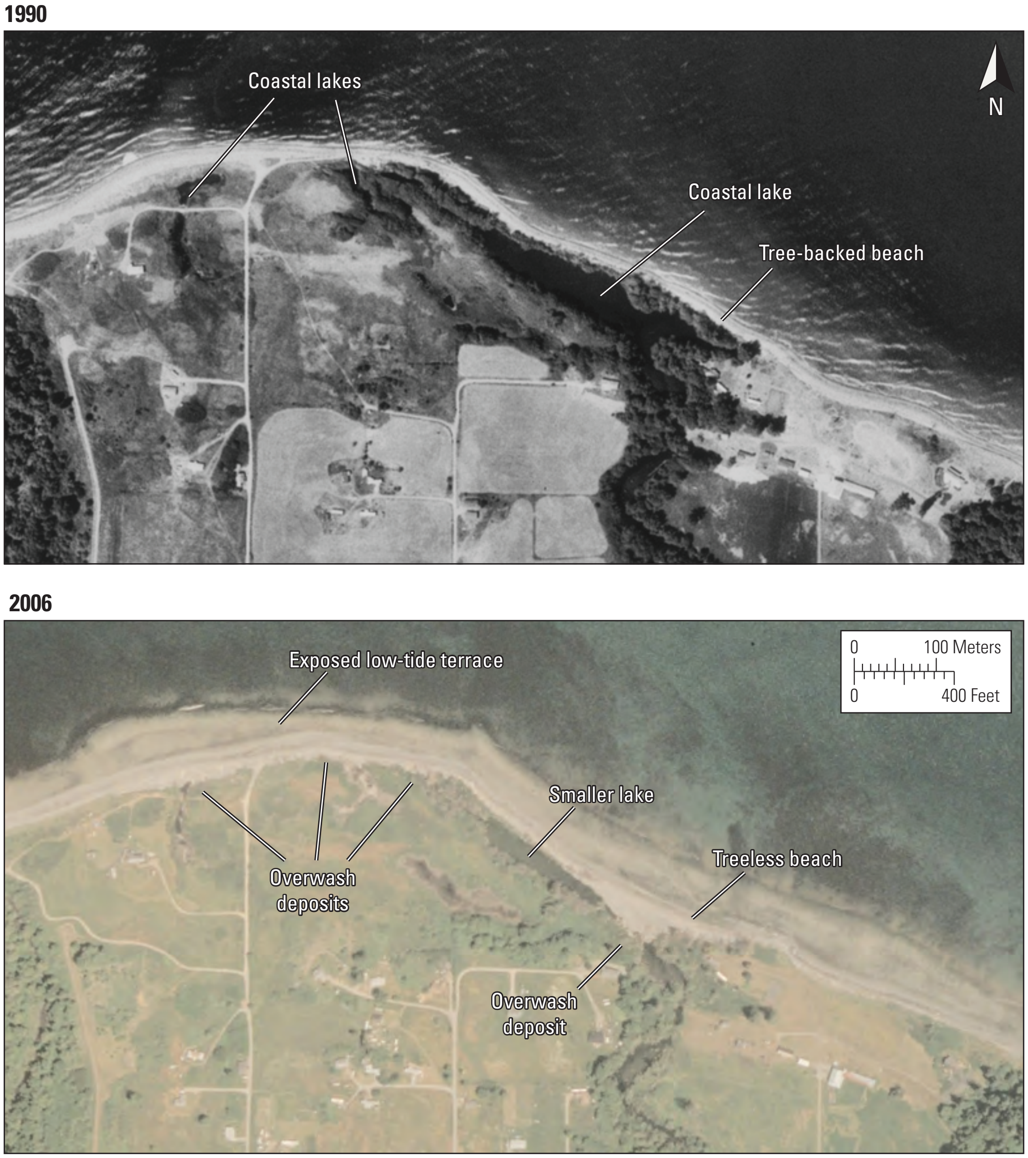


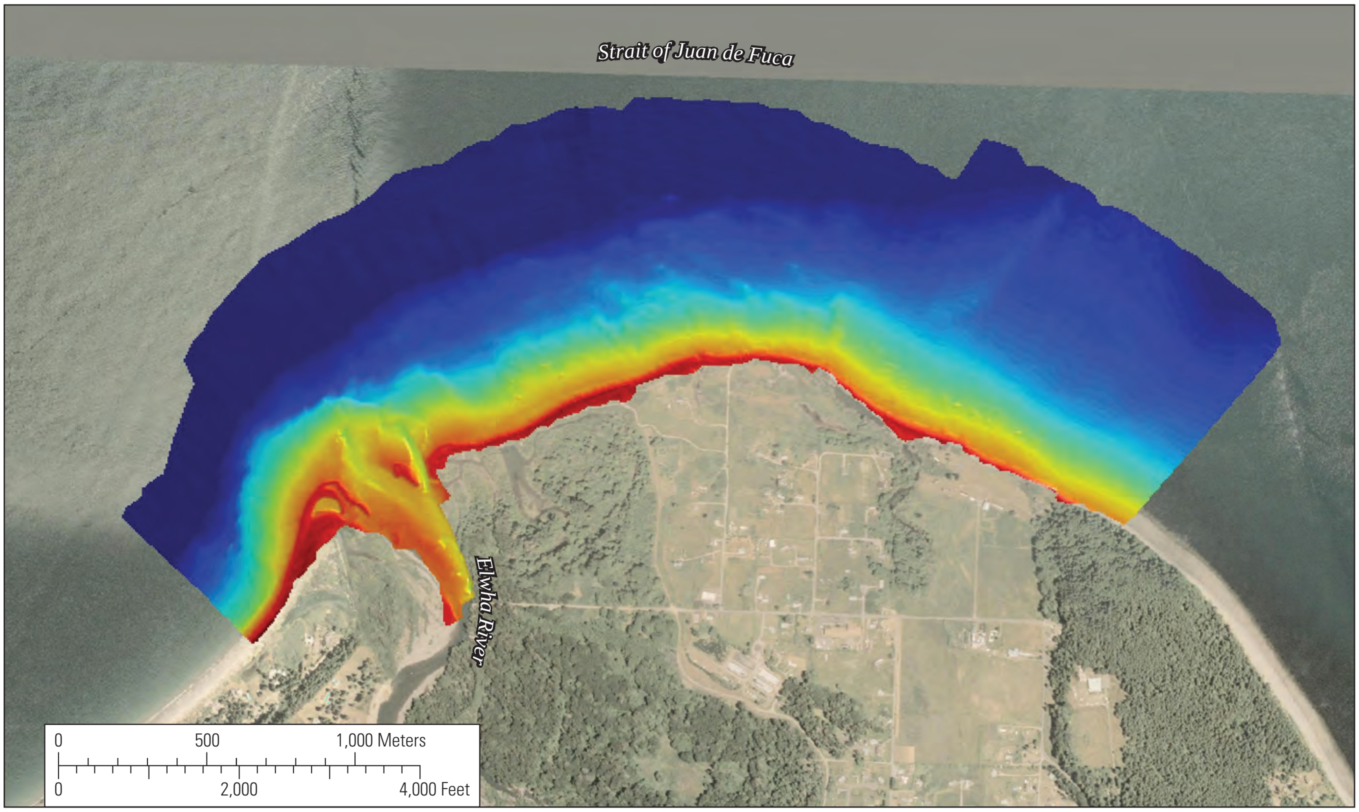
Figure 3.18. Aerial photographs of historical changes from shoreline erosion and overwash deposition resulting in coastal wetland loss in the eastern Elwha River delta, Washington. (After Warrick and others, 2009a).

Recent Coastal Monitoring

In response to the ongoing retreat of the Elwha River delta shoreline and because of the potential for restoration of this shoreline following dam removal, the Lower Elwha Klallam Tribe, the U.S. Geological Survey (USGS), and the Washington State Department of Ecology have collaborated on a coastal monitoring program since 2004. One of the primary tools used in this monitoring is high-resolution topographic surveying with real time kinematic global positioning systems (RTK-GPS). These RTK-GPS surveys require several scientific technicians to hike GPS receivers across the beach to measure topography and to drive small vessels with receivers and echo sounders through the coastal waters to measure seafloor bathymetry (fig. 3.19). The use of these surveying techniques results in high-resolution topographic maps of the study area (fig. 3.20) that can be used to detect change (Warrick and others, 2010).

These surveys show that the beaches of the Elwha River delta can be divided into three sections: (1) the steep, cuspate beach west of the river mouth (“west beach”); (2) the dynamic river mouth; and (3) the erosional beach east of the river mouth that has a distinct cobble low-tide terrace (“east beach”). The cuspate, west beach has changed little in position over the recent monitoring period (figs. 3.21 and 3.22), which is consistent with the historical analysis of this beach from aerial photographs discussed above. The east beach has eroded rapidly with a shoreward movement of the beach profile (figs. 3.21 and 3.22). The low-tide terrace portion (below approximately 1 m elevation NAVD 88) of the east beach is much more stable (fig. 3.21). The net effect of shoreline erosion of the east beach is landward movement of the beach face and broadening of the low-tide terrace. The cobble-faced low-tide terrace of the east beach is an armored portion of the beach resulting from beach erosion (Warrick and others, 2009a).

Figure 3.19. Topographic and bathymetric surveys using real time kinematic global positioning system (RTK-GPS) from (*A*) hiked backpack systems and (*B*) personal watercraft with integrated echo sounders along the of the Elwha delta coast, Washington.



Aerial photography from USDA - NAIP, 2006. Bathymetric data from Warrick and others (2010).

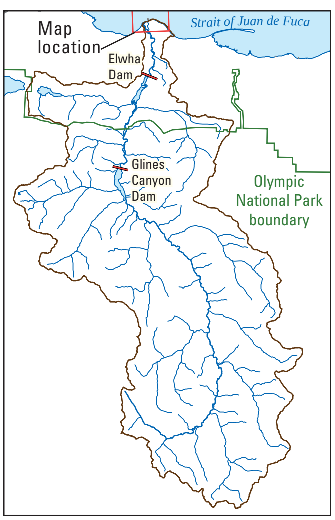


Figure 3.20. Topography and bathymetry from real time kinematic global positioning system (RTK-GPS) surveys of the Elwha River delta, Washington, during summer 2008. (Aerial photography from the U.S. Department of Agriculture National Agriculture Imagery Program (NAIP) survey of 2006.)

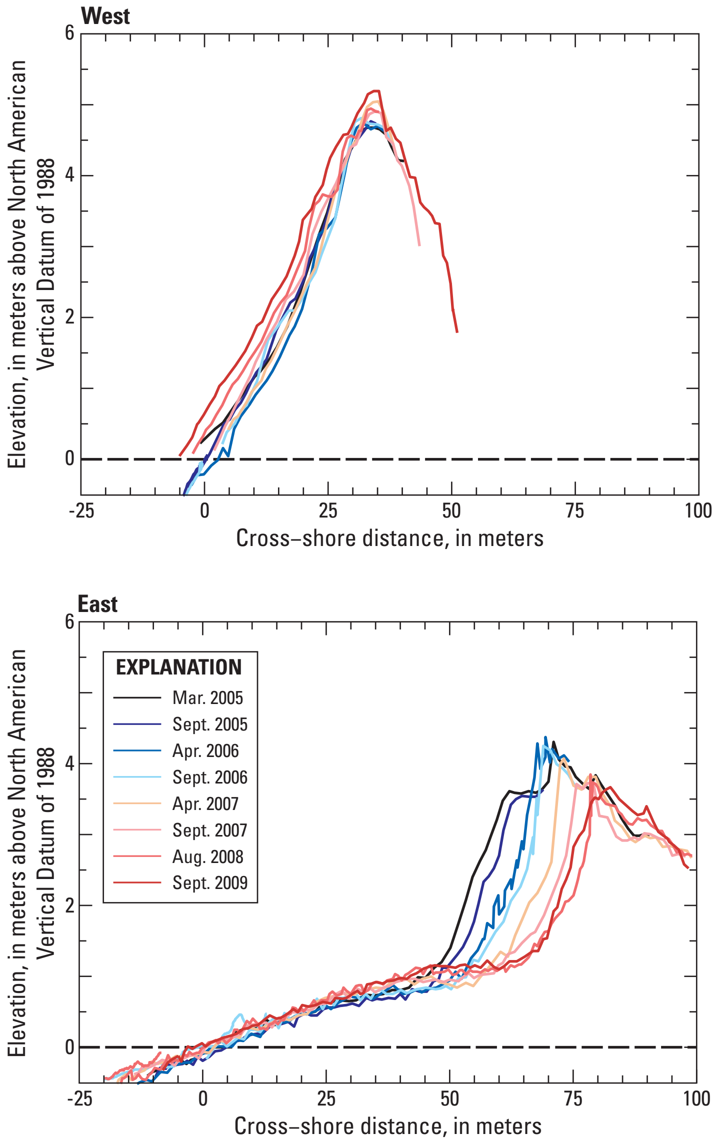


Figure 3.21. Example beach topographic profiles for the west and east beaches of the Elwha River delta, Washington, from surveys by the U.S. Geological Survey. The west beach has been relatively stable over time, whereas the east beach has eroded substantially.

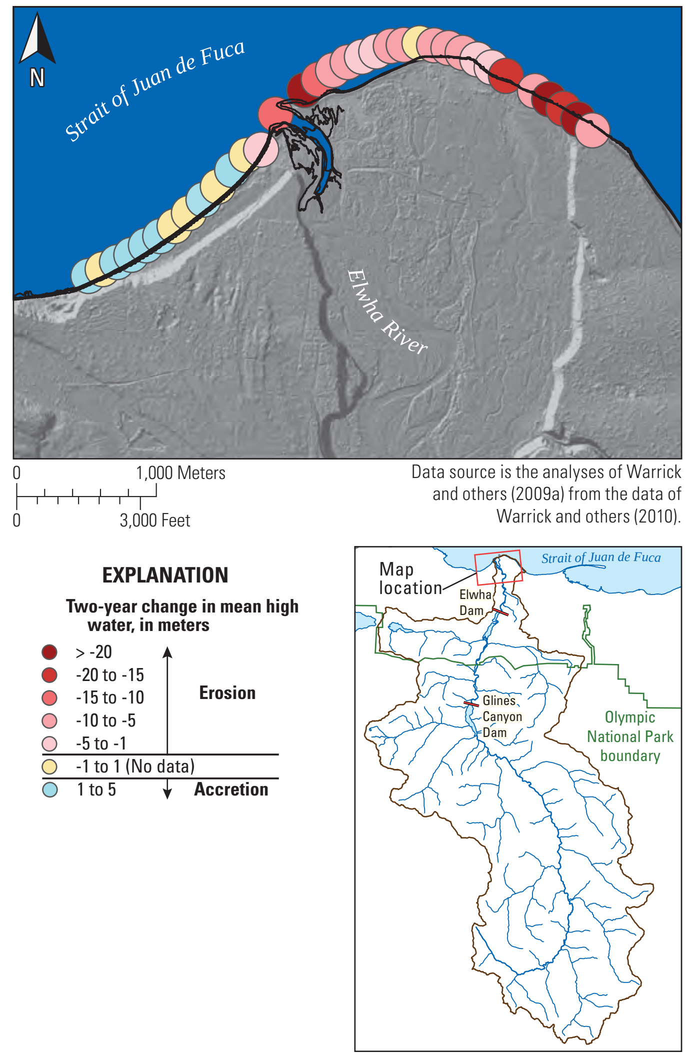


Figure 3.22. Shoreline change along the beaches of the Elwha River delta, Washington, during 2004–06 from real time kinematic global positioning system (RTK-GPS) surveys. (After Warrick and others, 2009a.)

The most dynamic part of the Elwha River delta shoreline is the river mouth, which is influenced by tidal flow, river discharge, and waves that move and reorganize sediment around the river mouth. Evidence of these changes comes from 3 years (2007–09) of topographic surveys in and around the river mouth (fig. 3.23). It is important to note that these surveys suggest that the river mouth region gained significant volumes of sediment in the time between the surveys (approximately 34,000 m^3 in 2007–08 and approximately 21,000 m^3 in 2008–09), even though the east beach underwent net erosion during the same time. These results suggest that the river is still contributing significant amounts of sediment to the coastal system (a conclusion which is consistent with the observations of Draut and others, 2011) and that much of the fluvial sediment in the delta is emplaced in the bar of the river mouth.

Sidebar 3.2 Mapping Grain Size with Digital Photos

Jonathan A. Warrick

Removal of the Elwha River dams will release large volumes of sediment into the lower Elwha River and the Strait of Juan de Fuca. We expect this flood of sediment to modify coastal landforms by accretion and to change fluvial and coastal habitat substrate from coarse to fine sediment. Simultaneous tracking of topographic change and sediment size may allow for the development of models for these changes that could be used for future restoration projects. Traditional methods for grain size analysis – including pebble counts and physical sampling – are labor intensive and costly. The use of digital photography alleviates many of these problems by providing a rapid sampling of the land or seafloor surface (Adams, 1979). The USGS has been developing and using digital photographic methods for the Elwha River study area since 2004 to assist in tracking the changes induced by dam removal.

To sample grain size on the beach or a river bar, photographs were taken at regular intervals across these landforms. Photographs must be taken orthogonally to the ground surface to limit distortion of surface features. Traditionally, the calculation of grain size parameters for each photograph was obtained from manual sizing and counts from each photograph, which required 1–2 hours of analysis time for each photograph. New methods suggested by Rubin (2004) have significantly shortened analysis time owing to an automated calculation of grain size based on spatial patterns in each photograph. These “cobble cam” techniques were applied to the Elwha River flood plain and coastal landforms by Warrick and others (2009b) with excellent results. Photographic sampling from Elwha River landforms show a broad diversity in grain size and sorting (fig. S3.2.1). Comparisons of the automated analyses with physical samples of the substrate show that errors in the analyses are only 14 percent, which are more than adequate to characterize the differences prevalent in these landforms. There are significant differences in the grain sizes across river bars, beach cusps, and the low-tide terrace to foreshore transition (see fig. S3.2.2). The successful development and implementation of these methods should allow for the quantification of changes throughout the Elwha River landforms as they occur in the future.

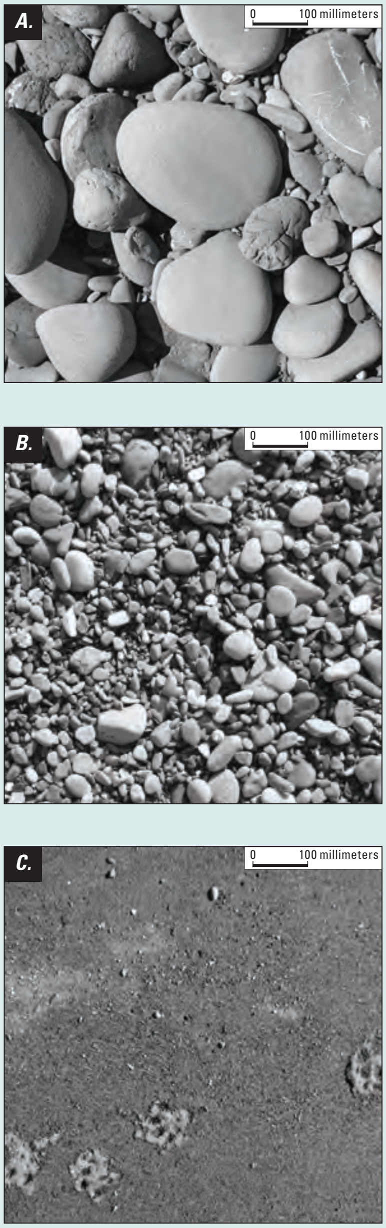


Figure S3.2.1 Examples of a range in grain size from (*A*) cobble (*C*) sand of the Elwha River delta beach, Washington. (After Warrick and others, 2009b).

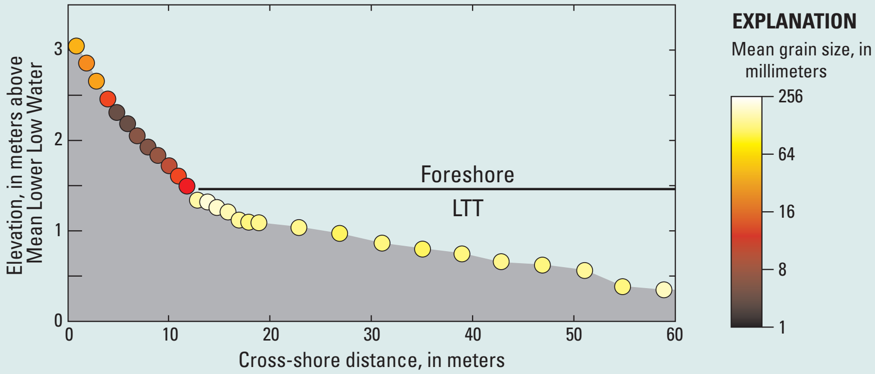


Figure S3.2.2 Example results of the mean sediment grain size from Cobble Cam photographic analyses across the east beach of the Elwha River delta (after Warrick and others, 2009b). A distinct grain size change is apparent between the foreshore and the low-tide terrace (LTT).

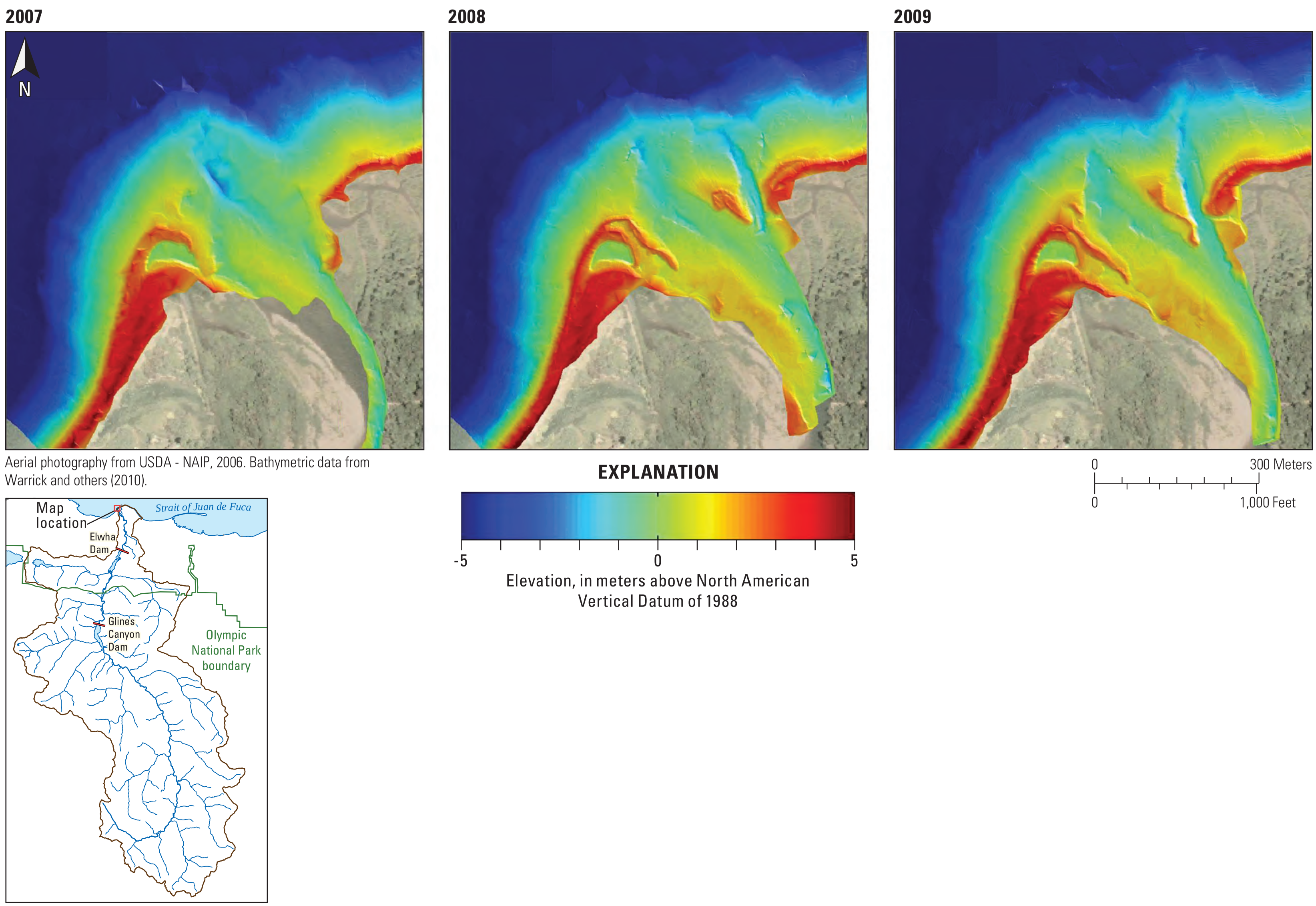


Figure 3.23. Topographic and bathymetric survey maps showing the evolution and growth of the river mouth bar of the Elwha River during 2007–09. All three surveys were conducted in the late summer (August–September) each year.

Summary

The Elwha River lies at the intersection of uplifting land and an eroding coast. In the past, movement of sand and gravel from the steep, eroding glaciated mountains provided the flood plain with sediment that formed spawning habitats for fish and provided a material to buffer the shoreline from energetic waves. Humans have altered this balance by trapping sediment behind dams and by modifying the river channel and shoreline, actions which have altered the position and form of the river channel, reduced sediment and woody debris fluxes through the lower river, and caused significant increases in coastal erosion of the beach east of the river mouth. Consequently, coastal wetlands also are shrinking around the Elwha River delta.

Restoration activities, including the placement of engineered log jams in the river channel, removal of dikes and other flood plain alterations, and the removal of two large dams, should further modify the geomorphology of this river and its coast, with the expectation of restoring some of the lost ecosystem functions. Consistent, long-term monitoring of these systems is needed to quantify the effectiveness of these restoration efforts.

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Chapter 4

Baseline Hydrologic Studies in the Lower Elwha River Prior to Dam Removal

By Christopher S. Magirl, Christopher A. Curran, Rich W. Sheibley, Jonathan A. Warrick, Jonathan A. Czuba, Christiana R. Czuba, Andrew S. Gendaszek, Patrick B. Shafroth, Jeffrey J. Duda, and James R. Foreman

Abstract

After the removal of two large, long-standing dams on the Elwha River, Washington, the additional load of sediment and wood is expected to affect the hydrology of the lower river, its estuary, and the alluvial aquifer underlying the surrounding flood plain. To better understand the surface-water and groundwater characteristics of the river and estuary before dam removal, several hydrologic data sets were collected and analyzed. An experiment using a dye tracer characterized transient storage, and it was determined that the low-flow channel of the lower Elwha River was relatively simple; 1–6 percent of the median travel time of dye was attributed to transient-storage processes. Water data from monitoring wells adjacent to the main-stem river indicated a strong hydraulic connectivity between stage in the river and groundwater levels in the flood plain. Analysis of temperature data from the monitoring wells showed that changes in the groundwater temperature responded weeks or months after water temperature changed in the river. A seepage investigation indicated that water from the river was moving into the aquifer (losing reach) between 1.7 and 2.8 kilometers from the river mouth. Surface-water measurements and temperature and salinity data collected throughout the estuary helped to characterize the magnitude and nature of water movement in and out of the estuary. Salinity and stage sensors positioned in the estuarine network showed a strong surface-water connection between the river and estuary waters east of the river. In contrast, there was a weaker connection between the river and estuarine water bodies west of the river.

Introduction

Removal of the Elwha and Glines Canyon Dams on the Elwha River on the Olympic Peninsula of northwest Washington is expected to affect aquatic ecology, sediment transport, and fluvial geomorphology along the river, estuary, and nearshore within the Strait of Juan de Fuca (see Duda and others, 2008; Kloehn and others, 2008; Warrick and others, 2009; Draut and others, 2011; Duda and others, 2011b, chapter 1, this report). Because the Elwha dams currently are operated in run-of-the-river mode, the underlying hydrologic regime in the river will change only slightly after dam removal (Duda and others, 2011b, chapter 1, this report). The fate of the 19 million m^3 of sediment entrapped behind both dams, however, is expected to have significant effects on sediment transport, sediment-size distribution, and river-bed elevation (Czuba and others, 2011, chapter 2, this report; Warrick and others, 2011a, chapter 3, this report), as well as the biological communities downstream of the reservoirs and in marine waters (Duda and others, 2011a, chapter 7, this report).

As a result of changes in areal extent and bed elevation within the river corridor, the dam-removal project also is expected to affect the underlying framework of water movement throughout the lower Elwha River and its estuary. For example, channel complexity (that is, the potential for water to follow multiple flow paths, including recirculation zones, down the river corridor) within the river will likely be affected as sedimentation and the addition of large woody debris from sources upstream of the former dam sites enhance channel-migration potential (Kloehn and others, 2008). Significant channel change occurred near the river mouth as a result of the December 2007 peak-flow event, with an estimated recurrence interval of 40 years (Draut and others, 2011). Changes in sediment-size distribution in the bed of the main-stem river also may alter hydraulic connectivity between the river and groundwater in the surrounding flood plain. In turn, groundwater levels in the lower Elwha River flood plain may rise as sedimentation in the main-stem river elevates the bed and associated water levels (Pacific Groundwater Group, 2005).

Water exchange and transport through the estuary complex also is expected to be affected. The movement of water through the lower Elwha River estuary is governed by channel morphology in the main-stem Elwha River, sedimentology of the underlying alluvium, ocean processes of tides and waves, and temperature- and solute-driven flows between freshwater and saltwater. Conceptually, the estuary is a transient storage facility that captures the outgoing flux of freshwater flowing from the river to the Strait of Juan de Fuca. The specific areas of water exchange, however, are temporally and spatially complex.

The Elwha River mouth and estuary form an important habitat for young salmonids and other fish and wildlife species (Shaffer and others, 2009; Duda and others, 2011a, chapter 7, this report), and adjacent terrestrial habitats support a diverse complex of wetland and riparian vegetation (Shafroth and others, 2011, chapter 8, this report). Within these water bodies, river water, groundwater, and seawater mix providing brackish conditions that change with tides, storms, and seasons. These water-level and salinity dynamics exert a strong influence over biotic communities in river mouth estuaries (Mitsch and Gosselink, 1993; Keddy, 2000; Shafroth and others, 2011, chapter 8, this report). Mixing of freshwater and saltwater offshore of the river mouth within the river plume is described more fully by Warrick and others (2011b), chapter 5, this report, and Warrick and Stevens (2011).

Previous Studies

Few studies have addressed the hydrology of the lower Elwha River. The Pacific Groundwater Group (2005) completed a groundwater model for the lower Elwha River flood plain to predict the response of groundwater elevations after a presumed 0.8-m post-dam-removal aggradation in the channel. The model was calibrated using groundwater data from wells distributed throughout the valley-wide flood plain. A strong hydraulic connection between river stage and the wells adjacent to the river channel was documented under low-flow and peak-flow conditions using groundwater data and model simulation results (Pacific Groundwater Group, 2005). The correlation in groundwater response to river stage decreased with distance from the main-stem river and was negligible along the coastline at the eastern boundary of the flood plain where tidal influences dominated groundwater elevations (Pacific Groundwater Group, 2005). The correlation in groundwater response to river stage decreased with distance from the main-stem river and was negligible along the coastline at the eastern boundary of the flood plain where tidal influences dominated groundwater elevations (Pacific Groundwater Group, 2005). Based on data from aquifer tests, Pacific Groundwater Group (2005) concluded that the hydraulic conductivity of the lower Elwha River aquifer was, on average, 220 meters per day (m/d), and ranged from 60 to 490 m/d.

Detailed velocity measurements were made at five transects near the river mouth (Curran and others, 2008). Similarly, recent monitoring of stream temperature noted an increase in mean annual stream temperature, and a decrease in diurnal temperature fluctuations, in the regulated reach downstream of Glines Canyon Dam compared with the unaltered upstream reach (Connolly and Brenkman, 2008).

This warming of stream temperature downstream of the dams was attributed to heat storage in the Lake Mills reservoir (Wunderlich and others, 1994). However, thermal data characterizing specific temperature distribution of free-flowing and quiescent water bodies located throughout the lower Elwha River corridor are rare.

Purpose and Scope

The U.S. Geological Survey (USGS), in collaboration with the Lower Elwha Klallam Tribe, the National Park Service, and other Federal and state agencies, has worked to characterize the hydrologic framework of the lower Elwha River. This chapter documents research and monitoring efforts that were completed before dam removal and have not been published elsewhere. Much of the data were collected with support from the USGS multi-disciplinary Coastal Habitats in Puget Sound (MD-CHIPS) initiative (Duda and others, 2011b, chapter 1, this report). The research, largely intended to establish baseline hydrologic data sets before dam removal, included efforts to (1) characterize transient storage in the lower river, (2) monitor flood-plain groundwater movement and hydraulic connection with the main-stem river as well as identify areas of exchange between surface water and groundwater, and (3) identify primary hydrologic fluxes and exchanges of groundwater and surface water of river and marine origin in the estuary complex (Duda and others, 2011b, chapter 1, this report). This estuary complex is biologically significant as a rearing habitat for some species of juvenile salmonids, especially Chinook salmon, which is on the Federal endangered species list (Duda and others, 2011a, chapter 7, this report).

Hydrologic Studies in the Lower Elwha River

Numerous measurements and data-monitoring efforts (table 4.1) were completed at several locations along the lower Elwha River flood plain (fig. 4.1) that enabled the characterization of key hydrologic aspects of the lower Elwha River. Transient storage in the river corridor was assessed using a dye tracer injected into the main stem during low flow. Groundwater movement was evaluated using water-elevation data from wells positioned throughout the wider valley flood plain. Surface-water/groundwater exchanges were identified using a seepage investigation (a technique used to identify river reaches that are gaining or losing surface-water flow to the aquifer; see, for example, Ely and others, 2008) and temperature data from the beds of surface-water bodies to identify regions of groundwater upwelling. Finally, a diverse assemblage of surface-water measurements, tidal data, thermal signatures, and specific conductance measurements were used to infer how water flows in and out of the estuary.

Due to its importance to aquatic habitat, considerable hydrologic analysis was focused on the estuary, which includes the network of surface-water bodies and interconnecting channels near the river mouth (fig. 4.2). The water bodies and channels east of the river mouth are collectively referred to as the east estuary. Following the naming convention established by Duda and others (2011b, chapter 1, this report), the east estuary consists of ES1, a water body closest to the river, ES2, a larger water body farther to the east that occupies a former river-mouth channel, and Bosco Creek, a spring-fed water source that originates in the flood plain east of the main-stem river, flows into the eastern edge of ES2, and is the only known fresh surface-water flow into the east estuary (fig. 4.2). The prominent channel that connects ES1 to the main stem Elwha River just upstream of the river mouth is referred to as the ES1 outlet channel. West of the river, two disconnected surface-water bodies comprise a smaller estuary assemblage. Just west of the river, a secondary distributary channel named the West Estuary Channel (WESC) (Duda and others, 2011b, chapter 1, this report) is, at times, hydraulically connected to the main-stem river. Farther west, beyond a flood-control levee is Dudley Pond, a water body disconnected from the main-stem river (fig. 4.2).

Transient Storage in the Lower Elwha River

Current methods of assessing in-stream habitat focus on physical measurements of specific habitat elements (for example, Fitzpatrick and others, 1998), but do not integrate across spatial scales or provide a direct measure of ecosystem function. Transient storage in a stream or river reach refers to any mechanism that slows down the bulk flow of water. Typically this results from in-channel features and exchange with shallow streambed sediments within the hyporheic zone (fig. 4.3). Therefore, there is a greater potential for transient storage in a stream that contains heterogeneous habitat elements at both large scales (for example, pool-riffle sequences and secondary channels) and small scales (heterogeneous substrate and in-channel vegetation).

Table 4.1. Surface-water and groundwater sampling sites along the lower Elwha River, Washington.

[**Abbreviations**: RKm, river kilometer; CT, conductivity and temperature sensors; CTD, conductivty and temperature sensors equipped with transducers to measure water depth; n/a, not applicable]

Decimal latitude	Decimal longitude	Site identifier	Data collection dates	Data collected
48.145808	-123.559370	TStn 1	8-1-09–12-18-09	Thermal
48.146438	-123.560386	TStn 2	8-1-09–12-21-09	Thermal
48.148155	-123.561795	TStn 3	8-1-09–12-11-09	Thermal
48.146982	-123.562848	TStn 4	8-2-09–8-8-09	Thermal
48.148670	-123.560036	TStn 5	8-2-09–8-8-09	Thermal
48.147375	-123.561944	TStn 6	8-2-09–8-8-09	Thermal
48.147515	-123.563160	ES1	7-28-07–3-29-10	CTD
48.147582	-123.561344	ES2	6-3-08–4-1-10	CT/CTD
48.145493	-123.567366	WESC	5-29-09–5-4-10	CT/CTD
48.144483	-123.568045	Dudley Pond	5-14-09–4-19-10	CTD
48.139917	-123.548006	AFN 256	11-9-07–8-27-09	Groundwater and thermal
48.135036	–123.554297	AFN 254	11-9-07–8-27-09	Groundwater and thermal
48.131425	–123.550747	MWGR	11-9-07–8-27-09	Groundwater and thermal
48.138014	-123.548883	AFN 259	11-9-07–8-27-09	Groundwater and thermal
48.147250	-123.563444	Acoustic Doppler velocity meter	6-3-08–9-2-08	Velocity/depth
48.145639	-123.563461	Estuary I/O flow	6-3-08–9-2-08	Discharge
48.144589	-123.556925	Bosco Creek	6-3-08–8-3-08	Discharge
48.119203	-123.553661	State hatchery outflow	9-24-07	Discharge
48.144314	-123.563133	RKm 0.4	9-24-07–9-26-07	Discharge
48.144111	-123.562589	RKm 0.4	9-24-07–9-26-07	Discharge
48.141756	-123.565631	RKm 0.8	9-25-07–9-26-07	Discharge
48.135597	-123.560611	RKm 1.7	9-25-07–9-26-07	Discharge
48.131853	-123.551247	RKm 2.2, old channel	9-26-07	Discharge
48.128217	-123.559203	RKm 2.6	9-24-07	Discharge
48.126889	-123.558706	RKm 2.8	9-24-07, 9-26-07	Discharge
48.121350	-123.554619	RKm 3.5	9-24-07	Discharge
48.118444	-123.554806	RKm 3.8	9-24-07	Discharge
48.114167	-123.553611	RKm 4.4	9-24-07–9-26-07	Discharge
48.142714	-123.563164	Tommy Boy	9-26-07	Discharge
48.120878	-123.555072	Trib 2	9-24-07	Discharge
48.129589	-123.554500	Trib 3	9-26-07	Discharge
48.131167	-123.551561	Trib 4	9-26-07	Discharge
48.136017	-123.558011	Trib 5	9-25-07–9-26-07	Discharge
48.137722	-123.559817	Trib 6	9-25-07–9-26-07	Discharge
48.143425	-123.562283	Tribal hatchery	9-26-07	Discharge
48.120171	-123.554580	Dye deployment	9-29-09	n/a
48.126030	-123.557110	Station 1	9-29-09	Dye concentration
48.131721	-123.557124	Station 2	9-29-09	Dye concentration
48.141685	-123.565316	Station 3	9-29-09	Dye concentration

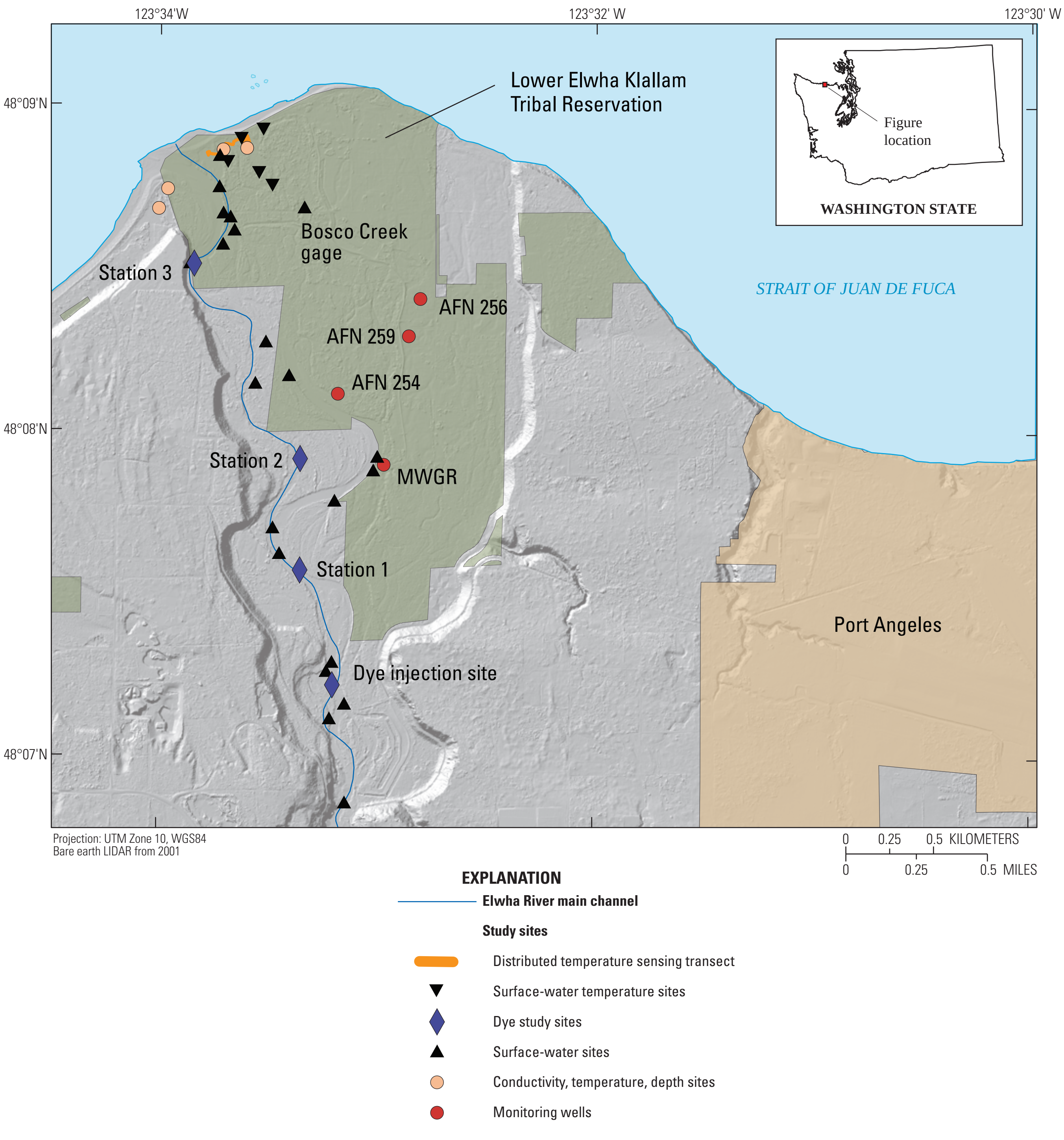

Figure 4.1. Locations of hydrologic monitoring sites, lower Elwha River and estuary, northwest Washington.

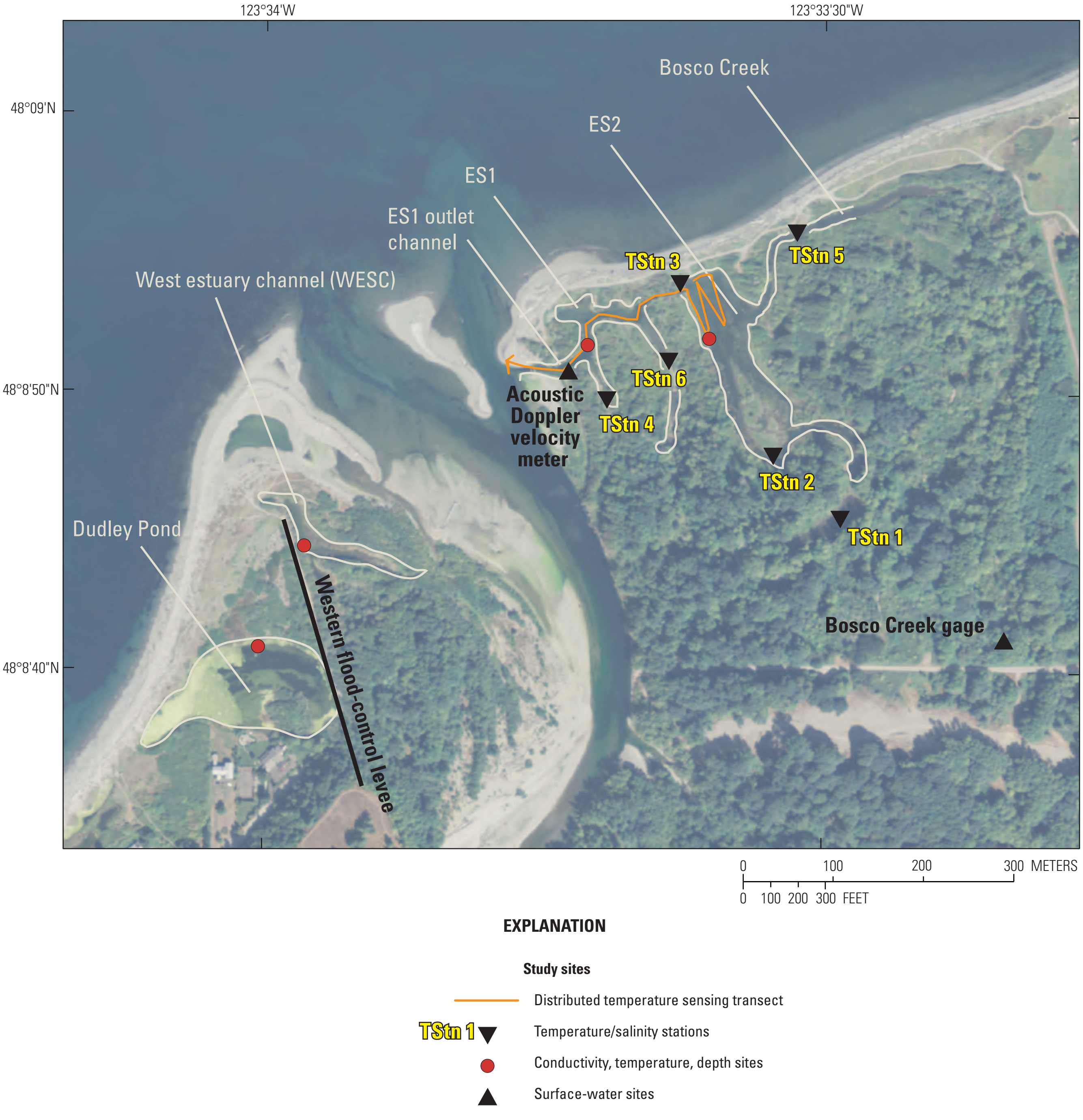

Figure 4.2. Locations of data collection sites near the Elwha River estuary, Washington.

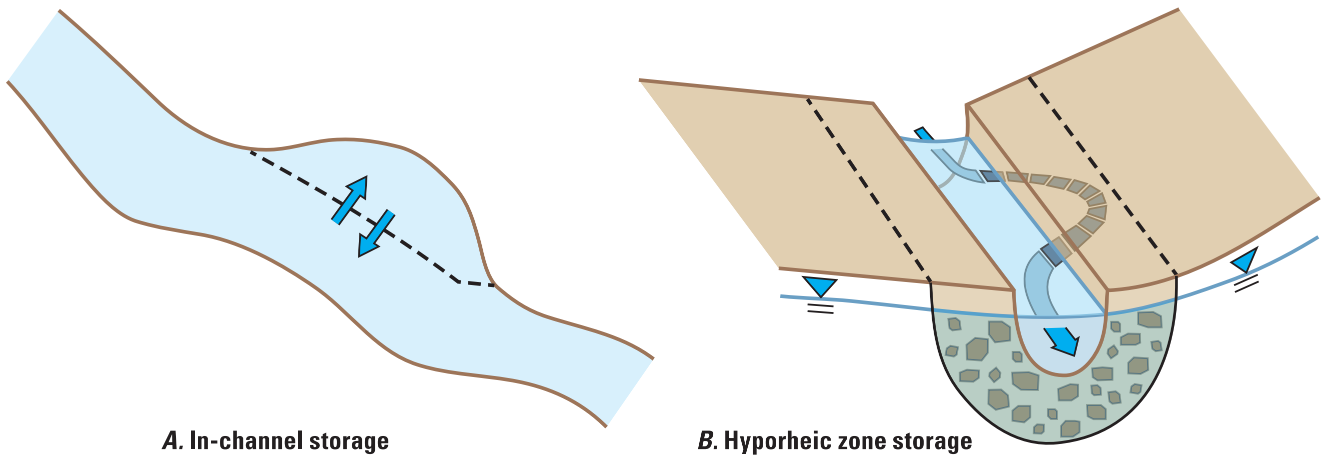

Figure 4.3. Conceptual diagram of transient storage, defined as the temporary retention of water due to (*A*) in-channel storage elements like pools and eddies, and (*B*) the hyporheic zone (shallow stream sediments). (Adapted from Runkel, 1998.)

Transient storage in streams has been shown to increase in the presence of large woody debris (Valett and others, 2002), in-stream vegetation (Harvey and others, 2003), channel meanders (Boano and others, 2006), and pool-riffle sequences (Gooseff and others, 2006), all of which contribute to the habitat needs of stream biota. Measures of transient storage in a reach, therefore, provide an indicator of habitat quality that is integrated across the spatial scales. Streams with high degrees of habitat heterogeneity (complexity) generally have a high potential to retain water in transient storage.

The amount of transient storage in a reach can be important for ecosystem functions of rivers and streams. For example, the longer that water remains in contact with stream sediment, the greater the opportunity for transformation and uptake of solutes in the reach becomes. Therefore, a complex channel with greater transient storage can lead to enhanced nutrient uptake (Valett and others, 1996; Butturini and Sabater, 1999; Bukaveckas, 2007) and increased stream metabolism (Mulholland and others, 1997). In the lower Elwha River, measurements of transient storage before and after dam removal will be an important tool for documenting changes in transient storage and its potential influence on solute uptake (for example, nutrients released from decaying salmon carcasses).

Reach-averaged transient storage commonly is measured by injecting a solute tracer into the stream and measuring the tracer concentrations downstream, yielding a solute breakthrough curve (fig. 4.4). If the tracer simply advects (that is, moves with the flow with little mixing or diffusion) through the reach because few habitat elements slow the main flow, the solute breakthrough curve will have a relatively high peak concentration and a quick return to background concentrations (fig. 4.4; solid line). Observation of a relatively lower peak concentration within the solute breakthrough curve and a slow return to background concentrations, indicates increased transient storage (fig. 4.4; dashed line). A solute-transport model that includes transient storage components to fit the field data from the tracer experiments can be used to quantitatively determine the amount of transient storage in a study reach. Previous work has shown that a complex channel structure (that is, multiple flow paths, pools, wood jams) increases the potential for transient storage, and that solute-injection experiments show promise for documenting this relation (Gooseff and others, 2007).

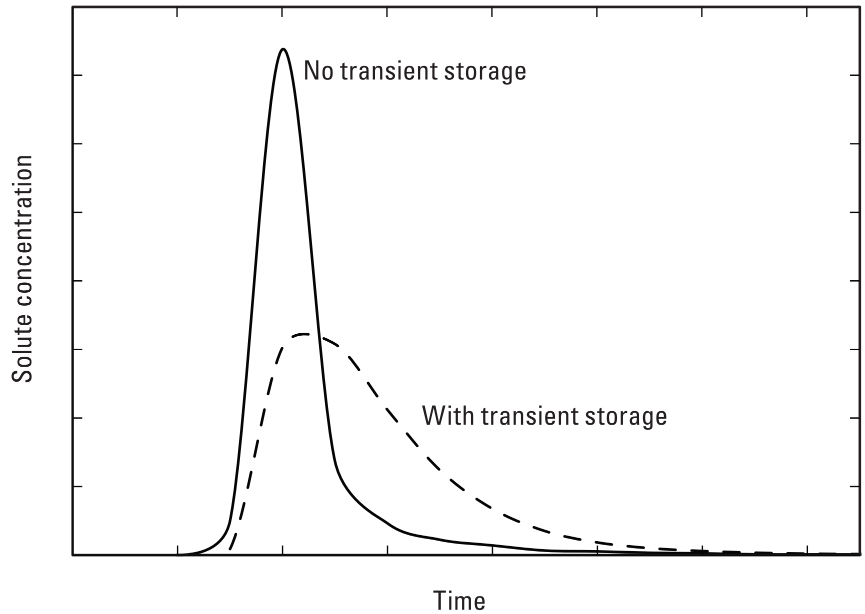

Figure 4.4. Theoretical downstream response to an instantaneous input of solute tracer. With little or no transient storage in the stream reach, the tracer is influenced only by advection and dispersion (solid line). A stream reach with significant transient storage will result in spreading of the peak with distance from the injection site (dashed line). (Adapted from Runkel, 1998.)

Transient storage was determined in the lower Elwha River using a dye-tracer experiment in which a slug of fluorescent dye was added to the main-stem river, while downstream detection instruments recorded dye concentration response over time. These temporal concentration data then were combined with a transient-storage model (Runkel, 1998) to assess overall transient storage for the study reach.

On September 29, 2009, a solute injection experiment using a non-toxic dye tracer, Rhodamine WT (RhWT), was completed on the lower Elwha River (fig. 4.1), timed to take advantage of autumnal low flow. The dye-injection site was in a section of the river with confined banks (about 3.9 km upstream of the river mouth), and the tracer was released into a riffle to enhance mixing (fig. 4.5). Another riffle farther downstream again mixed the dye throughout the width of the river. A total of 6.3 liters of 20 percent active ingredient Rhodamine WT was added to target a peak concentration of about 50 µg/L at the most downstream detection site, using the equations of Kilpatrick and Cobb (1985). To offset density differences and to improve lateral mixing, the tracer was added to the river simultaneously by three individuals spaced across the river at the injection site (fig. 4.5).

Concentration of dye was measured at three locations downstream of the injection site (fig. 4.1) using automated field fluorometers (SCUFA) that measured the fluorescence of the added tracer in the river with a reported detection limit of 0.04 µg/L (Turner Designs, Inc, 2004). Measurements of fluorescence were recorded at 1-minute intervals throughout the duration of the experiment to determine the solute breakthrough curves at each site. A SCUFA mount assembly for each instrument was secured to the riverbed using a cinderblock anchor attached to a float. The SCUFA then was affixed to this mount assembly so that the instrument was positioned at a point in the water column 0.75 m below the water surface. The three instruments were calibrated with simultaneous placement in the river for 10–15 minutes before and after the experiment to determine the bias in fluorescence readings between individual meters, and to assess potential instrument drift during the experiment.

Figure 4.5. Dye study in the lower Elwha River, Washington, September 29, 2009. (*A*) USGS hydrologist mixing the Rhodamine WT solution prior to slug injection; (*B*) three USGS hydrologists introducing the rhodamine into a riffle to enhance lateral mixing of the tracer; (*C*) dye tracer just downstream of the injection riffle, where the plume is spreading longitudinally and laterally in the slow moving water before reaching the next downstream riffle; and (*D*) looking downstream as the dye tracer approaches the first monitoring station. Main channel storage is evident on the left side of the channel in a slower moving backwater area, shown by the arrow. (Photographs taken by (*A*) James R. Foreman, U.S. Geological Survey and (*B, C, D*) Christopher A. Curran, U.S. Geological Survey.)

The three SCUFAs, referred to as Station 1, Station 2, and Station 3 (table 4.1), were placed 0.7, 1.5, and 3.0 km downstream of the injection site, respectively (fig. 4.1). River discharge was measured at Stations 1 and 2 during the experiment using a hand-held acoustic Doppler velocimeter, and at Station 3 using an acoustic Doppler current profiler mounted on a tethered boat. The theory of operation of these hydroacoustic technologies is well documented (see, for example, Yorke and Oberg, 2002). Standard USGS measurement protocols were followed for both methods, and measurement error was assumed to be less than 5 percent (Mueller and Wagner, 2009; Turnipseed and Sauer, 2010).

Rhodamine travel times to each SCUFA monitoring station were determined by plotting the tracer response at each SCUFA station compared to time (fig. 4.6), and the time when peak concentration was reached represented the average travel time to that monitoring station from the injection site (Kilpatrick and Wilson, 1989). Average travel times to each station from the injection point were 0.42, 0.93 and 1.57 hours, respectively. Measured river discharge was 9.46 m^3/s (334 ft^3/s) at Station 1, decreasing to 8.86 m^3/s (313 ft^3/s) at Station 2, and increasing to 9.85 m^3/s (348 ft^3/s) at Station 3 (table 4.2). For comparison, the daily discharge for that day as measured at USGS streamflow-gaging station 12045500 (Elwha River at McDonald Bridge near Port Angeles, Wash.) was 10.1 m^3/s (355 ft^3/s).

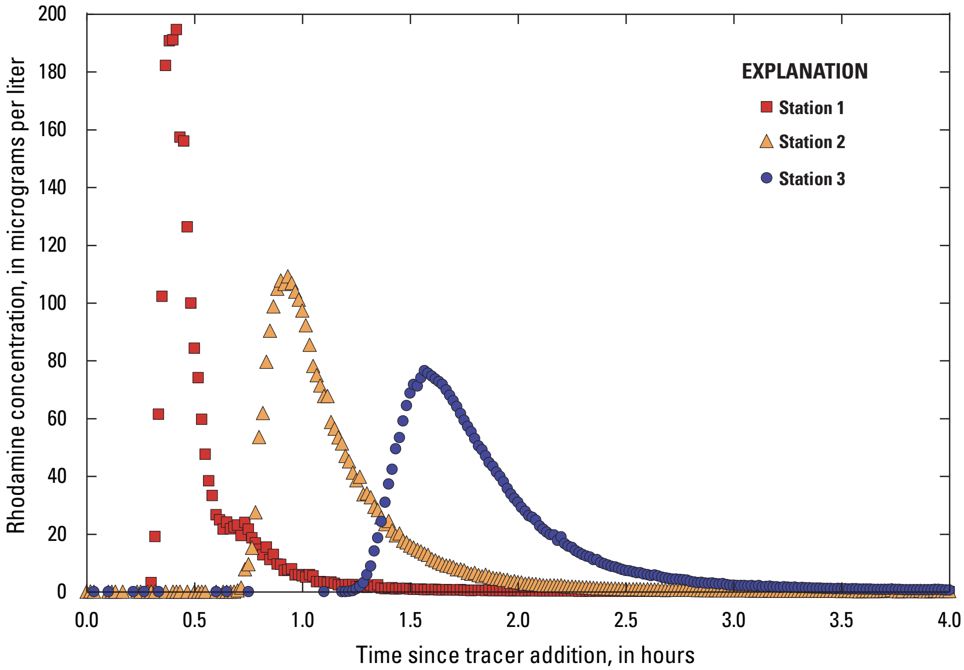

Figure 4.6. Measured rhodamine concentrations at monitoring stations 1, 2 and 3 after a slug injection of 6.3 liters of Rhodamine dye tracer in the lower Elwha River, Washington, September 29, 2009.

Table 4.2. Flow and cross section characteristics from manual discharge measurements, Elwha River, Washington, September 29, 2009.

[**Abbreviations**: m^3/s, cubic meter per second; m, meter; m^2, square meter; m/s, meter per second]

Station No.	Discharge (m^3/s)	Width (m)	Mean depth (m)	Area (m^2)	Velocity (m/s)
Station 1	9.46	46	0.58	26.8	0.37
Station 2	8.86	26	.43	11.5	.76
Station 3	9.85	33	.46	14.9	.67

Tracer breakthrough curves were simulated using the one-dimensional transport with inflow and storage (OTIS) model (Runkel, 1998) to estimate the amount of transient storage in the experimental reach. OTIS solves two differential equations developed by Bencala and Walters (1983) to calculate solute concentrations over time in the main channel and the storage zone of the stream reach. The changes in the river discharge measured through the study reach, presumably from water exchanges with the underlying aquifer, were incorporated into the modeling results. The model simulation results produced estimates of four hydrologic parameters used to calculate transient storage metrics. These output parameters (shown conceptually in fig. 4.7) include the dispersion coefficient for the reach (D), the main channel cross sectional area (A), storage zone area (A_s), and rate of exchange between the main channel and storage zone (α). From these parameter estimates, the metrics of transient storage can be determined, including the relative storage zone capacity (A_s/A) and the fraction of median travel time due to transient storage normalized against a 200-m long reach (F_{med}^{200}) which allows for future comparison with data from other rivers (Runkel, 2002). Discharge fundamentally alters solute transport within a stream; therefore, tracer studies ideally should be over a range of discharge. However, F_{med}^{200} has been shown to be a robust metric for assessing differences in transient storage across sites and through time (Runkel, 2002) and can be used as a primary indicator for tracking changes following dam removal. By calibrating this model for the study reach, metrics of transient storage are derived that serve as indicators of habitat complexity prior to dam removal.

To run the OTIS model, the tracer breakthrough curve at Station 1 was entered as the upstream model boundary condition. This input then was used to simulate the data at Station 2 and Station 3 through a non-linear least squares algorithm using the OTIS-P model (Runkel, 1998). OTIS-P uses the same numerical model as OTIS, but includes a parameter estimation subroutine to automatically fit the model to downstream tracer data by optimizing values for D, A, A_s.

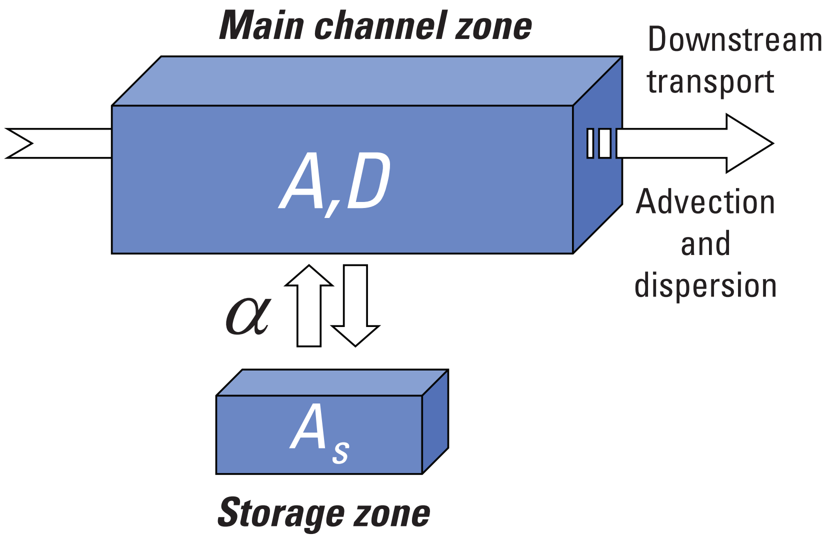

Figure 4.7 Conceptual diagram of one-dimensional transport with inflow and storage (OTIS) model showing advection and dispersion of solute (dye) within the main channel and exchange of solute into the storage zone. Four major model parameters are optimized to fit the calibration data including main-channel area (A), dispersion coefficient (D), storage zone area (A_s), and the exchange rate into the storage zone (α). (Adapted from Runkel, 1998.)

OTIS-P was executed for two model scenarios. The first scenario modeled the experimental reach as a single reach, with the Station 1 data used as the upstream boundary condition to simulate the response at Station 3. After the dams are removed, the entire reach between Stations 1 and 3 might remain intact as a single channel; therefore, this simulation will represent a baseline condition of the lower river before dam removal. The second scenario breaks up the experimental reach into two separate subreaches from (1) Station 1 to Station 2, and (2) Station 2 to Station 3, allowing transient storage properties to be assessed in each subreach. This sampling and modeling strategy provides some flexibility in the transient-storage analysis, and allows future studies to use the data collected even if the lower river channel experiences dramatic morphologic changes.

Transient storage model simulation results from OTIS-P simulations generally matched the measured Rhodamine curves for the single-reach scenario (fig. 4.8*A*), and the two-reach scenario (fig. 4.8*B*). Optimized parameters of transient storage from each model run indicate that little transient storage was in each reach (table 4.3). This observation is best illustrated by the values for F_{med}^{200}, the fraction of median travel time due to storage. In the single-reach scenario, F_{med}^{200} between Station 1 and 3 was 0.01, suggesting that approximately 1 percent of the travel time can be attributed to storage processes in the reach. For the two-reach scenario, F_{med}^{200} was higher in the first reach (0.06) when compared to the second reach (0.01). The relative size of the storage zone for the single-reach scenario, given by A_s/A, was 0.128, and for the two-reach scenario A_s/A was 0.237 and 0.160 for the upper and lower reaches, respectively. These A_s/A values indicate that the channel cross-sectional areas are larger than the theoretical storage-zone areas.

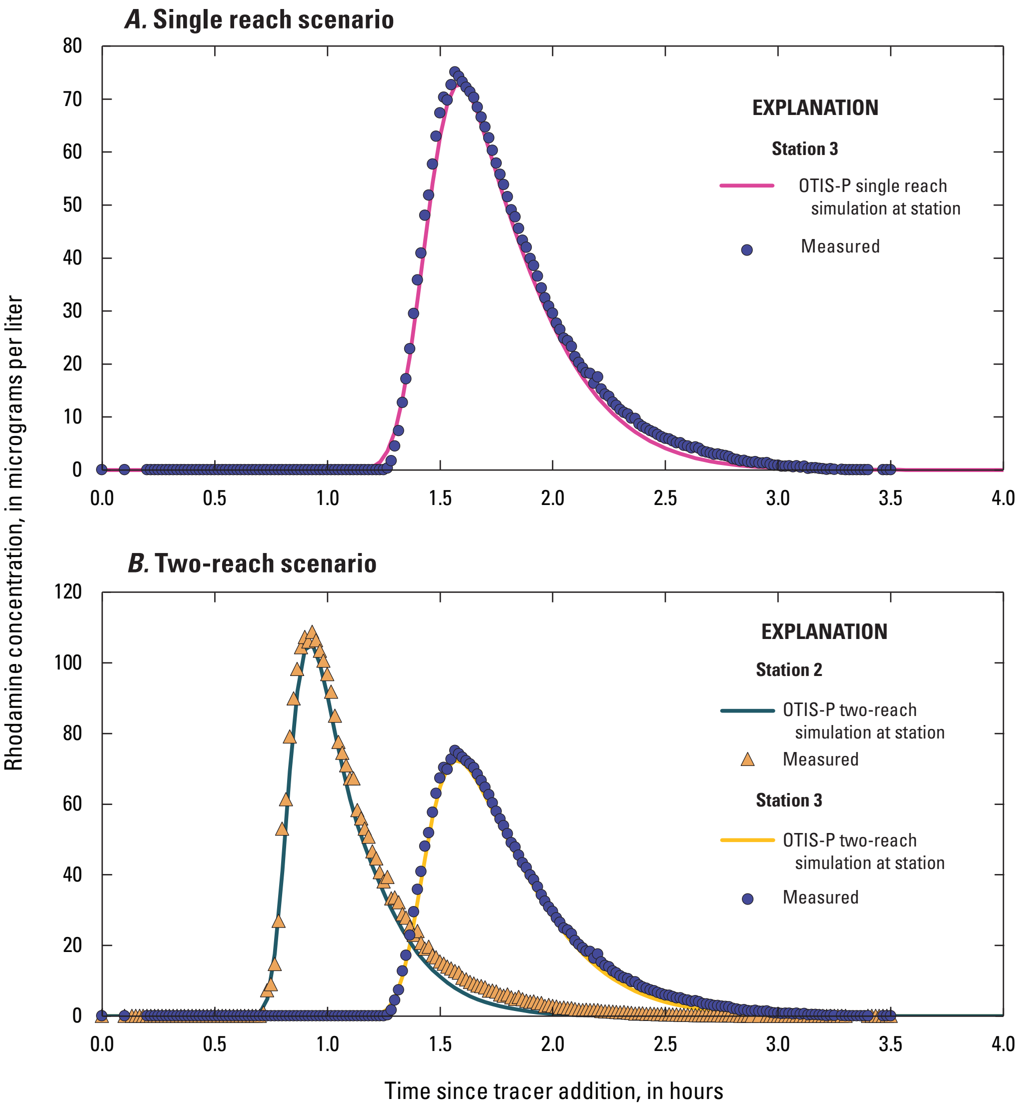

Figure 4.8 Measured and one-dimensional transport with inflow and storage (OTIS-P) simulated rhodamine breakthrough curves at (*A*) Station 3 for the single-reach scenario and at (*B*) Stations 2 and 3 for the two-reach scenario for the lower Elwha River, Washington.

Table 4.3. Optimized parameter estimates for the transient storage model using one-dimensional transport with inflow and storage (OTIS-P) and selected transient storage zone metrics, Elwha River, Washington.

[**Abbreviations**: *D*, dispersion coefficient; *A*, main channel cross-sectional area; A_s, storage zone cross-sectional area; α, storage zone exchange coefficient; A_s/A, relative storage-zone capacity; *DaI*, Damköhler number; F_{med}^{200}, fraction of median travel time due to storage normalized against a 200-meter long reach; m/s^2, meter per second squared; sec^{-1}; per second]

Reach	*D* (m/s^2)	*A* (m^2)	A_s (m^2)	α (sec^{-1})	A_s/A	F_{med}^{200}	*DaI*
Station 1 to 3	6.9	15.9	2.0	2.50E-04	0.128	0.01	8.9
Station 1 to 2	1.6	20.3	4.8	8.43E-04	.237	.06	7.2
Station 2 to 3	6.4	12.6	2.0	3.00E-04	.160	.01	4.7

To check how well the model simulated observed response in the river, the experimental Damköhler number (*DaI*) from each simulation was calculated. The *DaI* is a measure of how well the tracer experiment describes the transient storage processes of the experiment and is calculated by

$$DaI = \alpha \frac{(1 + A / A_s)L}{u}, \tag{1}$$

where *L* is the reach length (Wagner and Harvey, 1997). Values of *DaI* close to 1.0 are ideal, but values between 0.1 and 10.0 indicate that the tracer experiment adequately describes the transient storage processes of the experiment (Wagner and Harvey, 1997; Harvey and Wagner, 2000). The *DaI* results for this study were all greater than 1.0 (4.7 to 8.9, table 4.3), but were within this optimal range. Thus, the values of *DaI*, and the quality of the visual fit to field data, indicate that the modeling approach was reasonable for this experiment.

The amount of transient storage in the experimental reach is small, likely due to high water velocity and a relatively simple channel (table 4.2). Generally, from 1 to 6 percent of the median travel time in the lower reach can be attributed to transient storage processes. With a change in sediment supply and large woody debris after dam removal, sedimentation and channel reorganization in the lower Elwha River are expected to increase channel-structure complexity. These changes are expected to translate into measurable increases in travel times.

Groundwater and Surface-Water Interactions

Water levels and temperatures were collected from 2007 to 2009 using self-contained data recorders in four monitoring wells positioned throughout the Elwha River flood plain. These groundwater data, coupled with streamflow and thermal patterns measured in the main-stem river, allowed for the analysis of the relative connectivity of the aquifer to the river. A seepage investigation in September 2007 included near simultaneous discharge measurements at multiple locations along the river channel to determine the volumetric exchange of surface water and groundwater. Finally, to better quantify the spatial distribution of groundwater flow feeding into the east estuary, thermal patterns of the estuary bed were measured over a 20-hour period in September 2008.

Water Level Measurements near the Lower Elwha River

Four data recorders with vented pressure transducers were placed in established monitoring wells (fig. 4.1; table 4.1) to collect water level and temperature data from November 9, 2007, to August 27, 2009, at 1-hour intervals. Raw water-level data were adjusted to elevation in meters above sea level (NAVD 88) based on well-head elevation surveys by the Pacific Groundwater Group (2005). The two southern-most monitoring wells were on the river side of the eastern flood-control levee (Warrick and others, 2011a, chapter 3, this report) along the outer bend of a distributary channel that conveys overflow with increasing discharge. The farthest upstream of these two wells (MWGR; fig. 4.1) was about 2.7 km upstream of the river mouth, and the other well (AFN 254) was about 2.1 km upstream of river mouth. Of the other two monitoring wells, AFN 259 (fig. 4.1) was farther south and closer to the river, and AFN 256 was farther north and closer to the coastline.

Data collected at the four monitoring wells indicated a strong correlation between groundwater elevation and river stage throughout the year, regardless of the river discharge or the season (fig. 4.9). Data from AFN 259 and AFN 256, the two wells closest to the coastline, between June and September of 2008, demonstrated the strong coupling between groundwater elevation in the flood plain and stage in the river (fig. 4.10). Additionally, tidal influences also were apparent in these flood plain wells, although tidal influence seemed to be considerably smaller than the river influence. The response time between the peak river stage and the peak groundwater elevation ranged from hours to a few days. Continuous groundwater-elevation data collected from intermittent records from October 2002 through January 2005 showed similar characteristics (Pacific Groundwater Group, 2005).

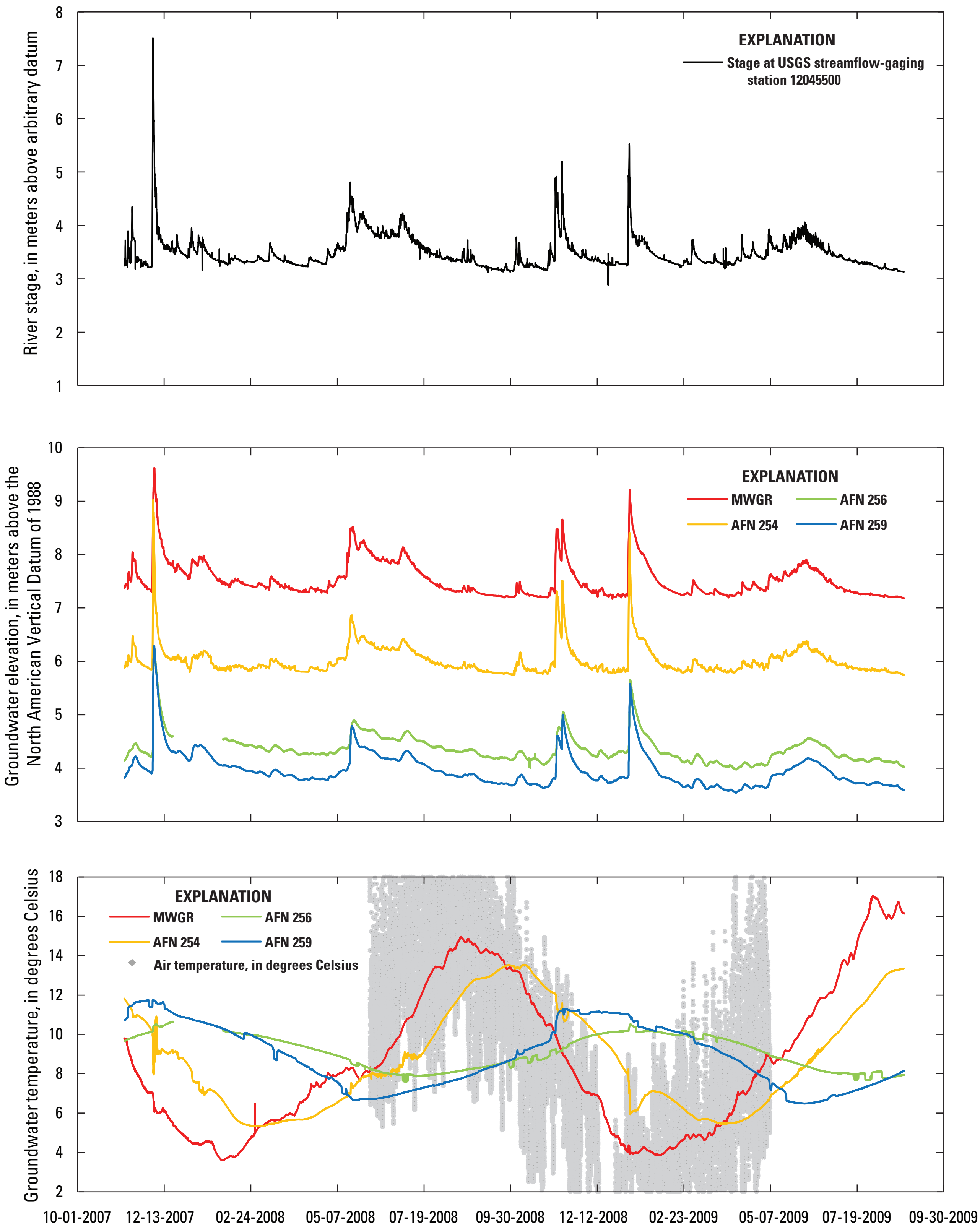

Figure 4.9 Groundwater elevations and temperatures from four monitoring wells in the Elwha River flood plain, in relation to river stage at U.S. Geological Survey streamflow-gaging station 12045500, Elwha River near Port Angeles, Washington, and air temperature at the Bosco Creek gaging station (12046523).

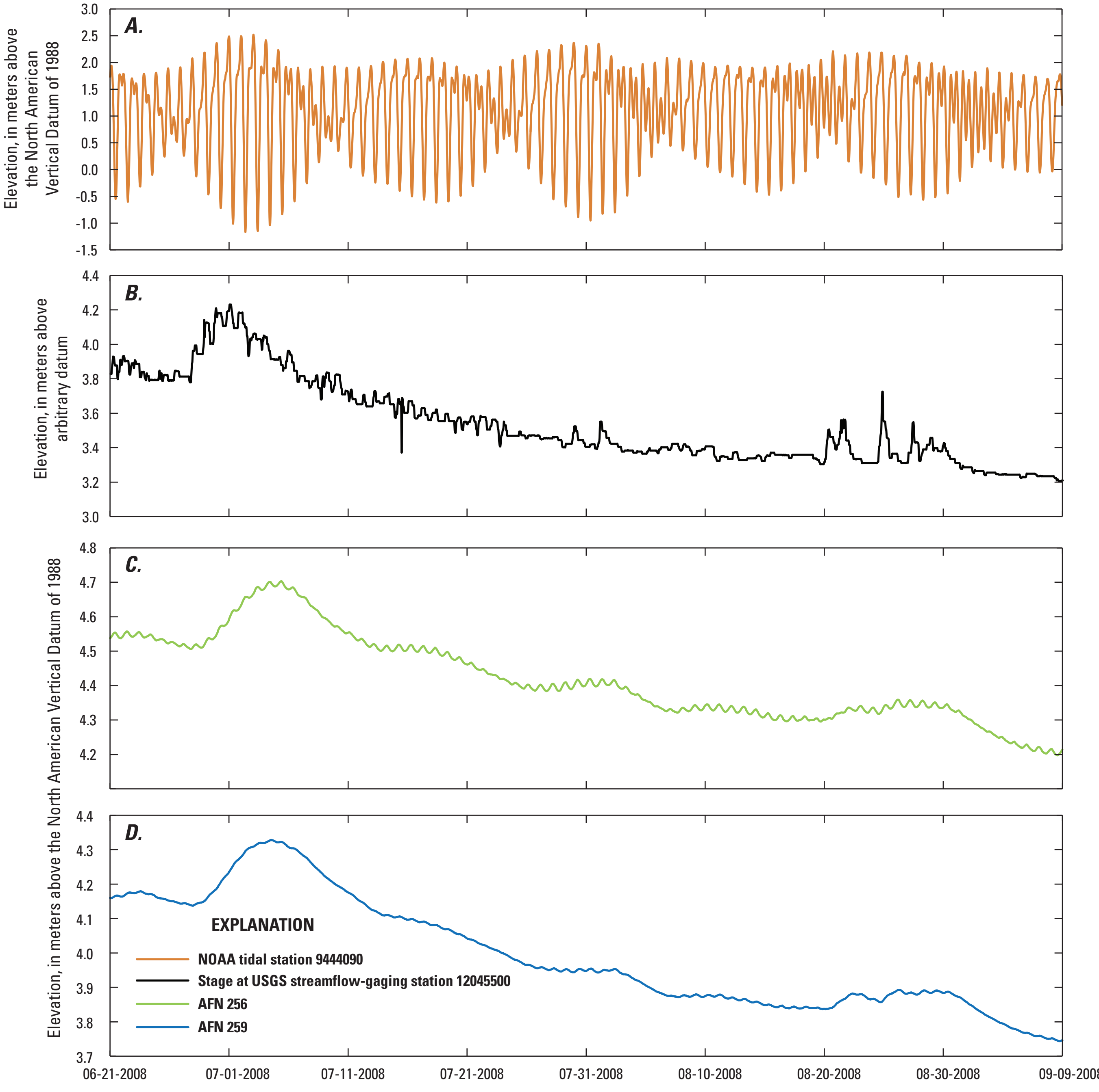

Figure 4.10. Water elevations representing (*A*) tidal stage at National Oceanic and Atmospheric Administration (NOAA) tidal station 9444090 in Port Angeles, Washington; (*B*) river stage measured at U.S. Geological Survey streamflow-gaging station 12045500, Elwha River near Port Angeles, Washington; (*C*) groundwater elevation in monitoring well AFN 256; and (*D*) groundwater elevation in monitoring well AFN 259. The data show the rapid response in groundwater elevation to changes in river stage. Both monitoring wells are at least 0.5 kilometers from the main-stem river and at least 1.0 kilometers from the coast.

Temperature data from the monitoring wells indicated that in contrast to the strong hydraulic connection of existed between the river and the aquifer, a relatively weak thermal connection existed between wells and, presumably, the main-stem river. Although daily water-temperatures measured directly in the lower Elwha River were not available, it was assumed that river temperature tracked air temperature, and air-temperature data were collected at 15-minute intervals with the BaroTROLL collector at USGS streamflow-gaging station 12046523, Bosco Creek near Port Angeles, Washington. These air-temperature data correlated well with the general trends in water temperature measured at MWGR (fig. 4.9). This new monitoring well is adjacent to the river channel and positioned 10–30 m from the distributary channel and likely is active for moderately large discharges. It seems likely that the delay or lag time between the temperature of water in MWGR and the river probably is on the order of a few days or weeks. At well AFN 254, about 100 m from the distributary channel, the warmest groundwater seems to occur about 1 month after the maximum air temperature (fig. 4.9). The occurrence of the warmest water at the flood-plain wells (AFN 259 and AFN 256) occurs 5–7 months after the maximum seasonal air temperatures (fig. 4.9).

Seepage Investigation in the Lower Elwha River

A seepage investigation from September 24 – 26, 2007, determined stream-flow gains and losses in the lower 4.4 km of the Elwha River. This investigation consisted of 18 discharge measurements collected at 8 sites on the main stem, and 4 sites on tributaries along the river (U.S. Geological Survey, 2007). Standard methods were followed for measuring discharge using a current meter for 8 measurements, an acoustic Doppler velocimeter for 6 measurements, and an acoustic Doppler current profiler for 4 measurements (Mueller and Wagner, 2009; Turnipseed and Sauer, 2010). For the seepage investigation, all discharge measurements were assumed to have an accuracy of 5 percent. Discharge measurements were made during autumn low flow to capture base-flow conditions when surface-water/groundwater interactions are most apparent.

During the 3-day seepage investigation, discharge at USGS streamflow-gaging station 12045500, the Elwha River at McDonald Bridge near Port Angeles, Washington, was constant at 10.25 m^3/s on September 24, 2007, and increased to 11.47 m^3/s on September 25, 2007, and remained constant for the rest of the seepage investigation (fig. 4.11). To allow direct comparison of discharge data between days, discharge measurements made on September 24, 2007, were adjusted upward by 1.22 m^3/s (the difference between background discharge of 11.47 m^3/s and 10.25 m^3/s) using the McDonald Bridge discharge values. Additionally, discharge measurements were adjusted for tributary inflow by subtracting all upstream tributary inflows from the main-stem Elwha River discharge measurements.

Elwha River discharge measurements that have been adjusted for variations in the hydrograph and tributary inflows show synoptic downstream variations in the flow that may be attributed to groundwater/surface-water exchange, tidal fluctuations in the estuary, measurement error, or some other factors (fig. 4.12). The variations in the discharge data at the section of river between 4.4 and 2.8 km upstream of the river mouth are within the uncertainty of the individual measurements; however, there may be a slight groundwater/surface-water exchange. Discharge measurements collected on the same day in the main-stem river between 2.8 and 1.7 km upstream of the river mouth indicated that about 10 percent of the Elwha River discharge seeped into the groundwater system. Discharge data downstream of 1.7 km from the river mouth indicated strong gains to the Elwha River on September 25, and no change on September 26 (fig. 4.12). However, the flow measurements on September 25 were made when the tide was falling, whereas the flow measurements on September 26 were made just after low tide. The differences in the adjusted flows near the river mouth between these 2 days likely were caused by the changing flow conditions in the estuary as water was stored and released to the river. Based on uncertainty of measurements, there were no clear trends indicating groundwater losses or gains in the lower 1.7 km of river or in the section of river upstream of a point 2.8 km from the river mouth.

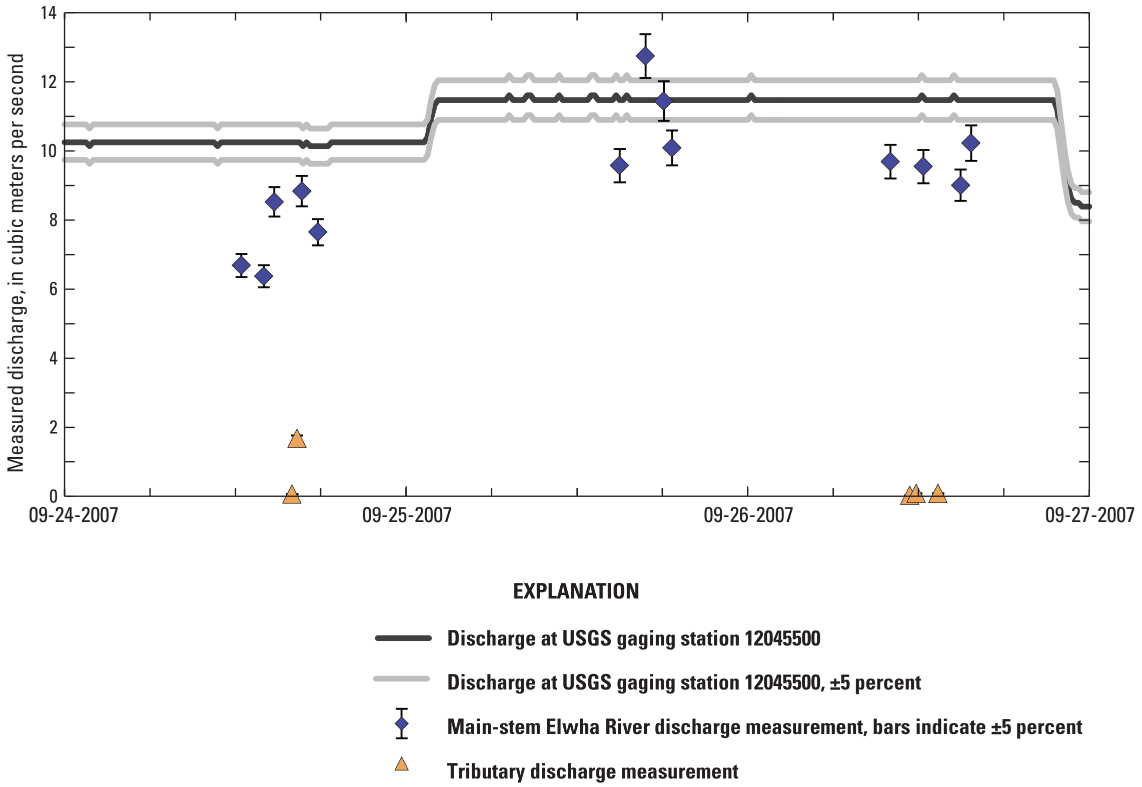

Figure 4.11. Discharges measured during the 3-day seepage investigation in September 2007, on the lower Elwha River in relation to the discharge at U.S. Geological Survey (USGS) streamflow-gaging station 12045500, Elwha River at McDonald Bridge near Port Angeles, Washington.

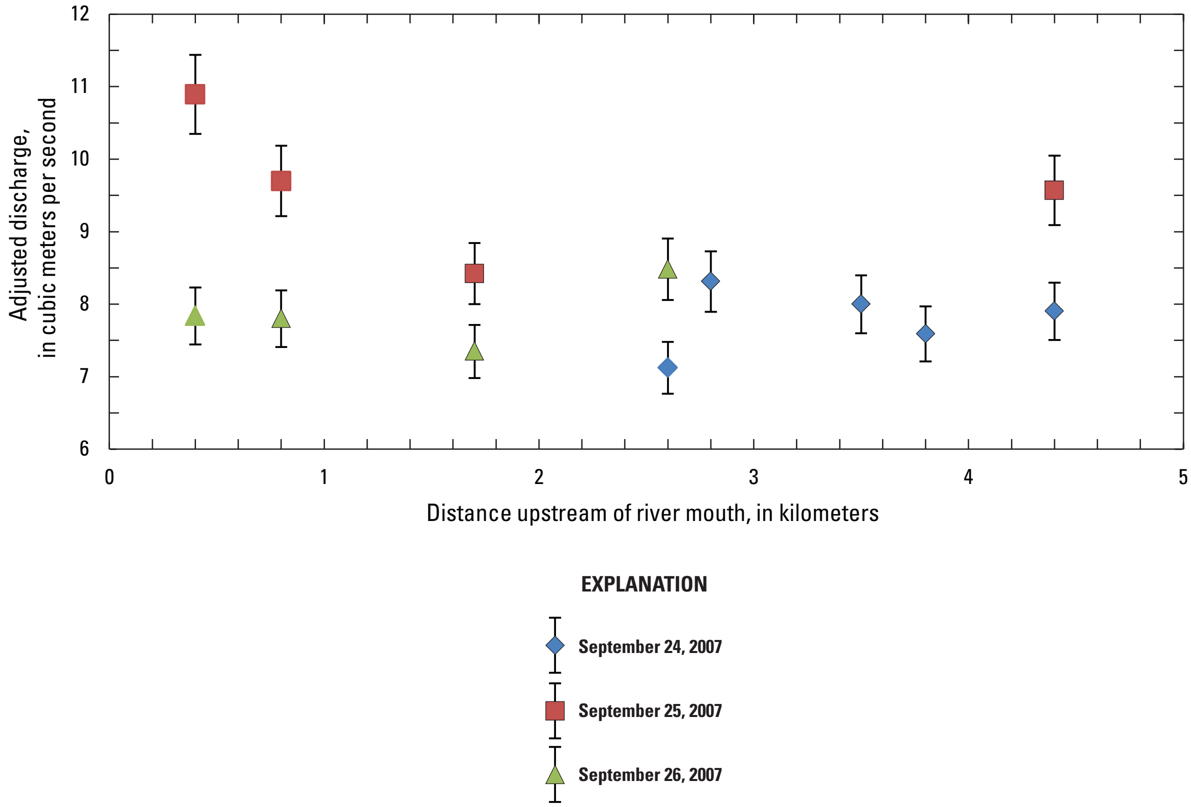

Figure 4.12. Adjusted synoptic discharge measurements in the lower Elwha River, Washington, for the September 2007 seepage investigation. Bars indicate ±5 percent.

Distributed Temperature Sensing in the Estuary

A water-temperature survey using fiber-optic distributed temperature sensing (DTS) technology (Selker and others, 2006; Tyler and others, 2009) was conducted in the east estuary in September 2008. The sampling sites covered much of the surface of the riverbed of the east estuary and ended in the main-stem river just upstream of the river mouth (fig. 4.2). The purpose of the temperature survey was to locate thermal signals (dampened diurnal fluctuations of surface-water temperatures) that could be attributed to groundwater discharge into the surface-water body. Water temperature also was used as a tracer to observe the circulation pattern of tidally induced mixing in the east estuary.

A field-portable Sensortran DTS unit was used with a 3-mm diameter steel and Kevlar reinforced single-ended fiber-optic cable dispensed from a portable reel containing about 1,500 m of cable. Temperature was recorded every 5 minutes at 1-m resolution along the cable length. The cable was calibrated on site with an ice bath at the DTS unit location. Using this arrangement, some degradation of precision occurs with distance from the DTS unit; however, DTS temperatures correlated well with Tidbit point sensors placed at multiple locations along the cable for comparison. Although 0.01 °C measurement precision has been demonstrated using similar DTS instruments (Selker and others, 2006), accuracy of measurements in both deployments were estimated to be 0.1 °C.

At the estuary site, DTS cable (900 m) was placed in the east estuary. The cable was laid over much of the bed of ES2, extended through connecting channels into ES1, extended farther through the ES1 outlet channel, and ended in the main-stem river just upstream of the river mouth (fig. 4.2). Turning points, used to guide the cable through the estuary complex without damage, were constructed of 2.5-cm steel pipe pounded into the substrate with a larger plastic-pipe sheath to protect the cable during placement. In most locations, the cable was placed on or near the bed substrate; an exception was in ES2 where thick algal mats and water depths often exceeding 1.2 m prevented the cable from resting directly on the bed. The estuary DTS cable was deployed over a 20-hour period and collected approximately 216,000 temperature measurements.

The weather during the DTS survey in the east estuary, on September 1–2, 2008, was warm and clear, with sun and little wind. Figure 4.13 shows the distribution of temperature along the DTS cable at transition times during the tidal cycle, when water flux into the estuary was changing. At 1400 on September 1, the water temperature in ES2 was a relatively warm 15 °C, but cold water of about 9 °C from the river was moving through the connecting channels into ES1 (fig. 4.13*A*). By 1700, at exactly high tide, cold water had ceased entering the connecting channel and nearly the entire cable was a relatively warm 16 °C (fig. 4.13*B*). This heating was caused by strong solar heating on September 1, 2008. By 2200, overall water temperature had decreased along the cable to about 14 °C in the ES1 and the ES1 outlet channels (fig. 4.13*C*). During the night and early morning, temperature generally dropped steadily in the smaller ES1 water body and connecting channels. Within ES2, however, the large thermal mass of the water body kept the water at about 15 °C throughout the night. Recorded temperature in ES2 also likely reflected the situation that algal mats suspended the DTS cable above the bottom of the water body. As a result, the temperature data collected in ES2 were from a depth location above the actual bed of the water body. Water temperature in the main-stem river fluctuated between 9 and 12 °C during the late afternoon and night, as warmer river water flowed over the top of the cooler marine water. Because the cable was laid on the bottom of Elwha River channel, cycles in the tide dictated whether relatively warm freshwater, or cool marine water, influenced thermal data in this location. By 1000 on September 2, as the tide approached its low point, solar heating had warmed most of the water in ES1 and ES2 (fig. 4.13*F*).

To analyze where the groundwater may have entered the estuary, the total variance of recorded temperature data over the 20-hour experiment is shown as a function of position along the cable (fig. 4.14). Due to the large thermal mass, and because the cable was suspended by an algal mat, overall thermal variance in ES2 was less than 2 °C. Nonetheless, assuming groundwater was cooler than 15 °C, no areas of groundwater influx to ES2 were identified with the experiment. Variance also was small between 180 and 200 m at a section of cable in the middle of ES1 (fig. 4.14). During the course of the experiment, temperature at this location in the cable ranged between about 13 and 14 °C, which is small compared to sections of the cable placed elsewhere where temperature fluctuated as much as 5 °C. The area of low variance is the short yellow section of cable in ES1 surrounded by red at 1700 on September 1 (fig. 4.13*B*), or the same yellow section of cable surrounded by cyan at 0700 on September 2 (fig. 4.13*E*). This small variance was interpreted to be a region of strong groundwater upwelling, probably associated with a relict fluvial channel of coarse sediment and relatively large hydraulic conductivity. Two other regions of decreased thermal variance were apparent near 155 and 76 m (fig. 4.14), also suggesting groundwater influx though the strength of the signal was smaller than in the region from 180 to 200 m.

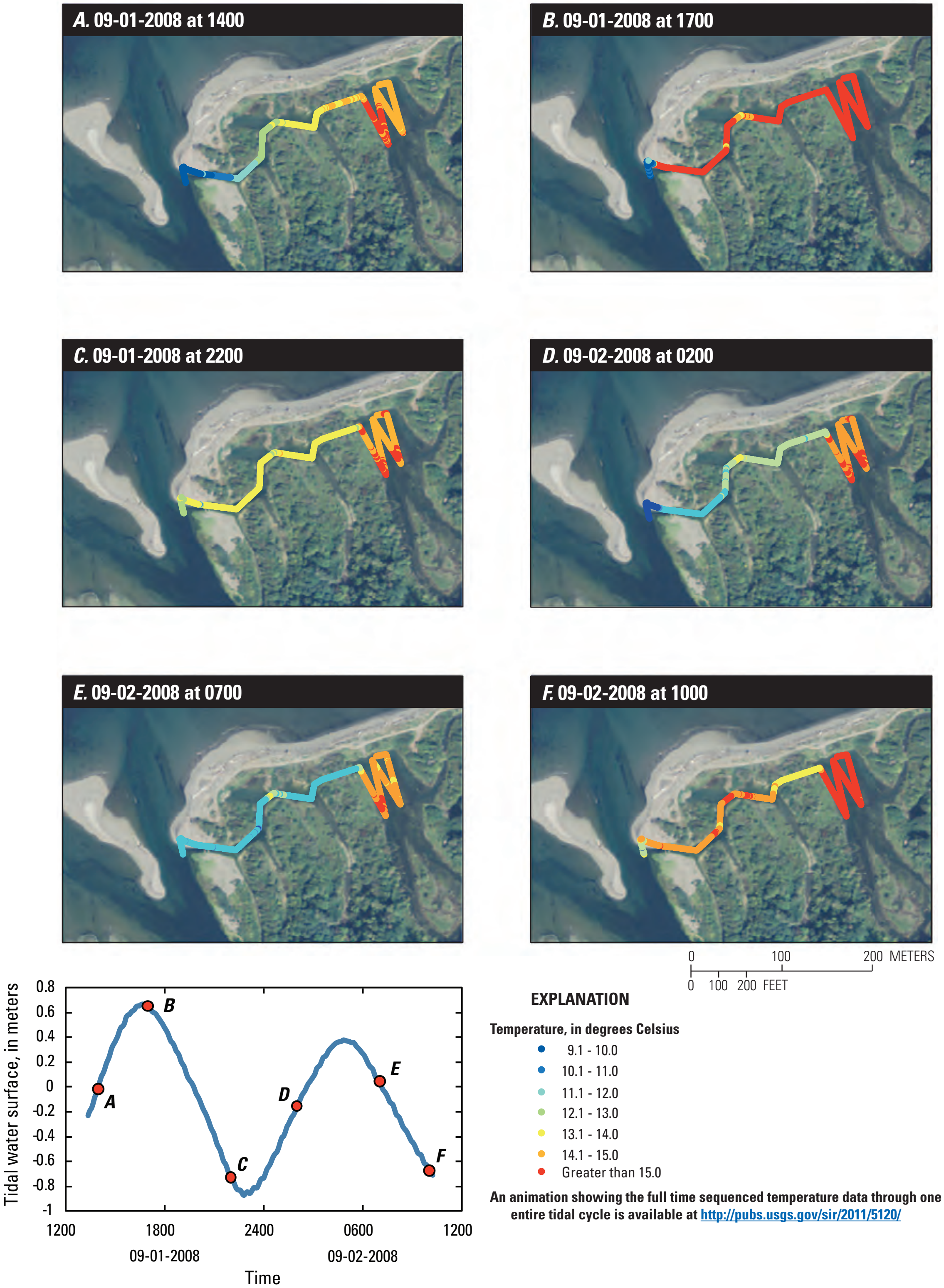

Figure 4.13. Photographs and graph showing temperature along a distributed temperature sensor (DTS) array positioned along the riverbed in the east estuary of the Elwha River between 1400 on September 1, 2008, and 1000 on September 2, 2008. Tidal stage at the National Oceanic and Atmospheric Administration tidal station 9444090 in Port Angeles, Washington, shows the influence of tides on surface-water exchange with the estuary.

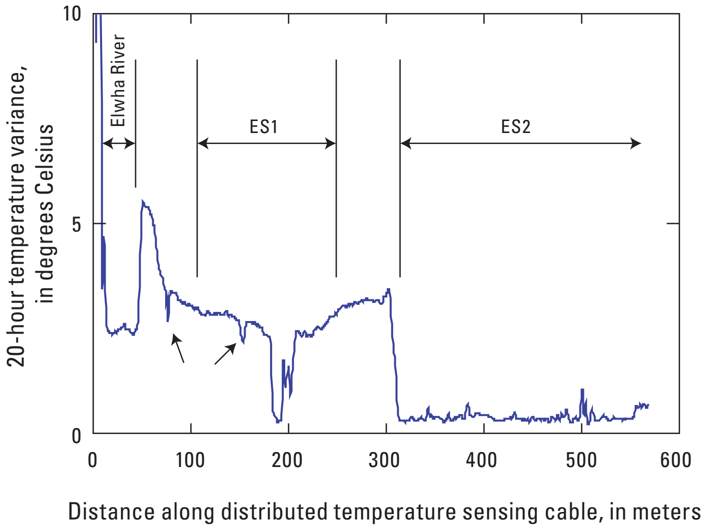

Figure 4.14. Variance in temperature fluctuation along the distributed temperature sensor array along the bed of the east estuary for the 20-hour period of data collection, Elwha River, Washington, September 1–2, 2008. Locations at 25 to 45 meters were in a section of the cable in the main stem of the Elwha River. Locations 320 meters and greater were sections of the cable in the ES2 water body. A prominent area of relatively small temperature variation from 180 to 200 meters likely was caused by strong groundwater upwelling. Similar reduced variation near 155 and 76 meters (indicated by arrows) might also reflect groundwater upwelling although the strength of upwelling was less pronounced than in the interval from 180 to 200 meters.

Water Data Collected within the Estuary

Qualitative observations of surface-water dynamics in the Elwha River estuary show the strong influence of tidal patterns and river flow dynamics (fig. 4.15), but do not explain the role of groundwater interactions, or quantify the relative contributions of water sources flowing to, and from, the estuary. By collecting temperature, salinity, stage, and surface-water velocity data at strategic locations in the estuary system, in addition to tidal information for the Strait of Juan de Fuca, a greater understanding of water movement through the complex was possible. Data recorders were placed in the four major coastal water bodies of the estuary (ES1, ES2, WESC, and Dudley Pond; fig 4.2). Continuous discharge values were calculated at the outlet channel of the east estuary. Finally, discharge at Bosco Creek, the primary surface-water inflow source to the east estuary, was periodically measured during the study to characterize surface-water fluxes.

Salinity, Temperature, and Depth Measurements in the Estuary

Water-temperature and salinity data were collected with self-recording electrical conductivity and temperature (CT) sensors, some of which also included transducers to measure water depth (CTD). These sensors were placed at four locations throughout the estuary (fig. 4.2; table 4.1). The sensors were attached to steel rods pounded into the estuary substrate and sampled at 15-minute or 1-hour intervals. The CTDs were in ES2 from June 2008 to April 2010, and in Dudley Pond from May 2009 to April 2010 (table 4.1). The CTs were placed in ES1 from July 2007 to March 2010, and in the WESC from May 2009 to April 2010 (table 4.1). Salinity data are presented in Practical Salinity Units (PSU), which is a relative salinity scale approximately equivalent to the total mass of salts in parts per thousand. Ocean water and freshwater typically are about 32 and 0 PSU, respectively.

Water levels in Dudley Pond and ES2 correlate with tidal fluctuations in the Strait of Juan de Fuca, represented by the water level at the NOAA tide gage in Port Angeles, Washington (fig. 4.16). However, water-level oscillations in Dudley Pond were smaller than ES2, and there was a lag in time between the water levels at Port Angeles and in Dudley Pond (fig. 4.16*A*). In contrast, the water level in ES2 correlates well with the high tides in the Strait of Juan de Fuca (fig. 4.16*B*), suggesting a direct surface-water connection with the river mouth.

Salinity levels in Dudley Pond were fairly low and steady, while WESC, on the river side of the levee, had higher levels and more fluctuations of salinity, suggesting more mixing of freshwater and seawater in WESC than in Dudley Pond (fig. 4.16). The east estuary (ES1 and ES2) had greater oscillations in salinity than the west estuary. Near the outlet of the estuary (ES1), there were daily oscillations between almost pure freshwater (about 0 PSU) and almost pure seawater (about 32 PSU). In ES2, salinity values commonly reached only one-third to one-half of those found in ES1 (fig. 4.16).

A longer record (spring to autumn 2009) of water temperature and salinity from the CTDs also demonstrated the differences between Dudley Pond on the west side and ES2 in the eastern estuary (fig. 4.17). During this sampling interval, there was a general warming trend until August, followed by a cooling trend through December. Dudley Pond was commonly from 2 to 5 °C warmer than ES2 during the summer (fig. 4.17A). The salinity levels in ES2 were more variable, and reached higher levels, than salinity levels in Dudley Pond (figs. 4.17*B* and 4.18).

Figure 4.15. Effects of tides on water levels in the Elwha River mouth and estuary, Washington. The date, time, and tidal stage, measured in meters above Mean Lower Low Water (MLLW) at National Oceanic and Atmospheric Administration tidal station 9444090 in Port Angeles, is shown for each photograph. (Photographs taken by Ian M. Miller (*E* and *F*) and Patrick B. Shafroth.)

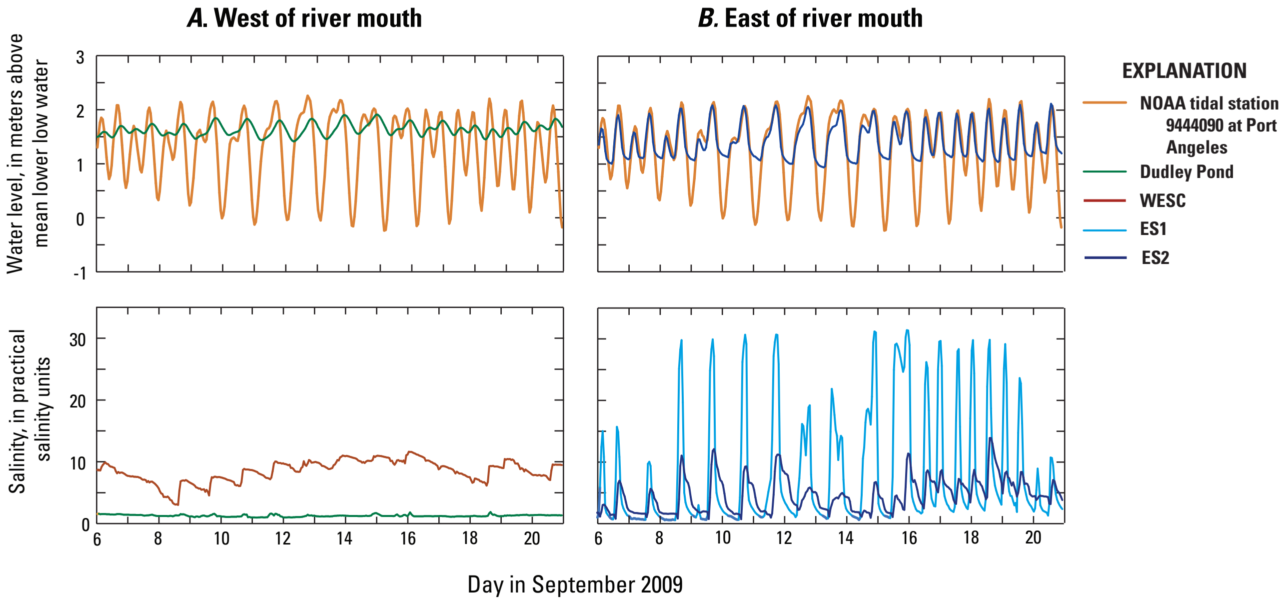

Figure 4.16. Surface-water levels and salinity over a 2-week (spring-neap) tidal cycle during low flow for (*A*) Dudley Pond and WESC in the west estuary and (*B*) ES1 and ES2 in the east estuary of the Elwha River, Washington, September 2009.

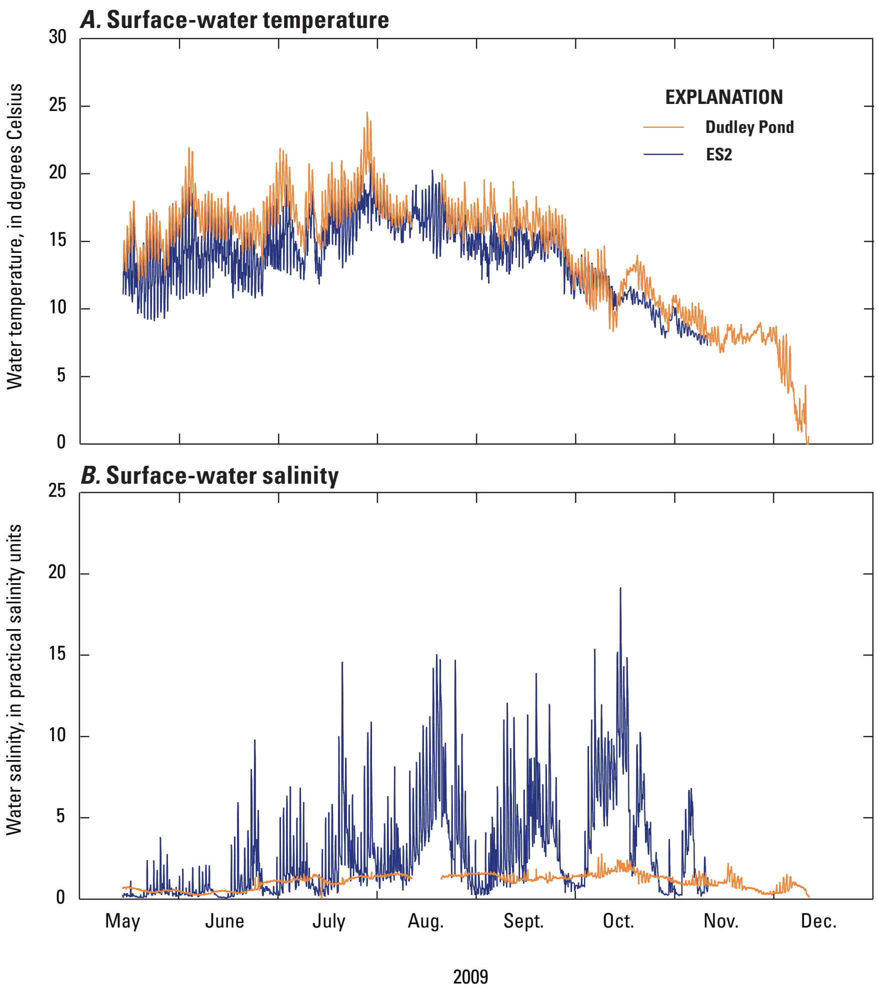

Figure 4.17. Surface-water (*A*) temperature and (*B*) salinity in Dudley Pond (west estuary) and ES2 (east estuary) in the Elwha River estuary, Washington, May–December 2009. Dudley Pond is consistently warmer, but it does not receive estuarine mixing during spring tides like ES2.

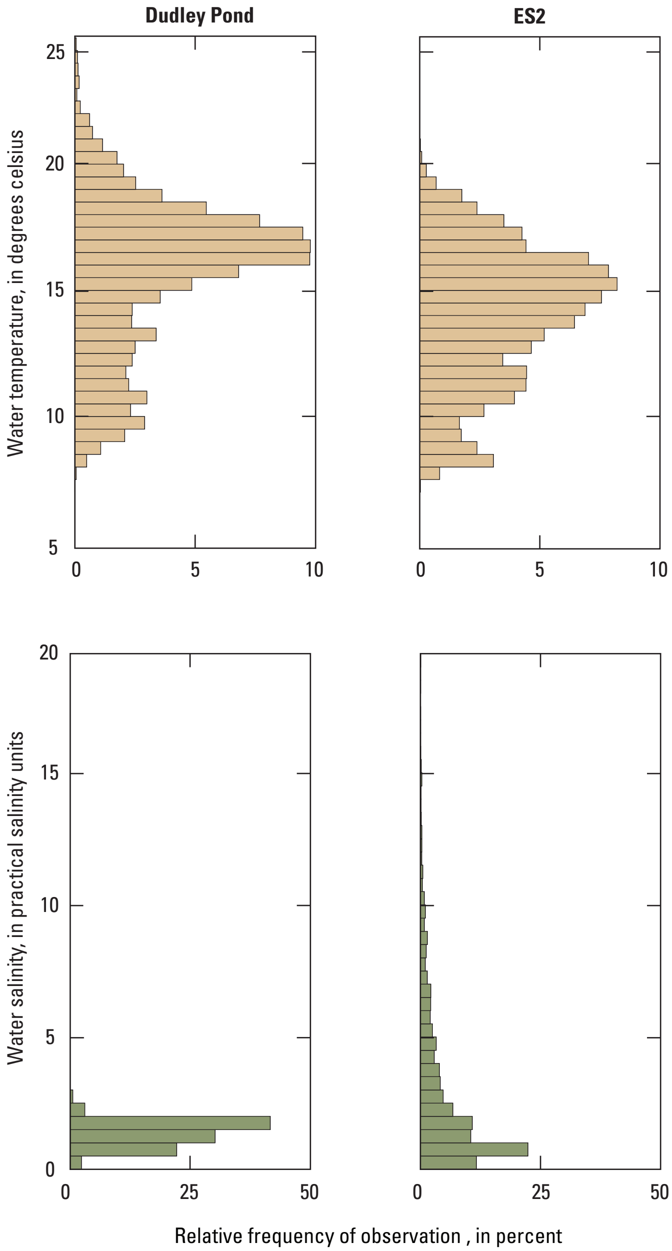

Figure 4.18. Frequency distribution plots showing comparison of water temperatures and salinities in Dudley Pond and ES2 in the Elwha River estuary, Washington, from May 14, 2009, to November 10, 2009.

Surface-Water Temperature Measurements in the East Estuary

To further characterize the temperature of surface water in the east estuary, six temperature data recorders were deployed in surface-water sites distributed throughout the estuary (fig. 4.2; table 4.1). These deployments started in early August 2009 during a cycle of large tidal variability. The six thermal recorders measured near-benthic temperature and the data helped explain overall spatial and temporal thermal trends in estuarine water. One “transect” of sensors was placed approximately parallel to the shoreline and included TStn 4 to the west, TStn 6 in ES1, and TStn 5 in Bosco Creek near where the creek enters ES2 (fig. 4.2). Another sensor transect was placed north-south in “finger” channels of ES2 (TStn 3 and TStn 2), and a secondary surface-water body to the south of ES2 (TStn 1; fig. 4.2). TStn 4, 5, and 6 were removed on August 8, 2009. TStn 1, 2, and 3 were removed in December 2009.

For thermal-recorder deployments, posts were pounded into the substrate of the bed of the water body, and a slotted plastic pipe was attached to the post. The bottom of the plastic pipe was positioned about 5 cm above the estuary substrate. The thermal recorders were placed near the bottom of the plastic pipe and secured with a nylon string hung from a bolt inserted near the top of the plastic pipe. TStn 4 was placed in a shallow well dug into the subaqueous substrate with a slotted-plastic casing. All thermal recorders were synchronized and recorded data at 15-minute intervals.

Overall, the thermal recorders operating in the east estuary showed diurnal fluctuations in temperature and a general cooling trend from summer into autumn and early winter (fig. 4.19). Additional temperature data collected at the Port Angeles tidal station (NOAA 9444090) showed that water temperature in the east estuary was closely coupled to seasonal air temperature, but was largely independent from marine water temperature in the Strait of Juan de Fuca. From August 3, 2009, to August 8, 2009, the lowest temperatures and the largest diurnal fluctuations were at TStn 5 (located where Bosco Creek enters ES2; fig. 4.2). Similarly, the next lowest recorded temperature values in the east estuary were measured at TStn 4, closest to the outlet channel (fig. 4.2). Diurnal temperature variation decreased from the summer months to the autumn.

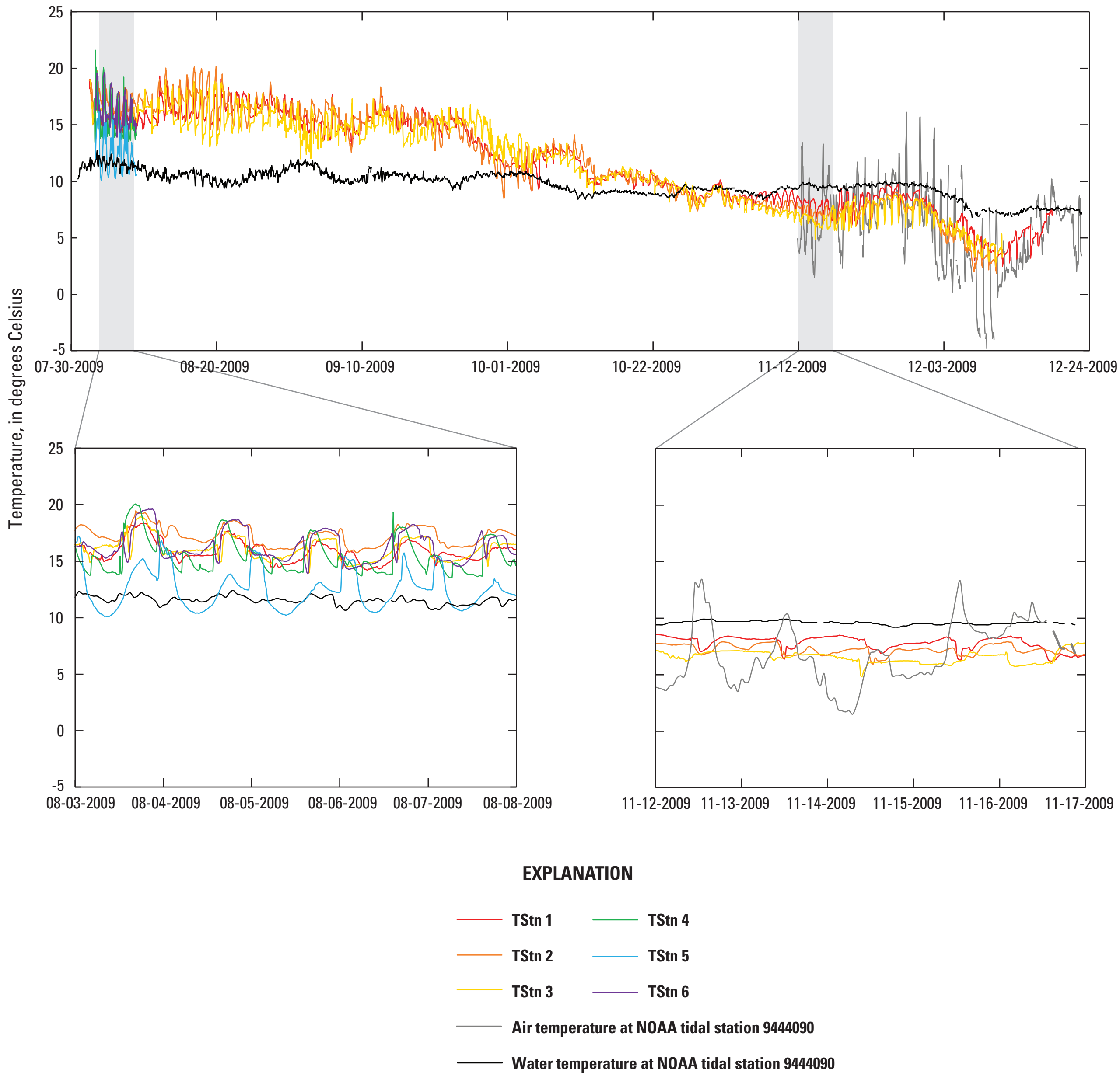

Figure 4.19. Surface-water temperature data collected in the eastern estuary of the Elwha River, Washington, July–December 2009.

Surface-Water Movement in the Estuary

To understand and quantify the flux of surface water to and from the east estuary, water velocity and depth were monitored in the ES1 outlet channel from June 3 to 13, 2008, and from August 3 to September 2, 2008. A 3-MHz acoustic Doppler velocity meter was mounted to the bottom of the ES1 outlet channel (fig. 4.2), and water velocity, stage, and temperature were recorded at 5-minute intervals. During the period of continuous acoustic Doppler velocity meter operation, discrete discharge and channel-geometry measurements were made at the ES1 outlet channel over incoming and outgoing flows, and used to develop index-velocity and stage-area relations needed to compute a continuous discharge record at the site. Positive discharge was defined as flow out of the east estuary; negative discharge as flow into the east estuary.

On four occasions during summer 2008 (June 3, June 13, July 16, and August 3), instantaneous discharge was measured at the Bosco Creek gaging station (12046523). In addition to periodic flow measurements, stage in Bosco Creek was continuously monitored during the study period with a 5-psi non-vented transducer, which was corrected for atmospheric pressure changes using a collocated barometric pressure recorder.

From June 3 to 13, 2008, the average net discharge of water from the ES1 outlet channel was 0.26 m^3/s. Surface-water discharge into the estuary from Bosco Creek was 0.09 and 0.08 m^3/s on June 3 and 13, respectively. The stage record at Bosco Creek showed a gradual decrease in stage during this period. Assuming that precipitation and evapotranspiration losses from the estuary were small, the Bosco Creek contribution accounted for about one-third of net surface-water flow to the estuary. The net discharge from the ES1 outlet channel was calculated to be 0.12 m^3/s for the sampling period August 3 to September 2, 2008. Average discharge from Bosco Creek, measured on July 16 and August 3, was 0.04 m^3/s, about one-third of the total net flux at the ES1 outlet channel in July and August. These results suggest that most of freshwater contributions to the estuary were from hyporheic flow, a mixture of shallow groundwater and subsurface river flow originating upstream and discharging into the estuary system. Overall, the discharge measurements at the ES1 outlet channel ranged from -0.30 to 1.12 m^3/s, demonstrating the close connection between tidal potential and surface-water from the estuary.

A 2-week record of these measurements in August 2008 shows the cyclical nature of water flux at the estuary outlet (fig. 4.20). The discharge of water into ES1 (vertical shading; fig. 4.20) occurred over a shorter duration, and at a smaller rate, than outflow. The resulting net discharge exiting ES1 to the river averaged about 14,000 m^3/d, or 0.17 m^3/s for the period recorded. Instantaneous peak discharge during rising-tide conditions exceeded 2 m^3/s, which is more than 10 times the net flow rate. This net outflow through the ES1 outlet channel represented the relative contributions of surface-water flow from Bosco Creek and groundwater flow into the estuary.

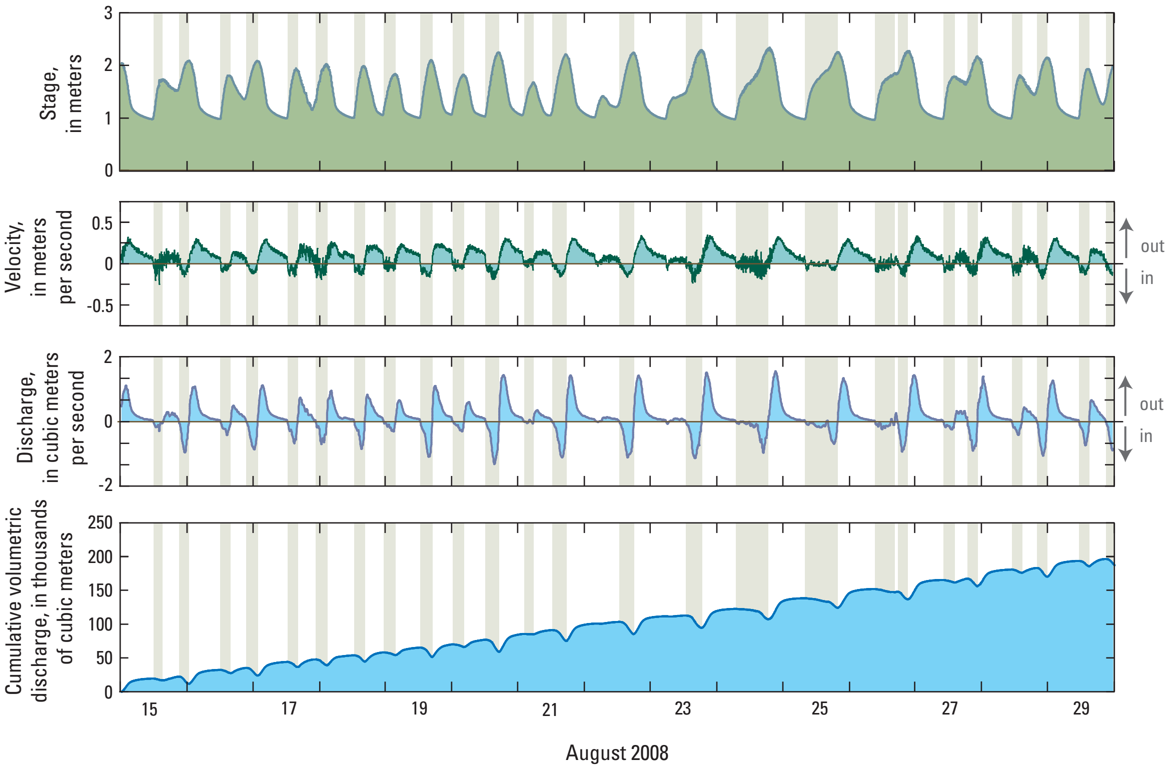

Figure 4.20. Measured stage, velocity, and discharge of water into and out of the east estuary through the ES1 outlet channel in the Elwha River estuary, Washington, during 2 weeks in August 2008. Shaded vertical bands show intervals of time with inflow into the estuary. The average net outflow was about 14,000 cubic meters per day (0.17 cubic meters per second).

Synthesis of Estuarine Hydrologic Data

Integrating the salinity, temperature, depth, and discharge data collected during this study allows insight into water movement and exchange in estuarine water bodies. During much of the year, water response in the estuary is governed by tidal cycles and groundwater and surface-water input from the aquifer and river. When storms affect the region, waves at the shoreline and high discharge within the river can alter estuarine water conditions for days or weeks.

General Water Movement within the Estuary Complex

The observations of water level, temperature, and salinity from the sensor network showed how hydrologic conditions varied across the estuary. For example, water level and salinity from measurements west and east of the river mouth revealed contrasting conditions during the summer of 2009. Although water levels in Dudley Pond and ES2 responded to tidal fluctuations in the Strait of Juan de Fuca, water level oscillations in Dudley Pond were smaller than ES2 (fig. 4.16). Furthermore, the western flood-control levee had hydraulically isolated Dudley Pond from direct connection with the river and Strait of Juan de Fuca, which limited water exchange to groundwater seepage and direct overwash of the beach berm from large waves. Groundwater exchange was responsible for most of the fluctuations in Dudley Pond water levels, shown by the lag between the water levels in the Strait of Juan de Fuca and in Dudley Pond (fig. 4.16*A*); in contrast, ES2 had a direct surface-water connection with the river mouth, shown by the close correlation in stage during the higher tides in the Strait of Juan de Fuca (fig. 4.16*B*).

These different hydrologic settings across the estuary also influenced the mixing of seawater throughout this system, shown by the substantial differences in the values and patterns of salinity across measurement sites. Dudley Pond, which was not connected to the Strait of Juan de Fuca by surface-water, had lower and more constant salinity, as opposed to WESC, on the river side of the levee, where there was more mixing of freshwater and seawater than in Dudley Pond (figs. 4.16 and 4.17). Similarly, the greater summer warming in Dudley Pond as compared to ES2 (fig. 4.17) also suggested a lack of surface-water connection to Dudley Pond.

Within the WESC, high water levels in the Strait of Juan de Fuca induced rapid increases of salinity which then decreased somewhat steadily until the next high water. This freshening effect between high waters likely was from hyporheic input. The greater oscillations in salinity in the east estuary (ES1 and ES2) resulted from tidal flows through the ES1 outlet channel (fig. 4.16). The spatial gradient in salinity values from ES1 to ES2 (greater daily oscillations in ES1 than ES2; fig. 4.16), demonstrated that the entire east estuary did not fully flush during each tidal cycle.

Surface-water temperatures in the eastern estuary (ES1 and ES2) followed the general decreasing trends measured in Dudley Pond (fig. 4.19). Diurnal variation in temperature also was greater in the summer months when clear skies allowed greater solar heating during the day and enhanced radiative cooling at night. As climatic temperatures decreased in the autumn, increased cloud cover reduced radiative heat transfer and diurnal temperature variation decreased. The cooler Bosco Creek temperatures (TStn 5; fig. 4.19) reflected the influence of spring-fed surface waters flowing down the creek. Near the outlet channel (fig. 4.2), temperatures were lower in TStn 4 and therefore were more strongly influenced by fluxes into the estuary from the river (fig. 4.19).

Different distributions of water temperature and salinity were measured within the 1 km that separates ES2 in the east estuary from Dudley Pond (fig. 4.18). Dudley Pond was warmer and less brackish than ES2. These differences were directly related to unique circulation patterns in the two water bodies, particularly the absence of hydraulic connection of surface water to Dudley Pond. Similarly, substantial differences were observed in fish diversity and abundance, invertebrate community composition, and vegetation structure between the east and west estuary (Shaffer and others, 2009; Duda and others, 2011a, chapter 7, this report; Shafroth and others, 2011, chapter 8, this report).

Effects of River Flooding and Morphologic Changes on Estuarine Hydrology

Heavy precipitation and snow melt create high discharge and stage on the lower Elwha River. Stage at the mouth of the Elwha River also rises during high tides, and these temporary changes in water levels influence water movement throughout the estuary.

The effects of the December 2007 peak-flow event, with a maximum value of 1,020 m^3/s and an estimated 40-year recurrence interval (Draut and others, 2011), were recorded in measurements in the east estuary. The high river flow lasted 2 days during which time the water levels in ES1 remained uncharacteristically high (fig. 4.21). The event reduced salinity in ES1 to nearly 0 PSU for 2 weeks, even though the pattern of fluctuation in stage in ES1 was strongly forced by tidal cycles immediately after this event (fig. 4.22).

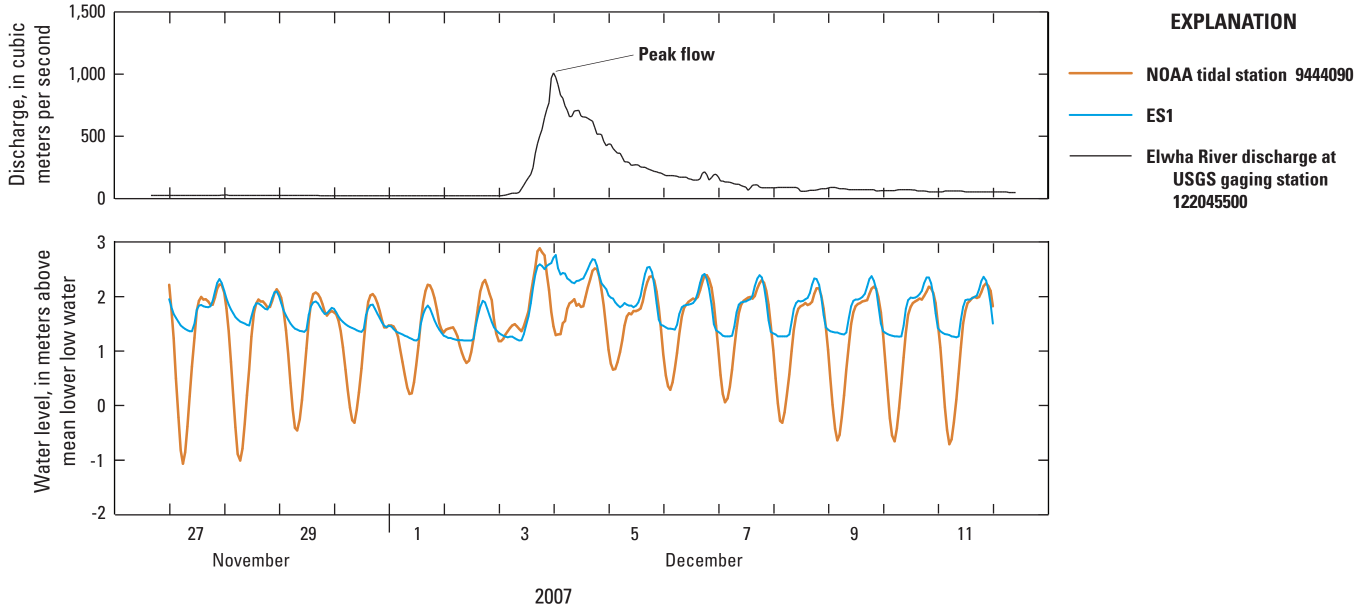

Figure 4.21. Discharge, stage in ES1 measured in the Elwha River, and tidal stage measured at National Oceanic and Atmospheric Administration (NOAA) tidal station 9444090 in Port Angeles, Washington, during the large peak-flow event of December 2007.

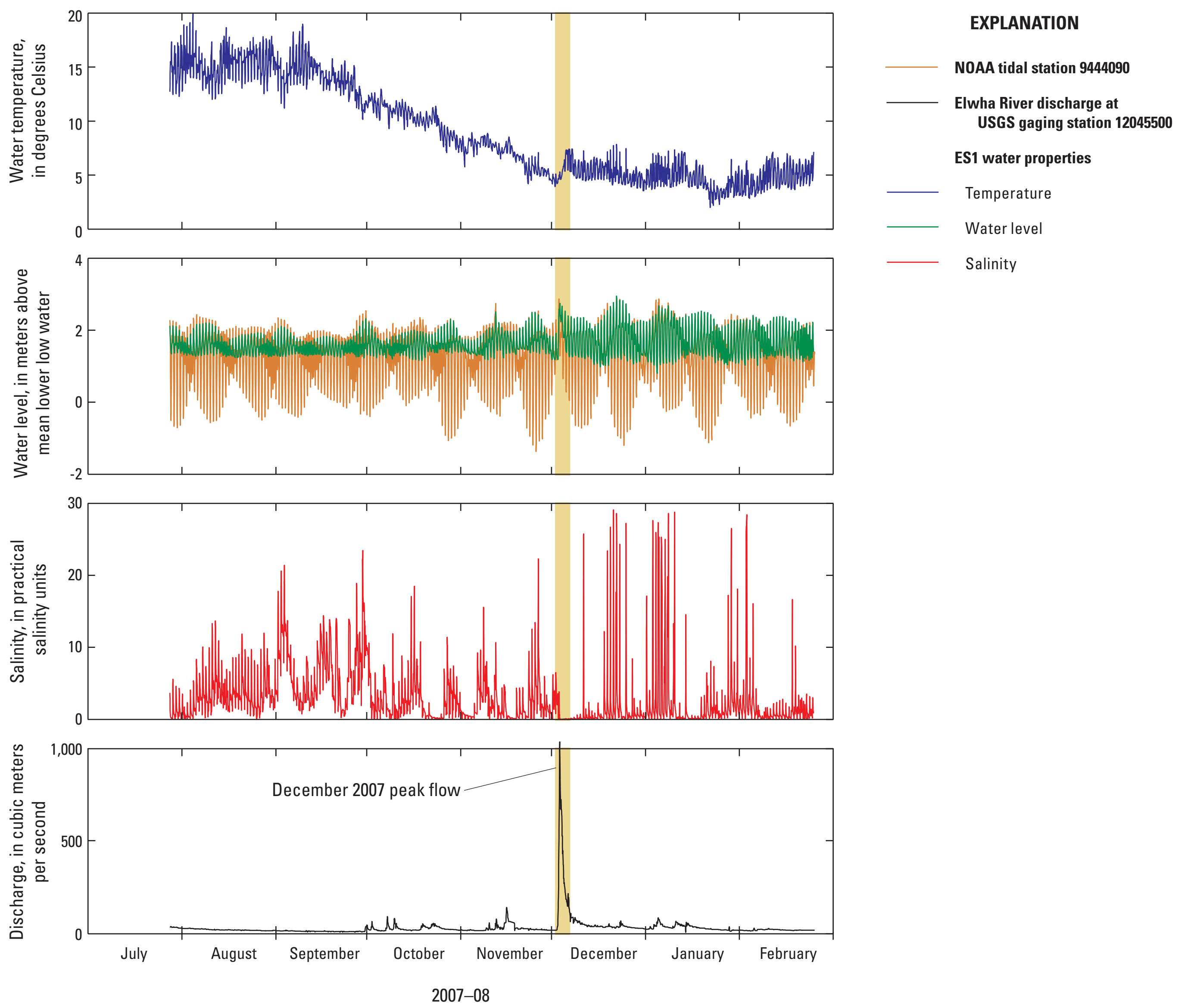

Figure 4.22. Hydrologic, temperature, and salinity conditions in the east estuary (ES1) related to the discharge in the Elwha River, Washington, from August 2007 to February 2008. The effects of the December 2007 peak-flow event on the east estuary are apparent, with decreased salinity in the east estuary for 2 weeks, and increased water levels and temperature.

This peak-flow event also modified the surface-water connection between the river mouth and ES1. During the event, 7–8 m of bank erosion and channel migration occurred in the river adjacent to the east estuary (Draut and others, 2011). This morphologic change altered the size and elevation of the ES1 outlet channel, increasing the hydraulic connection between the river and ES1. These changes were reflected in the increased amplitude of water levels in ES1 after the event (fig. 4.22). After the 2007 peak-flow event, there was approximately 1 m greater range in the cycle of stage in ES1; the stage peaks were higher, and the stage troughs were lower (fig. 4.23). Similarly, water salinity in ES1 had more variability following the 2007 peak-flow event, further reflecting larger overall volumetric exchange of water through the ES1 outlet channel (fig. 4.22).

Although the December 2007 peak-flow event expanded the ES1 outlet channel, future high flows or channel reorganization might reduce the size of this connecting channel or block it altogether. Such a cut-off event would greatly reduce estuarine exchanges in ES1 and ES2, dampening fluctuations in water level, temperature, and salinity to levels currently observed in the WESC or Dudley Pond. A blockage also would limit access to rearing habitat for young fish. For example, a large sediment deposit completely blocked access to ES1 and ES2 following a peak-flow event in November 2006. This particular estuary blockage changed the community composition and density of salmonids in 2007 (Duda and others, 2011a, chapter 7, this report).

Conceptual Model of Water Exchange in the Estuary

The observations of water conditions and flow from 2007 to 2009 provided insight into the general framework of water exchange within the river mouth and estuary. The extent and timing of these water exchanges are related to the connections of surface water and groundwater to these water bodies, and it is apparent, for the east estuary at least, that these surface-water connections can change dramatically after high flows or by channel-evolution processes.

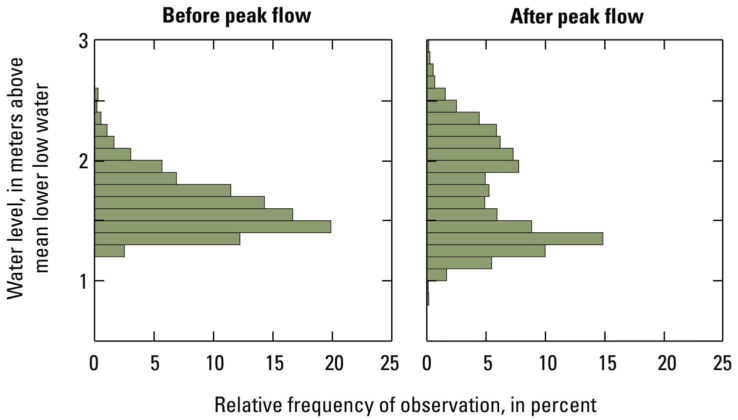

Figure 4.23. Frequency distribution plots showing a comparison of water levels in ES1, in the east estuary of the Elwha River, Washington, for the 80 days before (September 7, 2007, to November 29, 2007) and after (December 8, 2007, to February 25, 2008) the large river peak-flow event of December 2007. The wider distribution of water levels after the peak-flow event indicates an increase in the surface-water connectivity between the tidal river mouth and east estuary.

A conceptual diagram (fig. 4.24) schematically shows the movement of water through the estuary over a tidal cycle under the current pre-dam-removal conditions. Water movement, in turn, affects salinity patterns and overall water quality in the estuary (fig. 4.24). Between high and low tide, the river exports water through the river mouth. The river discharge rate and evacuation from ES1 and ES2 is greatest during the falling (ebbing) tide (fig. 4.24*B*). During the rising (flooding) tide, saline ocean water enters the river mouth and spreads into connecting estuarine channels causing increases in water salinity within the estuary (fig. 4.24*D*). The rate of water flux through these estuarine channels is governed by geometry in the ES1 outlet channel and stage in ES1, ES2, and the river. Moreover, overall magnitude of estuarine-channel water flux changes with the regular morphologic evolution of the river-mouth region (Warrick and others, 2011a, chapter 3, this report). With the imminent removal of the two dams on the Elwha River and the arrival of increased sediment load and concomitant morphological changes, the Elwha River estuary likely will experience a dynamic increase in the rate of change and evolution. At some point after dam removal, when the fluvial transport of sediment and large woody debris stabilizes, the rate of morphological change in the estuary would, presumably, reach a new equilibrium.

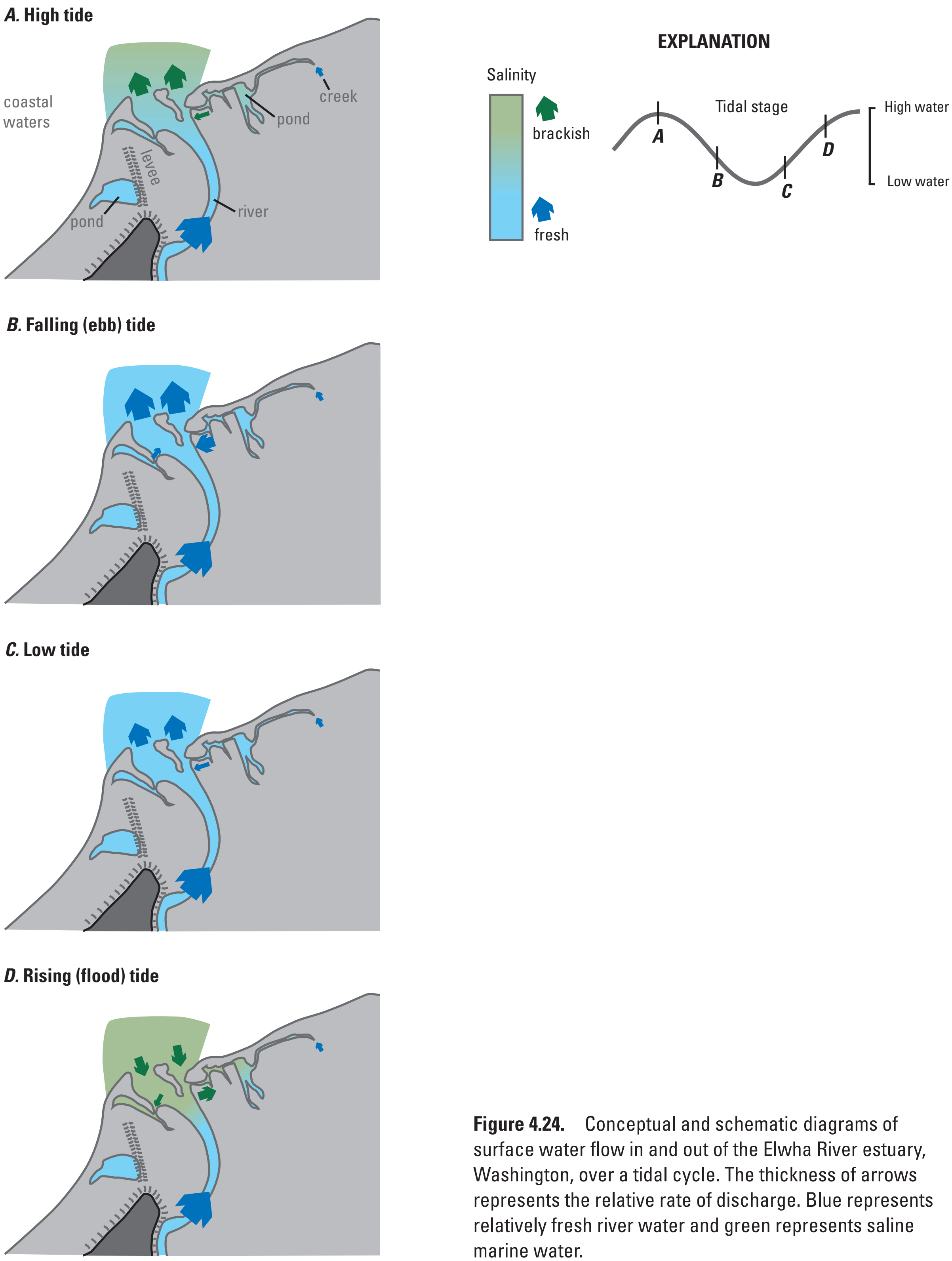

Figure 4.24. Conceptual and schematic diagrams of surface water flow in and out of the Elwha River estuary, Washington, over a tidal cycle. The thickness of arrows represents the relative rate of discharge. Blue represents relatively fresh river water and green represents saline marine water.

Summary

To better understand the hydrology of the lower Elwha River and its estuary before the removal of two dams, the U.S. Geological Survey, working with collaborators, collected baseline hydrologic data and interpreted the current hydrologic state of the lower-river system. In particular, transient storage was measured in the lower river, groundwater response and river interactions with the groundwater aquifer were characterized, salinity and stage were measured and analyzed in estuarine water bodies, and the movement of water through the estuary was analyzed and characterized.

Transient storage has been measured in many streams throughout the United States; however, due to logistical constraints, these studies are typically conducted on small headwater streams. Relatively few transient-storage studies have been conducted on larger systems comparable to the Elwha River or used to quantify changes in transient storage following a large dam removal, although theoretical models have predicted how a channel will change after small dam removals. Results from this study represent a unique dataset to assess changes in a large river before and after dam removal and the subsequent effects on channel shape and sediment supply. The dye experiment showed that the low-flow channel of the lower 3.9 kilometers of the Elwha River was relatively simple, and 1 to 6 percent of the median travel time of dye was attributed to transient-storage processes.

The monitoring of groundwater elevation in four wells located along the lower Elwha River showed a strong correlation between stage in the river and groundwater levels in the flood plain. A tidal signal also was observed in the groundwater-elevation data, but groundwater fluctuations from tides were significantly smaller than fluctuations due to changes in river stage. Water temperature data collected in the monitoring wells showed a delay of days to months between the changes in temperature in the well adjacent the river and temperature in wells farther from the river.

A seepage investigation along the lower 4.4 kilometers of river in September 2007 showed a 10 percent loss of river flow to the groundwater aquifer at a section of the river 1.7 to 2.8 kilometers upstream of the mouth. No clear trends of losses or gains were observed along the rest of the study reach.

A distributed temperature sensing cable deployed for 20 hours in the east estuary on September 1–2, 2008, characterized bed temperature of ES2, ES1, the main-stem river, and the interconnecting channels of the east estuary. Both ES1 and ES2 were warmed by solar heating during the day, and the water temperature in ES2 remained a relatively constant 14 to 15 °C. ES1 was cooled during the day and night as tidally influenced water from the river flowed into the estuary. ES2 remained relatively warm during the night hours due to its distance from the river and larger size. An analysis of the temperature variance at particular points along the cable enabled the identification of one pronounced location in the middle of ES1 where groundwater upwelling regulated the temperature of the estuary bed. The location probably is an old relict river channel. Two other possible upwelling sites were located along the cable, although the variance data could not conclusively confirm these upwelling sites.

Salinity and depth data were collected at locations in ES1, ES2, the WESC, and Dudley Pond from 2007 to 2010 helping to explain how water moved through the estuary. During summer low-flow conditions in the river, stage and salinity in ES1 and ES2 were found to fluctuate significantly in response to the rising (flooding) and falling (ebbing) tides. Salinity in ES1 in September 2009 ranged between 1 and 32 Practical Salinity Units (PSU), and salinity in ES2 ranged between 1 and 15 PSU. In contrast, fluctuations of stage and salinity in the WESC were much less pronounced. Salinity in the WESC ranged between about 3 and 11 PSU. Changes in stage were relatively small in Dudley Pond, reflecting the fact that this water body is hydraulically isolated from the river and the Strait of Juan de Fuca. Similarly, salinity in Dudley Pond was no more than 2 PSU during the low-flow conditions of late summer 2009. As the estuary was influenced by storm conditions in autumn and winter, overall salinity as well as salinity fluctuations increased in the estuary. In contrast, temperature and temperature fluctuations decreased as the weather became dominated by temperate rain storms and cloudy conditions.

An acoustic Doppler velocity meter deployed in the ES1 outlet channel in the summer of 2008, coupled with discharge and stage measurements, allowed the measurement of the total discharge of surface water flowing from the east estuary. Although discharge moved both in and out of the estuary, depending on river stage governed by tidal cycles, the net flow of surface water was out of the east estuary. In June, the net outflow from the east estuary averaged 0.26 m^3/s. In August and September, the net outflow averaged 0.12 m^3/s. Contemporaneous measurements of discharge in Bosco Creek showed that the surface-water inflow into the east estuary was about one-third the net outflow through the ES1 outlet channel, both in June and in August to September. It was presumed that the remaining two-thirds of extra inflow originated from groundwater flow into the estuary.

The river's influence on estuarine salinity and stage increased dramatically in autumn and winter as peak-flow

events in the main-stem river, coupled with tidal effects, pushed large volumes of fresh water into the east estuary. For example, a large peak-flow event in the river in December 2007 reduced salinity in ES1 to nearly zero for two weeks.

The data collected within and adjacent to the estuary helped to explain some of the complex interactions of saline marine water, fresh river water, and fresh groundwater under seasonal conditions. This insight led to the formulation of a simple conceptual model of water movement in the estuary that will help guide future research efforts focused on understanding river and estuary response to dam removal.

The underlying hydrologic framework of the lower Elwha River, its mouth, and the estuarine system of coastal ponds and interconnected channels reflects a complex set of fluvial, marine, geologic, climatic, and anthropogenic influences. These hydrologic processes will likely change after the removal of the two, long-standing Elwha River dams. Although the opportunity to study and understand the evolution of these hydrologic processes during and after dam removal is unprecedented, the collection of baseline hydrologic data and the formulation of fundamental scientific understanding of the existing processes are essential to maximize the potential insight gained from a dam-removal project of exceptional size and complexity.

This chapter documents efforts by the U.S. Geological Survey and its collaborating partners to monitor and interpret the hydrologic system within the lower Elwha River and its estuary. More importantly, the basic research characterizing the hydrologic framework of the dam-influenced Elwha River contained herein will provide future scientists the opportunity to study processes and river and estuary responses not envisioned before the project.

Acknowledgments

Michael McHenry, Matthew Beirne, Mitch Boyd, all with the Lower Elwha Klallam Tribe, contributed expertise, advice, and data to support this study.

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Chapter 5

Coastal Processes of the Elwha River Delta

By Jonathan A. Warrick, Andrew W. Stevens, Ian M. Miller, and Guy Gelfenbaum

Abstract

To understand the effects of increased sediment supply from dam removal on marine habitats around the Elwha River delta, a basic understanding of the region's coastal processes is necessary. This chapter provides a summary of the physical setting of the coast near the Elwha River delta, for the purpose of synthesizing the processes that move and disperse sediment discharged by the river. One fundamental property of this coastal setting is the difference between currents in the surfzone with those in the coastal waters offshore of the surfzone. Surfzone currents are largely dictated by the direction and size of waves, and the waves that attack the Elwha River delta predominantly come from Pacific Ocean swell from the west. This establishes surfzone currents and littoral sediment transport that are eastward along much of the delta. Offshore of the surfzone the currents are largely influenced by tidal circulation and the physical constraint to flow provided by the delta's headland. During both ebbing and flooding tides, the flow separates from the coast at the tip of the delta's headland, and this produces eddies on the downstream side of the headland. Immediately offshore of the Elwha River mouth, this creates a situation in which the coastal currents are directed toward the east much more frequently than toward the west. This suggests that Elwha River sediment will be more likely to move toward the east in the coastal system.

Introduction

Dam removal on the Elwha River will result in fundamental increases in sediment delivery to the Strait of Juan de Fuca (Strait). To understand and predict the effects of this increased sediment supply on marine habitats around the delta, a basic understanding of coastal processes is necessary. This chapter synthesizes information about the physical setting of the Elwha River delta to provide a general understanding of the coastal processes that will determine future changes to the coastal ecosystems in and around the Elwha River delta after dam removal.

Coastal Setting

The Elwha River discharges into the Strait of Juan de Fuca, a waterway between the Salish Sea and the Pacific Ocean (fig. 5.1). The Elwha River delta is sheltered from much, but not all, of the waves and swells from the Pacific Ocean, although it is exposed to the abundant and energetic exchange of water between the Salish Sea and the Pacific Ocean. This setting—sheltered from the ocean and yet exposed to strong tidal and estuarine circulation—is a defining feature of the Elwha River delta.

Another defining feature is the broad and shallow submarine delta that extends several kilometers offshore from the present shoreline (fig. 5.1*B*). The origins of this submarine delta are discussed in Warrick and others (2011, chapter 3, this report). This feature strongly influences the waves and coastal currents that act upon this section of coast. Acoustic, video, and physical sampling techniques have shown that the submarine delta seafloor is composed of coarse sand to boulders (Warrick and others, 2008). No fine-grained sediment has been deposited on the submarine delta, although deposits of fine-grained sediment are expected following dam removal (Warrick and others, 2008).

Another element of the Elwha River delta setting is the fundamental difference between processes and currents within the surfzone and within the coastal zone offshore of the surfzone (fig. 5.2). The fundamental difference between these zones is the relative importance of waves in dictating currents. Near the beach, breaking waves provide a dominant influence on currents (Komar, 1998). Waves in the surfzone are not only responsible for the back-and-forth motions of water as each wave passes, but also can drive currents along the beach when waves approach at an oblique angle (fig. 5.2). These wave-generated currents are the primary drivers of alongshore beach sediment movement in littoral cells (Komar, 1998).

Offshore from the surfzone, waves still impart substantial back-and-forth movements in the water-column and along the seafloor, but the currents primarily are driven by tidal, wind-generated, or other coastal currents. This may result in a situation, such as shown in figure 5.2, in which currents in the surfzone and the coastal zone are in opposing directions. This situation would cause differences in the transport directions of water and sediment in these zones. Although shear will not always be present between the surf- and coastal zones, transport within these zones will be produced by different, and largely independent, processes. For the Elwha River delta, the surfzone extends only several to tens of meters off of the shoreline depending on wave height and tidal stage. More information about these processes is available in the reviews of Nittrouer and Wright (1994) and Komar (1998).

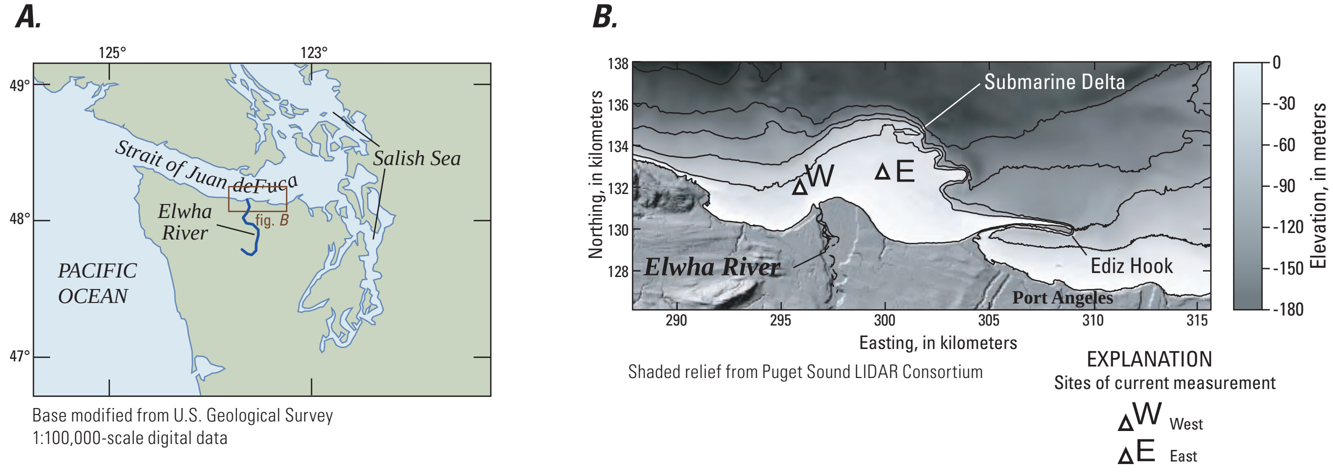


Figure 5.1. Elwha River and Strait of Juan de Fuca, near Port Angeles, Washington. (*A*) The Strait of Juan de Fuca is the primary waterway between the Salish Sea and the Pacific Ocean. (*B*) The local topography and bathymetry of the Elwha River delta.

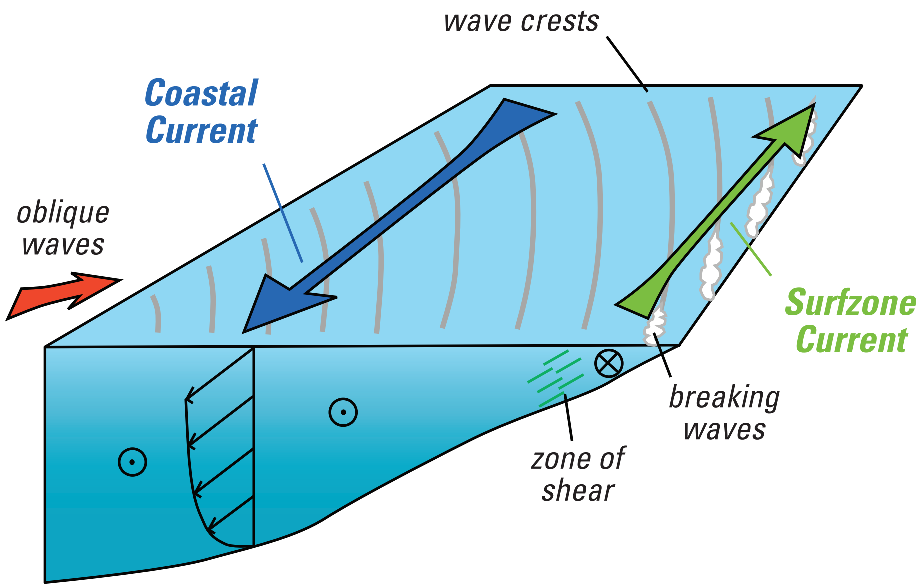


Figure 5.2. Schematic diagram showing alongshore currents of nearshore waters. Surfzone currents (green arrow) are caused by breaking waves when they have oblique approach angles. The coastal currents (blue arrow) offshore of the surfzone are influenced by tidal, wind-driven, or other regional currents and may or may not be in the same direction as the surfzone currents.

Waves

The Strait of Juan de Fuca has an intermediate wave climate between the energetic outer coast exposed to the Pacific Ocean and the more placid conditions of the Salish Sea (Shipman, 2008), although few measurements of the wave conditions within the Strait of Juan de Fuca are available to corroborate this understanding. To better characterize waves, the U.S. Geological Survey (USGS) installed acoustic Doppler current profilers (ADCP) along the seafloor at two sites at about 10 m water depth, and sampled waves hourly between January 26 and May 2, 2006 (fig. 5.1*B*). Although limited in their duration, these measurements provide important information about the general wave conditions at the delta.

Waves heights at the Elwha River delta regularly exceeded 1 m and occasionally exceeded 2 m (fig. 5.3*A*). In contrast, waves of the outer coast of Washington exceeded 10 m during this same winter-to-spring season (Ruggiero and others, 2010). The wave heights at the Elwha River delta rose and fell rapidly in response to locally generated wind waves and swell from in the Pacific Ocean (fig. 5.3). Wave heights in excess of 1 m generally lasted for only a day or less (fig. 5.3), and wave heights were most commonly less than 0.5 m (fig. 5.4*A*).

The narrow shape of the Strait of Juan de Fuca causes waves to be focused into a narrow northwestern directional band. For example, more than 90 percent of the wave energy that approached the Elwha River delta during our observations came from the northwest (fig. 5.5). Warrick and others (2009a) provide further evidence that northwest wave directions dominate all seasons of the year.

The waves at the Elwha River delta also are derived from a combination of local wind-generated waves from the Strait and swell from the Pacific. Because of the shape and size of the Strait of Juan de Fuca, the Elwha River delta “fetch” (the distance over which wind can blow to generate waves) is about 70 km. One effect of this 70-km fetch is that waves generated within the Strait will be limited to periods of less than 8 seconds (Komar, 1998; U.S. Army Corps of Engineers, 2002). Waves with longer periods come from Pacific Ocean swell, where the fetch is much longer. Our records of waves at the Elwha River delta show combined influences from low period (4–8 seconds), locally generated wind waves and longer period (greater than 8 seconds) ocean swell (fig. 5.3*B*). The most common condition measured consisted of wave periods in excess of 8 seconds (fig. 5.4*B*), which indicates that Pacific Ocean swell dominate wave conditions more frequently than wind waves.

Because the waves at the Elwha River delta originate from the northwest direction, these waves break at oblique angles along much of the shoreline, and these obliquely breaking waves drive surfzone currents and littoral sediment transport. Observations of wave breaking angles have been obtained from airborne light detection and ranging, or “lidar,” of the ocean water surface, historical aerial photography of the beach, and numerical modeling of wave propagation in the Strait (fig. 5.6). Combined, these observations show that the northwest wave approach imparts normally aligned waves on the western side of the delta and oblique wave breaking angles on the eastern side (fig. 5.6). Thus, transport of littoral sediment should be negligible west of the river mouth and eastward on the eastern side of the river mouth. This conceptual model for littoral sediment transport is consistent with recent measurements of littoral sediment movement, which suggests that sediment can move more than 100 m/d along the east beach under the strongly oblique wave directions (see Miller and others, 2011, sidebar 5.1, this report).

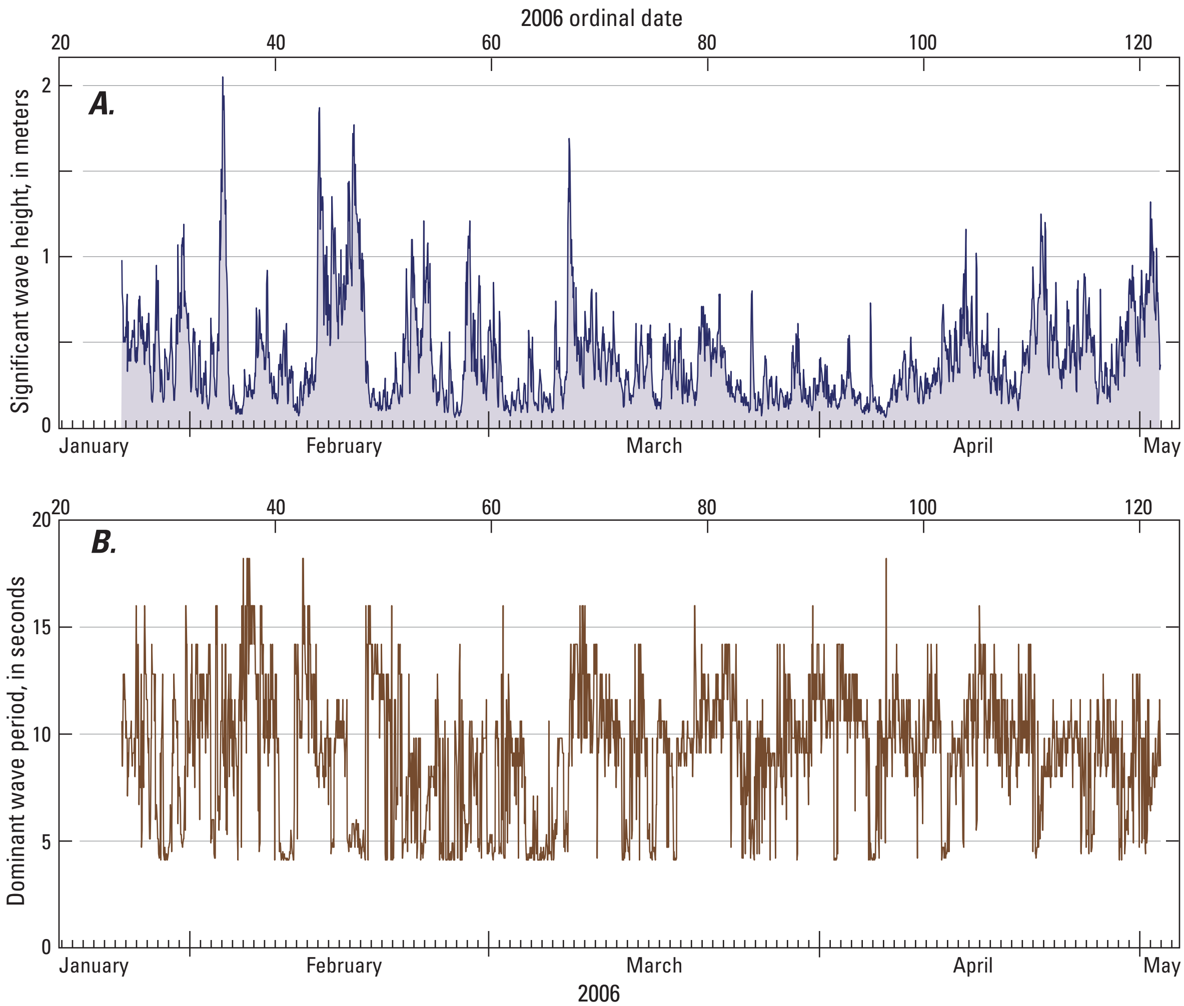


Figure 5.3. Wave height and wave period measurements from the western side (site W) of the Elwha River delta, Washington, during 2006.

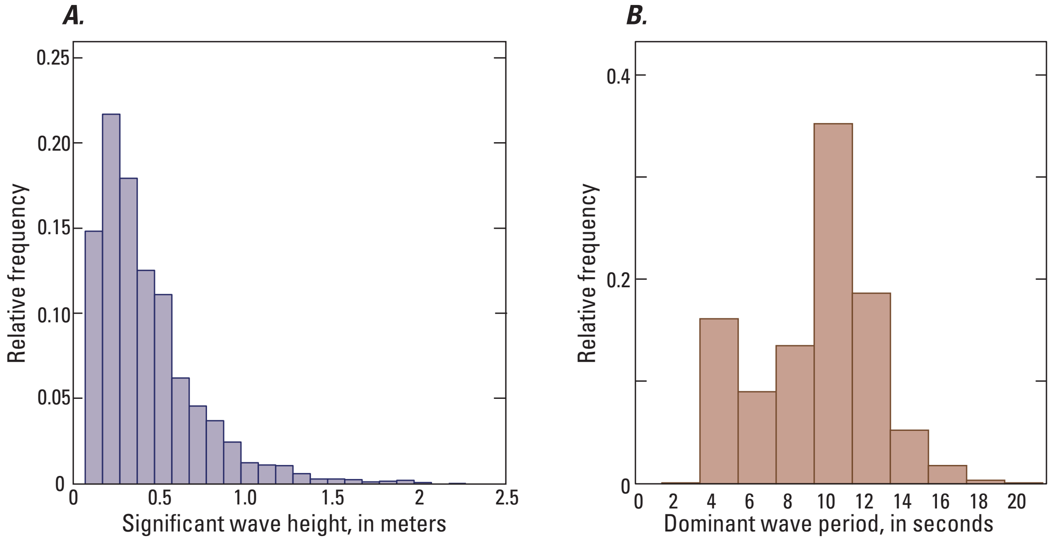


Figure 5.4. Histograms of significant wave height and dominant wave period from observations at the western side (site W) of the Elwha River delta, Washington. Waves generally are less than 1 meter with periods greater than 8 seconds, indicating Pacific Ocean origins. Location of site is shown in figure 5.1.

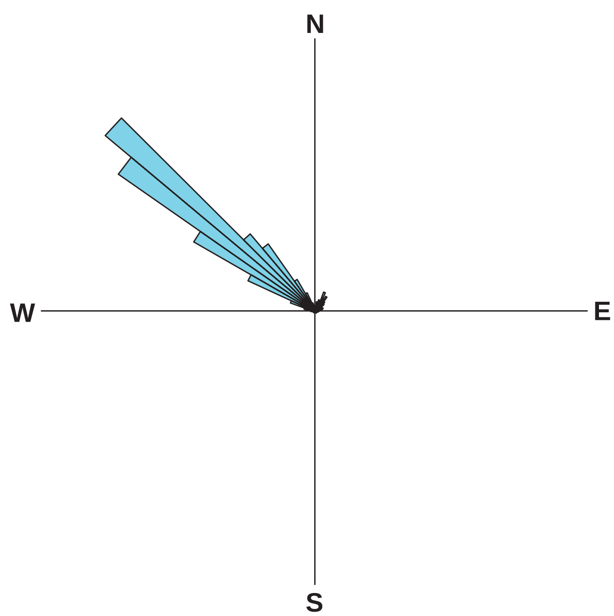


Figure 5.5. Rose diagram showing dominant wave direction from observations at the western side (site W) of the Elwha River delta, Washington. More than 90 percent of the observations were dominated by waves from the northwest, which is the direction of approach for waves generated in either the Pacific Ocean or the western Strait of Juan de Fuca.

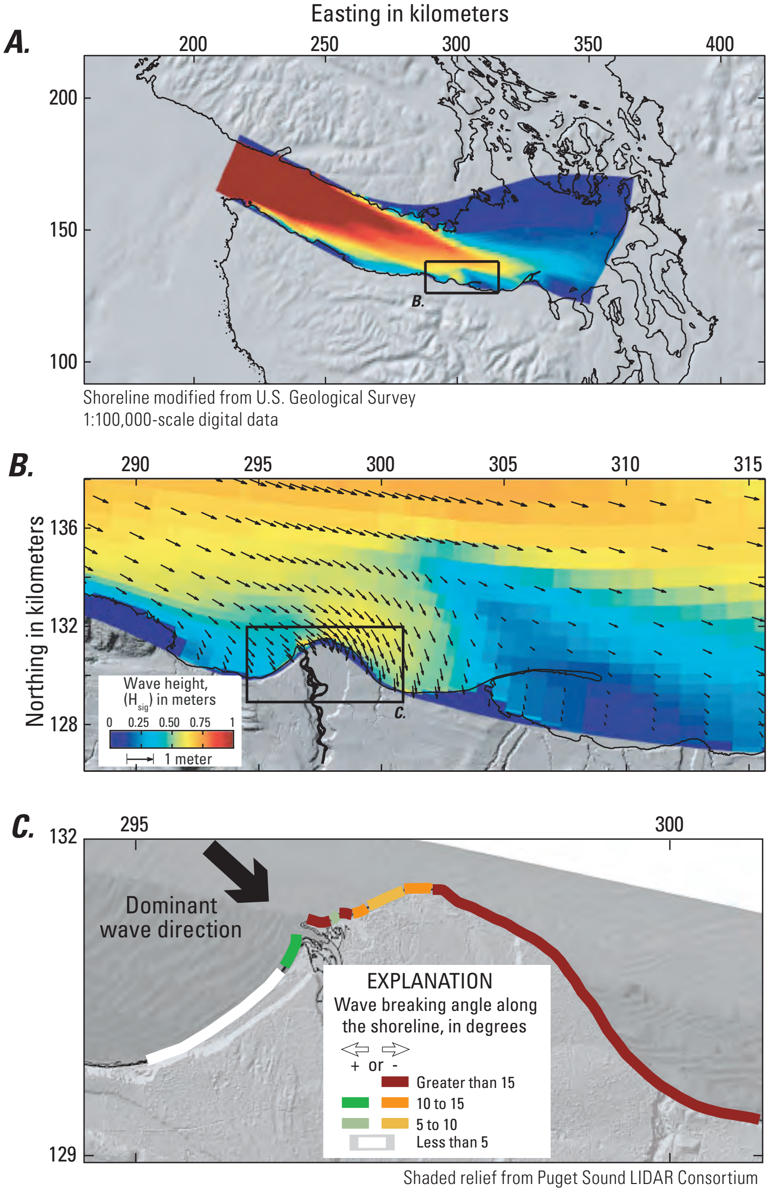


Figure 5.6. Wave patterns for the Strait of Juan de Fuca and Elwha River delta, Washington, from measurements and model simulation results. (*A*–*B*) The transformation of wave heights and directions in the Strait from 6 meter Pacific Ocean swell as predicted by the SWAN numerical model (after Gelfenbaum and others, 2009). Arrows in (*B*) show modeled wave directions. (*C*) Measurements of the angle of breaking waves along the Elwha River delta shoreline during northwest swell (black arrow) from lidar and aerial photographs (after Warrick and others, 2009).

Sidebar 5.1 Tracking the Movement of Cobble on the Elwha Beach

Ian M. Miller and Jonathan A. Warrick

It is challenging, and yet important, to measure and understand sediment transport rates and directions on a mixed grain-size beach like the one along the Elwha River delta. To directly measure these sediment movements, Radio Frequency Identifier (RFID) tags have been placed into native pebbles, cobbles, and small boulders, and their movements have been tracked over various sites, time scales, and wave conditions. RFID tags (fig. S5.1*A*) are small passive transponders used frequently in tracking fish and applied more recently to the study of bedload transport in rivers and beaches. The device contains a small chip and a capacitor, which will absorb energy transmitted by a nearby radio antenna and, using the power stored in the capacitor, transmit a unique numerical code back to the antenna. Small (23 mm long), durable glass encapsulated RFID tags were implanted directly into native beach clasts to create the tracers; the only evidence of the RFID tag is a small epoxy cap visible on the surface of the rock (fig. S5.1*B*). An easily carried mobile antenna was used to re-locate and identify individual clasts buried in the beach (fig. S5.1*C*–*D*).

Application of the RFID technique resulted in 80–100 percent recovery of the tracers placed on the beach. To the east of the Elwha River mouth, tracer trajectories indicate that there is consistent movement to the east (fig. S5.2). By contrast, tracer samples placed on the beach west of the river mouth moved in both alongshore directions (Miller and others, 2011). Wave height, and therefore the energy contained in the waves, seems to be the best predictor of the overall velocity of a sample of tracers on the beach over a 24 hour interval (fig. S5.2). The velocities of these tracer sediments also have been surprisingly high. During the 0.64 m wave heights of February 6-7, 2009, gravels and cobbles were measured moving more than 100 m during a tidal cycle (fig. S5.2*C*).

Figure S5.1. Methods used to track cobble movement on the beach with Radio Frequency Identifier (RFID) tagged clasts, Elwha River delta, Washington. (*A*) A single 23-millimeter long RFID tag. (*B*) A set of completed tracers ready for placement. The epoxy cap is barely visible on each tracer and is the only permanent indication of the RFID tag inside. (*C*) The mobile reader unit, with a hand-held antenna. The battery and unit are carried in a backpack. (*D*) A recovered clast (at the base of the orange stake) was buried approximately 25 centimeters.

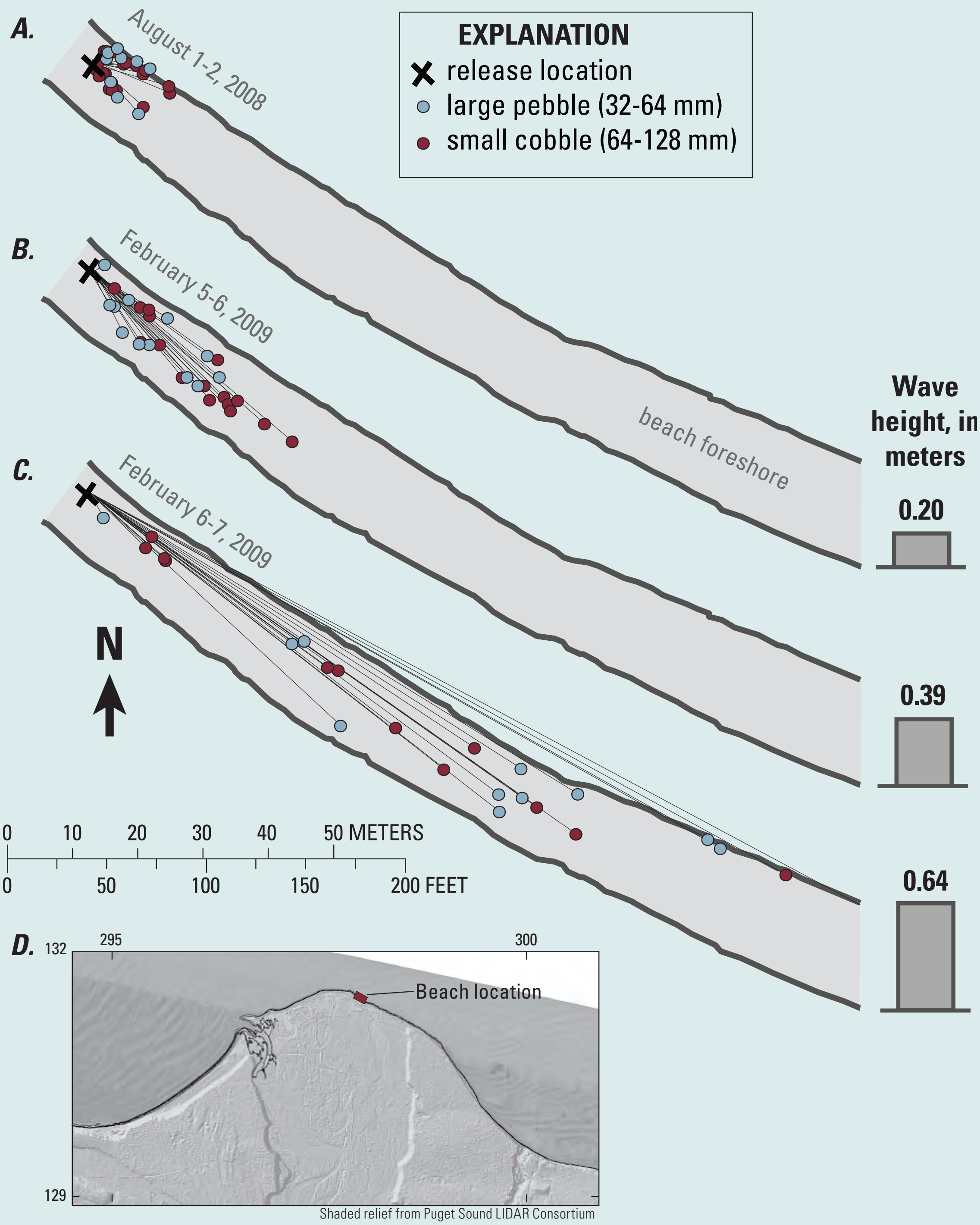


Figure S5.2. Diagrams and map showing examples of three 24-hour releases of the Radio Frequency Identifier tracers over a range of wave conditions, Elwha River delta, Washington. (*A*) August 1–2, 2008; (*B*) February 5–6, 2009; (*C*) February 6–7, 2009, (*D*) location of active foreshore and release site. Wave heights from visual estimates on August 1–2, 2008, and a pressure transducer located offshore of this site during the February 2009 measurements.

Tides and Water Levels

The water levels within and outside of the Elwha River delta dictate the estuarine flow in and out of the river mouth, the elevation of beach processes such as swash, and the potential for coastal flooding. These water levels respond to tidal and atmospheric conditions that can change greatly over the course of a day. The Strait of Juan de Fuca has a mixed, mesotidal regime (Mofjeld and Larsen, 1984; Finlayson, 2006). The "mixed" classification indicates that diurnal and semi-diurnal cycles in water levels (that is, cycles that occur approximately once and twice per day, respectively) influence the overall water level in approximately equal amounts. The "spring tidal range" is defined as the difference between the mean higher high water (MHHW) and the mean lower low water (MLLW), which are the multiple year averages of the higher high tide of each day and lower low tide of each day (Komar, 1998). The spring tidal range of the nearest tidal gauge in Port Angeles, Washington (National Oceanic and Atmospheric Administration station 9444090) is 2.15 m, which makes this region "mesotidal." The spring tidal range generally increases with distance into the Salish Sea, and the inner parts of the Puget Sound are macrotidal with spring tidal ranges greater than 4 m (Finlayson, 2006).

An example of water levels at nearby Port Angeles is shown in figure 5.7. These data reveal two important characteristics. First, the water levels reside at the higher elevations for longer intervals of time than the lower elevations (fig. 5.7). The low tides are fairly brief in time and have punctuated drops and rises. This is a common phenomenon within the Strait of Juan de Fuca and the Salish Sea, and results from the mixing of diurnal and semi-diurnal tidal components (Finlayson, 2006). Thus, the distribution of water levels at the Elwha River delta is skewed toward the higher elevations (fig. 5.8).

The second important characteristic is that water levels near Port Angeles can deviate from expected levels, especially during storm surges. Storm surges within the Strait of Juan de Fuca and the Salish Sea result from drops in atmospheric pressure as storms reside over the region, and these effects generally are greatest in winter (Finlayson, 2006). The net effect of storm surge is that water levels can be much higher than expected, and water level deviations of as much as 0.5 m (fig. 5.7) are common for the winter storms in the Strait (Finlayson, 2006).

Storm surges within the Strait of Juan de Fuca also occur during times of high winds, and therefore high waves, because the storms that cause surge also cause high winds. The combination of these conditions—storm surge and high waves—often results in substantial littoral sediment transport and coastal erosion during winter storms (Finlayson, 2006).

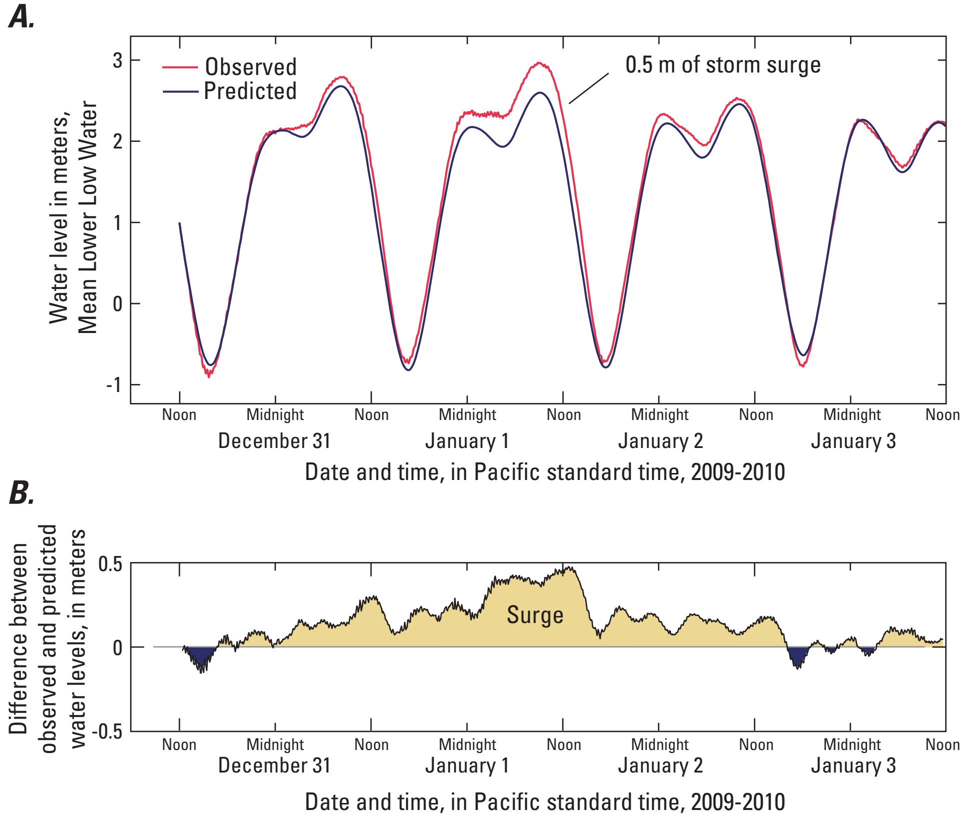


Figure 5.7. Water levels from the National Oceanic and Atmospheric Administration tidal gauge in Port Angeles, Washington (station 9444090) showing typical spring tidal range coupled with storm surge from the January 1, 2010, storm.

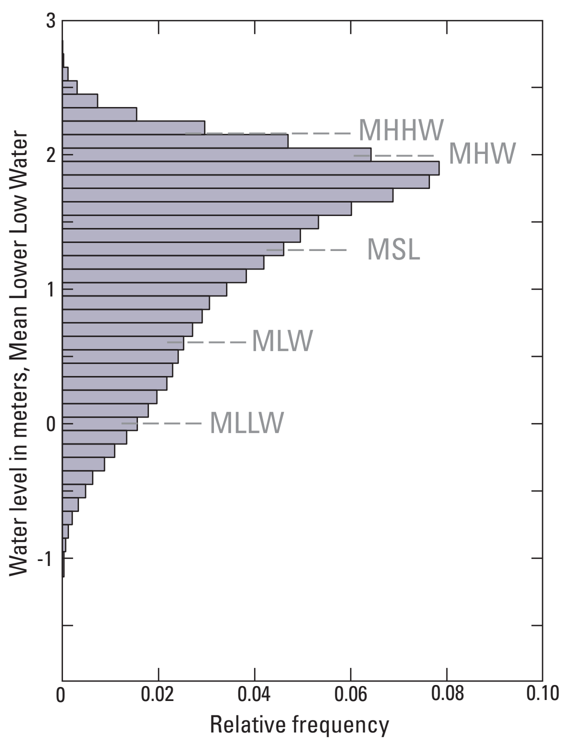


Figure 5.8. Relative frequency of water levels and tidal statistics at the National Oceanic and Atmospheric Administration tidal gauge in Port Angeles, Washington (station 9444090) showing distribution skewed toward high water levels. Data are from an 18.6 year tidal epoch.

Coastal Currents

The Strait of Juan de Fuca is the primary channel connecting the Salish Sea with the Pacific Ocean, and because of this physical setting the Strait serves as a conduit for tidal and estuarine exchange between these two water bodies. To provide the water necessary for the meso- to macrotidal conditions within the Salish Sea, strong tidal currents with speeds on the order of 100 cm/s flow in and out through the Strait daily (Holbrook and others, 1980). Much like the “mixed” diurnal and semi-diurnal characteristics of the water levels in the Strait, the strong tidal currents have diurnal and semi-diurnal components with somewhat more importance of the semi-diurnal (twice daily) cycling (Holbrook and others, 1980).

The large volume of river discharge that enters into the Salish Sea also dictates coastal currents in the Strait of Juan de Fuca. Freshwater discharged into the Salish Sea forms a freshened, buoyant layer along the upper 50 to 150 m of water depths in the Strait, and this freshened surface layer exits toward the Pacific Ocean at an average rate of about 10 cm/s (Holbrook and others, 1980; Thomson and others, 2007). Because this “estuarine” outflow of freshened water has strong vertical gradients in density (freshwater is less dense than seawater), it is characterized as a “baroclinic” flow. This flow of freshened water varies considerably over seasonal timescales, and is most intense during the spring freshet from snowmelt runoff (Masson and Cummins, 2004; Thomson and others, 2007).

To evaluate the effects of these coastal currents on circulation around the Elwha River delta, the USGS has conducted current meter measurements (fig. 5.1*B*) and developed a numerical circulation model described in Gelfenbaum and others (2009). These combined data show that the coastal currents near the Elwha River delta are strongly tidal (fig. 5.9). Most (about 90 percent) of the variance in the currents near the Elwha River delta can be explained by tidal ebbs and floods (Warrick and Stevens, 2011). These tidal currents have diurnal and semi-diurnal components, with slight dominance in the semidiurnal, or twice daily, cycling of currents.

These observations also suggest that coastal currents do not flow in a simple, uniform manner around the delta, but rather the coastal currents generate eddies as they flow past the headland of the delta (fig. 5.10). The shape of the Elwha River delta causes the flow to separate near the tip of the delta headland during ebbing and flooding, which causes a return flow, or “eddy”, on the downstream side of delta (fig. 5.10). Warrick and Stevens (2011) determined that the headland-induced eddies are regular features in the coastal currents at the Elwha River delta during spring and neap tides. These eddies also explain why there is little agreement between currents measured at opposite sides of the delta even when differences in the orientation of the currents and in the timing of the tidal responses were considered (fig. 5.9; Warrick and Stevens, 2011).

One implication of these headland-induced eddies is that coastal currents near the shore on both sides of the delta are preferentially directed toward the delta tip. For example, the coastal waters immediate offshore of the river mouth are subject to currents that flow toward the northeast more frequently and at faster current speeds than those toward the opposite, or southwest, direction (fig. 5.10). In this situation, the waters discharged by the river are subject to northeast-directed currents, which strongly affect the location of the Elwha River plume and its ecological effects.

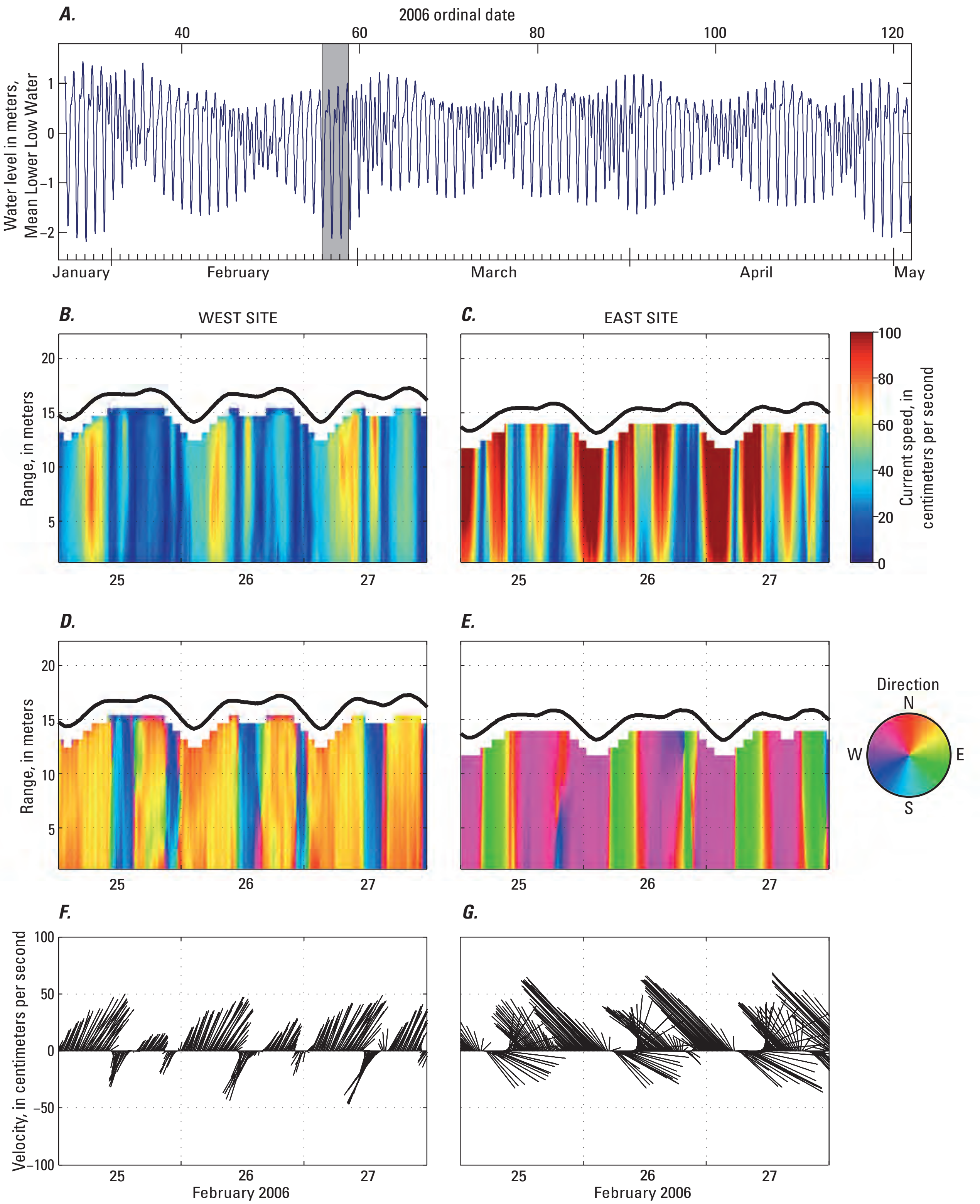
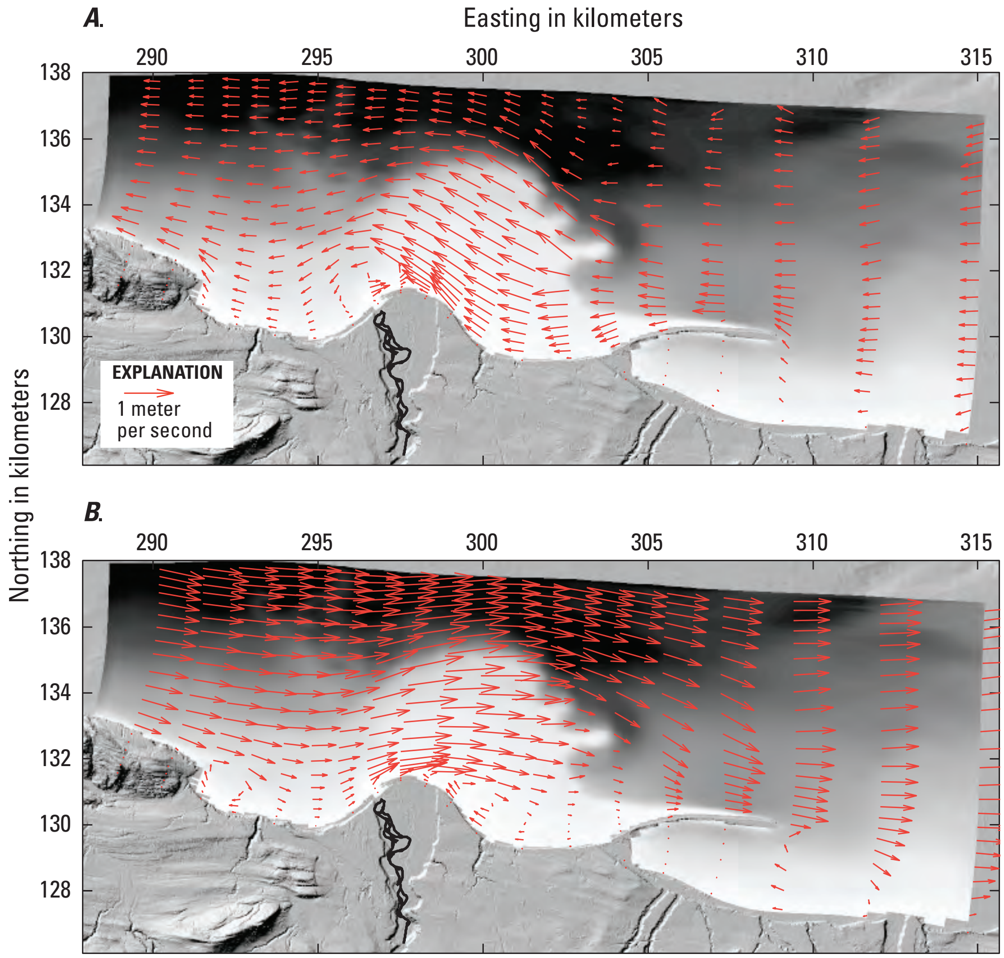


Figure 5.9. Coastal currents measured offshore of the Elwha River delta, Washington, at the West and East sites. (*A*) A four month tidal record and highlights, with a gray bar. (*B–G*) the 3-day records. Currents are shown as (*B, C*) depth-dependent speed, (*D, E*) depth-dependent direction, and (*F, G*) depth-averaged current. Although currents at both sites are strongly tidal, the magnitude, direction, and persistence of these tidal currents are not consistent.



An animation showing the simulated currents over one tidal cycle is available at http://pubs.usgs.gov/sir/2011/5120/

Figure 5.10. Coastal currents offshore of the Elwha River delta, Washington, from numerical simulations during (*A*) tidal ebbing and (*B*) tidal flooding. Note the tidal currents in excess of 1 meter per second and eddies on the downstream side of the deltaic headland.

Sidebar 5.2 Kelp-Mediated Sediment Transport

Ian M. Miller, Stephen P. Rubin, and Jonathan A. Warrick

Traditional models of sediment transport indicate that sediments will move in response to water flowing around the particle. In mixed sedimentary environments like the Elwha River delta coastal zone, however, larger clasts serve as substrate for kelps, and by doing so they make themselves easier to transport over large distances.

Numerous cases of kelp-mediated sediment transport were observed during surveys in the Elwha River coastal zone, where kelps have been attached to transported sediments of various sizes, including pebbles as small as a few centimeters to small boulders that are 40 cm or more in diameter (fig. S5.3). In some instances, the kelp-attached sediment was observed in motion due to high tidal or wave-induced currents. In other instances, trenches or disturbed sea-floor adjacent to a kelp-covered boulder were observed, indicating recent movement (fig. S5.3*B*). Windrows of kelp on the Elwha River delta beach were observed after big swells. Each kelp attached to a cobble clearly indicates that it recently was transported from deeper water (fig. S5.3*A*).

Two possible mechanisms may facilitate kelp-mediated clast transport: (1) the addition of drag as a result of the increased surface area of the kelp, or (2) the lift from buoyant kelps transmitted to the sediment through the kelp's holdfast. Non-buoyant understory kelps, such as *Cymathere triplicata* and *Pterygophora californica* are prolific in the Elwha River delta coastal zone, and have been observed attached to mobile sediments. Bull kelp (*Nereocystis leutkeana*) is one of a few buoyant species of kelp observed in the Elwha coastal zone, Bull kelps have been observed rafting cobbles above the seafloor along the Elwha coastal zone. The dominance of mixed grain substrate in the Elwha coastal zone, combined with the abundance of kelp species may, in fact, indicate that this kelp-mediated sediment transport plays an important role in transporting sediments and shaping coastal and littoral habitats.

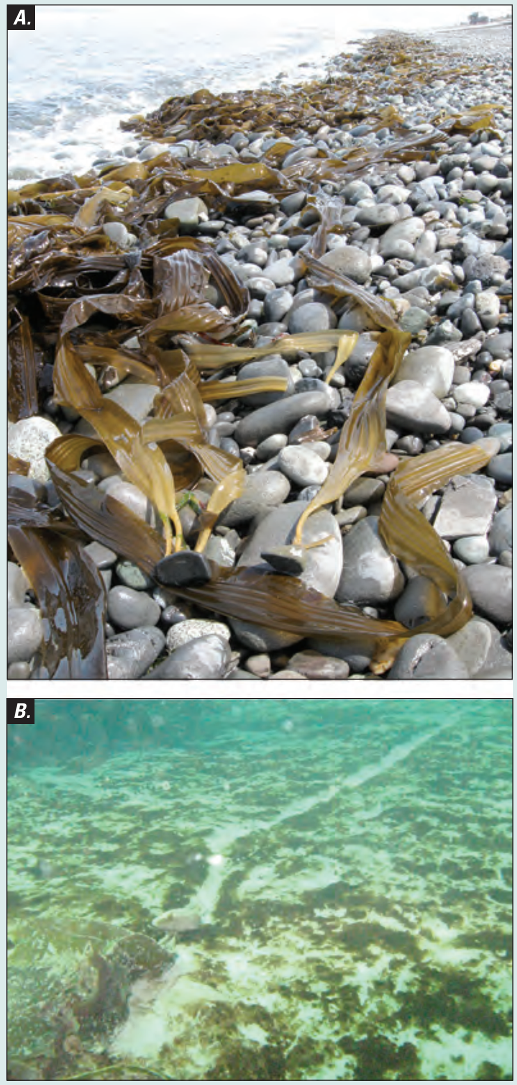


Figure S5.3. Examples of kelp-mediated clast movement from the Elwha River delta, Washington, coastal zone. (*A*) Numerous kelp-attached clasts litter the beach on June 21, 2009, about 250 meters east of the river mouth. Water level in this photograph is estimated to be 1.0 meter above Mean Lower Low Water (MLLW). The dominant kelp in these windrows is *Cymathere triplicata*, a common kelp at depths between -3.0 and -8.0 meters offshore of the delta. The two kelp-attached clasts in the near-field are estimated to be 5 centimeters along their long axes. (*B*) Active movement of an unidentified kelp attached to a cobble through algal debris at about -5.0 m MLLW depth offshore of the base of Ediz Hook on August 11, 2008. The clast under motion in this photograph is estimated to be 15 centimeters along the long axis. (Photographs taken by Ian Miller, University of California, Santa Cruz.)

Elwha River Coastal Plume

It is important to understand the patterns and dynamics of the coastal plume of suspended sediment from the Elwha River (fig. 5.11), because this plume will define the region of the coastal effects from dam removal. Numerous physical processes will influence this plume, however, which makes estimates complicated. For example, once turbid river waters reach the Strait, the freshwater of the river will be positively buoyant (that is, it will float), whereas suspended sediment particles will begin to sink toward the seafloor (Nittrouer and Wright, 1994; Hill and others, 2007). The rate of particle sinking will be related to the size, abundance, and chemistry of these particles, although for most river plume settings most of the fine sediment (silt and clay) will reside in the buoyant freshwater plume for an hour or more. Therefore, the buoyant plume can impart an initial and important transport direction and speed to the discharged sediment (Hill and others, 2007).

Repeat measurements of the position and characteristics of the Elwha River buoyant plume by Warrick and Stevens (2011) shows that the plume commonly hugs the eastern shoreline as it spreads toward the northeast, as shown in figure 5.11. The Elwha River plume is regularly directed toward the northeast because of the northeast-dominated currents that are offshore of the Elwha River mouth, which in turn are caused by eddies (fig. 5.10).

As the Elwha River discharges into the Strait, it freshens the upper 1–3 m of the coastal water column (fig. 5.12). The depth of the plume generally increases with the strength of the local current, because stronger currents mix the buoyant plume deeper (Warrick and Stevens, 2011). The rate of water discharge from the river mouth into the plume—especially during low river flow—also is a function of the tidal stage. Falling, or "ebbing," tides export river water at higher rates as described more fully in Magirl and others (2011, chapter 4, this report).

Using this information about the coastal currents and buoyant river plume, the USGS has developed a numerical model that tracks the freshwater and suspended sediment discharged by the river (Gelfenbaum and others, 2009). Model simulation results indicate that suspended sediment discharged from the river should be dynamic and largely mimic the patterns of the buoyant plume (fig. 5.13). Because the buoyant plume is directed toward the east more frequently than toward the west, the suspended sediment effects of dam removal—including but not limited to water column turbidity and sedimentation on the seafloor—should be greatest and most persistent on the eastern side of the river mouth.

Figure 5.11. Aerial photograph of the plume discharging toward the northeast from the mouth of the Elwha River, Washington. (Photograph from the Washington State Department of Natural Resources, 1994.)

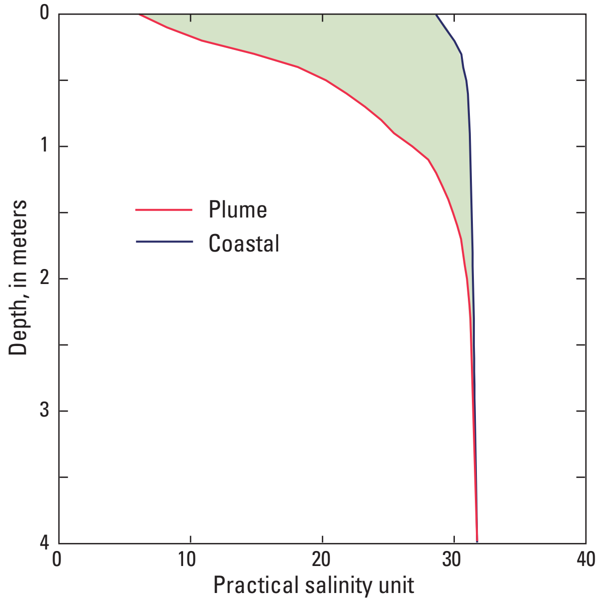


Figure 5.12. Salinity inside the buoyant plume (plume) and in coastal waters directly outside the plume (coastal), Elwha River, Washington. Measurements were taken at 1800 Pacific Standard Time on June 9, 2007, at about 0.5 kilometers offshore of the river mouth. The freshening of the upper 1.5 meters of water from the river water is shaded green. (After Warrick and Stevens, 2011.)

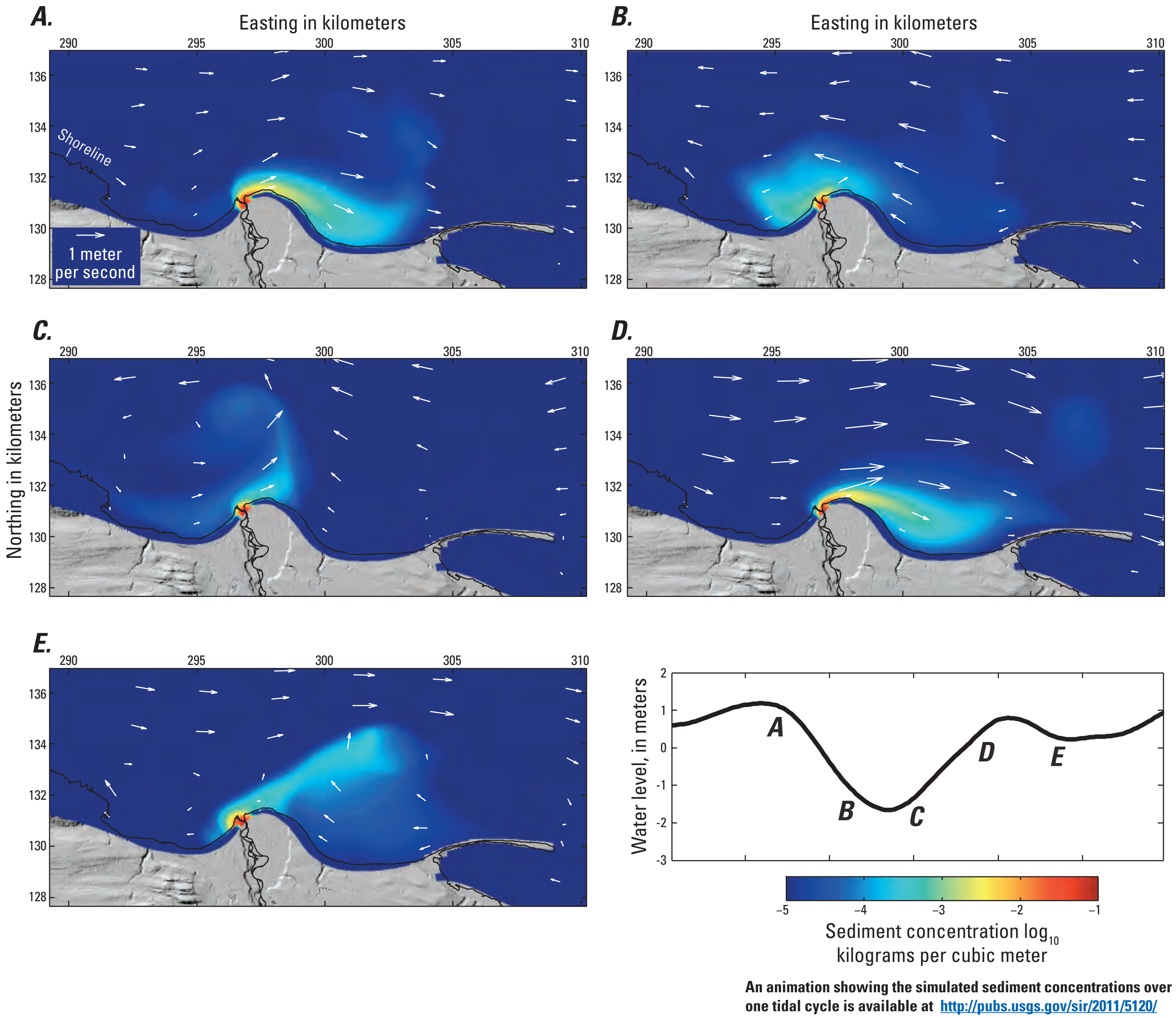


Figure 5.13. Evolution of coastal turbidity from the discharge of suspended sediment from the Elwha River, Washington, as determined by model simulation results (after Gelfenbaum and others, 2009). Arrows indicate depth-averaged flow direction and magnitude.

Summary

The coastal zone of the Elwha River delta is strongly influenced by waves from the Pacific Ocean and currents that respond to the exchange of water between the ocean and the Salish Sea. The shape and morphology of the submarine delta and shoreline also have substantial effects on how these waves and currents vary within this coastal system. Waves approach largely from the northwest, which results in oblique breaking wave directions along much of the shoreline. These oblique wave breaking angles will move littoral sediment from the river mouth toward the east. The coastal currents are largely tidal, but these currents also induce eddies at the tip of the delta that influence dominant current directions. The coastal currents immediately offshore of the river mouth are directed toward the east more frequently than to the west, which influences the direction of transport for suspended river sediment discharged by the river.

Because of these patterns, most of the sediment introduced from dam removal likely will be transported toward the eastern side of the river mouth. It is, perhaps, no coincidence that since the dams were emplaced, the eastern beach is the region of greatest coastal erosion (see Warrick and others, 2011, chapter 3, this report). This does not mean that no significant effects may occur from the river sediment toward the west of the river mouth. In fact, there are occasions during which coastal currents offshore of the river mouth are either negligible or toward the west. During these conditions, the Elwha River plume is expected to spread westward toward Freshwater Bay. If a high sediment discharge occurs during a westward spreading plume, fine sediment will be deposited in Freshwater Bay.

Although there is a good understanding of the basic physical conditions of the Elwha River delta nearshore and coastal zones, these conditions and processes have not been fully integrated into a predictive sediment transport model for the post-dam removal conditions. During the post-dam removal phase there will be a unique opportunity to measure the direction and magnitude of transport from this massive release of sediment. These observations can be used to develop, test, and improve analytical capabilities, so that future projects can benefit from these insights. Development and implementation of a strong coastal monitoring program during the post-dam removal phase and coordination of these observations with numerical modeling efforts can help in understanding the marine effects of dam removal.

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Chapter 6

Nearshore Biological Communities Prior to Removal of the Elwha River Dams

By Stephen P. Rubin, Ian M. Miller, Nancy Elder, Reginald R. Reisenbichler, and Jeffrey J. Duda

Abstract

Increases in sediment delivery to coastal waters are expected following removal of dams on the Elwha River, Washington, potentially increasing sediment deposition on the seafloor and suspended sediment in the water column. Biological communities inhabiting shallow, subtidal depths (3–18 m) near the mouth of the Elwha River, between the west end of Freshwater Bay and the base of Ediz Hook, were surveyed in August and September 2008, to establish baselines prior to dam removal. Density was estimated for 9 kelp taxa, 65 taxa of invertebrates larger than 2.5 cm any dimension and 24 fish taxa. Density averaged over all sites was 3.1 per square meter (/m^2) for kelp, 2.7/m^2 for invertebrates, and 0.1/m^2 for fish. Community structure was partly controlled by substrate type, seafloor relief, and depth. On average, 12 more taxa occurred where boulders were present compared to areas lacking boulders but with similar base substrate. Four habitat types were identified: (1) Bedrock/boulder reefs had the highest kelp density and taxa richness, and were characterized by a canopy of *Nereocystis leutkeana* ***(bull kelp) at the water surface and a secondary canopy of perennial kelp 1–2 m above the seafloor; (2) Mixed sand and gravel-cobble habitats with moderate relief provided by boulders had the highest density of invertebrates and a taxa richness nearly equivalent to that for bedrock/boulder reefs; (3) Mixed sand and gravel-cobble habitats lacking boulders supported a moderate density of kelp, primarily annual species with low growth forms (blades close to the seafloor), and the lowest invertebrate density among habitats; and (4) Sand habitats had the lowest kelp density and taxa richness among habitats and a moderate density of invertebrates. Uncertainties about nearshore community responses to increases in deposited and suspended sediments highlight the opportunity to advance scientific understanding by measuring responses following dam removal.***

Introduction

Two dams on the Elwha River have reduced sediment transport from the upper watershed to the lower river and coast for nearly 100 years (Duda and others, 2011, chapter 1, this report). Increases in sediment delivery to coastal waters are expected following dam removal, potentially increasing sediment deposition on the seafloor and suspended sediment in the water column. Large increases are expected initially (3–5 years after initiation of dam removal; Czuba and others, 2011, chapter 2, this report) as sediments that have accumulated behind the dams are released. Thereafter, sediment supply likely will decrease but should remain higher than before dam removal due to restored transport of sediments from the upper watershed. Much of the sediment currently impounded by the dams is silt, sand, and clay, which are readily transportable (Czuba and others, 2011, chapter 2, this report). Spatial patterns of suspended and deposited sediment in coastal waters will depend on local physical processes including waves and currents (Warrick and others, 2011b, chapter 5, this report). The amount, timing, and temperature of river water discharged into coastal waters are expected to be little changed by dam removal because the dams have been largely operated as “run of the river” (Duda and others, 2011, chapter 1, this report).

Sediment deposition and suspended sediment can have a variety of effects on nearshore plants and animals. Sediment deposition can affect organisms directly through burial, which can reduce light, oxygen, nutrients, and waste removal; scour, which can injure or dislodge organisms; and replacement of hard, stable substrate with finer particles that inhibit settlement for some organisms (Airoldi, 2003). Sediment deposition also can indirectly affect communities by altering outcomes of competitive and predator-prey interactions among species with different tolerances and responses to sedimentation (Airoldi and Cinelli, 1997; Airoldi and Virgilio, 1998). Suspended sediment increases turbidity, which reduces light penetration and can negatively affect photosynthetic organisms. Seaweeds and sea grasses require high ambient light, account for a large portion of the primary production in nearshore waters, and create three-dimensional structures inhabited by various species (Mumford, 2007); therefore, effects on them may propagate to other parts of the community. Turbidity also can influence competition among plant species with different light requirements, or affect competitive or predatory abilities of animals that depend on vision (Beauchamp and others, 1999). Direct effects of suspended sediment include damage to fish and invertebrate gills, and clogging or damage to feeding structures of filter feeders (Newcombe and MacDonald, 1991). Beds of seaweeds or sea grasses can dampen current velocities, thereby increasing sedimentation and decreasing suspended sediment within the beds (Madsen and others, 2001).

Substrate characteristics, including particle size, stability, and relief, are important for structuring benthic communities. Stable substrates of large particle size, for example bedrock and boulders, support various species adapted to attach to the substrate’s surface (Witman and Dayton, 2001). Fine sediments such as sand and silt do not provide attachment points but do permit burrowing and support various species adapted to living in the sediment (Lenihan and Micheli, 2001). Substrates of intermediate particle size (for example, gravel and cobble) provide some attachment space on their surfaces, but are subject to overturning and displacement by waves and currents, and support different communities compared to more stable rocky substrates (Scheibling and others, 2009). Propagules of some large seaweed species can start to grow on small rocks such as gravel or cobble. Drag on the seaweed increases as it grows, and depending on the size of the rock, the seaweed and the rock it is attached to may be lifted and transported (see Miller and others, sidebar 3.2, this report). A mixture of particle sizes offers a variety of habitats in close proximity. In high current environments, large substrates such as cobble and boulders can dampen current speeds, allowing retention of fine sediments that would otherwise wash away. The presence of fine sediments among larger substrates promotes coexistence of species adapted to different particle sizes. Seafloor relief in rocky habitats provides sloped and vertical surfaces that support different communities than horizontal surfaces. Relief also affects communities by modifying flow patterns (Witman and Dayton, 2001). Effects of sedimentation on benthic communities likely will vary among habitats with different substrate characteristics.

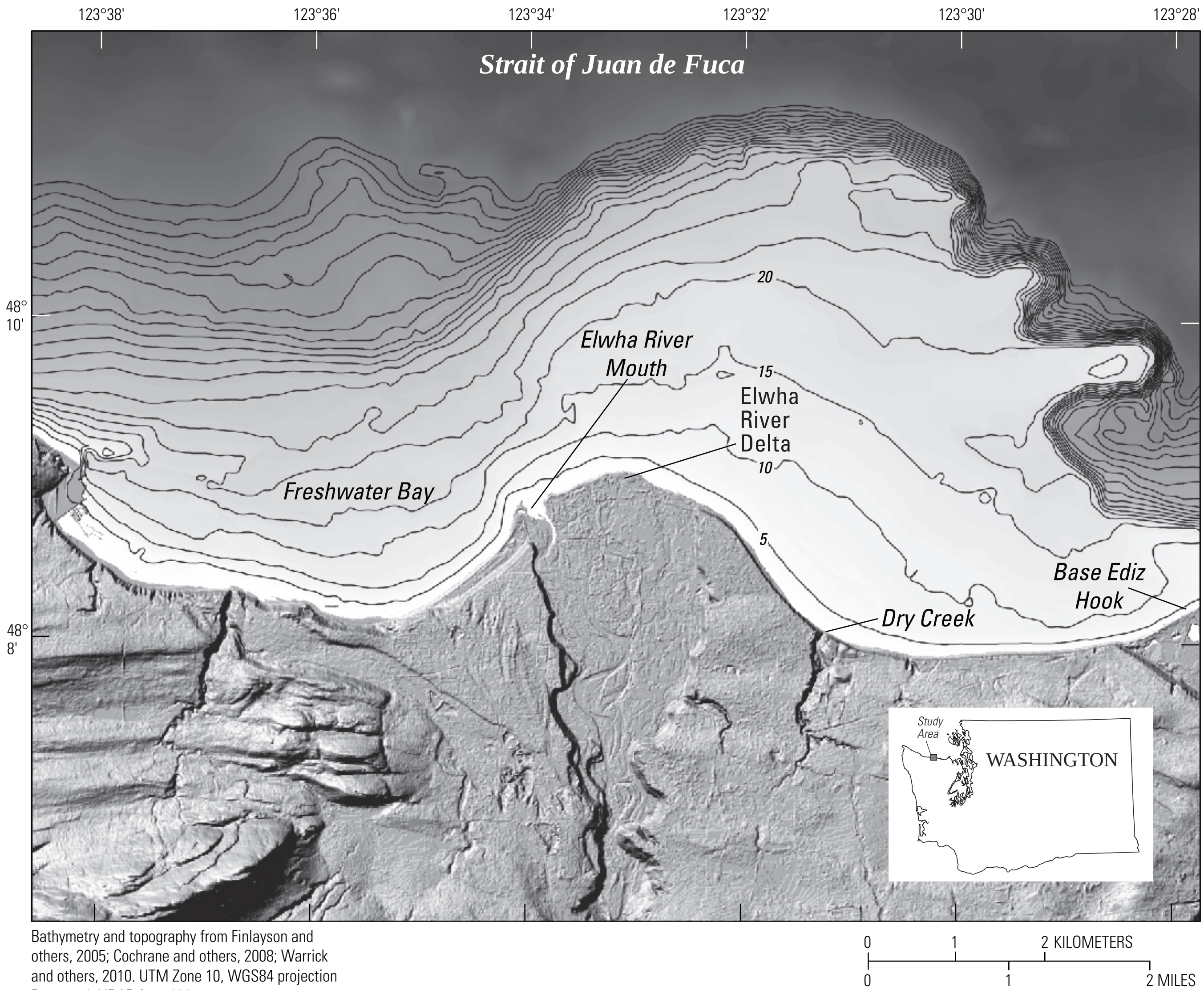
Purpose and Scope

This study was initiated in 2008 to characterize nearshore biological communities prior to removal of the Elwha River dams. The intent was to establish a baseline to measure changes following dam removal. In this chapter, two questions are addressed: (1) What communities currently are present in shallow subtidal areas potentially affected by dam removal? (2) What role does substrate play in structuring these communities?

Study Area

The study area includes shallow depths (3–18 m below mean lower low water [MLLW]) extending from the west end of Freshwater Bay to the base of Ediz Hook, a distance of 15 km (fig. 6.1). This range of depths should include most depths with sufficient light for photosynthesis and where kelp and other benthic macroalgae are dominant given suitable substrate for attachment (Whitman and Dayton, 2001; Mumford, 2007). The length of coast that will be affected by dam removal is not certain, but we expect that the boundaries west and east of the river mouth will include the affected area. They coincide with the boundaries of the Elwha nearshore defined by Shaffer and others (2008) except that their eastern boundary, the tip of Ediz Hook, is 5 km farther east.

Prominent features of the study area are Freshwater Bay, the Elwha River delta, which extends north into the Strait of Juan de Fuca, and the area east of the delta, which is sheltered by the delta to the west and Ediz Hook to the east. High bluffs border much of Freshwater Bay and the shore between Dry Creek and Ediz Hook. Bathymetry is steepest offshore of the river mouth, less steep in mid Freshwater Bay, and most gradual offshore of the eastern flank of the delta (fig. 6.1). Geomorphology of the study area is described in detail in Warrick and others, 2011a, chapter 3, this report.



Bathymetry and topography from Finlayson and others, 2005; Cochrane and others, 2008; Warrick and others, 2010. UTM Zone 10, WGS84 projection Bare earth LIDAR from 2001

Figure 6.1. Southern waters of the Strait of Juan de Fuca near the mouth of the Elwha River, Washington. Five-meter bathymetry contours also are shown.

Previous Work

Sonar and video surveys were conducted in 2005 in Freshwater Bay and offshore of the Elwha River mouth to map bathymetry and characterize substrate type and distribution (Warrick and others, 2008). Sonar backscatter data and video footage were used to classify substrate as hard (bedrock or boulders), mixed (gravel and cobble), or soft (sand or sand waves) (Cochrane and others, 2008). Most of the hard substrate was in Freshwater Bay, particularly in mid Freshwater Bay close to shore (fig. 6.2*A*). Mixed substrate predominated offshore of the river mouth and in parts of Freshwater Bay. Areas of soft substrate occurred at the west and east ends of Freshwater Bay. The areas farthest offshore were characterized by a combination of mixed and soft substrate. The substrate classifications were integrated with other information, including delineation of bedrock outcrops and counts of boulders visible in the raw backscatter data, to form a substrate type and morphology map (fig. 6.2*B*; Warrick and others, 2008). Bedrock outcrops were apparent in mid Freshwater Bay close to shore. Boulder abundance was high (greater than 10,000/km^2) close to shore and intermediate (10–10,000/km^2) farther from shore in Freshwater Bay. Boulder abundance was low (less than 10/km^2) in a band extending from outer Freshwater Bay to the Elwha River delta.

Aerial photographs taken during 1989 to 2004 were used to map the annual spatial extent of overstory kelp (*Nereocystis leutkeana* [bull kelp] and *Macrocystis integrifolia* [giant kelp], which form a canopy at the water surface) in the Strait of Juan de Fuca (Berry and others, 2005). Kelp was most persistent (present nearly every year) in the area of hard substrate (bedrock outcrops and abundant boulders) in mid-Freshwater Bay (fig. 6.2*C*). Kelp also occurred frequently just to the northeast of the river mouth, where the sonar survey detected little hard substrate in 2005 (Warrick and others, 2008), and in a few patches between Dry Creek and the base of Ediz Hook. Rigg (1915) mapped kelp beds in the Strait of Juan de Fuca in 1911 and 1912.

The entire study area was surveyed in 2006 to map *Zostera marina* (eelgrass), and Freshwater Bay was resurveyed in 2009 (Norris and Fraser, 2009). In both years, a large *Z. marina* bed (23 ha) was present at the west end of Freshwater Bay in a sand-dominated area (fig. 6.2*A–B*). In both years a much smaller bed (2 ha) was present in east-central Freshwater Bay. This bed was inshore from the area surveyed for substrate and west of the sandy area identified by the sonar survey at the east end of Freshwater Bay. *Z. marina* was not detected anywhere else in the study area.

Scuba surveys by Seavey and Ging (1995) in 1994 characterized existing marine resources potentially affected by removal of the Elwha River dams. They identified the macroalgae community as a particularly important resource that could be affected by increased sediment load after dam removal. Their study area included depths from 0–15 m and extended from mid Freshwater Bay east to Dry Creek; however, they concentrated their sampling in shallow water (0–9 m) close to the river mouth and did more sampling east than west of the mouth because of their expectation that sediment deposition would be greatest in these areas. They detected high abundance and diversity of macroalgae. Percent cover of macroalgae was greater than 75 percent for 40 percent of their quadrates and greater than 50 percent for 73 percent of their quadrates (*n* quadrates = 836; area of each quadrate = 0.84 m^2). They reported 40 macroalgae taxa (genus or species level identifications) that occurred in at least one quadrate. They also reported 57 taxa of invertebrates and fish that occurred in at least one quadrate and provided density estimates for 15 invertebrate species. They did not provide density estimates for macroalgae. An important component of their results was a set of distribution maps for five macroalgae species and *Z. marina*, for percent cover of macroalgae, and for substrate types (10 classes). *Z. marina* was present in 6 percent of their quadrates.

Shaffer (2000) sampled seasonally (every 3 to 4 months) from March 1996 to April 1997 in two overstory kelp beds (3–6 m depth) in Freshwater Bay, one dominated by *N. leutkeana* and the other by *M. integrifolia*, as well as in a pair of *N. leutkeana* and *M. integrifolia* beds farther west in the Strait of Juan de Fuca. The percentage of cover between the substrate and 1 m above the substrate for three kelp species (*N. leutkeana*, *M. integrifolia*, and *Pterygophora californica*), fleshy red algae, and total vegetation was reported. The density for the three kelp species, *Haliotis kamtschatkana* (northern abalone), *Strongylocentros franciscanus* (red urchins), and *Strongylocentrotus droebachiensis* (green urchins) (*n* quadrates per bed per season = 10; area of each quadrate = 1.0 m^2) also was reported. Percentage of cover of total vegetation in the Freshwater Bay beds was lowest in December (35–55 percent) and highest in June and September (65–95 percent). *P. californica* was the densest of the three kelps in all seasons in the *N. leutkeana* bed in Freshwater Bay; *M. integrifolia* was always densest in the other Freshwater Bay bed. *H. kamtschatkana*, *S. franciscanus* and *S. droebachiensis* were present in the *N. leutkeana* bed (density less than or equal to 2 per m^2 per species) but were nearly absent from the *M. integrifolia* bed in Freshwater Bay.

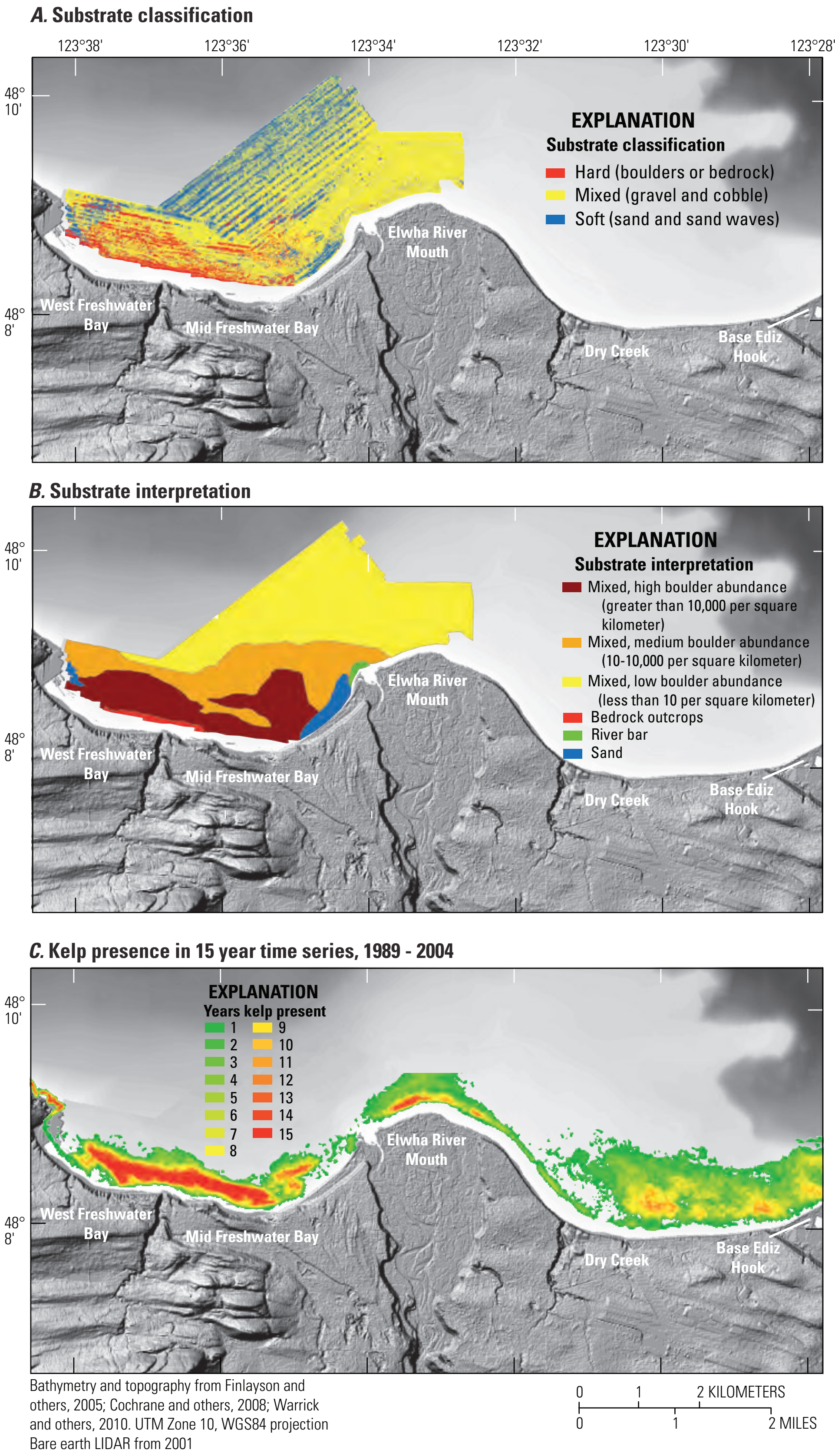


Figure 6.2. Substrates, and the extent of canopy forming kelp, near the mouth of the Elwha River, Washington. (*A*) Substrate classifications from acoustic backscatter data (Cochrane and others, 2008). (*B*) Substrate type and morphology formed by integrating substrate classifications and other information including delineation of bedrock outcrops and counts of boulders visible in the raw backscatter data (Warrick and others, 2008). (*C*) Presence and persistence of surface canopy forming kelp from annual aerial photographs taken in 1989–2004 (Berry and others, 2005).

In 2005 and 2006, the Washington Department of Fish and Wildlife (WDFW) established four permanent sites, marked with metal fence posts driven into the substrate, to assess effects of removal of the Elwha River dams, particularly effects on commercially important shellfish species (Michael Ulrich, Washington Department of Fish and Wildlife, written commun., February 26, 2010). Three sites consisted primarily of large and small boulders, cobble, and solid rock. Two sites were located in Freshwater Bay and considered treatment sites potentially affected by dam removal, and the third was located near Green Point, 21 km east of the Elwha River mouth, and considered a control site unlikely to be affected. One treatment site and the control site were established in 2005 and resurveyed in 2006 and 2007. The other treatment site was established in 2006 and resurveyed in 2007. A treatment site characterized by soft substrate was established offshore of the Elwha River mouth and surveyed in 2006. Sites were rectangular with areas ranging from 140 to 250 m^2. Site depths ranged from 11 to 15 m. Census counts were made for commercially important species including *H. kamtschatkana*, *S. franciscanus*, *S. droebachiensis*, and *Parastichopus californicus* (California sea cucumbers) at the rocky sites, and *Panopea generosa* (geoduck clams) and *Tresus capax* (horse clams) at the soft substrate site. Presence or absence was recorded for other animal and plant species. *S. franciscanus* and *S. droebachiensis* were measured for size. The two rocky treatment sites were recently repurposed and will not be available for resurveying following dam removal; however, pre-removal data from these sites may prove valuable when comparing data collected at other sites with similar habitat.

Study Design

Our study design is based on a Before-After-Control-Impact approach (Underwood, 1994), with sampling effort concentrated in the area expected to be affected by dam removal and in two control areas where effects of dam removal are expected to be minimal. The control areas are offshore of Low Point and Green Point, 20 and 21 km west and east of the Elwha River mouth, respectively. The data reported herein constitute part of the “before” component of the study, and repeated sampling is expected following dam removal (“after”). Analysis of future changes in the control areas compared to changes near the Elwha River should allow for effects of dam removal to be separated from other potential effects such as interannual variability and climate change. Because the purpose of this chapter is to characterize areas potentially affected by dam removal, results from Low and Green points will not be presented.

Site Selection

Sites were randomly selected after stratifying the study area according to three factors: distance from the Elwha River mouth, water depth, and substrate type. Five distance bands were oriented north to south and spaced approximately evenly from the west to the east end of the study area, with the central band straddling the Elwha River mouth (fig.6.3). Three depth strata—3–6, 9–12, and 15–18 m below MLLW—were delineated (fig. 6.3) using the best available bathymetry data (Cochrane and others [2008] for Freshwater Bay and offshore of the river mouth; Finlayson [2005] elsewhere). The intersection of the cross-shore bands and the depth strata produced 15 “bins,” within which areas with different substrate types were identified based on substrate classifications by Cochrane and others (2008) from sonar surveys in Freshwater Bay and offshore of the Elwha River mouth (classifications were hard, mixed, or soft; fig. 6.2*A*), or on the presence of overstory kelp in the two cross-shore bands east of the river mouth that lacked substrate classifications from sonar. Berry and others (2005) mapped the spatial extent of overstory kelp for 15 years during 1989–2004 (fig. 6.2*C*). We inferred substrate type from overstory kelp frequency of occurrence by assuming that kelp were attached to hard substrate; categories were hard (kelp present in 9–15 years), mixed (kelp present in 1–8 years), and soft (kelp never present). Warrick and others (2008) determined that for depths less than 15 m, overstory kelp was present in 58 percent of the area classified as hard from sonar data, but was present in 18–23 percent of the area classified as mixed or soft from sonar data, providing some support for inferring substrate type from kelp presence.

At least one site was randomly selected within each cross-shore band by depth stratum by substrate type combination using a geographic information system. Not all three substrate types were present in every cross-shore band by depth combination (fig. 6.3). One site in the cross-shore band near Dry Creek was placed outside of its intended depth strata; for analyses, it was grouped with the stratum closest to it in depth.

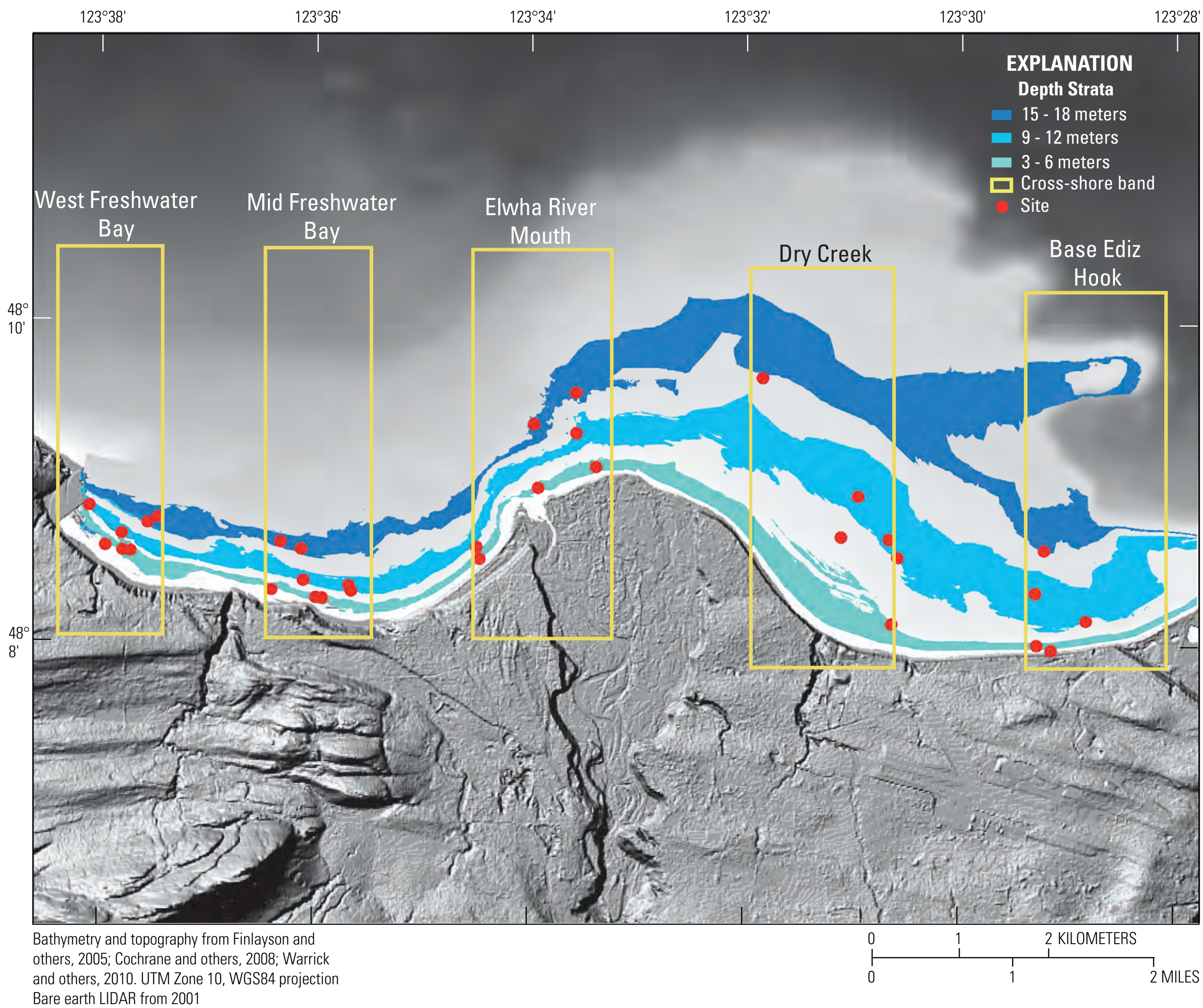


Figure 6.3. Three depth strata and five cross-shore bands at varying distances from the mouth of the Elwha River, Washington. Sites were randomly selected within each of the 15 distance by depth stratum "bins." Data were collected in at least two 30 meter-long shore-parallel transects at each site. One site in the Dry Creek band was placed outside of its intended depth strata; for analyses, it was grouped with the stratum closest to it in depth (3–6 meters).

Data Collection

Data collection methods were adapted from protocols developed by the Partnership for Interdisciplinary Studies of Coastal Oceans (2011) for monitoring ecosystems associated with rocky reef habitats (Carr and others, 2001). Data were collected at 30 m-long transects. Swath surveys were conducted to estimate density. Individual organisms present in a swath of fixed width along the entire transect were identified to the lowest practicable taxonomic level and counted. Measurements taken at discrete points along the transect were used to estimate percent coverage of different substrate types and seafloor relief categories.

We navigated to each randomly selected site using a Wide Area Augmentation System-enabled Global Positioning System (±5 m accuracy) and dropped the boat anchor. Two teams of two divers descended the anchor line and established transects in opposite, approximately shore-parallel directions using compass bearings taken at the surface (fig. 6.4*A*). Water visibility was measured by each team at the anchor as the distance at which one diver could see two fingers held up by the other. Visibility ranged from 4–14 m (mean=8 m) among sites. Depth at transect starting and ending points was recorded from dive computers.

Fish Swath

The fish survey was initiated 10 m from the anchor. One diver swam slightly ahead of the other and counted all conspicuous fish within a 2 m wide by 2 m high swath. The second diver reeled out a measuring tape along the transect line while watching for fish.

Invertebrate and Kelp Swaths

At the 40 m mark, the tape was secured and both divers proceeded back along the transect, one identifying and counting invertebrates in a swath on one side of the tape and the other doing the same for kelp on the other side of the tape (fig. 6.4*B*). Fish observed during the invertebrate or kelp surveys but not observed during the fish survey (typically small, cryptic individuals touching the substrate) were noted and included in fish density estimates. Invertebrate and kelp surveys were either 1 or 2 m wide based on the surveyor's judgment of whether the survey could be completed in a single dive. Surveys were only 2 m wide when organisms were sparse, visibility good, and depth shallow. Invertebrates greater than 2.5 cm (any dimension) for which individuals could be recognized (not encrusting or colonial species) were counted. *N. luetkeana* and *P. californica* with stipes greater than 30 cm long were counted. Other kelp species with combined stipe and blade lengths greater than 24 cm were counted. When *Z. marina* was observed, shoots present in the kelp swath were counted.

A variable area sampling method was used for swath surveys when high densities of invertebrates or kelp were encountered. The transect was divided into three 10 m increments (0–10, 10–20, 20–30). The surveyor counted individuals of a taxon until 30 individuals were counted or the end of the 10 m segment was reached. If 30 were counted before the end, the surveyor recorded the distance along the transect to that point and stopped counting the taxon until the start of the next 10 m segment.

Uniform Point Contact Survey

The uniform point contact (UPC) survey was started when invertebrate and kelp surveys were finished. Both divers participated, each working on separate 10-m segments of the 30 m transect. The UPC involved classifying sediment grain size and seafloor relief at 60 points spaced every 0.5 m along the transect (fig. 6.4*C*). Sediment directly under each point was classified as sand (less than 0.2 cm), gravel-cobble (0.2–25 cm), boulder (25 cm-1 m), or bedrock (greater than 1 m diameter). Relief was measured as the greatest elevation difference within a rectangle 1 m across the transect by 0.5 m along the transect and centered on the transect point (fig. 6.4*C*). Relief classifications were less than 0.1 m, 0.1–1 m, and greater than 1 m.

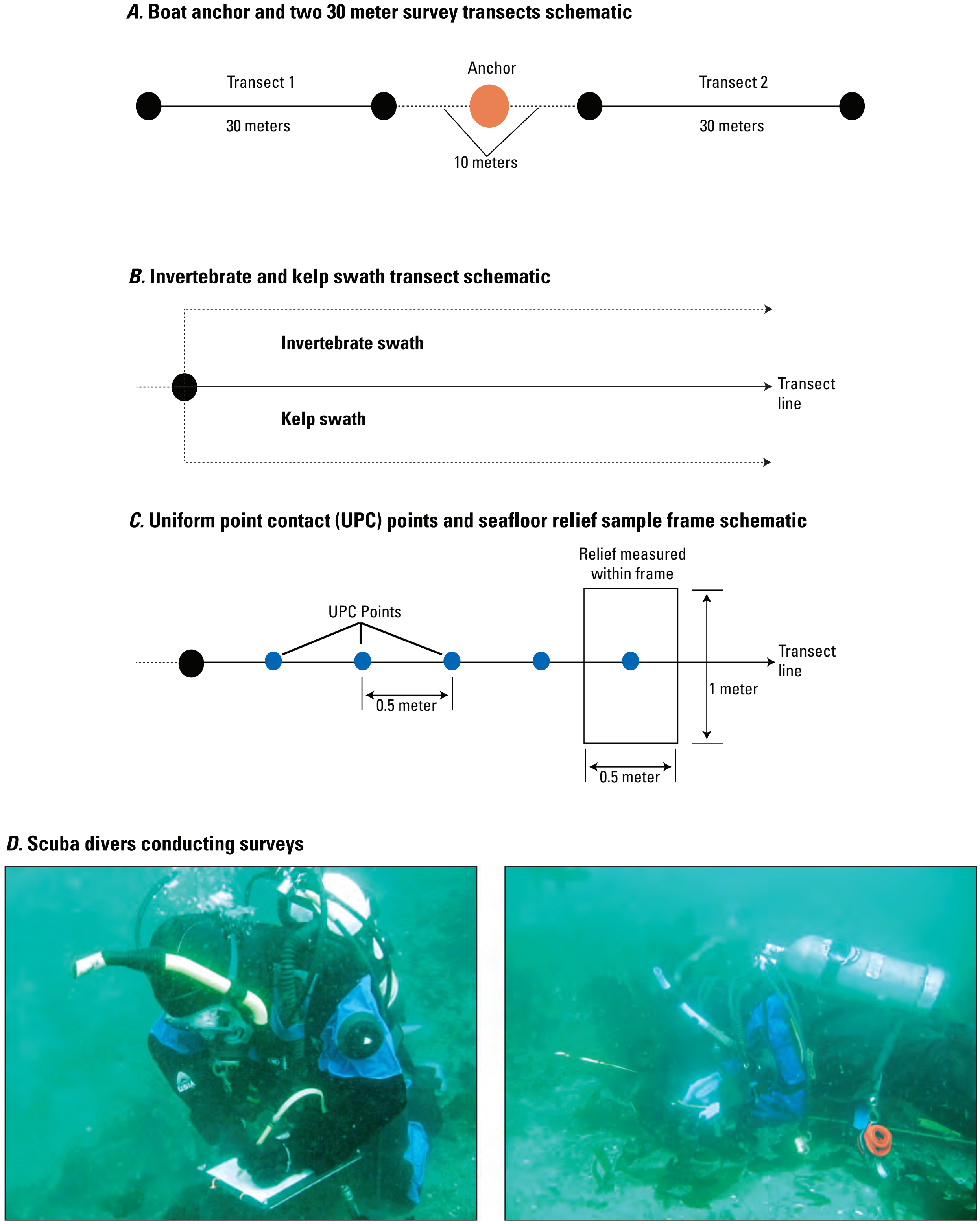


Figure 6.4. Schematic diagrams and photographs showing establishment of transects at a site. (*A*) Boat anchor and two 30 meter transects at one site; (*B*) invertebrate and kelp swaths along a transect; (*C*) uniform point contact (UPC) points spaced every 0.5 meter along a transect, and 0.5- by 1.0-meter rectangles centered on each UPC point. Seafloor relief was measured as the maximum elevation difference within a rectangle; (*D*) Scuba divers at work.

Other Aspects of Data Collection

If divers had adequate air and allowed bottom time, a second transect was done along the same bearing (third or fourth transect at the site), in which case that transect was started 20 m from the end point of the previous transect to maintain a 20 m spacing interval between transects.

Nine sites were surveyed between August 7 and 11, 2008, 14 sites between August 21 and 25, 2008, and 10 sites between September 5 and 7, 2008, for 75 transects completed at 33 sites (table 6.1). Additional sites were surveyed at control locations during these times (data not presented). Diving occurred on dates with low tidal exchanges to minimize current in August–September to optimize favorable weather and wind conditions. Local current predictions generated from current harmonics measured by instruments deployed around the Elwha River delta (Warrick and others, 2011b, chapter 5, this report) assisted with planning dives to coincide with reduced currents. All surveys were conducted during daylight hours. Surveys in 2009 were accomplished using the same methods as in 2008. Data from 2009 are preliminary and are not reported here.

Table 6.1. Completed transects and sites by cross-shore band and depth and substrate stratum, east, west, and at the mouth of Elwha River, Washington, August and September 2008.

[Site locations are shown in figure 6.3]

Cross-shore band	Depth stratum	Substrate stratum	Number of transects	Number of sites
West Freshwater Bay	3–6 meters	Hard	2	1
		Mixed	2	1
		Soft	2	1
	9–12 meters	Hard	2	1
		Soft	3	1
	15–18 meters	Hard	2	1
		Mixed	2	1
Mid-Freshwater Bay	3–6 meters	Hard	6	3
	9–12 meters	Hard	2	1
		Mixed	4	2
	15–18 meters	Hard	2	1
		Mixed	2	1
Elwha River Mouth	3–6 meters	Hard	2	1
		Mixed	2	1
		Soft	3	1
	9–12 meters	Mixed	2	1
		Soft	4	1
	15–18 meters	Mixed	4	2
Dry Creek	3–6 meters	Mixed	3	1
		Soft	2	1
	9–12 meters	Mixed	4	2
		Soft	2	1
	15–18 meters	Soft	2	1
Base Ediz Hook	3–6 meters	Mixed	2	1
		Soft	4	1
	9–12 meters	Hard	2	1
		Mixed	2	1
	15–18 meters	Soft	2	1
Total			**75**	**33**

Data Analyses

Scores were assigned to sediment grain size and relief classifications. Substrate classified as sand, gravel-cobble, boulder, or bedrock received a score of 1, 2, 3, or 4, respectively. A mean and standard deviation (SD) were then computed for each transect as the mean and SD of the substrate scores for each of the 60 UPC points. Relief classified as less than 0.1 m, 0.1–1 m, or greater than 1 m received a score of 1, 2, or 3, respectively; mean and SD relief scores were computed as for substrate.

Analyses were based on site means from data averaged over transects at a site. For purposes of analysis, data were frequently grouped by the cross-shore band and depth strata used in site selection, but to represent substrate type UPC classifications were used rather than the three *a priori* substrate strata.

Pearson's correlation coefficient (r; Sokal and Rohlf, 1995), which takes a value between -1 and +1, was used as a measure of association between two variables (for example, kelp density and mean substrate score). Values near 1 indicate that high values of one variable are associated with high values of the other. For example, an r-value near 1 for the correlation between kelp density and mean substrate score would indicate that sites with high kelp density also had high mean substrate scores. Values near -1 indicate that high values of one variable are associated with low values of the other, and values near 0 indicate no association between the two variables. Student's t-test (Sokal and Rohlf, 1995) was used to determine whether the mean (for example, mean kelp density) for one subset of sites (for example, deep sites) was different from the mean for a second subset of sites (for example, shallow sites). The notation $P<0.05$ was used to indicate that the correlation between two variables or the difference between two means was greater than what would be expected by chance (less than 5 times out of 100 by chance).

Substrate Composition and Relief

Substrate composition at sites ranged from entirely sand (grain size less than 0.4 cm) to mostly coarse grain sizes (boulders or bedrock) (fig. 6.5*A*). Bedrock primarily was at three shallow sites in mid-Freshwater Bay. Boulders were present in Freshwater Bay and offshore of the Elwha River mouth but were rare east of the river mouth. Areas of nearly pure sand were close to shore at the west and east ends of Freshwater Bay and at the base of Ediz Hook. Gravel-cobble substrate dominated at many of the deeper sites, particularly offshore of the Elwha River and Dry Creek.

Seafloor relief at sites ranged from nearly flat (100 percent less than 0.1 m) to moderate (mostly greater than 0.1 m) (fig. 6.5*B*). High relief (greater than 1 m) occurred only at a few sites in Freshwater Bay. Relief greater than 0.1 m was common offshore of the river mouth and to the west but was nearly absent at sites to the east of the river mouth.

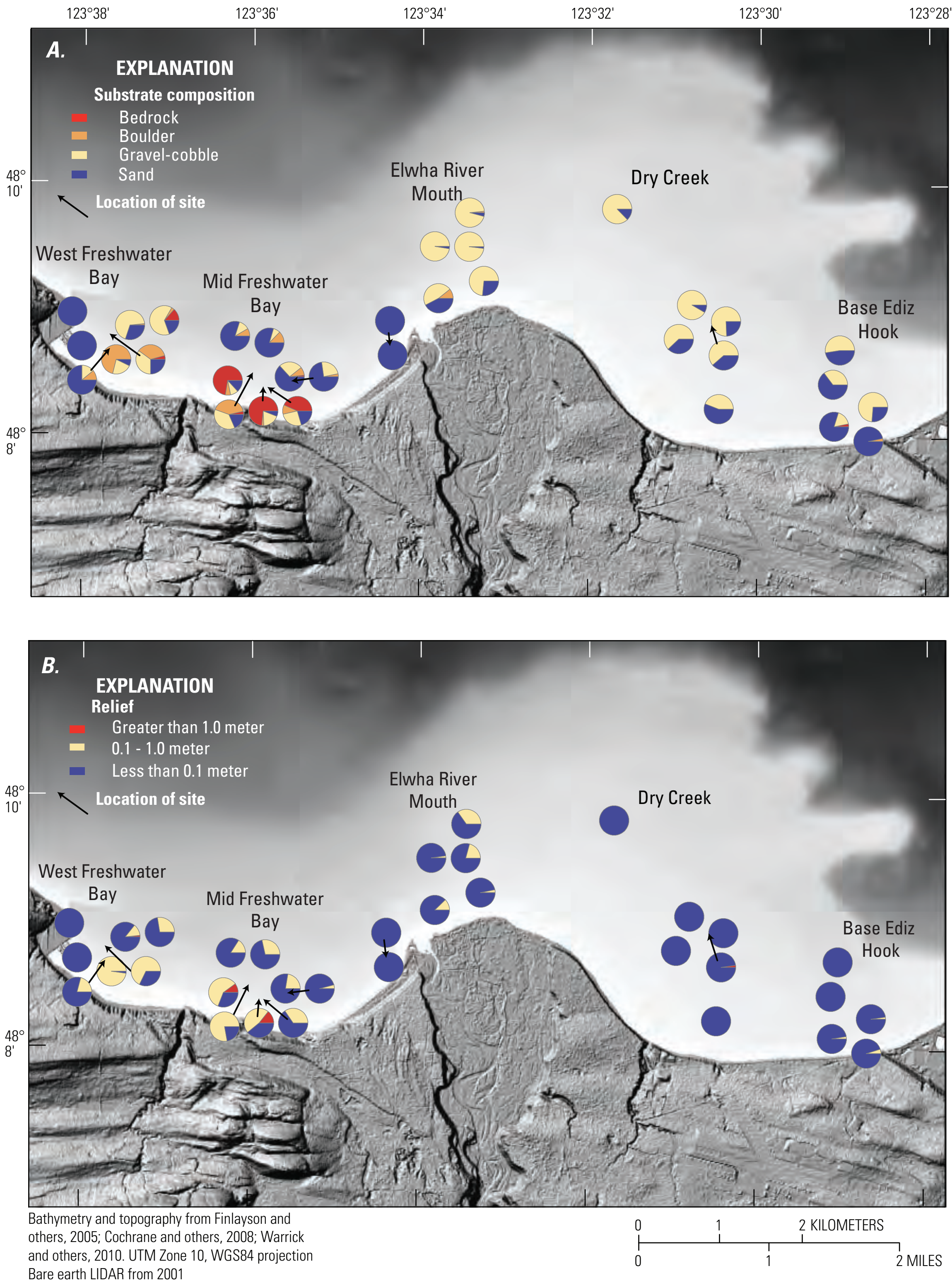


Figure 6.5. Substrate and relief at sites near the mouth of Elwha River, Washington, August and September 2008. (*A*) Substrate composition from classifications made at points spaced every 0.5 meter along 30-meter transects (usually 2 transects and 120 points per site). (*B*) Seafloor relief from classifications made at the same points as substrate classifications. At each point, maximum relief (elevation difference) within a 0.5- by 1.0-meter rectangle centered on the point was measured and binned as shown in the explanation.

Number of Taxa and Mean Density

Density was estimated for 9 taxa of brown algae, 65 invertebrate taxa, and 24 fish taxa (table 6.2). Organisms often were identified to species (67 taxa), but sometimes identification could be made only to genus (7 taxa) or to broader taxonomic categories (24 taxa). All the brown algae were from the order *Laminariales* which contains the large brown algae commonly known as kelp (O'Clair and Lindstrom, 2000; Gabrielson and others, 2006).

Mean density of organisms (averaged over all sites) was 3.10/m^2 for kelp, 2.69/m^2 for invertebrates, and 0.078/m^2 for fish (table 6.2). Mean density within invertebrate subgroups (phyla, subphyla, or classes) ranged from 1.40/m^2 for *Polychaeta* (polychaete worms) to 0.10/m^2 or less for *Cnidaria* (anemones and jellyfish), *Porifera* (sponges), and *Urochordata* (tunicates). High polychaete density was due to high densities of worms that build and live in soft (non-calcareous) tubes constructed from mucous combined with sediment or other local materials (families *Chaetopteridae*, *Maldanidae*, *Onuphidae*, *Sabellidae*, and *Terebellidae*; Lamb and Hanby, 2005). All of these worms were combined into one broad taxonomic category due to difficulties with consistently identifying to lower taxonomic levels. Some of the soft tube worms detected in the study area are shown in figure 6.6. Densities were low for *Cnidaria*, *Porifera*, and *Urochordata* partly because we could not reliably count individuals for the many colonial or encrusting species from these groups. Mean density within fish families ranged from 0.026/m^2 for *Cottidae* (sculpins) to 0.005/m^2 or less for several other families. Fish were combined into broad taxonomic categories (for example, flatfish) either because individuals swam away before they could be identified or because they were difficult to identify visually.

Z. marina (eelgrass) was present only at the west end of Freshwater Bay at two shallow sites with high percentages of sand. Shoot density was 8.9/m^2 at a site with 100 percent sand and 0.5/m^2 at a site with 75 percent sand.

Table 6.2. Number of taxa and mean density (averaged over all sites) from swath surveys for brown algae, invertebrates, and fish at sites near the mouth of the Elwha River, Washington, August and September, 2008.

[**Subgroup:** Order for brown algae, family for fish and mixed for inverebrates (P = phylum, SP = subphylum, C = Class). **Density:** Calculated as number per square meter]

Major group	Subgroup	Common name	Number by taxonomic level				Density
			Species	Genus	Higher	Total	
Brown algae	*Laminariales*	Kelp	**8**	**1**	**0**	**9**	3.10
Invertebrates	*Cnidaria* (P)	Anemones, jellyfish	7	2	1	10	0.10
	Crustacea (SP)	Barnacles, crabs, shrimps	8	0	6	14	0.37
	Echinodermata (P)	Sea cucumbers, stars, urchins	12	0	1	13	0.35
	Mollusca (P)	Chitons, clams, octopi, snails	13	4	5	22	0.42
	Polychaeta (C)	Polychaete worms	0	0	2	2	1.40
	Porifera (P)	Sponges	0	0	1	1	0.02
	Urochordata (SP)	Tunicates	2	0	1	3	0.03
	Total		**42**	**6**	**17**	**65**	2.69
Fish	*Bothidae, Pleuronectidae*	Flatfish	0	0	1	1	0.011
	Chimaeridae	Ratfish	1	0	0	1	0.003
	Cottidae	Sculpins	5	0	1	6	0.026
	Gadidae	Cods	1	0	0	1	0.005
	Hexagrammidae	Greenlings, lingcod	3	0	0	3	0.012
	Pholididae	Gunnels	0	0	1	1	0.015
	Stichaeidae	Pricklebacks, warbonnets	1	0	1	2	0.002
	Other		6	0	3	9	0.004
	Total		**17**	**0**	**7**	**24**	0.078
Total			**67**	**7**	**24**	**98**	**5.87**

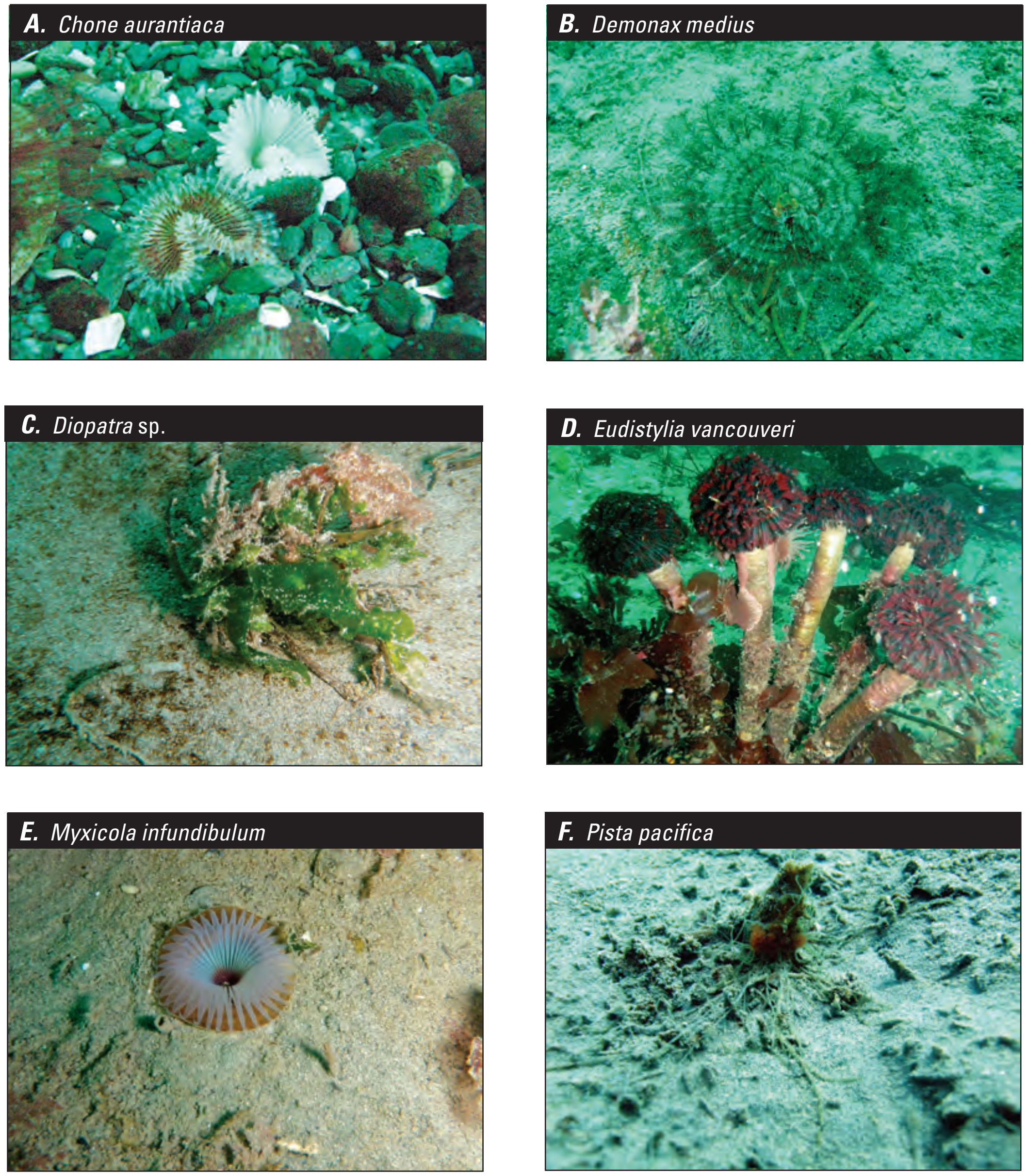
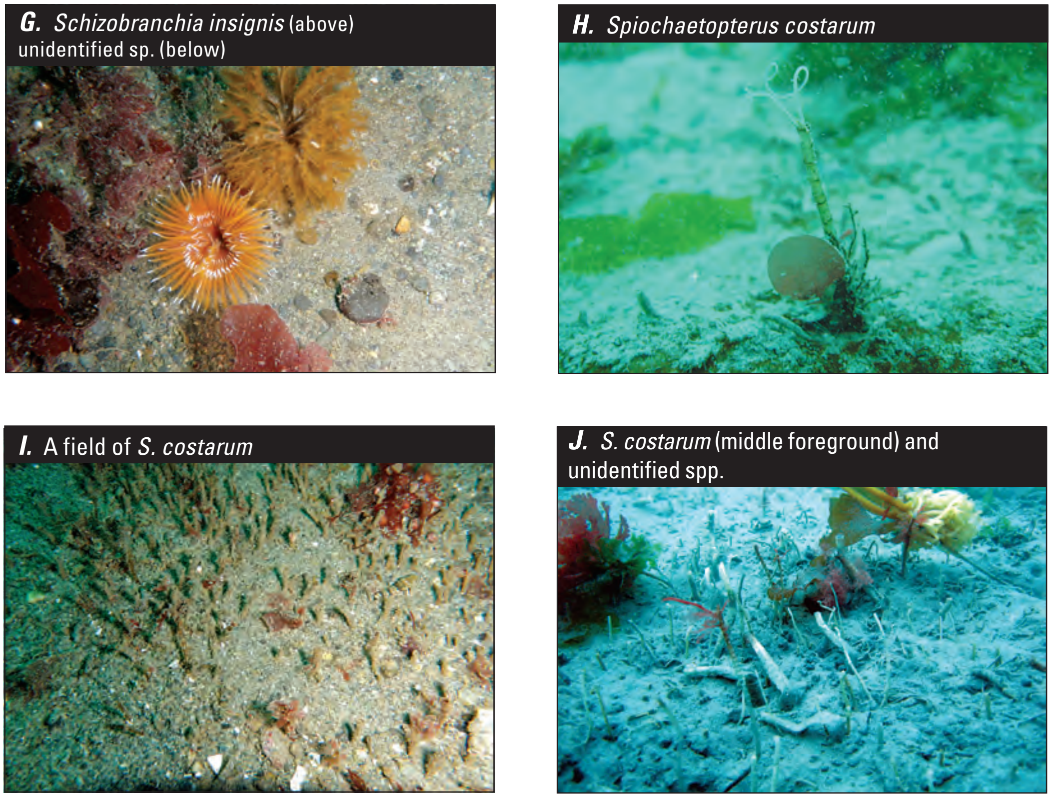


Figure 6.6. Soft tube worms commonly observed near the mouth of the Elwha River, Washington. The tentacular crown, visible for *Chone aurantiaca* (*A*) and several of the other species shown in the figure, serves the dual function of filter feeding and respiration. The crown can be retracted into the tube when the worm senses danger. *Diopatra* worms (*C*) festoon their tubes with local materials, sometimes completely concealing the tube as shown. *Pista pacifica* (*F*) feeds on detritus and small invertebrates on the surface of the substrate by extending the long bucal tentacles, visible here, from its tube. (*J*) The small and large diameter whitish tubes harbor unidentified species.



G. *Schizobranchia insignis* (above) unidentified sp. (below)

H. *Spiochaetopterus costarum*

I. A field of *S. costarum*

J. *S. costarum* (middle foreground) and unidentified spp.

Figure 6.6.—Continued

Abundant Species

Ten species of kelp were observed at the study sites and counted during swath surveys (fig. 6.7). *N. luetkeana* (bull kelp) has a thin, flexible stipe (stem) that extends to or near the water surface, and blades (leaves) that are held there by pneumatocysts (floats). *P. californica* has a rigid stipe that extends upward 1–2 m and blades that can form a dense canopy at that level. The other eight species have short stipes, and their blades lie on or near the seafloor. Counts of *Saccharina latissima* and *Saccharina subsimplex* were combined (as *Saccharina* spp.) because we could not always distinguish small individuals. Both species were observed at the study sites based on characteristics of large individuals. *M. integrifolia* (giant kelp) was observed in the study area at the water surface but not on transects at our sites. Mean density for kelp taxa ranged from 0.91/m^2 for *P. californica* to 0.09/m^2 for *Alaria marginata* (fig. 6.8*A*).

Of the invertebrates identified to species, the 12 species with the highest mean densities across all sites included two crab species, two sea urchins, two sea stars, and one species each of sea cucumber, snail, jellyfish, clam, tunicate, and chiton (fig. 6.9). Densities for these species were an order of magnitude lower than for the kelp species (fig. 6.8*B*). Densities ranged from 0.09/m^2 for *Cancer oregonensis* (pygmy rock crab) to 0.02/m^2 for *Cryptochiton stelleri* (gumboot chiton). Combined density was 0.10/m^2 for the remaining invertebrates identified to species (the species with the 13th through the 42nd highest densities).

Of the fish identified to species, the seven species with the highest mean densities were *Hexagrammos decagrammus* (kelp greenling; 0.007/m^2), *Gadus macrocephalus* (Pacific cod; 0.006/m^2), *Hydrolagus colliei* (spotted ratfish; 0.003/m^2), *Ophiodon elongatus* (lingcod; 0.003/m^2), *Jordania zonope* (longfin sculpin; 0.002/m^2), *Blepsias cirrhosus* (silverspotted sculpin; 0.002/m^2), and *Hexagrammos stelleri* (whitespotted greenling; 0.002/m^2) (fig. 6.10). Combined mean density was 0.006/m^2 for the remaining fish identified to species.

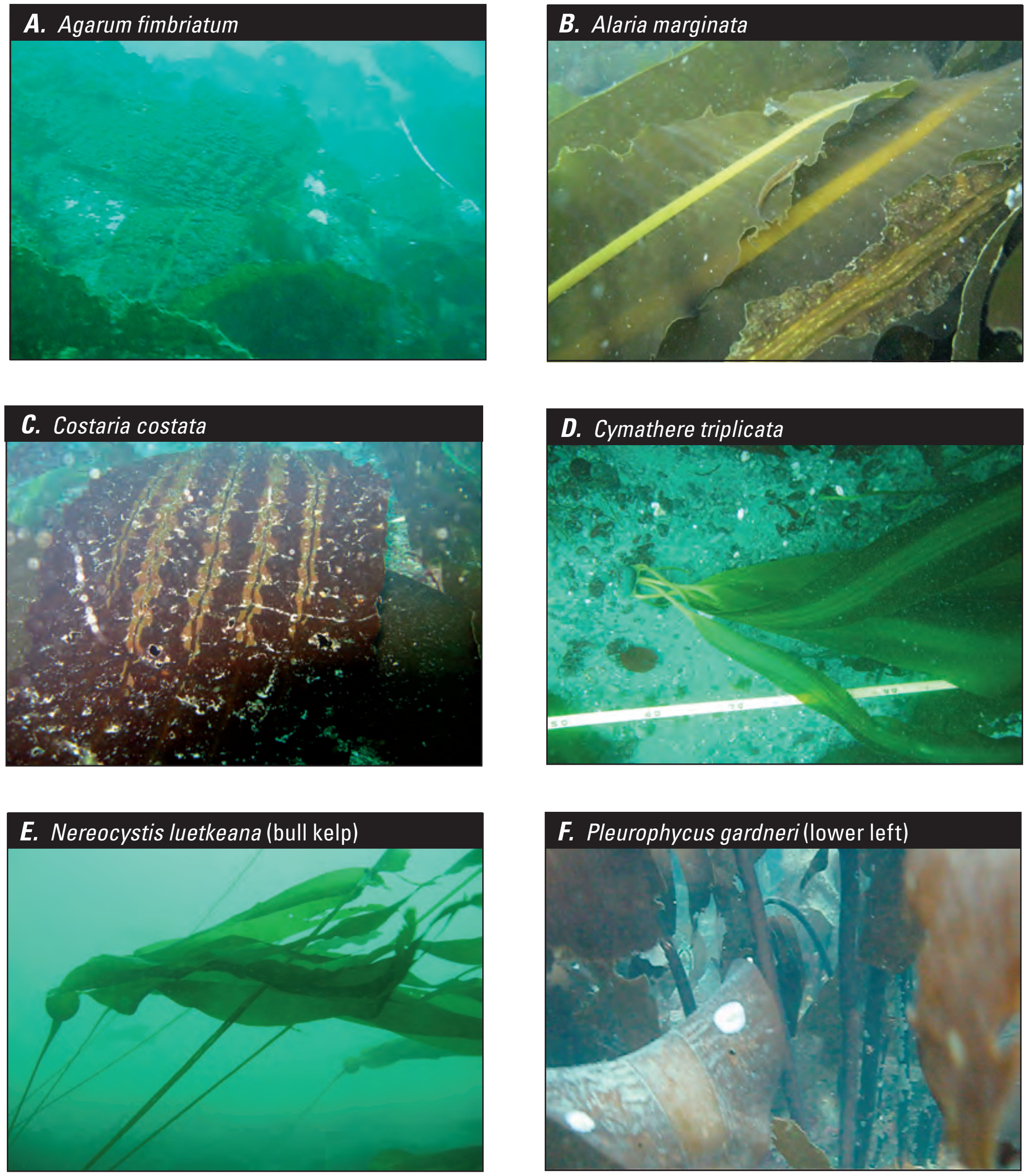


Figure 6.7. Nine kelp taxa (eight species and one genus comprising two species) surveyed for density near the mouth of the Elwha River, Washington. (*B*) Two individuals of *Alaria marginata* are shown (the two blades with light midribs in the center of the image); a *Synchirus gilli* (manacled sculpin) is perched on the upper of the two blades. (*D*) At least five individuals of *Cymathere triplicata* are attached to a single cobble. (*H*) Either *Saccharina latissima* or *S. subsimplex*. These two species sometimes cannot be distinguished without dissection.

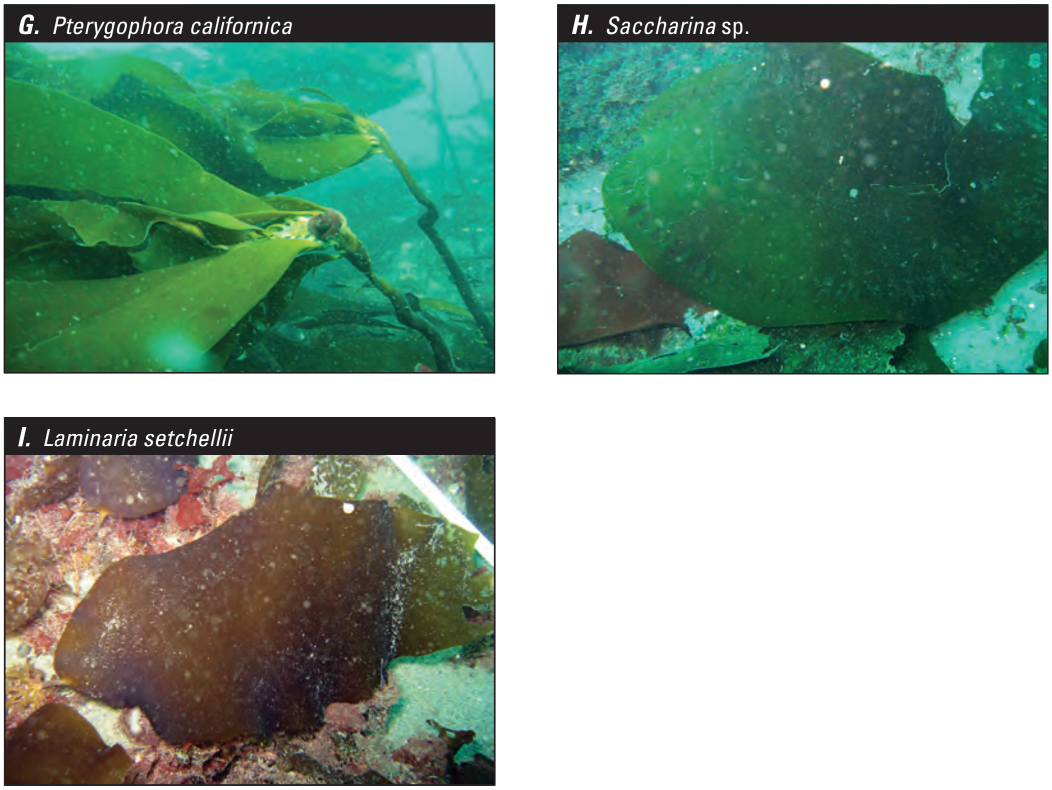


Figure 6.7.—Continued

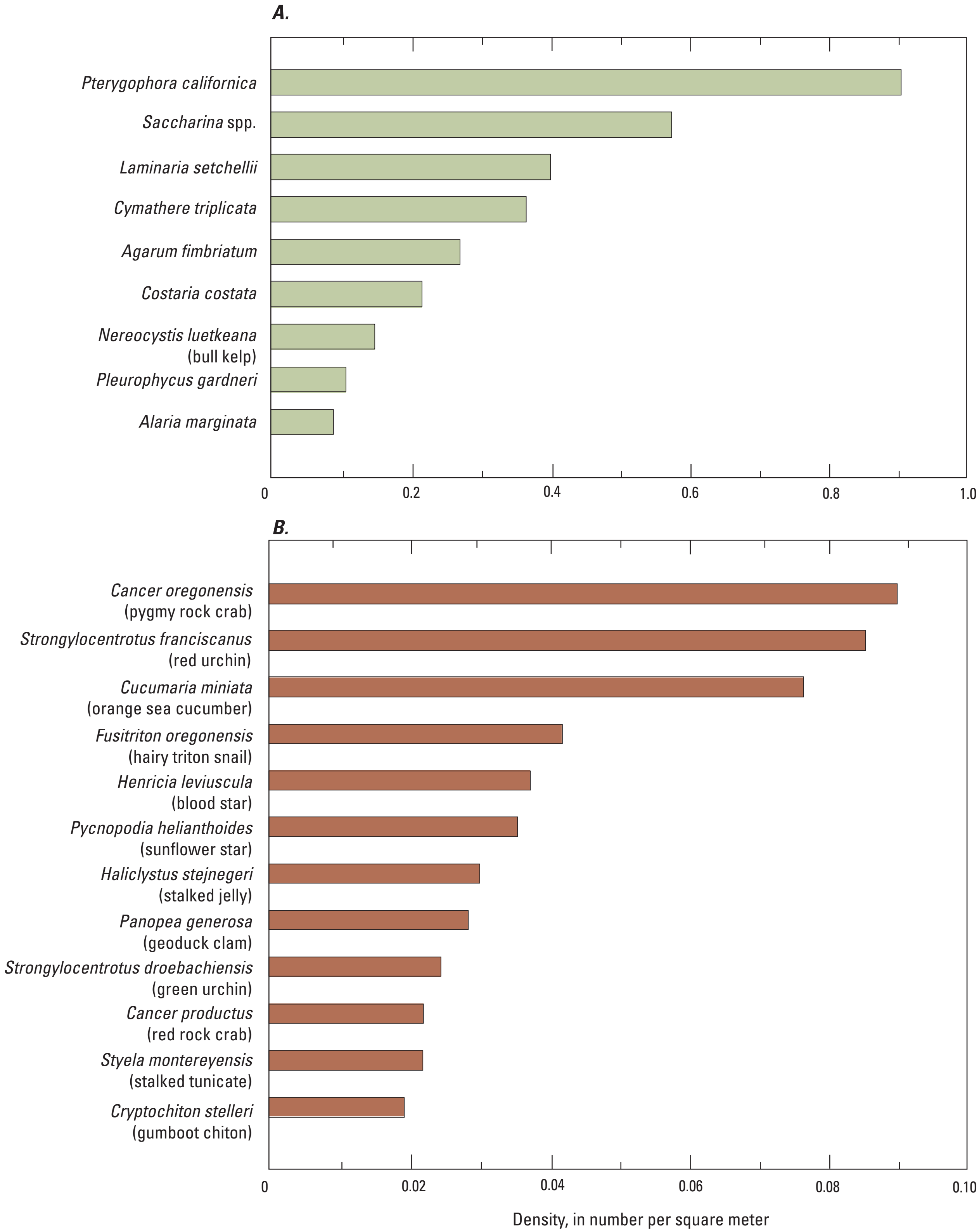


Figure 6.8. Mean density (averaged over all sites) for (*A*) 9 kelp taxa and (*B*) the 12 most common invertebrate species near the mouth of the Elwha River, Washington, August and September 2008. Note that densities are an order of magnitude lower for invertebrates than for kelp. *Saccharina* spp. includes two species, *S. latissima* and *S. subsimplex*, that were difficult to distinguish in the field.

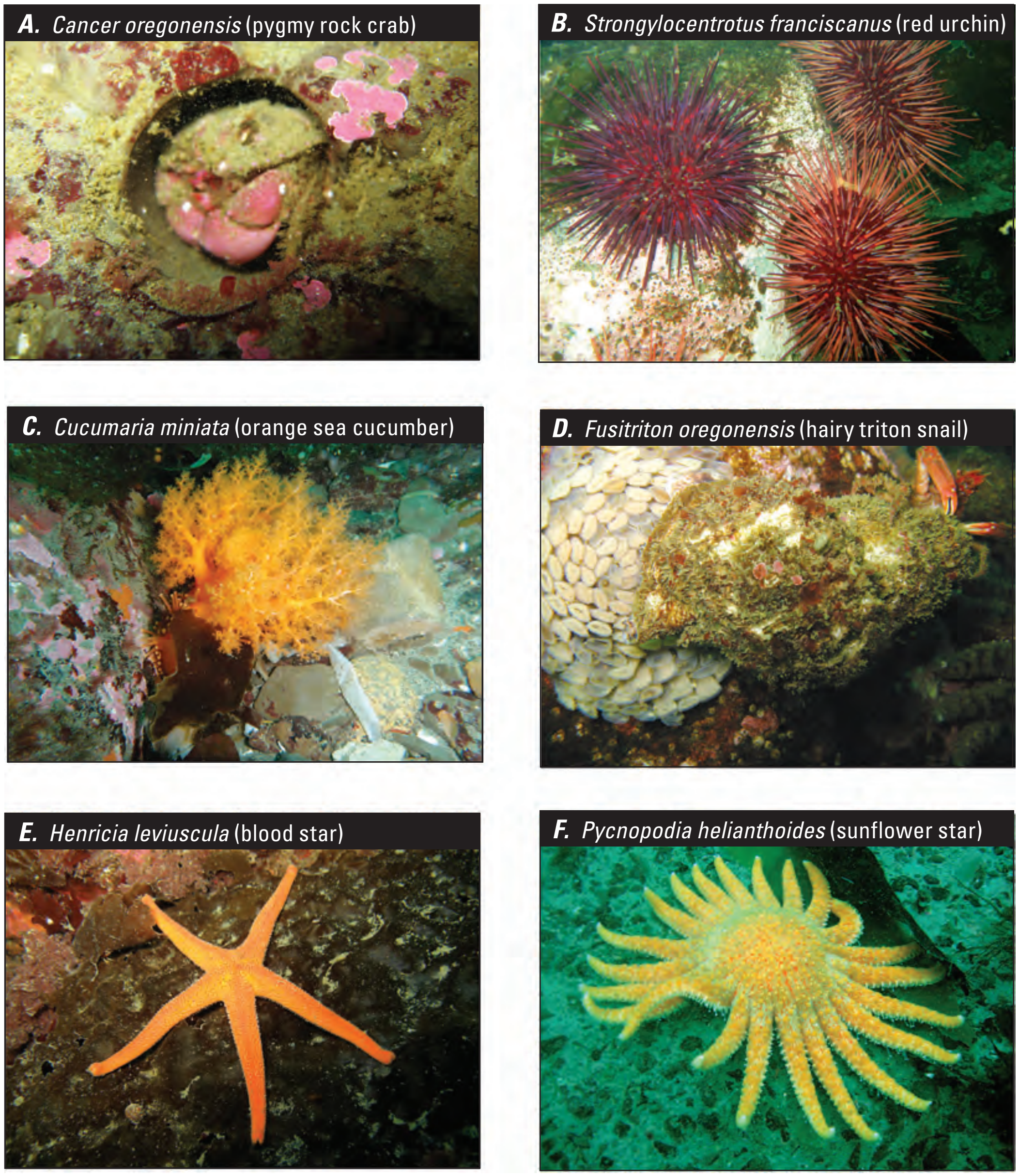


Figure 6.9. Twelve invertebrate species with the highest densities near the mouth of the Elwha River, Washington. Species are shown in order of decreasing density (fig. 6.8*B*). *Cancer oregonensis* (pygmy rock crab; *A*) is shown in a hole excavated in soft bedrock by a burrowing clam. *Haliclystus stejnegeri* (stalked jellyfish; *G*) adopts a sessile lifestyle by attaching to seaweed. *Panopea generosa* (geoduck clam; *H*) is shown with the tips of its siphons protruding from the sand.

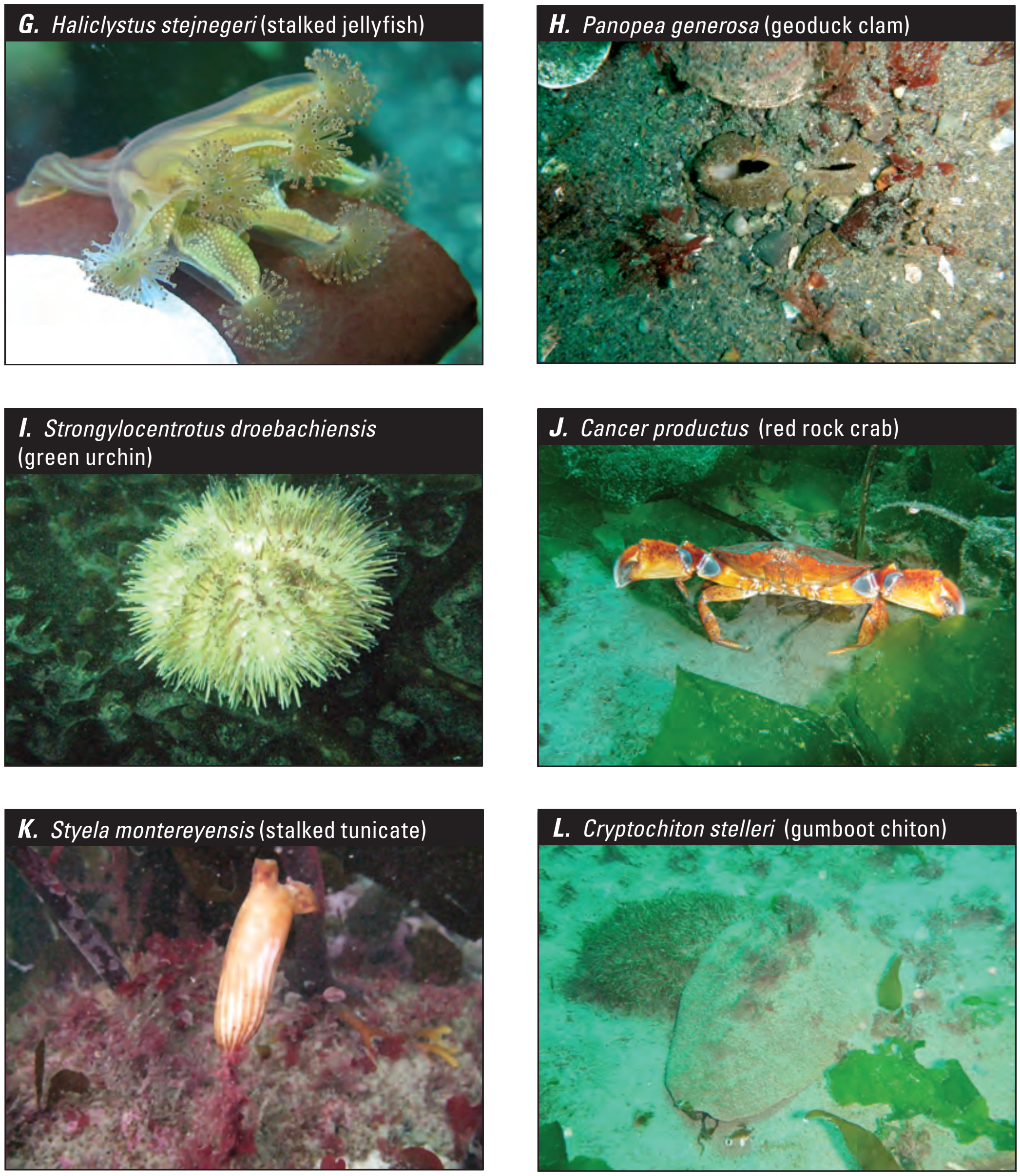
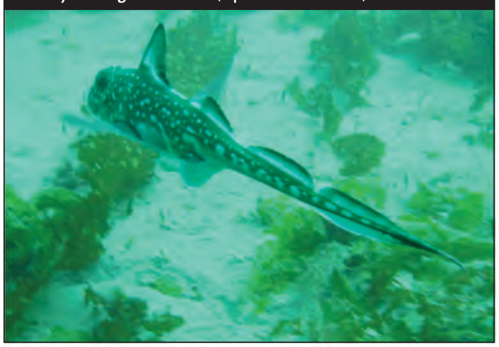


Figure 6.9.—Continued

A. *Hexagrammos decagrammus* (kelp greenling)

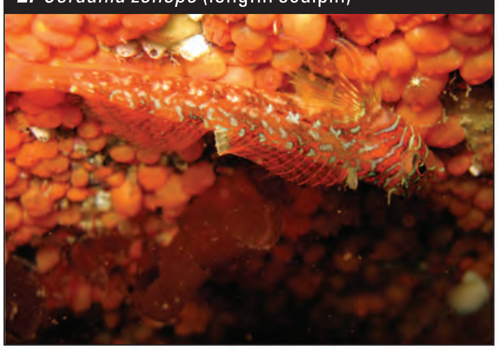
B. *Juvenile Gadus macrocephalus* (Pacific cod)

C. *Hydrolagus colliei* (spotted ratfish)



D. *Juvenile Ophiodon elongatus* (lingcod)

E. *Jordania zonope* (longfin sculpin)



F. *Blepsias cirrhosus* (silverspotted sculpin)

G. *Hexagrammos stelleri* (whitespotted greenling)

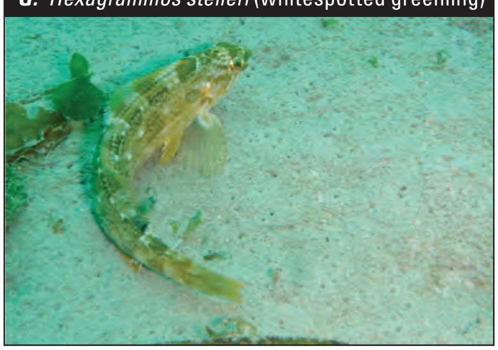


Figure 6.10. Seven fish species with the highest densities near the mouth of the Elwha River, Washington. Species are shown in order of decreasing density.

Spatial Patterns of Density and Taxa Richness

Kelp density was highest in mid- and west Freshwater Bay close to shore (fig. 6.11*A*) at sites with high percentages of hard substrate (bedrock or boulders; fig. 6.5*A*). Kelp density was low off Dry Creek and at two sandy sites southwest of the Elwha River mouth. The site with the highest invertebrate density, 18.5/m^2, was in mid-Freshwater Bay (fig. 6.11*B*). Most invertebrates at this site were polychaete worms with soft tubes (worm density=17.9/m^2). Invertebrate density was 5.8/m^2 at the site with the next highest invertebrate density. Invertebrate density was generally high offshore of the Elwha River mouth and to the west, and generally low off Dry Creek. Fish density was less variable across the study area than either kelp or invertebrate density (fig. 6.11*C*). Taxa richness (total number of taxa for kelp, invertebrates, and fish combined) was higher off the Elwha River mouth and to the west than to the east (fig. 6.11*D*).

Mean density and taxa richness within cross-shore band by depth "bins" are shown in figure 6.12. Kelp density was highest at shallow depths in mid-Freshwater Bay. Invertebrate density was generally higher at mid-depth and deep sites than at shallow sites. Taxa richness tended to be higher at deep sites than at shallow or mid-depths, although the highest taxa richness occurred at shallow depths in mid-Freshwater Bay. Invertebrate density and taxa richness were generally higher offshore of the river mouth and to the west than to the east.

Spatial and depth-related patterns of density varied among kelp taxa (fig. 6.13). *P. californica* achieved high densities at shallow depths in Freshwater Bay and occurred in nearly all combinations of depths and cross-shore bands. *N. luetkeana* also achieved high densities at shallow depths in Freshwater Bay and occurred widely at lower densities, although less widely than *P. californica*. Understory kelp (species with short stipes and with blades lying close to the substrate) exhibited a variety of spatial and depth-related patterns. *A. marginata* was detected at shallow and mid-depths but not at deep sites, and off the Elwha River mouth and to the east but not to the west. In contrast, *Agarum fimbriatum* was most abundant at deep sites, nearly absent at shallow sites, and was detected off the Elwha River mouth and to the west but not to the east. *Cymathere triplicata* was present at shallow and mid-depths but not deep sites in nearly all cross-shore bands. *Laminaria setchellii* was most abundant at deep sites off the mouth and was absent at shallow depths. *Saccharina* spp. was detected across all depths and cross-shore bands with no clear preference for particular depths or regions. Similarly, *Costaria costata* and *Pleurophychus gardneri* were widely distributed although both species were rare at shallow sites east of the river mouth.

Density patterns also varied among abundant invertebrates that were identified to species (figs. 6.14 and 6.15). *C. oregonensis* (pigmy rock crab), the species with the highest mean density across all sites (fig. 6.8*B*), also had the most restricted distribution. It primarily was detected at shallow depths in mid-Freshwater Bay (fig. 6.14), and within that cross-shore band by depth statum it was detected at only the three sites with high percentages of bedrock (fig. 6.5*A*) where it attained a high density (1/m^2). The limited distribution of *C. oregonensis* may have been due to an association with holes in the bedrock. Burrowing clams excavate holes in the relatively soft rock (fig. 6.16*A*). After the clams die, their holes provide daytime hiding places for *C. oregonensis* (fig. 6.9*A*) from which the crabs venture at night to feed (Jensen, 1995).

Most of the other 11 species of invertebrates with high mean densities (fig. 6.8*B*) could be placed into four distribution categories. *Pycnopodia helianthoides* (sunflower star) and *Cancer productus* (red rock crab) were widespread, occurring at nearly all combinations of depths and cross-shore bands (figs. 6.14 and 6.15). *S. franciscanus* (red urchin), *Cucumaria miniata* (orange sea cucumber), *Fusitriton oregonensis* (hairy triton snail), *S. droebachiensis* (green urchin), and *Styela montereyensis* (stalked tunicate) occurred off the Elwha River mouth and to the west, but not to the east, at all depths. *Haliclystus stejnegeri* (stalked jellyfish) was detected primarily at shallow depths where it often attached to vegetation (fig. 6.9*G*). *P. generosa* (geoduck clam) occurred primarily at deep sites. *Henricia leviuscula* (blood star) and *C. stelleri* (gumboot chiton) were harder to place into one of these four distribution categories. *H. leviuscula* was detected off the river mouth, to the west at all depths, and to the east but only at deep sites off Dry Creek. *C. stelleri* was widespread, but less so than *P. helianthoides* or *C. productus*.

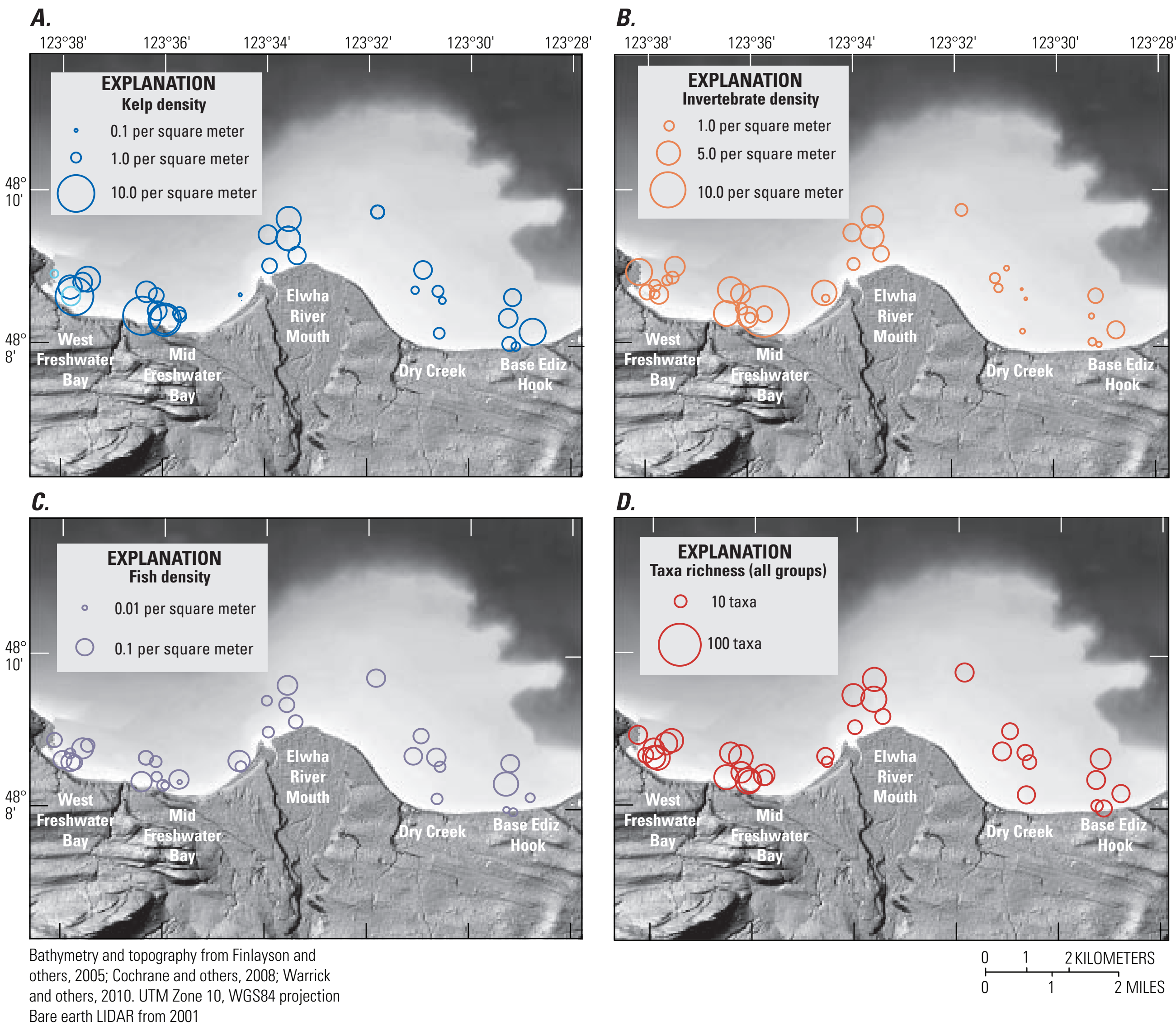


Figure 6.11. Density of (*A*) kelp, (*B*) invertebrates, and (*C*) fish, and (*D*) combined taxa richness for all three groups, at each of the 33 sampling sites near the mouth of the Elwha River, Washington, August and September 2008. Symbol size is proportional to density or taxa richness.

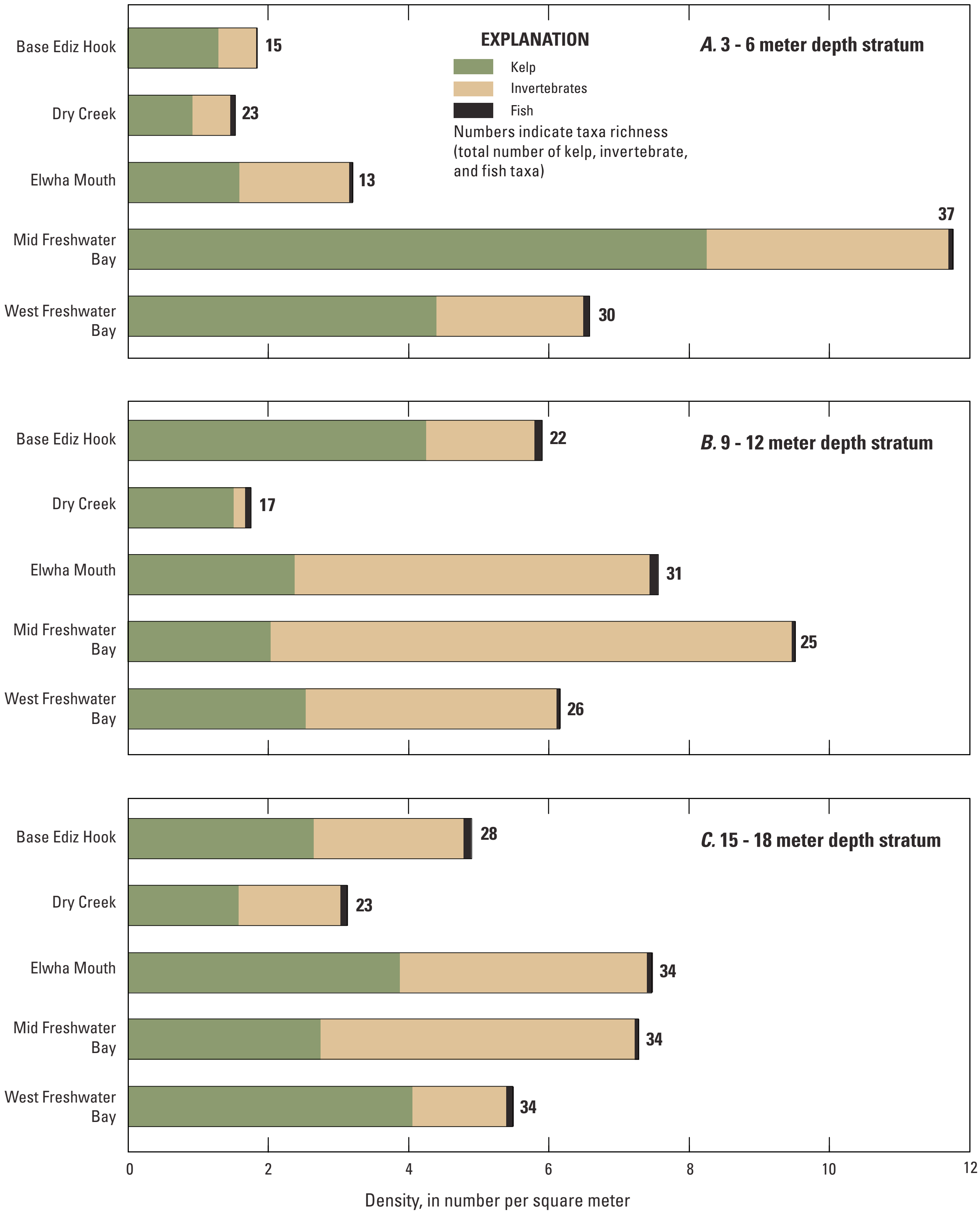


Figure 6.12. Mean densities of the major taxonomic groups (kelp, invertebrates, and fish) in each cross-shore band for (*A*) shallow, (*B*) mid-depth, and (*C*) deep sites near the mouth of the Elwha River, Washington, August and September 2008.

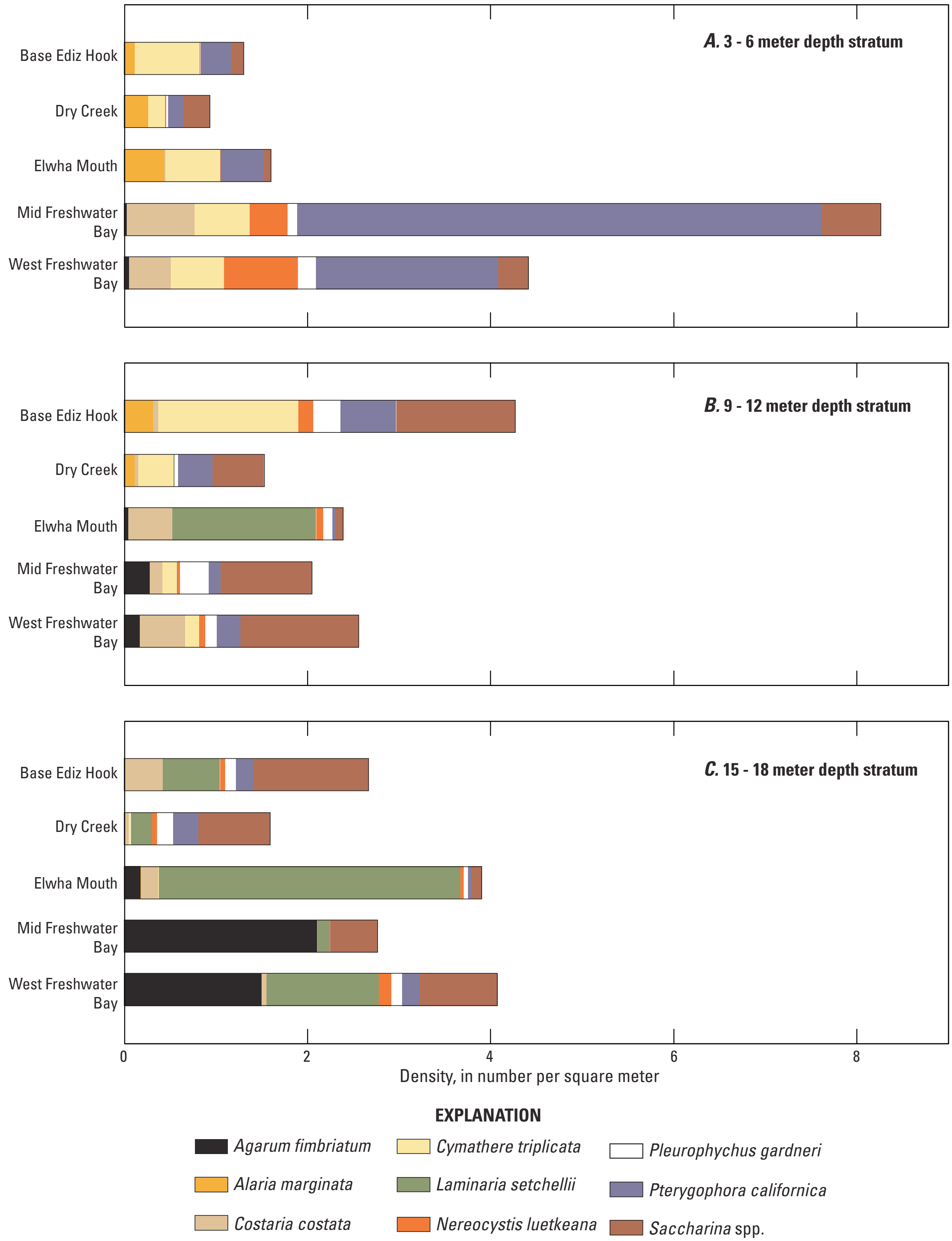


Figure 6.13. Mean densities of the nine kelp taxa in each cross-shore band for (*A*) shallow, (*B*) mid-depth, and (*C*) deep sites near the mouth of the Elwha River, Washington, August and September 2008.

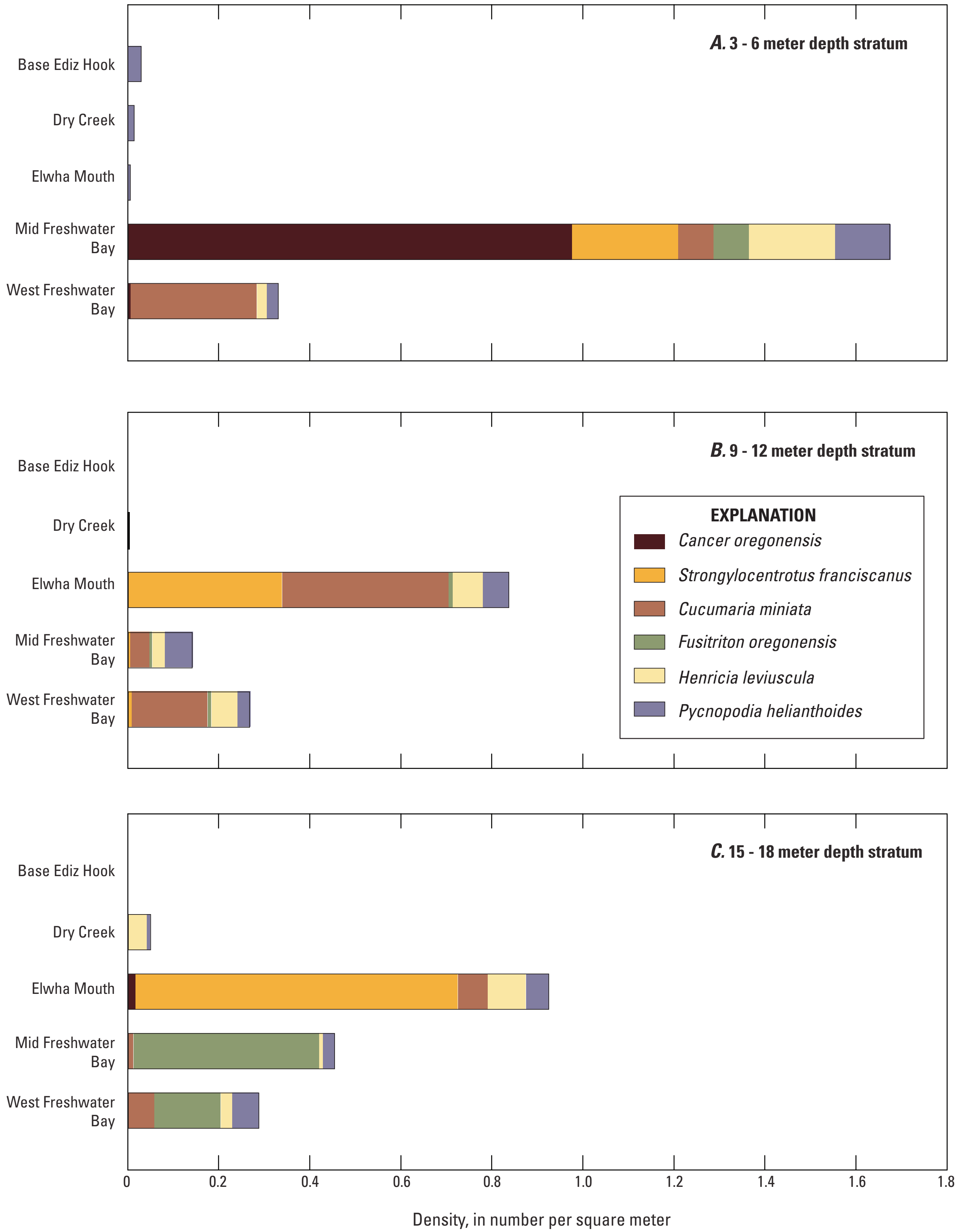


Figure 6.14. Mean densities of the six most common invertebrate species (see fig. 6.8*B*) in each cross-shore band for (*A*) shallow, (*B*) mid-depth, and (*C*) deep sites near the mouth of the Elwha River, Washington, August and September 2008.

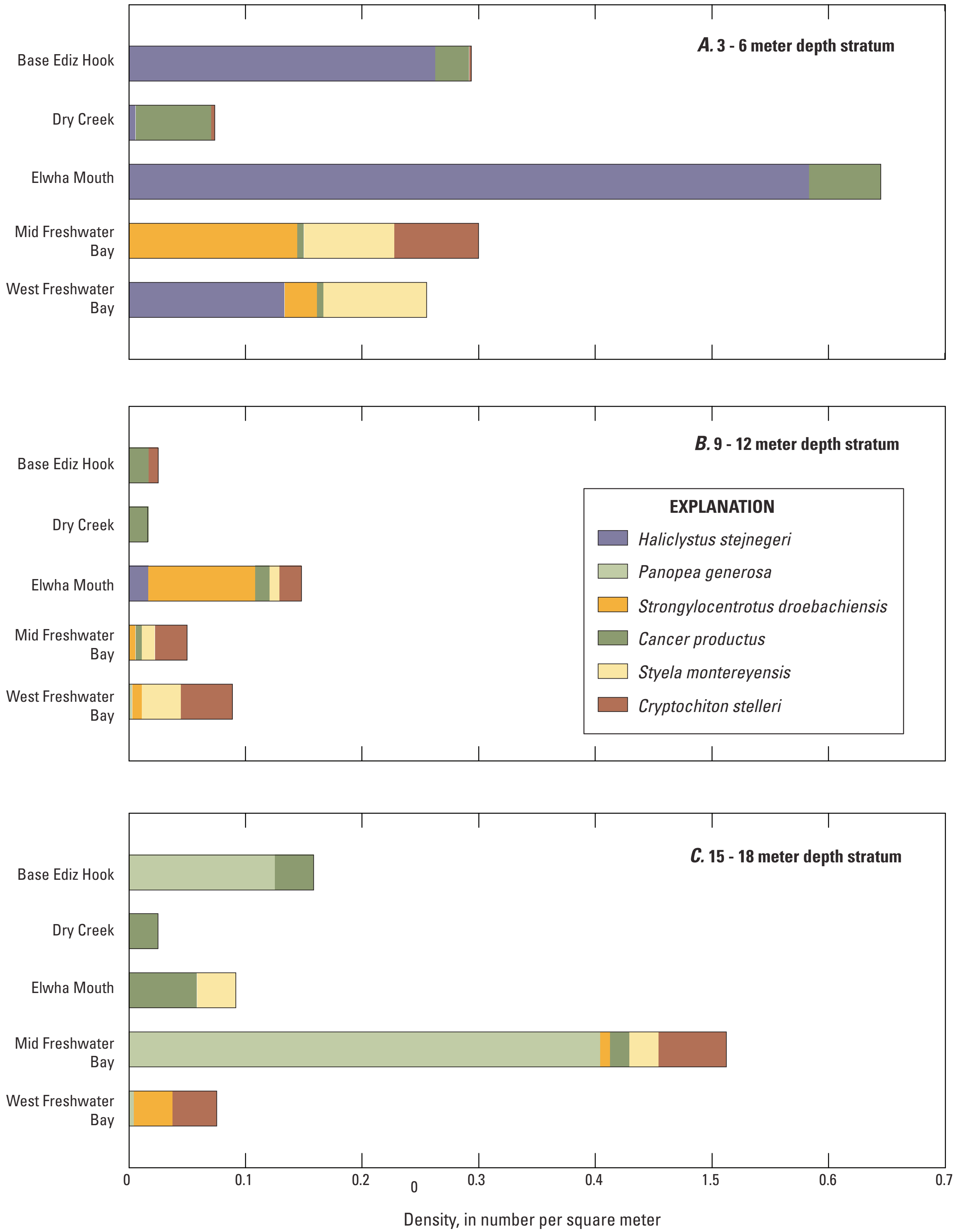
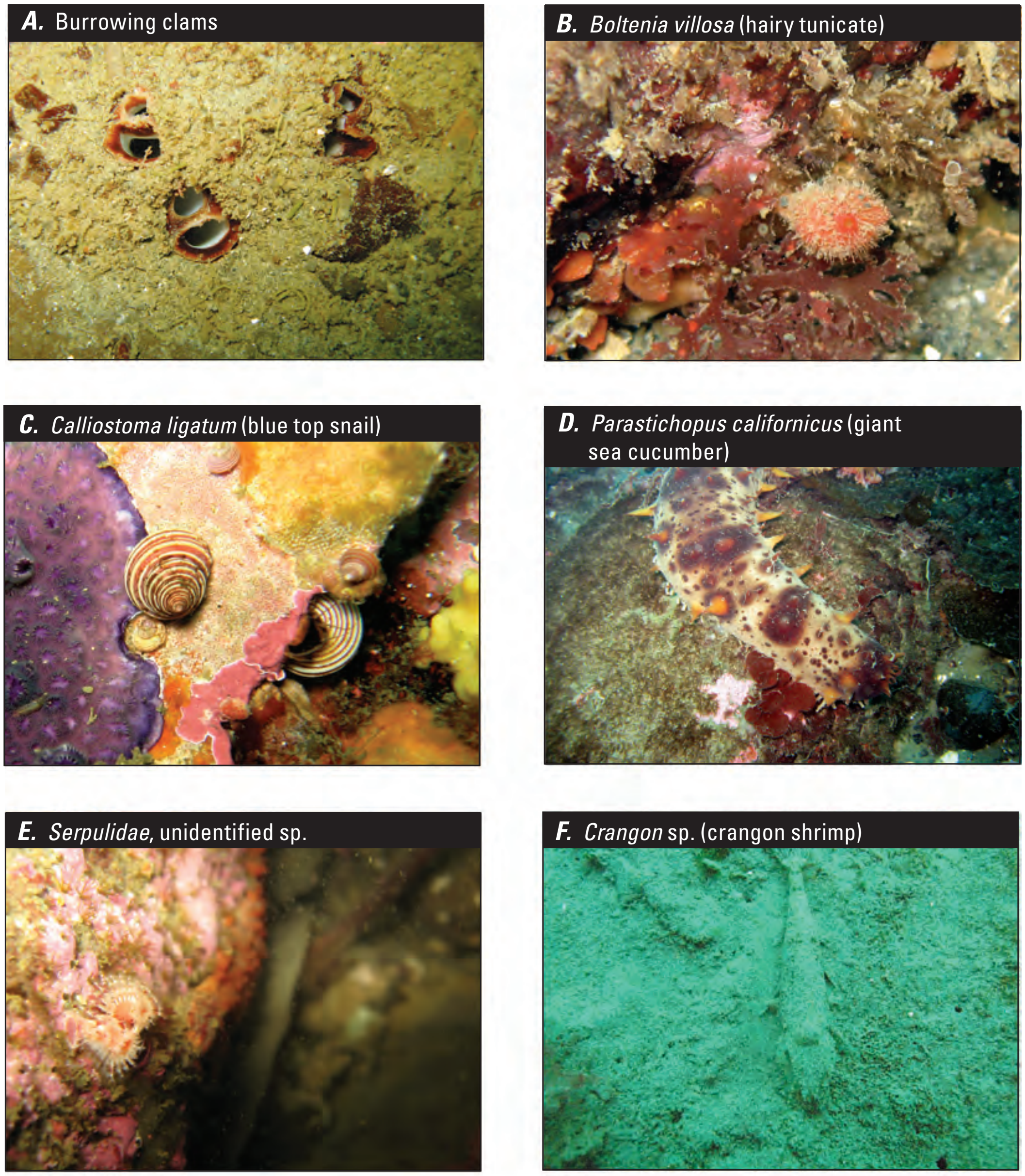


Figure 6.15. Mean densities of the next six most common invertebrate species (see fig. 6.8*B*) in each cross-shore band for (*A*) shallow, (*B*) mid-depth, and (*C*) deep sites near the mouth of the Elwha River, Washington. August and September 2008.



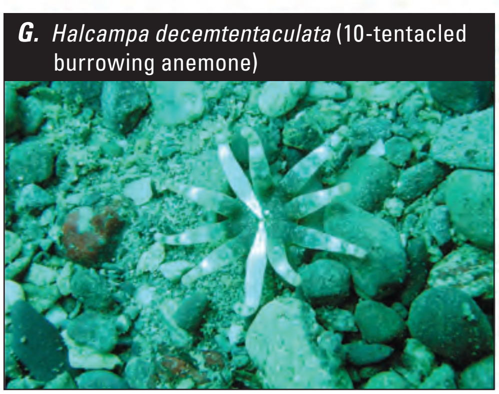


Figure 6.16. Additional invertebrates of interest (invertebrates other than those shown in fig. 6.9) at sites near the mouth of the Elwha River, Washington. Burrowing clams (*A*) excavate holes in soft bedrock. Taxa shown in photographs *B-E* were common at moderate relief sites and were not common at low relief sites (table 6.3). Worms in the family Serpulidae (*E*) have hard tubes made of calcium carbonate and tentacular crowns for feeding and respiration. Crangon shrimp (F) were uncommon at moderate relief sites and common at low relief sites (table 6.3). Of the invertebrates identified to genus or species, *Halcampa* spp. (*H. decemtentaculata* and *H. crypta*) was the only group to attain a mean density greater than 0.1/m^2 at sites classified as having mixed substrates and low relief.

Table 6.3. Taxa common at moderate relief sites and uncommon at low relief sites, and taxa uncommon at moderate relief sites and common at low relief sites near the mouth of the Elwha River, Washington, August and September 2008.

[Only the 22 sites where mean substrate score ranged from 1.25 to 2.15 (fig. 6.20*B*) were included. Mean relief score ranged from 1.12 to 1.35 at moderate relief sites (*n* = 9) and from 1.00–1.05 at low relief sites (*n* = 13). **Percent occurrence:** Percentage of total moderate relief sites or total low relief sites where a taxon was detected]

Occurrence-relief category	Percent occurrence		Taxon
	Moderate Relief	Low Relief	
Common-moderate, uncommon-low	88.9	0.0	*Cucumaria miniata* (orange sea cucumber)
	88.9	15.4	*Agarum fimbriatum* (an understory kelp)
	77.8	15.4	*Henricia leviuscula* (blood star)
	77.8	15.4	*Parastichopus californicus* (giant sea cucumber)
	66.7	0.0	*Serpulidae* (calcareous tube worms)
	55.6	0.0	*Boltenia villosa* (hairy tunicate)
	55.6	0.0	*Fusitriton oregonensis* (hairy triton)
	55.6	7.7	*Calliostoma* spp. (top snails)
	44.4	0.0	*Strongylocentrotus droebachiensis* (green urchin)
	44.4	7.7	*Styela montereyensis* (stalked tunicate)
Uncommon-moderate, common-low	11.1	76.9	*Alaria marginata* (an understory kelp)
	0.0	46.2	*Crangon* spp. (crangon shrimps)
	0.0	30.8	*Hexagrammos stelleri* (whitespotted greenling)

Density and Taxa Richness in Relation to Substrate and Relief

Results presented for relations of organism density and taxa richness to substrate and relief characteristics were obtained from analyses of means and SDs of substrate and relief scores rather than from analyses of substrate and relief composition. Mean substrate score is an index of average grain size of sediment. It ranged from 1.0 for sites with 100 percent sand to 3.4 for the site with the most bedrock (fig.6.17*A*). SD substrate score is an index of heterogeneity of sediment grain size. It ranged from 0.0 for sites with only one grain size (100 percent sand) to 1.2 for the site with the most even mixture of the four grain size classes. Similarly, mean and SD relief score are indexes of average relief and variability of relief, respectively (fig.6.17*B*). The analyses were based on means and SDs because they lend themselves to simple statistical techniques and straightforward interpretations of average tendencies and heterogeneity.

Kelp density increased with increasing mean substrate score and was highest at shallow sites with high percentages of boulders or bedrock (fig. 6.18*A*). The positive correlation between kelp density and mean substrate score was greater than would be expected by chance for all sites, for shallow sites (r = 0.91, P<0.05), and for mid-depth sites (r = 0.68, P<0.05), but not for deep sites. Invertebrate density was unrelated to mean substrate score when all sites were considered (fig. 6.18*B*) but was positively correlated with mean substrate score at shallow sites (r = 0.80, P<0.05). Fish density was unrelated to mean substrate score for all sites (fig. 6.18*C*) and within each depth stratum. Taxa richness increased with increasing mean substrate score for all sites (fig. 6.18*D*) and shallow sites (r = 0.81, P<0.05). SD substrate score was less correlated with kelp density and taxa richness than was mean substrate score for analyses conducted with all sites included, suggesting that heterogeneity of sediment grain size was less closely associated with kelp density and taxa richness than was average sediment grain size.

Relations of density and taxa richness to seafloor relief were similar to those for sediment grain size. Kelp density was positively correlated with mean relief score for all sites (fig. 6.19*A*) and shallow sites (r = 0.95, P<0.05). Invertebrate density was unrelated to relief for all sites (fig. 6.19*B*) but was positively correlated with relief at shallow sites (r = 0.80, P<0.05). Fish density was unrelated to relief (fig. 6.19*C*).

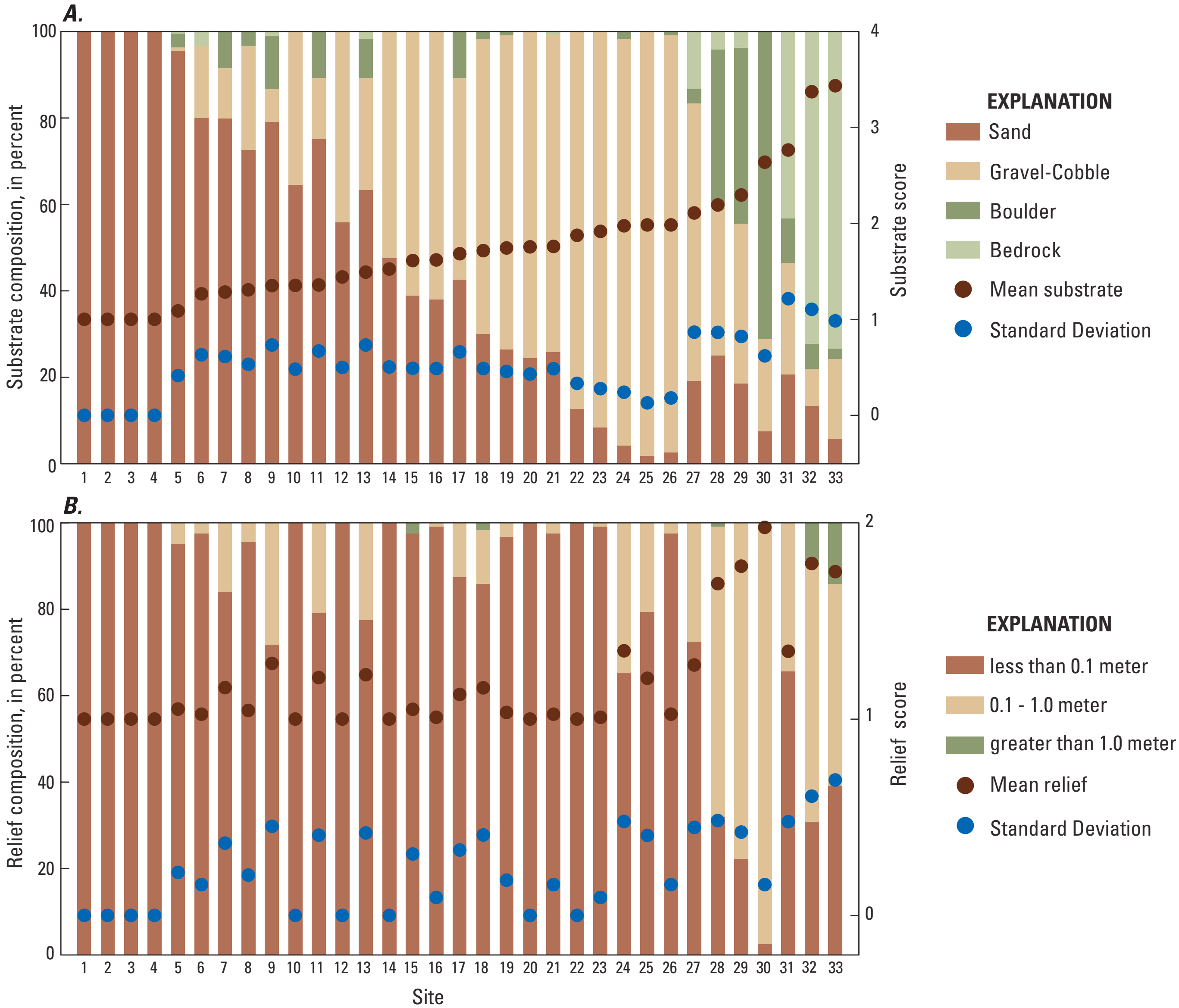


Figure 6.17. (*A*) Substrate composition and mean and standard deviation (SD) substrate score and (*B)* relief composition and mean and SD relief score for each site near the mouth of the Elwha River, Washington, August and September 2008. Sites are ordered from lowest to highest mean substrate score.

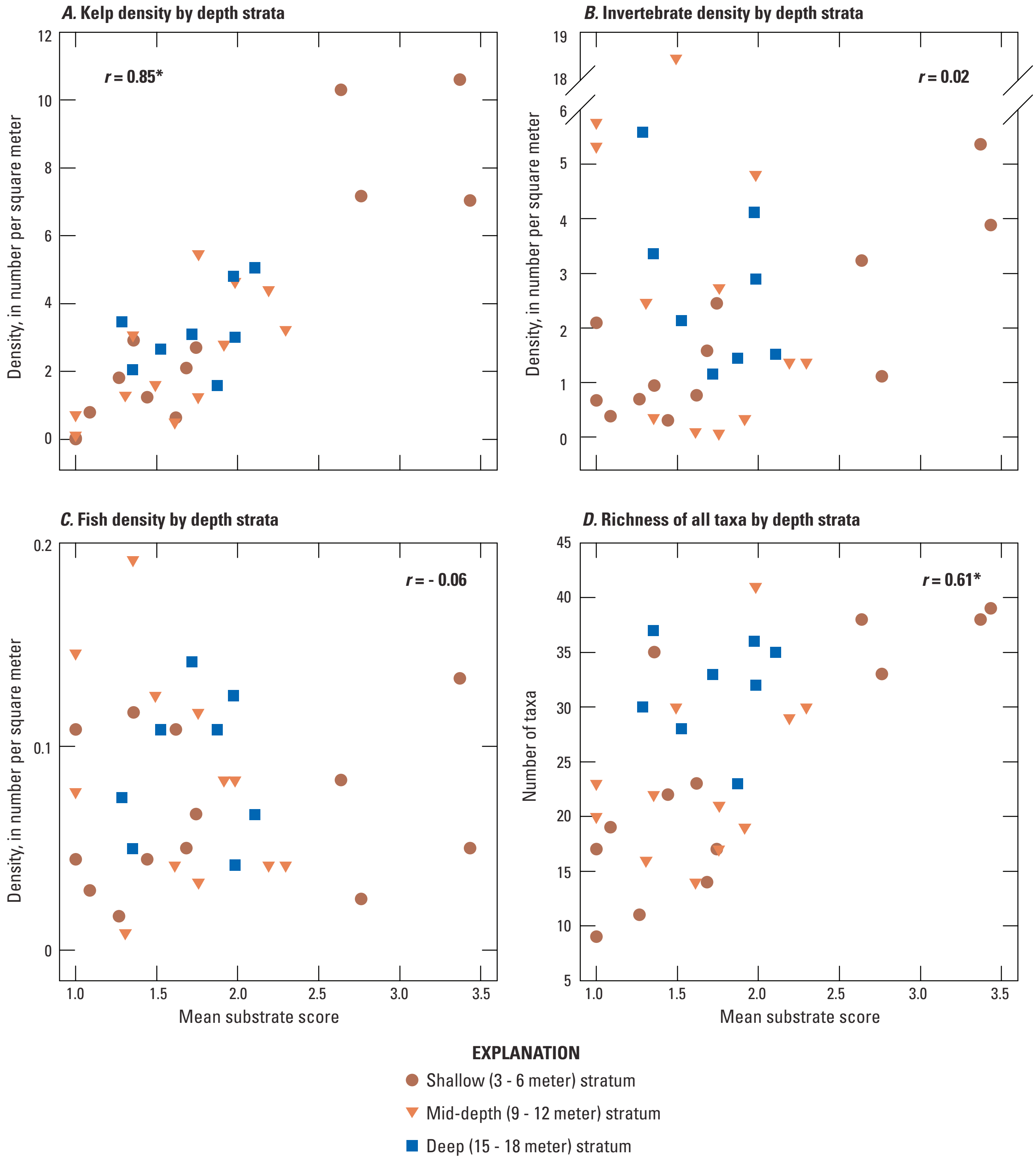


Figure 6.18. Density of (*A*) kelp, (*B*) invertebrates, and (*C*) fish, and (*D*) taxa richness for all three groups combined, in relation to mean substrate score at sites near the mouth of the Elwha River, Washington, August and September 2008. Each data point represents a site (n=33). Pearson's correlation coefficient (r) is a measure of association between two variables, for example, density and mean substrate score. Asterisks indicate that the correlation is stronger than would be expected by chance (P<0.05).

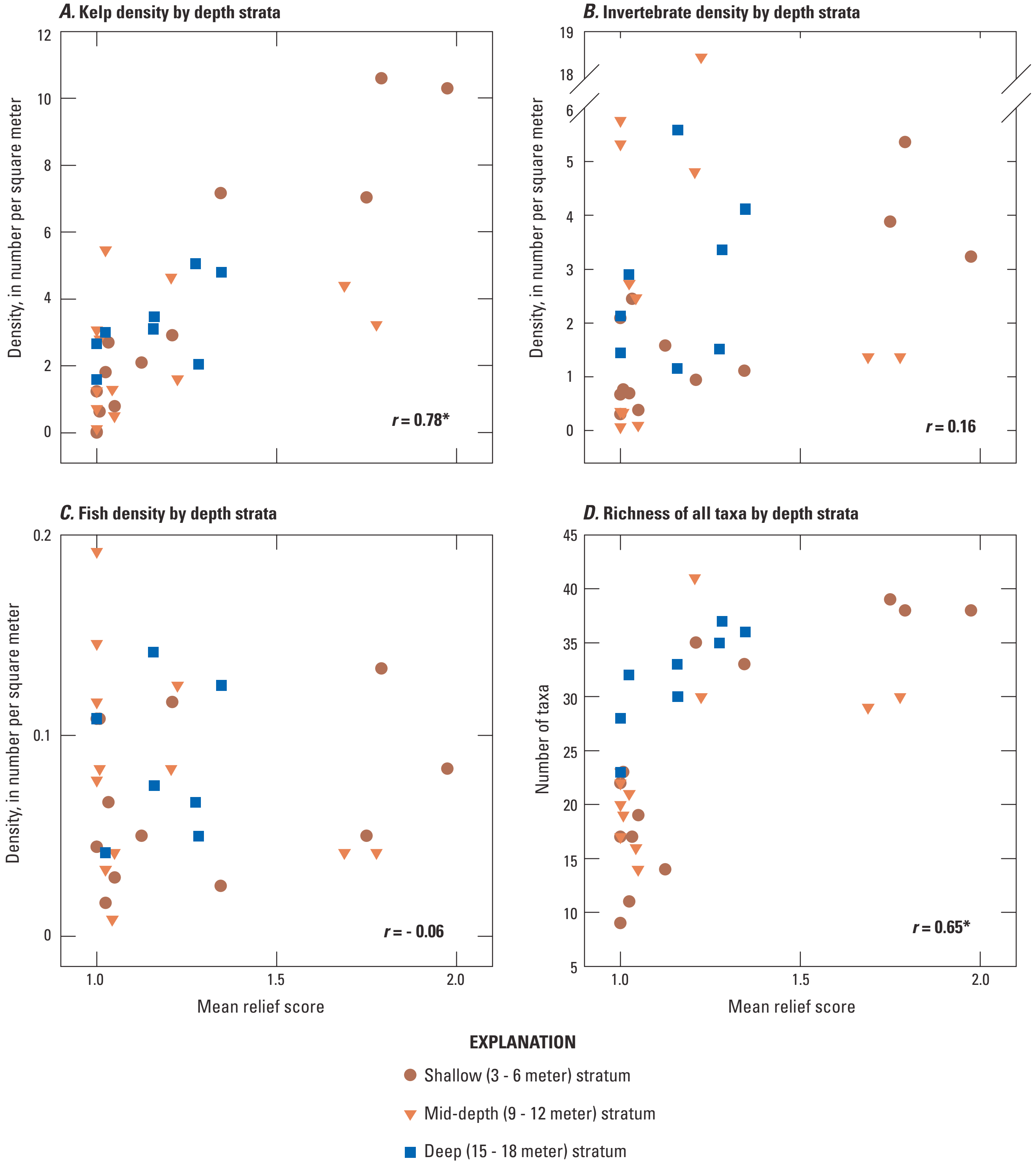


Figure 6.19. Density of (*A*) kelp, (*B*) invertebrates, and (*C*) fish, and (*D*) taxa richness for all three groups combined, in relation to mean relief score at sites near the mouth of the Elwha River, Washington, August and September 2008. Each data point represents a site (n=33). Pearson's correlation coefficient (r) is a measure of association between two variables, for example, density and mean substrate score. Asterisks indicate that the correlation is stronger than would be expected by chance (P<0.05).

Taxa richness was positively correlated with relief for all sites (fig. 6.19*D*), shallow sites ($r = 0.84$, $P<0.05$), and deep sites ($r = 0.84$, $P<0.05$). In comparison to mean relief score, SD relief score was less correlated with kelp density and nearly equivalently correlated with taxa richness ($r = 0.66$ between SD relief score and taxa richness) for analyses conducted with all sites included. This suggests that heterogeneity of relief was no more closely associated with kelp density or taxa richness than was average relief.

Mean substrate score was positively correlated with mean relief score (fig. 6.20*A*), indicating that sites with fine sediment also had low relief and sites with coarse sediment had high relief. Therefore, it is not surprising that mean substrate and mean relief were similarly related to density and taxa richness. However, mean substrate score was uncorrelated with mean relief score for the subset of sites with mean substrate scores between 1.25 and 2.15 (fig. 6.20). Some of these sites had low relief (mean relief scores between 1.00 and 1.05) whereas others had moderate relief (mean relief scores between 1.12 and 1.35; fig. 6.20*B*). Density and taxa richness were compared between the moderate and low relief sites to determine whether there was an effect of relief independent from the effect of sediment grain size.

Kelp density was similar between moderate and low relief sites (fig. 6.21*A*). Invertebrate density was higher at moderate relief sites than at low relief sites (fig. 6.21*B*). Mean invertebrate density at moderate relief sites was 1.6/m^2 higher than at low relief sites when one site was excluded (the moderate relief site where density was 18.5/m^2) and 3.3/m^2 higher than at low relief sites when all sites were included. Fish density was similar between moderate and low relief sites (fig. 6.21*C*). The difference in taxa richness between moderate and low relief sites was striking (fig. 6.21*D*). On average, 12 more taxa occurred at moderate relief sites than at low relief sites.

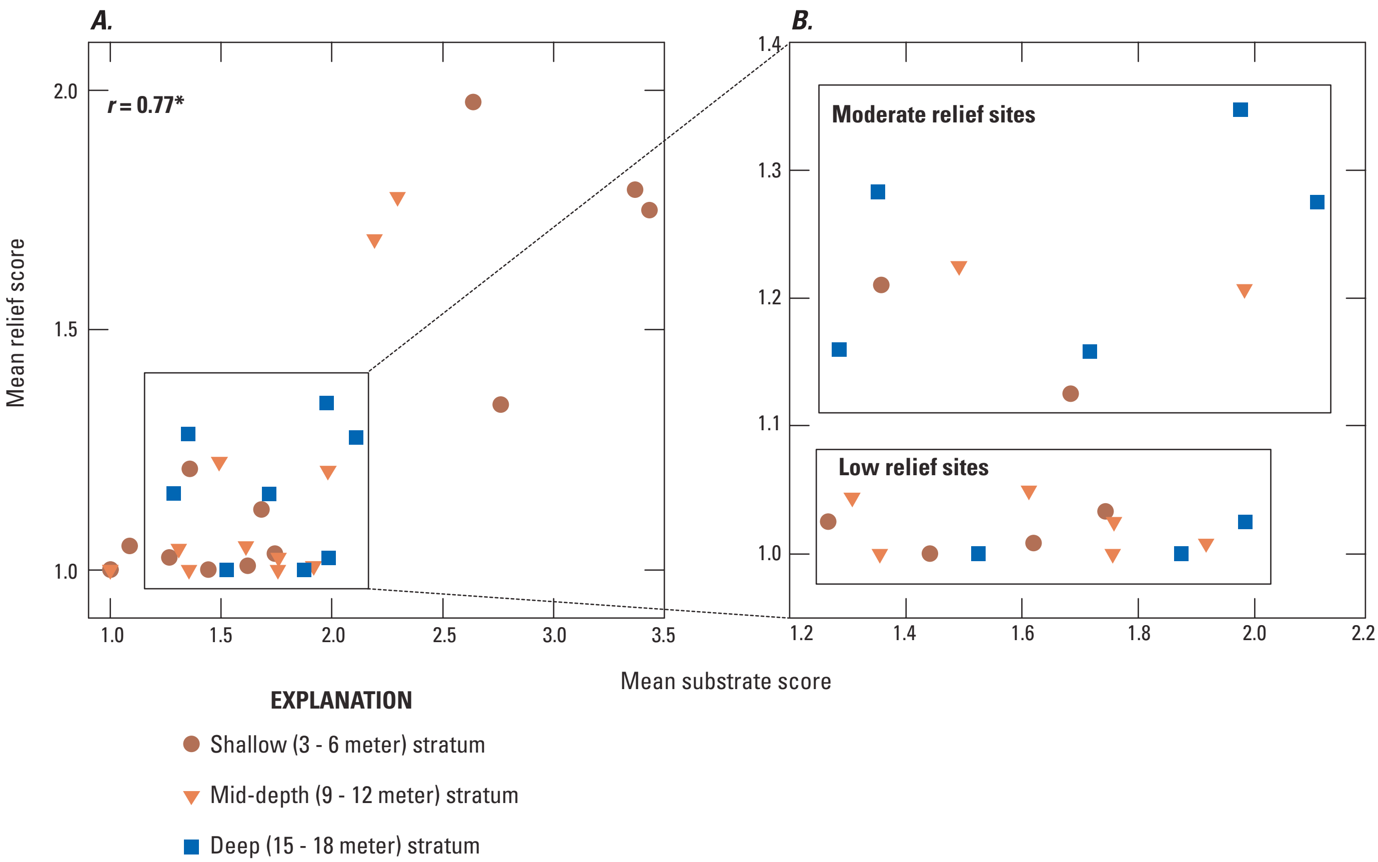


Figure 6.20. Mean relief score in relation to mean substrate score at sites near the mouth of the Elwha River, Washington, August and September 2008. (*A*) All 33 sites. (*B*) The 22 sites with mean substrate scores between 1.25 and 2.15.

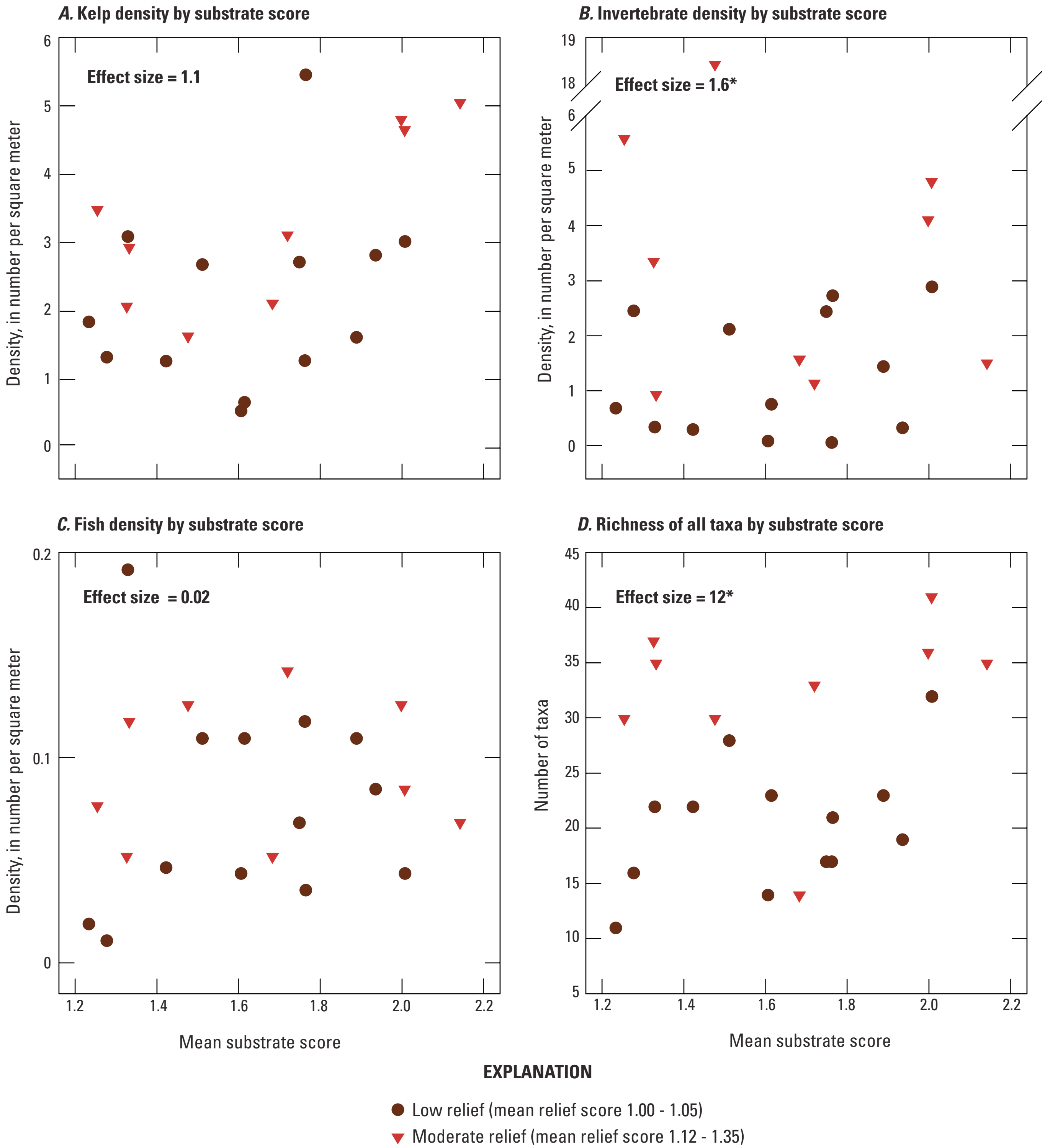


Figure 6.21. Density of (*A*) kelp, (*B*) invertebrates, and (*C*) fish, and (*D*) taxa richness for all three groups combined, in relation to mean substrate score at sites near the mouth of the Elwha River, Washington, August and September 2008. Only the 22 sites with mean substrate scores between 1.25 and 2.15 are shown. Effect size is the difference between the mean for moderate relief sites and the mean for low relief sites. Asterisks indicate that effect size is greater than would be expected by chance ($P<0.05$). Note that for invertebrates, effect size and its asterisk were calculated by excluding the moderate relief site where density was 18.5, much higher than at any other site. Effect size was 3.3 with that site included.

Taxa Associated with Seafloor Relief

The finding that taxa richness was higher at moderate relief sites than at low relief sites was intriguing. It suggests that some taxa preferentially occurred at sites with at least moderate relief, thereby increasing taxa richness at those sites. Further analysis was performed to screen for taxa that were common at the moderate relief sites but uncommon at low relief sites, or the opposite.

Ten taxa were classified as common at moderate relief sites and uncommon at low relief sites, and 3 taxa were classified as uncommon at moderate relief sites and common at low relief sites (table 6.3; fig. 6.7*A*–*B*; fig. 6.9*C*–*E*, *I*, and *K*; fig. 6.10*G*; fig. 6.16*B*–*F*). Thus, more taxa occurred mainly at moderate relief sites than at low relief sites, contributing to the increase in taxa richness with relief. Relief at moderate relief sites usually was provided by a few boulders perched on top of sand or gravel-cobble substrate. Several of the taxa detected mainly at moderate relief sites, including *Serpulidae*, *Boltenia villosa*, and *S. montereyensis*, are sessile and require hard, stable substrate for attachment. Other taxa, notably *F. oregonensis*, were nearly always observed on boulders or bedrock. Thus, the presence of boulders may have allowed the occurrence of species that require or prefer hard substrate, in addition to species requiring or tolerating finer substrates, thereby increasing taxa richness. Of the five common invertebrate species that were detected offshore of the river mouth and to the west but not to the east (figs. 6.14 and 6.15), four (*C. miniata*, *F. oregonensis*, *S. montereyensis*, and *S. droebachiensis*) were detected mainly at moderate relief sites. Their distribution pattern may have been due to the availability of seafloor relief offshore of the river mouth and to the west and the lack of such habitat to the east (fig. 6.5*B*).

Habitat Types and Associated Taxa

The study sites were qualitatively classified into four habitat types based on physical and biological characteristics (table 6.4 and fig. 6.22).

Bedrock/Boulder Reef

Three shallow sites in central Freshwater Bay were located on a bedrock reef. One shallow site in west Freshwater Bay was included in the same habitat type as the bedrock sites because of its predominantly hard substrate (boulders) and similar species at similar densities. Bedrock/boulder reef habitat supported beds of *N. luetkeana* and *P. californica* that respectively formed nearly continuous canopies at the water surface and 1–2 m above the seafloor, and a diverse assemblage of understory kelp and invertebrates (fig. 6.23*A*–*B*). It also supported a density of kelp more than twice that for any other habitat type and the highest taxa richness among habitat types (fig. 6.24).

Sand and Gravel-Cobble with Moderate Relief

Most of the mid-depth and deep sites in Freshwater Bay and offshore of the Elwha River mouth had mixed sand and gravel-cobble substrate with sufficient boulder abundance to provide moderate relief (fig. 6.23*C*–*D*). The relative fractions of sand and gravel-cobble varied considerably within this classification. From a biological perspective, the presence of large boulders, which provided hard, stable substrate and relief that some organisms seemed to depend on, was more important than the composition of the base substrate. Kelp and invertebrate assemblages included species that were rare at sites lacking relief. The mixed substrate-moderate relief habitat type supported the highest density of invertebrates among habitats with taxa richness nearly equivalent to that for bedrock/boulder reefs (fig. 6.24).

Sand and Gravel-Cobble with Low Relief

The substrate was composed of mixed sand and gravel-cobble with little relief at most sites east of the Elwha River mouth, at one deep site offshore of the river mouth, and at one mid-depth site in central Freshwater Bay (fig. 6.23*E*–*F*). The relative fractions of sand and gravel-cobble varied within this classification but varied less than for the moderate relief classification. Kelp was dominated by understory species, notably *A. marginata* that did not occur west of the river mouth. This habitat supported a density of kelp close to that for the moderate relief classification, but density of invertebrates was the lowest among habitat types and taxa richness was considerably lower than for moderate relief (fig. 6.24).

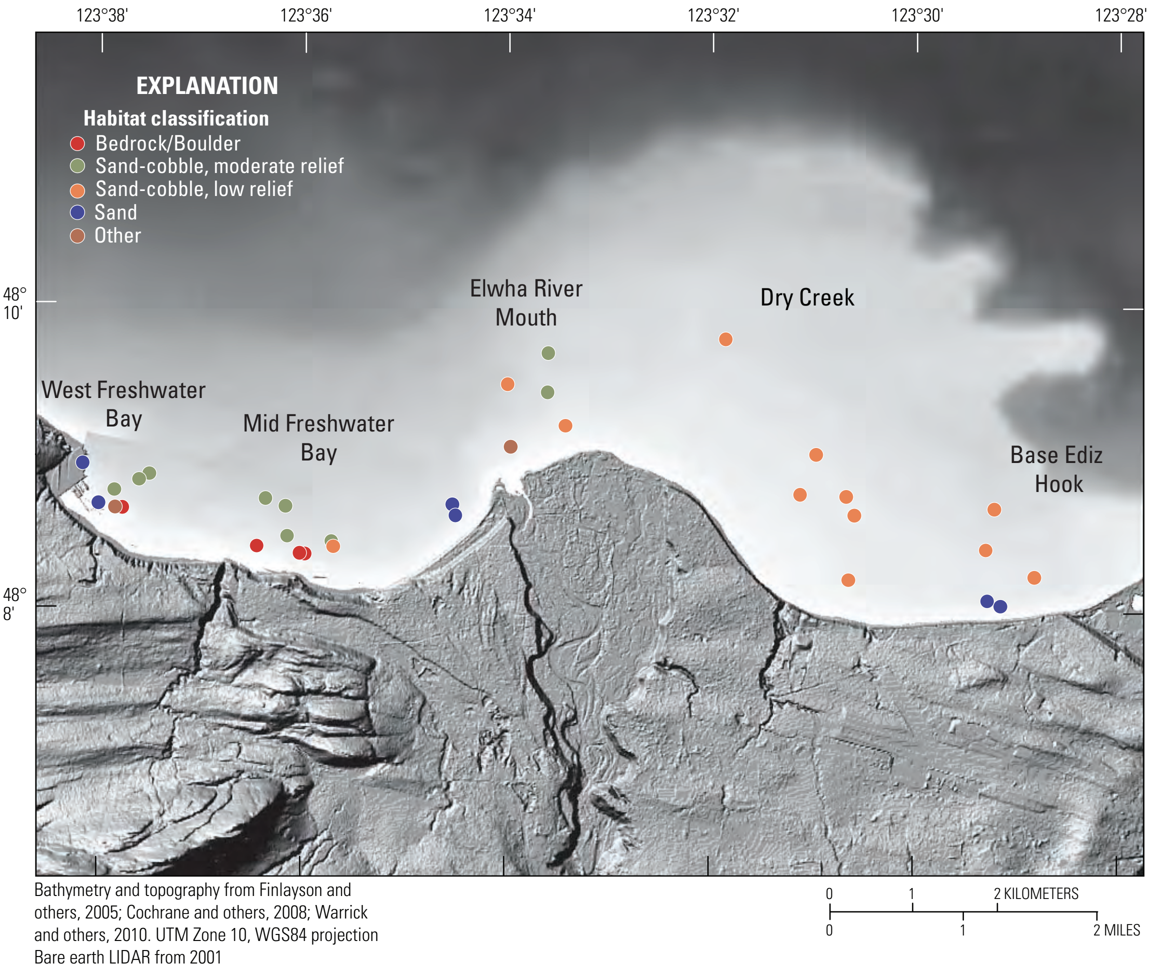


Figure 6.22. Qualitative classification of four habitat types at sites near the mouth of the Elwha River, Washington. The "other" category contains atypical sites that did not fit well into any of the four types. Descriptions of physical and biological characteristics for each habitat type are shown in table 6.4.

Figure 6.23. Opposite page. Four habitat types that were qualitatively classified at sites near the mouth of the Elwha River, Washington. (Characteristics of the habitat types are listed in table 6.4, and locations of the habitat types are shown in fig. 6.22). (*A*–*B*) Bedrock/boulder reef supporting a kelp forest. (*A*) View under the canopy formed by *Pterygophora californica. P. californica* stipes (stems) are attached to bedrock encrusted with pink coralline algae; orange mysid shrimp are visible in the water column. (*B*) View above the *P. californica* canopy. The vertically oriented "ropes" are actually *Nereocystis leutkeana* (bull kelp) stipes that extend from the bedrock to pneumatocysts (floats) at the water surface. Several blades (leaves) grow from each pneumatocyst forming a nearly unbroken canopy at the surface. (*C*–*D*) Mixed sand and gravel-cobble substrate with moderate relief provided by a few boulders. (*C*) A boulder on substrate composed primarily of sand. Macroalgae and a *Cryptochiton stelleri* (gumboot chiton) are visible on the boulder. (*D*) Gravel-cobble substrate and a boulder covered with *Strongylocentrotus franciscanus* (red urchins). (*E*–*F*) Mixed sand and gravel-cobble substrate lacking boulders and therefore showing low relief. (*E*) A mid-depth site offshore of Dry Creek; *Cymathere triplicata* is visible. (*F*) A mid-depth site at the base of Ediz Hook; *C. triplicata*, *Costaria costata*, and *Saccharina* sp. are visible. (*G*–*H*) Sand substrate. (*G*) A *Zostera marina* (eelgrass) bed at the shallower of the two sandy sites in west Freshwater Bay; a *Polyorchis penicillatus* (redeye medusa) is visible in the water column. (*H*) The deeper of two sandy sites in east Freshwater Bay; a *Cancer magister* (Dungeness crab) is visible in the foreground.

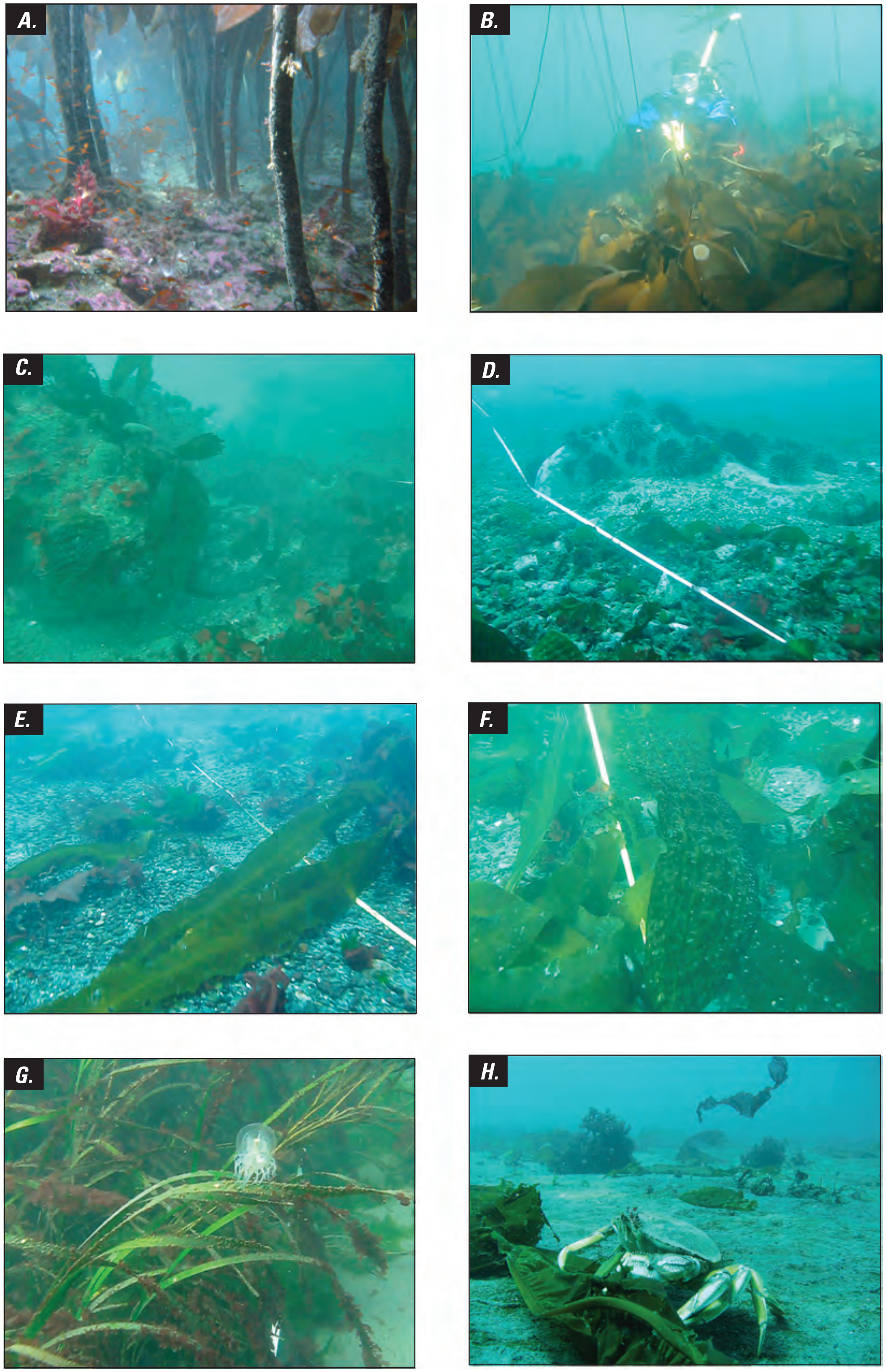

A.
B.
C.
D.
E.
F.
G.
H.

Table 6.4. Qualitative classification of four habitat types based on physical and biological characteristics at sites near the mouth of the Elwha River, Washington.

[Site locations are shown in figure 6.22. Note that two atypical sites were classified as "other" and are not included in this table. Taxa identified to genus or species are listed in order of decreasing mean density. All such taxa with mean density greater than a minimum value (0.17 per square meter for kelp, 0.10 per square meter for invertebrates, 0.005 per square meter for fish) are listed. Of the taxa identified to a broader taxonomic level than genus, the taxon with the highest mean density is listed after the semicolon]

Characteristic	Habitat type			
	Bedrock-boulder reef	Sand and gravel; moderate relief	Sand and gravel; low relief	Sand
		Physical		
Number of sites	4	9	12	6
Depth strata	Shallow	Medium-deep	Shallow-deep	Shallow-medium
Depth[1]	5.3 (3.8–6.1)	13.6 (9.1–16.7)	10.9 (5.4–16.8)	6.4 (3.0–10.4)
Substrate composition[2]	47.2, 22.5, 18.4, 11.8	2.6, 12.6, 49.1, 35.6	0.1, 0.4, 64.7, 34.8	0.6, 0.6, 2.9, 95.9
Substrate score[3]	3.1 (2.6–3.4)	1.8 (1.3–2.3)	1.7 (1.3–2.0)	1.1 (1.0–1.3)
Relief composition[4]	34.5, 59.4, 6.0	65.6, 34.1, 0.3	98.6, 1.2, 0.2	98.8, 1.3, 0.0
Relief score[5]	1.7 (1.3–2.0)	1.3 (1.2–1.8)	1.0 (1.0–1.0)	1.0 (1.0–1.1)
Boulder abundance	Low-high	Medium	Very low	Very low
		Taxa assemblage		
Kelp	*Pterygophora californica, Nereocystis leutkeana, Costaria costata, Cymathere triplicata, Saccharina* spp., *Pleurophychus gardneri*	*Agarum fimbriatum, Saccharina* spp., *Costaria costata, Pleurophychus gardneri*	*Saccharina* spp., *Cymathere triplicata, Pterygophora californica, Alaria marginata*	*Cymathere triplicata*
Invertebrates	*Cancer oregonensis, Cucumaria miniata, Strongylocentrous franciscanus, Henricia leviuscula, Strongylocentrous droebachiensis, Styela montereyensis, Pycnopodia helianthoides;* calcareous tube worms (*Serpulidae*)	*Cucumaria miniata, Strongylocentrous franciscanus, Calliostoma* spp., *Fusitriton oregonensis;* soft tube worms	*Halcampa* spp.; soft tube worms	Soft tube worms
Fish	*Hexagrammos decagrammus, Jordania zonope;* unidentified sculpins	*Hexagrammos decagrammus, Gadus macrocephalus* juveniles; unidentified sculpins[6], gunnels[6]	*Gadus macrocephalus* juveniles, *Hexagrammos decagrammus, Hydrolagus colliei*; unidentified sculpins	*Ophiodon elongatus* juveniles, Blepsias cirrhosus (only in eelgrass); flatfish
Eelgrass				West Freshwater Bay only

[1]Mean (range) in meters, referenced to mean lower low water.

[2]Mean percentage bedrock, boulder, gravel-cobble, and sand, respectively.

[3]Mean (range); substrate score is an index of average sediment grain size.

[4]Mean percentage in three categories: less than 0.1 meter, 0.1–1.0 meter, and greater than 1 meter, respectively.

[5]Mean (range); relief score is an index of average seafloor relief.

[6]Unidentified sculpins and gunnels tied for the highest mean density among taxa identified to a broader taxonomic level than genus.

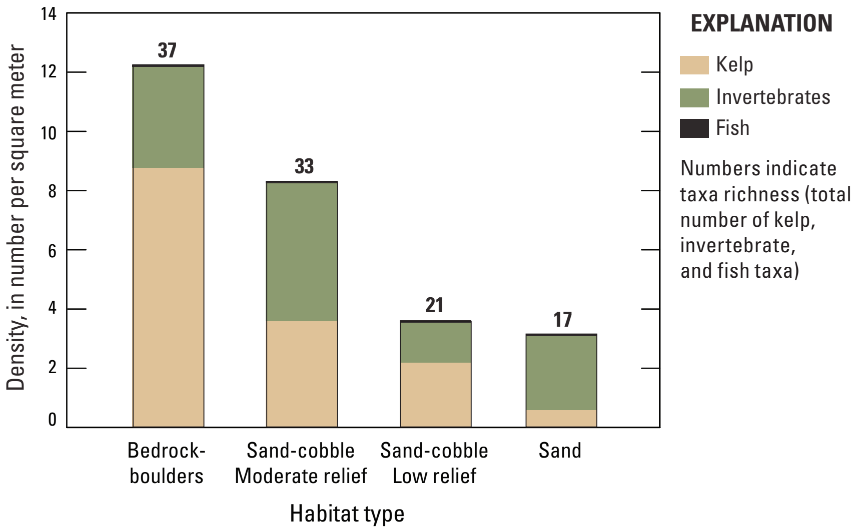


Figure 6.24. Mean densities for kelp, invertebrates, and fish for four qualitative habitat type classifications at sites near the mouth of the Elwha River, Washington, August and September 2008. (Characteristics, locations, and photographs of the habitat types are shown in table 6.4, figure 6.22, and figure 6.23, respectively.)

Sand

Sites dominated by sand were widely dispersed from west to east. Two sites were at the west end of Freshwater Bay, two were at the east end, and two were at the base of Ediz Hook. Eelgrass was interspersed with patches of bare sand at the shallow site in west Freshwater Bay (fig. 6.23*G*). Eelgrass was absent from the other five sandy sites, although it was present at one of the sites classified as "other." Small amounts of substrate allowing kelp attachment (for example, boulders or woody debris), or cobbles with kelp attached that were imported from other areas (Warrick and others, 2011, Chapter 5, sidebar 5.2, this report), were sometimes present at sandy sites, accounting for the occurrence of kelp at low density. Invertebrate density was more twice that for the mixed substrate-low relief habitat and taxa richness was the lowest among habitat types (fig. 6.24).

Other

Two sites were classified as "other" because they did not fit well into any of the four habitat types. The shallow site directly offshore of the Elwha River mouth was a moderate relief site, but it had the lowest mean relief score among the moderate relief sites (fig. 6.20*B*) and a taxa richness considerably lower than for any of the other moderate relief sites (fig. 6.21*D*). A shallow site in west Freshwater Bay was classified as "other" because it was on a boundary between habitat types. One transect was in sand with eelgrass, the other on boulder reef.

Key Findings

The results of this study indicate that biological communities in the study area were partly controlled by substrate, relief, and depth. Communities differed markedly among hard substrate (bedrock or boulders), mixed substrate (sand and gravel-cobble), and sand. The presence of relief (boulders) in areas with mixed substrate increased species richness by providing habitat for species that otherwise would not be represented. Kelp proliferated at shallow depths (3–6 m) given suitable substrate but also was abundant at deep sites (15–18 m). Community composition of kelps and invertebrates varied with depth.

The distribution of substrate, relief, and depth within the study area created a mosaic of habitats and communities. Freshwater Bay was particularly rich. Present at shallow sites in Freshwater Bay, from west to east, were an eelgrass bed, a bedrock/boulder reef supporting a dense and diverse kelp forest, and a sandy area lacking eelgrass but supporting *Cancer magister* (Dungeness crab; fig. 6.23*H*) and juvenile *O. elongatus* (lingcod; fig. 6.10*D*) among other species. At greater depths in Freshwater Bay, mixed sand and gravel-cobble substrate and the presence of boulders supported a distinct community nearly as diverse as the kelp forest. Mid-depth and deep sites directly offshore of the Elwha River mouth merit attention. Diversity was high at these sites (fig. 6.11*D*). Substrate was almost entirely (greater than or equal to 94 percent) gravel-cobble that appeared stable and may have provided habitat for species needing hard substrate even when boulders were absent. Tidal currents here are among the highest in the study area (Warrick and others, 2011b, chapter 5, this report). The distribution of substrate types important for structuring biological communities in Freshwater Bay and offshore of the river mouth is well characterized by the substrate morphology map of Warrick and others (2008) (fig. 6.2*B*).

Mixed substrate lacking boulders predominated at sites east of the Elwha River mouth. The community here was

Sidebar 6.1 Fine Sediments and the Role of Benthic Amphipods

Ian M. Miller, Jeffrey J. Duda, Nancy Elder, Reginald R. Reisenbichler, and Stephen P. Rubin

Although the focus of this study was on benthic invertebrates larger than 2.5 cm, qualitative observations were made of smaller-bodied invertebrates that may play a role in the marine biological community or may serve as useful indicators of change following dam removal, including brittle stars, mysid shrimp, and small tube worms. Of particular interest are benthic amphipods of the genus Ampelisca. These small, tube-building amphipods were abundant at a few sites that included sediments finer than sand.

The genus *Ampelisca*, typically associated with bottoms of soft sediments (Dauvin and BellanSantini, 1996), is globally distributed and can be abundant in shallow coastal environments. In Chile, for example, a species of this genus is associated with sediments composed of more than 50 percent silt and clay (Carrasco and Arcos, 1984). Almost all members of this genus build tubes constructed of sediment grains, using the tube as a protection against predatory grazing. During feeding the amphipods extend part of their body from the top of the tube and use their antennae to collect detritus, plankton or other epibenthic invertebrates from the surrounding seafloor or water column (Sheader, 1998). Their tube-building habit modifies the substrate and influences the exchange of material between sediments and the water column. One study from the east coast of the United States suggests that *Ampelisca* tubes provide structure to sea-floor sediments, making them more desirable as a recruitment substrate for commercially important quahog clams (*Mercenaria mercenaria*) (Mackenzie and others, 2006). The abundant amphipods also provide food for a number of larger animals including gray whales (*Eschrichtius robustus*) (Oliver and Slattery, 1985; Darling and others, 1998). *Ampelisca* are thought to be a primary food source for a variety of fish species (Carrasco and Arcos, 1984), in some cases contributing as much as 88 percent of the prey base (Franz and Tanacredi, 1992).

Field observations made during this study and sonar-derived backscatter interpretation from a previous study (Warrick and others, 2008) suggest that sediments finer than sand are rare within the Elwha coastal zone. Sediments usually were composed of coarser sand or a mix of sand and gravel even at sites classified as "soft". Although fine sediments were generally lacking in the Elwha River nearshore area, tubes of *Ampelisca* were observed at one shallow site at the western edge of Freshwater Bay (fig. S6.1A), in clearings between dense thickets of eelgrass (*Zostera marina*). The substrate at this site was primarily sand, but it may have included silt and clay because these grain sizes often are associated with eelgrass in Puget Sound (Mumford, 2007)

Water visibility, measured by divers, likely can be used as a proxy for the presence of fine sediments, because re-suspension of mud and silt can cloud the water. Mean visibility for all shallow sites at the two control areas (Green Point and Low Point) was lower than for shallow sites in the Elwha coastal zone (fig. S6.2), suggesting the presence of fine sediments in control area substrates. The reduced visibility at Green Point and Low Point was not simply the product of a single low visibility dive skewing the data. Out of 85 visibility measurements made in 2008, the lowest 10 occurred in these two areas.

Ampelisca was observed in abundance at shallow sites offshore of Green Point (fig. S6.1B). Gray whales were observed at these sites exhibiting behavior associated with bottom feeding—repeatedly diving and creating clouds of silt visible at the surface (fig. S6.3A). A year later at the same location trenches were observed in the seafloor, approximately 3 m long, 1–2 m wide and 0.2–0.5 m deep, which were interpreted as gray whale feeding excavations. Along the edges of these trenches, the sediment tubes of Ampelisca and inactive or dead amphipods were observed on the bed

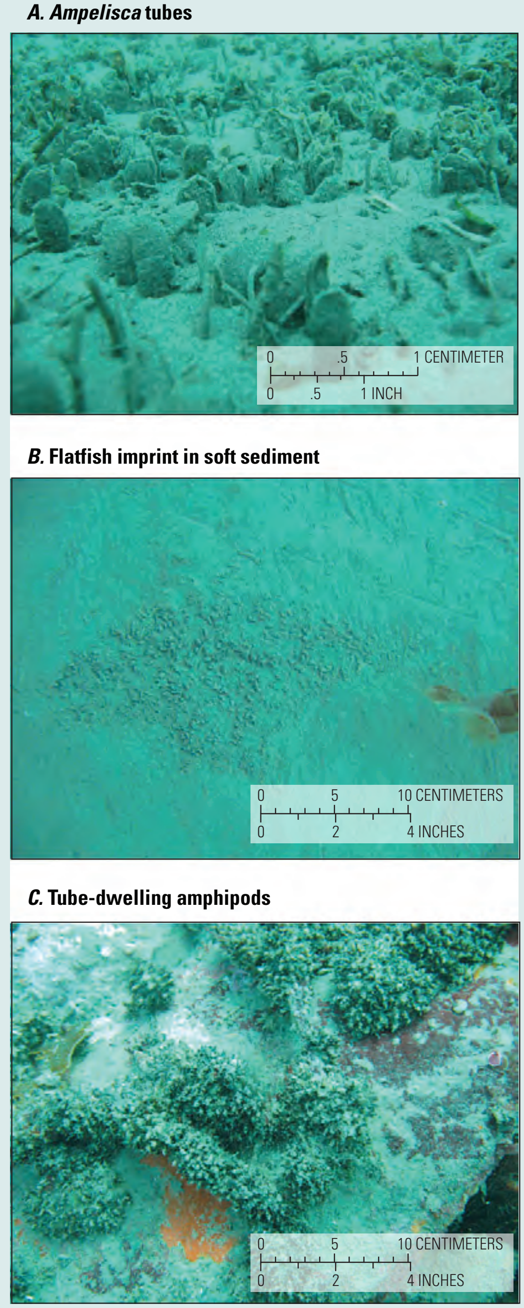


Figure S6.1. (*A*) *Ampelisca* tubes photographed at macro-range in an eelgrass bed at the west end of Freshwater Bay, September 9, 2009. (*B*) Outline of a flatfish in soft sediment off of Green Point, Siebert Creek, Strait of Juan de Fuca. Within the outline are numerous tubes of the amphipod *Ampelisca*, August 12, 2009. (*C*) Tube-dwelling amphipods, perhaps of the genus Ameplisca, attached to a small boulder off shore of Low Point, Lyre River, Strait of Juan de Fuca, September 10, 2009.

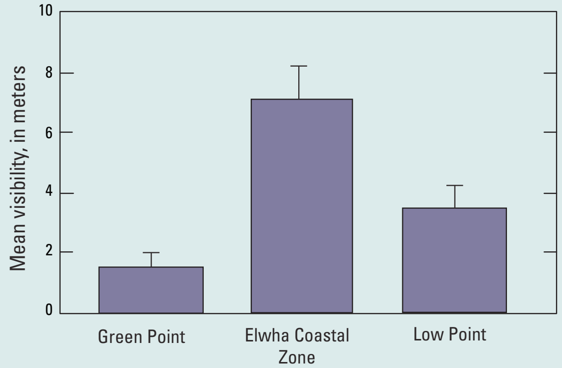


Figure S6.2. Mean visibility at shallow sites for two control areas (Green Point and Low Point) and the Elwha coastal zone. Error bars are one standard deviation.

***A.* Gray whale off of Green Point**

***B.* Edge of whale bite-mark**

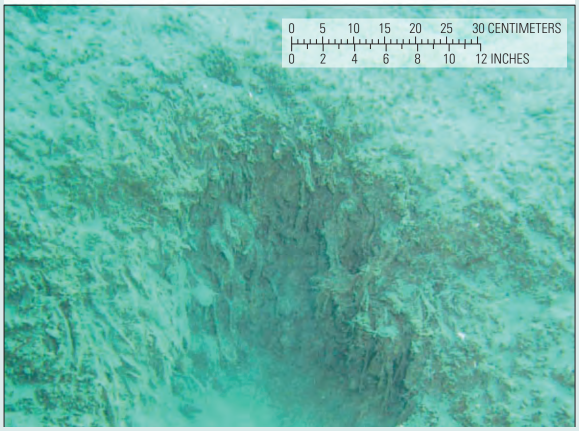


Figure S6.3. (*A*) A gray whale feeding off Green Point, September 4, 2008. Plumes of sediment trailed from this whale as it surfaced after feeding along the bottom. (*B*) An excavated edge cut into the soft substrate off shore of Green Point, likely due to gray whale feeding, September 11, 2009. This illustrates the extent to which soft sediment is used as habitat. Amphipods, annelid worms, and other small invertebrates were observed on and around the excavated bed in these pits.

(fig. S6.3*B*). Previous research in the Strait of Juan de Fuca (Oliver and Slattery, 1985) documented the effects of gray whale feeding on benthic community structure, showing the colonization of feeding excavations by motile scavengers, followed by the return of sessile tube worms in about 2 months. Amphipods also were observed in abundance at Low Point, but with a slightly different growth form, with clumps of tubes attached to boulders (fig. S6.1*C*). It is not clear if these amphipods also are a species of *Ampelisca*, although the tubes apparently were constructed of sediment and individual amphipods were observed at the opening of the tubes.

The role played by *Ampelisca* in supporting upper trophic levels in the Strait of Juan de Fuca is not clear, but this species has been a major dietary item of gray whales feeding elsewhere in the northern Pacific (Oliver and others, 1984; Oliver and Slattery 1985). Whether the removal of the dams on the Elwha River will increase fine sediments in the substrate around the Elwha delta is not known. The configuration of the Elwha delta and its exposure to waves and tidal currents might make it unsuitable for the settlement of silts, mud or clay. We hypothesize, however, that if dam removal results in a shift toward a substrate composed at least partially of fine sediments, development of habitat types and biological communities that are currently sparse may be facilitated in the Elwha coastal zone. We hypothesize that if fine sediments released in the Elwha coastal zone remain after dam removal, they may play a role in restructuring marine habitats by providing one of the essential ingredients for the success of abundant populations of *Ampelisca*.

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less dense and diverse than to the west but nonetheless abundant and distinct. Seavey and Ging (1995) thoroughly surveyed shallow depths between the river mouth and Dry Creek and reported high percentage coverage of macroalgae (74–100 percent) for most of this area. We confirmed their finding that *A. marginata* (fig. 6.7*B*) and *C. triplicata* (fig. 6.7*D*) are dominant kelp species in this area. These species are annuals whose blades grow during the spring and summer months (Mumford, 2007). Therefore, these species avoid damage or dislodgement by winter wave action and can thrive on finer substrate (gravel-cobble) than is required by perennial species like *P. californica* (fig. 6.7*G*). The invertebrate community at mixed substrate sites east of the river mouth was dominated by *Halcampa* spp. (burrowing anemones; fig. 6.16*G*), the most abundant taxon among those identified to genus or species, and soft tube worms including *Eudistylia vancouveri* (northern feather duster worm, fig. 6.6*D*), which was more abundant east than west of the river mouth. *H. Colliei* (spotted ratfish; fig. 6.10*C*) was common in mixed substrate habitat east of the river mouth. Sand habitat was also present east of the river mouth in shallow water at the base of Ediz Hook.

Implications for Effects of Dam Removal

Considerable influx of sediment to the nearshore is expected initially (for 3–5 years after start of dam removal) due to release of sediments that have accumulated in the reservoirs. Thereafter, sediment influx will likely decrease as the natural (pre-dam) rate of sediment delivery is re-established. This temporal pattern of sediment delivery raises a number of questions about biological community responses. What will happen to habitats and biota shortly after dam removal and in the long term? Will the marine system return to its natural, pre-dam condition? How long will restoration (recovery) take? What "services" (for example, kelp production) will be lost or impaired? What services will be enhanced? Our pre-removal survey establishes a baseline for evaluating short- and long-term consequences of dam removal. Follow-up surveys should help answer the questions posed above.

The spatial extent, vertical thickness, frequency, duration, timing (seasonality), and grain size of deposited sediments following dam removal will be important for determining effects on nearshore communities (Airoldi, 2003). Sediment deposition is expected to be greatest east of the Elwha River mouth due to the prevailing direction of coastal currents; however, sedimentation also may occur to the west in Freshwater Bay because currents are occasionally negligible or directed westward (Warrick and others, 2011b, chapter 5, this report). Thus, the community associated with mixed substrates lacking relief (boulders) that currently predominates east of the river mouth may experience the greatest sediment loads following dam removal, but kelp forest and relief-dependent communities to the west also may be affected. Furthermore, each of these communities may respond differently even to similar sedimentation levels. Finally, suspended sediment from riverine inputs as well as from re-suspension of deposited sediments may affect biological communities. These uncertainties highlight the opportunity to advance scientific understanding by measuring responses following dam removal.

Whatever changes come, they will likely involve tradeoffs, favoring some species and disfavoring others in the short term and over the long term as restoration occurs. Kelp may be particularly intolerant of sedimentation because their spores require sediment-free surfaces for settlement, their microscopic gametophyte and young sporophyte life stages are susceptible to burial, and they require high light levels for growth (Mumford, 2007). Even so, some kelp species can persist in areas with fine sediment (Dayton, 1985), perhaps depending on the depth or timing of coverage by fine sediment and the life history traits of the kelp (timing of reproduction; annuals compared to perennials; physiological adaptations). Sediment influx following dam removal may reduce kelp abundance overall, but may favor some species over others. Organisms that can propagate vegetatively, thereby bypassing vulnerable juvenile life stages, may be relatively resistant to sedimentation (Airoldi, 2003). Some turf-forming seaweed can trap sediments, which may discourage grazing or give them a competitive advantage over organisms less tolerant to sedimentation (Airoldi, 2003). If localized accumulations of sediment are great enough, some areas could convert from hard substrate to soft substrate habitat causing dramatic shifts in the benthic community. If dam removal results in accumulations of very fine sediments (silt, mud, or clay), which are currently rare in the Elwha coastal zone (Warrick and others, 2008), a group of small invertebrates that provide food for a number of other organisms may benefit (sidebar 6-1).

Summary

Benthic communities inhabiting shallow, subtidal depths near the mouth of the Elwha River were biologically diverse. They included 10 kelp species with differing growth forms and habitat associations, and a wide variety of invertebrate and fish species. Substrate type, seafloor relief, and water depth were important determinants of organism density and species composition. As such, the spatial distribution of these characteristics created a rich mosaic of habitats and associated communities. The removal of two dams on the Elwha River will release large quantities of sediment, some of which will be discharged into the Elwha River nearshore. The greatest influx is expected during the first 3–5 years after start of dam removal from sediment that has accumulated in the reservoirs. Thereafter, sediment input should decrease but remain higher than before dam removal owing to restored connectivity with the upper watershed. Measuring community responses to short and long term changes in deposited and suspended sediments following dam removal offers an unprecedented opportunity to gain insight relevant to managing these important marine resources.

Acknowledgments

We thank the many people that provided help or support including Mike McHenry and Doug Morrill (Lower Elwha-Klallam Tribe); Kurt Fresh (National Marine Fisheries Service); Sam Brenkman, Patrick Crain, and Steve Fradkin (National Park Service); Mike Ferguson (Porthole Dive Charters); Lieutenant Christian Polyak (U.S. Coast Guard); Guy Cochrane, Tim Elfers, Pete Dal Ferro, Guy Gelfenbaum, Jamie Grover, Mike Hayes, Marshal Hoy, Rusty Rodriguez, Andrew Stevens, Glen VanBlaricom, Jon Warrick, and Lisa Wetzel (U.S. Geological Survey); Kevin Britton-Simmons (University of Washington Friday Harbor Laboratories); Don Rothaus, Anne Shaffer, and Michael Ulrich (Washington Department of Fish and Wildlife); and Tom Mumford (Washington Department of Natural Resources).

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Chapter 7

Aquatic Ecology of the Elwha River Estuary Prior to Dam Removal

By Jeffrey J. Duda, Matthew M. Beirne, Kimberly Larsen, Dwight Barry, Karl Stenberg, and Michael L. McHenry

Abstract

The removal of two long-standing dams on the Elwha River in Washington State will initiate a suite of biological and physical changes to the estuary at the river mouth. Estuaries represent a transition between freshwater and saltwater, have unique assemblages of plants and animals, and are a critical habitat for some salmon species as they migrate to the ocean. This chapter summarizes a number of studies in the Elwha River estuary, and focuses on physical and biological aspects of the ecosystem that are expected to change following dam removal. Included are data sets that summarize (1) water chemistry samples collected over a 16 month period; (2) beach seining activities targeted toward describing the fish assemblage of the estuary and migratory patterns of juvenile salmon; (3) descriptions of the aquatic and terrestrial invertebrate communities in the estuary, which represent an important food source for juvenile fish and are important water quality indicators; and (4) the diet and growth patterns of juvenile Chinook salmon in the lower Elwha River and estuary. These data represent baseline conditions of the ecosystem after nearly a century of changes due to the dams and will be useful in monitoring the changes to the river and estuary following dam removal.

Introduction

As juvenile salmon migrate from freshwater rearing areas, they undergo the physiological, behavioral, and life history changes necessary for transition to life in the ocean. Estuaries are recognized as an important part of this migration (Simenstad and others, 1982; Bottom and others, 2005b). Some salmon species, such as Chinook (*Oncorhynchus tshawytscha*) and chum (*O. keta*) can spend extended periods (weeks to months) in the estuarine environment, whereas others such as pink salmon (*O. gorbuscha*) largely forego extended use of estuarine areas in favor of ocean rearing (Groot and Margolis, 1991). It is hypothesized that salmon use estuarine habitats because they provide a higher growth potential, facilitate the physiological transition to saltwater conditions, and reduce the risk of predation (Quinn, 2005). All of these factors may play a role in higher growth and survival during the subsequent marine life history phase (Ruggerone and others, 2009). Estuarine fish assemblages fluctuate seasonally, based on migration timing, habitat structure and connectivity, and physical factors such as salinity and temperature. Estuaries vary in size (see Duda and others, 2011, figs. 1.8–1.10, chapter 1, this report) and in the relative influence of freshwater and marine drivers important for ecological processes. This affects the amount, complexity, biological productivity, and physical conditions of estuarine habitats, which varies among river systems. Although the role of estuaries has been deemed important for some Puget Sound Chinook populations (Simenstad and others, 1982; Beamer and Larsen, 2004), the significance of the Elwha River estuary and nearshore (coastal marine water zone extending from high tide elevation to the limit of the photic zone) to Chinook salmon populations is relatively unknown. We set out to describe the existing conditions of some key ecological components of the lower Elwha River and its estuary prior to dam removal.

Research efforts focused on collecting physical (water chemistry; see also Magirl and others, 2011, chapter 3, this report, for salinity and temperature data) and biological data from throughout the estuary and adjacent riverine and nearshore areas in the Elwha River study area (fig. 7.1) prior to dam removal. Water samples were collected monthly to measure nutrient concentrations from stations dispersed in the lower river (2 sites), in the estuary (4 sites), and in the nearshore (1 site). Seasonal use of the estuary and nearshore by salmon during their migration to the marine environment also was estimated. We studied fish distribution, abundance, and migration timing for all salmonids using the Elwha River estuary and examined the diet and growth rates of juvenile Chinook salmon. The aquatic and terrestrial macroinvertebrate communities of the estuary also were surveyed. These animals play a central role as prey for salmon and are indicators of water quality that should be responsive to physical changes associated with dam removal. Other studies are being conducted on fish ecology by colleagues elsewhere in the watershed (for examples, see Burke and others, 2008; Connolly and Brenkman, 2008; McHenry and Pess, 2008; Pess and others, 2008, Brenkman and others, 2008a, b; Duda and others, in press), in the nearshore proximal to the Elwha River, and in the Strait of Juan de Fuca (for example, Shaffer and others, 2009). Juvenile Chinook salmon samples from these studies were obtained to assist in estimating several life history traits, including age and growth (otolith microstructure), feeding (stomach contents), and genetics (fin clips). This information should help provide a better understanding of Chinook salmon life histories and help inform adaptive management of the population following dam removal and recolonization of the Elwha River watershed.

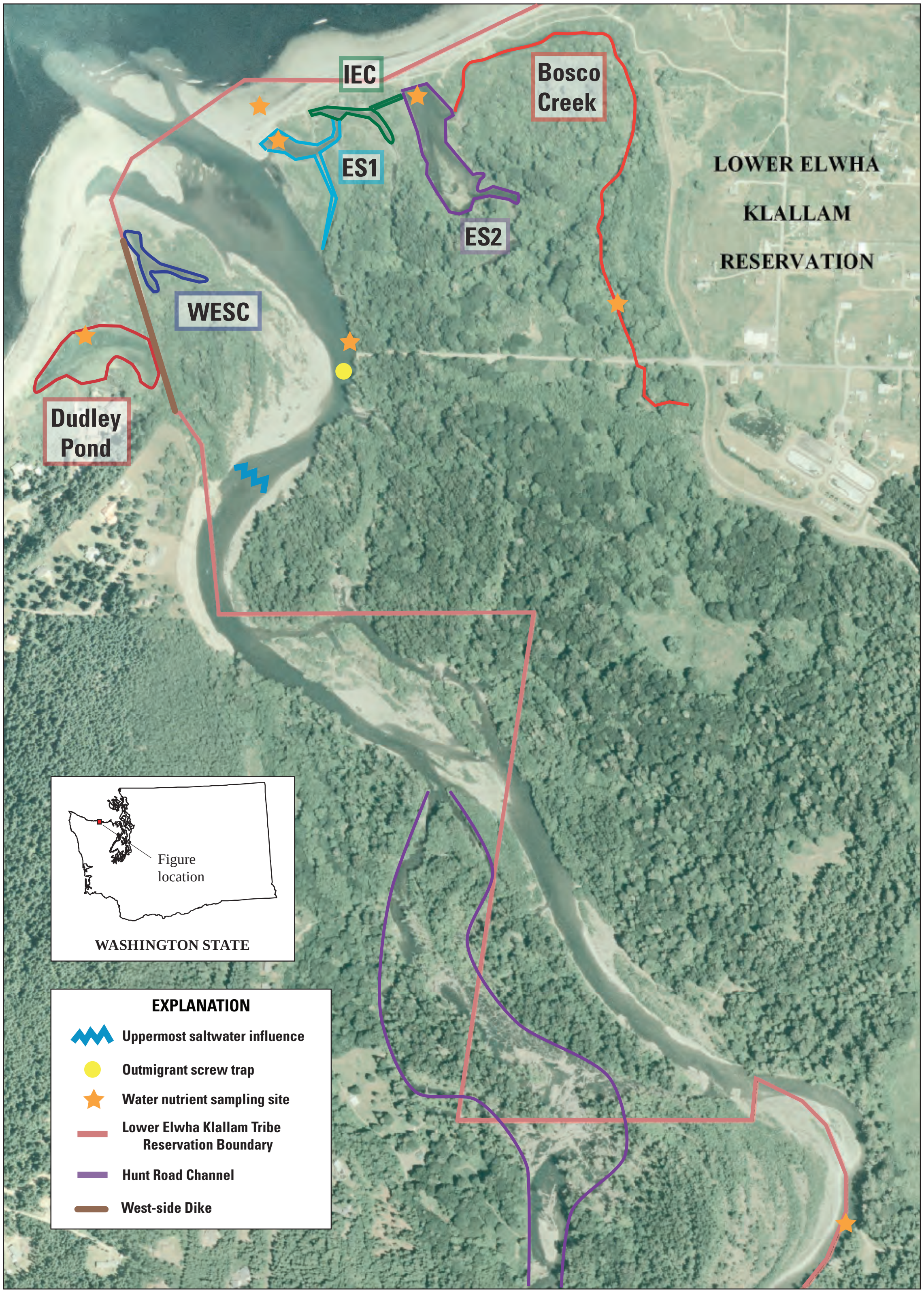


Figure 7.1. Sampling locations where water nutrients, benthic and terrestrial insects, and fish samples were collected in the Elwha River, Washington, study area.

Water Nutrients

Inorganic nutrients are important constituents for cell growth and often limit population growth and drive competitive interactions; therefore, they play a central role in determining presence and abundance patterns of biota. The availability of essential nutrients in the water column can limit the amount of biological productivity and thus the character of the entire ecosystem. Measurement of nutrients through time also can reflect patterns in seasonal biological productivity, as nutrients are diminished during periods of high growth (for example of algae and other primary producers during the summer growing season) and are higher during periods of biological dormancy. The availability of nutrients in any watershed depends on both biological and physical properties. Natural forces such as rainfall, geology, atmospheric deposition, and sedimentation drive levels of available nutrients, which are then consumed by autotrophic organisms and further modified by allochthonous energy inputs from the surrounding landscape. Other factors largely caused by human activity, such as resource extraction, fertilization, and industrial pollution also can affect the nutrient levels in a watershed.

Previous water nutrient sampling of main stem, side channel, and tributary habitats throughout the Elwha River watershed has indicated that, for its size and condition, the Elwha River is oligotrophic, or low in nutrients (Munn and others, 1999; Duda and others, in press). Although these earlier studies had broad spatial coverage of the watershed, they were temporally restricted to base (summer) flows and did not include water samples from the estuary or nearshore. This chapter provides comparable nutrient samples from the estuary and nearshore and a time series over a 16-month period. This baseline assessment of spatial and temporal trends in nutrient levels of the water column will be useful for comparisons during and following dam removal.

Water Sample Collection and Laboratory Analysis of Nutrient Concentrations

Water samples were collected monthly from seven locations in the lower Elwha River and estuary (fig. 7.1). Sites included one main stem (*Lower Elwha River – main stem*) and one side channel (*Lower Elwha River – side channel*) location in the lower Elwha River; the groundwater fed outflow of the tribal hatchery (*Bosco Creek*) that serves as a primary source of surface water to the east estuary; the tidally influenced lentic sloughs east of the river mouth in the east estuary (*ES1* and *ES2*); the disconnected lake to the west of the river mouth (*Dudley Pond*); and the surf zone (*Nearshore – Strait of Juan de Fuca*) 25 m east of the river mouth in the Strait of Juan de Fuca. Samples were collected across a range of tidal conditions at approximately 30-day intervals, with interruptions in sample collection at all sites in November 2006 (inclement weather) and July 2007 (processing error).

Water samples were collected for analysis of total nitrogen (TN), total phosphorous (TP), dissolved inorganic nitrogen (nitrate [NO_3^-], ammonia [NH_4^+]) and phosphate (PO_4^{-3}). Water collection (500 mL) and sample bottles (60 mL) were acid washed (10 percent HCl) prior to sample collection. At each site, all sampling equipment and bottles were rinsed three times with ambient water. Samples were collected at the same depth for each site, generally 60 percent of the water column depth at least 1 m from shore. An inverted water collection bottle was submerged, filled, and capped at depth, to ensure that no water was collected from the surface. Samples for dissolved nutrients were filtered in the field using a 0.45-µm pore size syringe filter. Once collected, samples were kept on ice in the field and frozen in the laboratory prior to analysis by the University of Washington Marine Chemistry Laboratory, Seattle, Wash. This procedure and laboratory were the same that were used by Morley and others (2008) and Duda and others (in press) for samples collected throughout the Elwha River watershed.

Total nitrogen and total phosphorous were analyzed using the persulfate digestion method of Valderrama (1981). The procedure of Armstrong and others (1967) was modified for the analysis of NO_3 and NO_2. Water samples were passed through a cadmium (Cd) column where the nitrate (NO_3) was reduced to nitrite (NO_2). This NO_2 was then diazotized with sulfanilamide and coupled with N-(1-naphthyl)-ethylenediamine to form an azo dye. The sample was then passed through a 15 mm flowcell and absorbance measured at 540 nm. The procedure is the same for the NO_2 analysis less the Cd column. Nitrate concentration equals the (NO_3 + NO_2) concentration minus the NO_2 concentration. A modification of the Slawyk and MacIsaac (1972) procedure was used for the analysis of ammonium (NH_4). Water samples were treated with phenol and alkaline hypochlorite in the presence of ammonia (NH_3) to form idophenol blue (Berthelot reaction). Sodium nitroferricyanide was used as a catalyst in the reaction. Precipitation of calcium (Ca) and magnesium (Mg) hydroxides was eliminated by the addition of sodium citrate complexing reagent. The sample stream then was passed through a 55 °C heating bath, then through a 50 mm flowcell and absorbance was measured at 640 nm. Phosphate was analyzed using a modification of the Bernhardt and Wilhelms (1967) method. Ammonium molybdate was added to water samples to produce phosphomolybdic acid, which then was reduced to phosphomolybdous acid following the addition of dihydrazine (or hydrazine) sulfate. The sample was passed through a 50 mm flowcell and absorbance was measured at 820 nm.

Spatial Patterns of Water Nutrients

Fourteen water samples were collected from June 2006 until September 2007 at most locations (table 7.1). The concentration of each nutrient by month was graphed and smoothed curves were created using a cubic spline interpolation in SPSS Inc. (2010). The data are presented in the form of sparklines, a method of data visualization that uses small, high-resolution graphics coupled with words and numbers (Tufte, 2006). By creating small, time series graphics for each location and water chemistry constituent (with maximum, average, and minimum values provided), spatial and temporal trends in water chemistry can be compared in a single figure.

The time series showed seasonal variations in nutrient levels and differences among sites (figs. 7.2*A* and 7.2*B*). Concentrations of nutrients (TN, TP, PO_4 and NO_3), at the nearshore zone of the Strait of Juan de Fuca generally were higher than sites in the Elwha River and its estuary. An exception was Bosco Creek with the highest NH_4 concentration, possibly due to the hatchery operations at the source of the surface water supply, which concentrates the animal waste byproducts associated with large-scale fish production. Higher concentrations of TP, TN, NO_3, and PO_4 were measured in Bosco Creek than in the other five sites of the lower Elwha River and estuary. Following Bosco Creek and the nearshore, the two locations in the east estuary (ES1 and ES2) had the next highest levels of nutrient concentrations. Samples from the Elwha River (main stem and side channel) were consistently the lowest of all sites for most nutrients. This result is consistent with previous studies documenting the oligotrophic nutrient status of the river (Munn and others, 1999; Duda and others, in press). Levels of nitrogen and phosphorous generally increased along the gradient from freshwater–estuarine–marine waters, which is a result consistent with expectations.

Temporal Patterns of Water Nutrients

The highest temporal variability in water nutrient patterns was in the Strait of Juan de Fuca and Bosco Creek, which was indicated by the widest confidence intervals around their mean values across all samples (table 7.1). Oscillations at both these sites were during months that did not follow a consistent seasonal pattern (figs. 7.2*A* and 7.2*B*). The estuarine and lower Elwha River samples, however, displayed a more typical seasonal pattern. During spring and summer, when primary production in the river and estuary increases with increasing day length, levels of most nutrients were lower than levels during winter, when primary productivity wanes. Interestingly, Dudley Pond in the western part of the estuary, which is disconnected from the Elwha River by a dike, was out of phase with the east estuary during the autumn and winter periods, especially for TP, PO_4, and NH_4.

Table 7.1. Average concentration of total and dissolved nutrients at seven sites in the lower Elwha River, its estuary, and the nearshore surf zone of the Strait of Juan de Fuca, Washington, 2006 and 2007.

[All values are in micrograms per liter (standard deviation). Locations are shown in figure 7.1. Samples were collected at approximately 30 day intervals from June 2006 to September 2007, with interruptions in sample collection at all sites in November 2006 and July 2007. **Abbreviations:** MS, main stem; SC, side channel; ES1, east estuary 1; ES2, east estuary 2]

Location	Total phosphorus	Total nitrogen	Dissolved phosphate	Dissolved nitrate	Dissolved nitrite	Dissolved ammonia
Lower Elwha – MS	13 (4)	83 (48)	2 (2)	16 (17)	0.2 (0.1)	2 (3)
Lower Elwha – SC	12 (4)	105 (60)	2 (2)	24 (17)	0.2 (0.2)	4 (5)
Bosco Creek	32 (12)	275 (81)	14 (8)	76 (54)	3 (2)	79 (51)
ES1	16 (6)	139 (73)	5 (5)	28 (35)	0.2 (0.2)	13 (12)
ES2	18 (3)	178 (87)	6 (3)	43 (55)	0.6 (0.7)	18 (18)
Dudley Pond	16 (4)	163 (91)	3 (2)	7 (7)	0.3 (0.2)	10 (11)
Nearshore	49 (18)	284 (123)	30 (16)	132 (82)	2 (2)	12 (12)

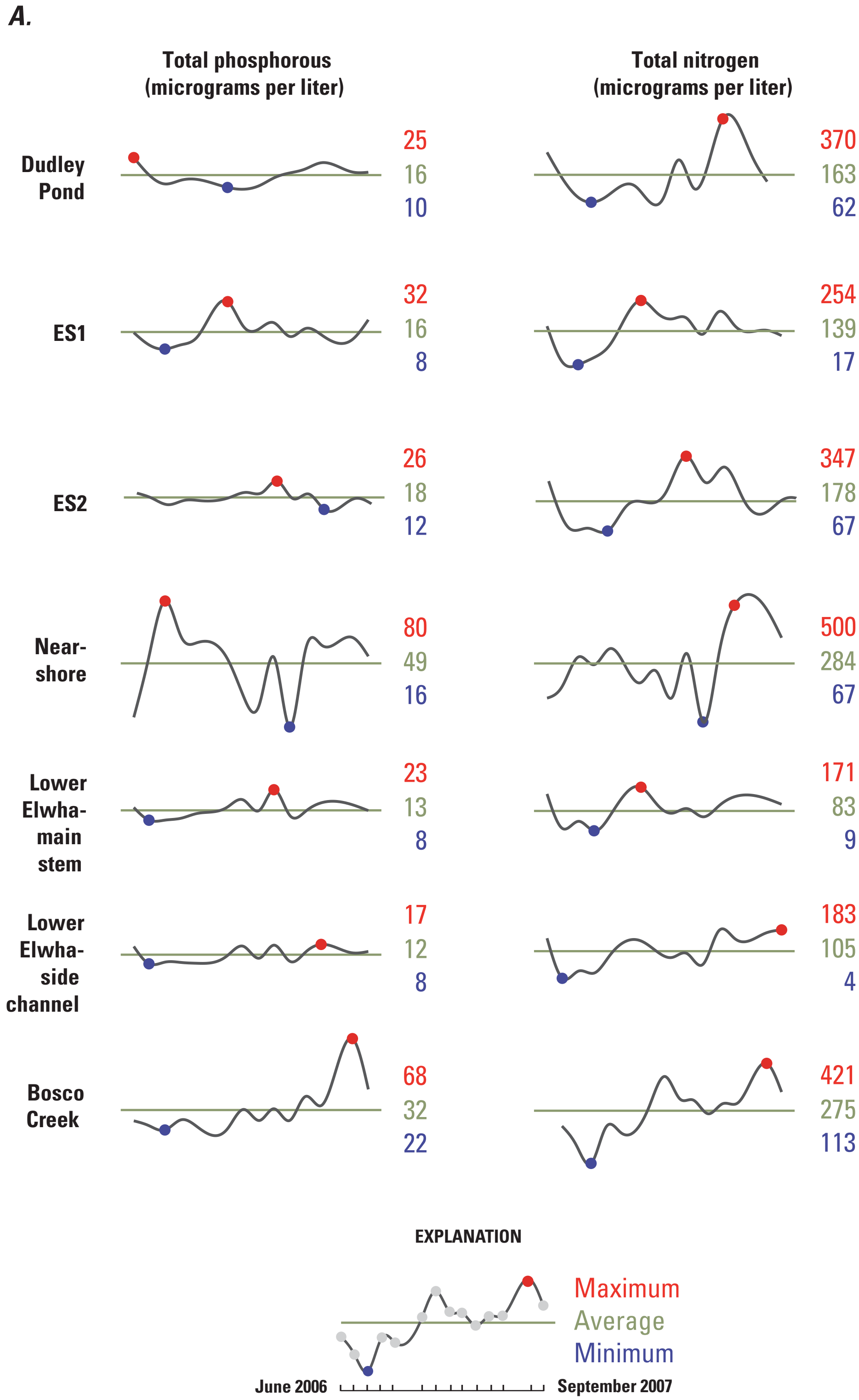


Figure 7.2. Sparkline graphs showing time series of water chemistry species (*A*) total phosphorous and total nitrogen and (*B*) phosphate, nitrate, and ammonia, from the lower Elwha River, its estuary complex, a spring fed tributary draining into the east estuary, and the nearshore surf zone of the Strait of Juan de Fuca near the mouth of the Elwha River, Washington, 2006–07.

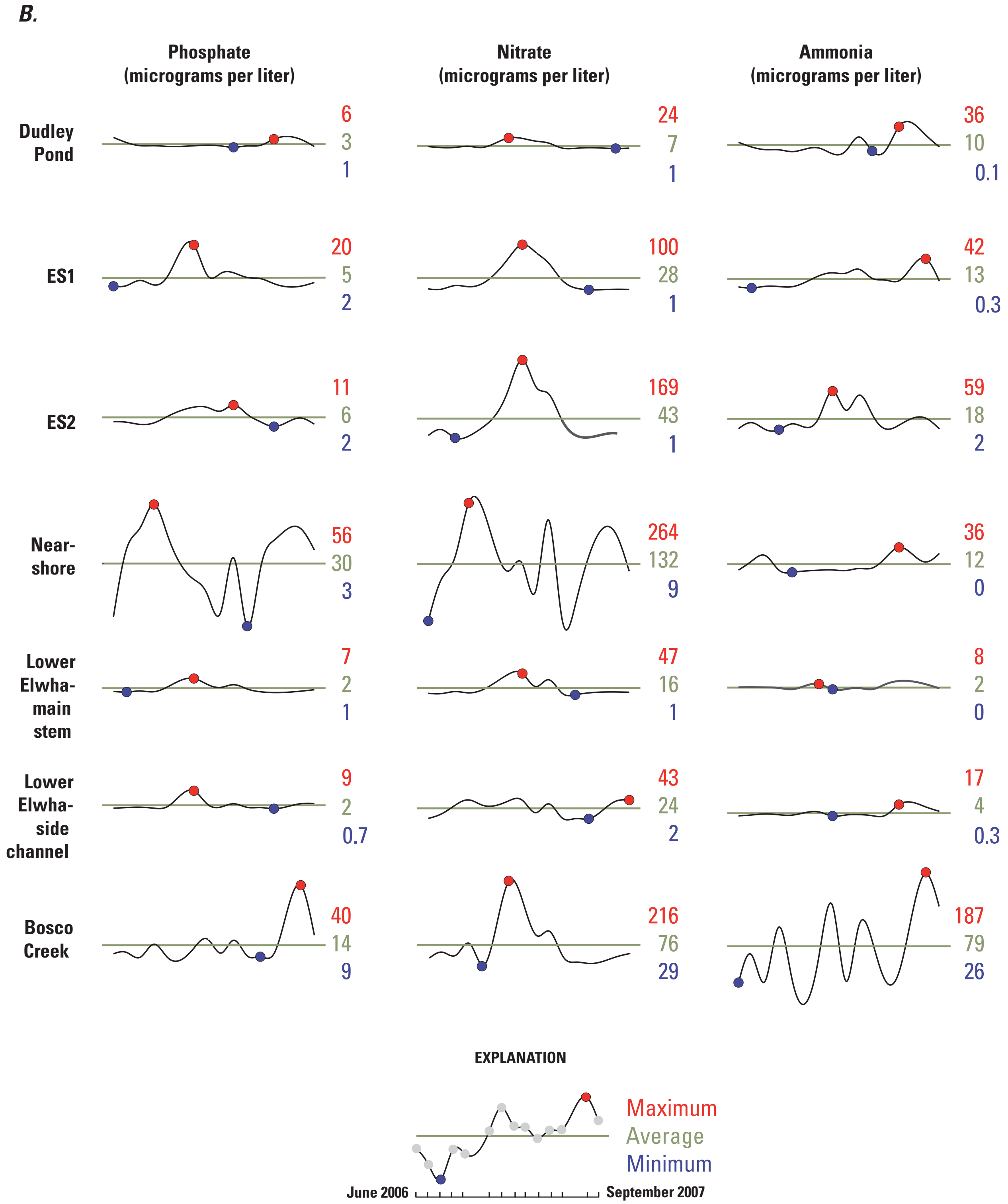


Figure 7.2—Continued

Nutrients and Dam Removal

The Elwha river is oligotrophic (Munn and others, 1999; Duda and others, in press), as measured values of nitrogen and phosphorous throughout the watershed are at low levels that are likely contributing to relatively low levels of primary productivity (Morley and others, 2008). This assessment, based on synoptic measurements taken during summer low flows, probably is affecting the nutrient status of the estuary. Results from the 2 freshwater Elwha River sites were similar to those reported by Duda and others (in press) for 12 other lower Elwha River sites during summer low flows (from 2005 and 2006). In that study, sites in the regulated sections of the Elwha River (downstream of the Glines Canyon and Elwha Dams) were significantly different from the upper unregulated section for phosphate, but not dissolved inorganic nitrogen. Duda and others (in press, fig. 5) reported that phosphate and dissolved inorganic nitrogen values in regulated sections of the Elwha River were significantly lower than in other unregulated rivers of the Pacific Northwest with salmon runs. However, they could not explain the significant differences in phosphate between the regulated and unregulated sections of the Elwha River.

How the levels of these nutrients in the lower Elwha River and its estuary will respond after dam removal is not known. Significant changes are expected in sediment supply, bed particle size, and nutrients supplied by salmon, all of which could affect the nutrient status of the sites. The supply of sediment is expected to increase dramatically in some places during and following dam removal (Czuba and others, 2011, chapter 2, this report; Konrad, 2009). Immediately following dam removal, fine-grained sediments stored in the reservoir bottoms and the deltas will be released and transported downstream.

Previous research has indicated that sediments released from reservoirs can change the nutrient status downstream (Stanley and Doyle, 2002). For example, phosphorous often is retained in reservoir sediments and the release of these sediments during dam removal could increase nutrient levels downstream during the short term. In the longer term, the reservoirs will not trap phosphorous, which will be transported to downstream reaches. The net effect of this material on nutrient levels after dam removal when sediment is transported to the lower Elwha River, estuary, and nearshore, is not clear. Based on an analysis of Lake Mills reservoir and delta sediments, Cavaliere (2010, p. 54) reported that, "The Elwha River sediments from Lake Mills have limited P (phosphorous) available for algal and plant growth. The water quality after the dams are removed is unlikely to be harmed by the release of excess P (phosphorous)…" from reservoir sediments. The low levels of nutrients held in reservoir sediments coupled with high rates of suspended sediment transport could limit any release of stored nutrients to the lower Elwha River and estuary.

If following dam removal and recolonization of upper parts of the watershed causes salmon to return to the river in larger numbers than the current populations, as projections suggest, then at least during some times of the year the nutrient status of the river may change. Based on estimates of future salmon returns following full recovery, 1,275–10,900 kg of nitrogen and 210–1,350 kg of phosphorous derived from salmon could be input annually into the Elwha River (estimates based on assumptions of Munn and others [1999] and projected spawners presented by Ward and others [2008]). If phosphorous and nitrogen are limiting primary productivity in the Elwha River, then increases in these nutrients may have important implications for the Elwha River ecosystem. It is not clear that an increase in salmon numbers will result in clear increases in water column nutrients, but salmon-derived nutrients may provide significant temporal increases in the biomass and growth rates of resident biota (for example, Bilby and others, 1996; Schuldt and Hershy, 1995; Wipfli and others, 1998; Chaloner and Wipfli, 2002; Duda and others, in press).

Aquatic and Terrestrial Macroinvertebrate Assemblages of the Elwha River Estuary

The current configuration of the Elwha River estuary has been relatively static for much of the period since dam construction, especially in recent decades (see Warrick and others, 2011, chapter 3, this report). The sediment deficit and related effects of the dams that led to the simplified, incised channel of the lower river (Pohl, 2004; Draut and others, 2008, 2011; Kloehn and others, 2008) also has contributed to the stability of the estuarine complex. This is most notable in the age and size classes of woody vegetation on the east estuary, particularly red alder and red elderberry (see Shafroth and others, 2011, chapter 8, this report). Effects to the estuary following dam removal will include sediment deposition, a likely associated increase in channel formation, and perhaps over the long term the development of multiple river mouths. This could result in dramatic changes to plant community associations and age class distributions, which can affect macroinvertebrate community assemblages.

The estuary is an important habitat for some juvenile salmon during their migration to the ocean in large part because of the food supply available, especially aquatic and terrestrial invertebrates (Simenstad and others, 1982). The Lower Elwha Klallam Tribe initiated a baseline assessment of macroinvertebrate assemblages in the estuary, including benthic macroinvertebrates and terrestrial insects in the emergent and shrub transition-habitat zones. Describing the current conditions of the macroinvertebrate communities is an important monitoring goal, for their sensitivity to environmental changes and importance as a prey source for juvenile salmon.

Benthic Macroinvertebrates of the Elwha River Estuary

Sampling and Analysis Methods for Benthic Macroinvertebrates

Samples of benthic macroinvertebrates residing in the sediments throughout the Elwha River estuary were collected to establish baseline data on species diversity, relative abundance, and availability to juvenile salmonids throughout their migration season (spring and summer). Sampling occurred on three occasions (May, July, and September) in 2007. During each occasion, 22 samples were collected from sites in the east and west estuary. This sampling was near our juvenile salmon beach seining sites in the estuary. The 22 samples were distributed among the east estuary (ES1 [3 locations], ES2 [8 locations], the channel connecting ES1 and ES2 [2 locations; hereafter referred to as the intraestuarine channel [IEC]), and the west estuary (Dudley Pond [4 locations] and WESC [5 locations]) (fig. 7.1).

A spring-loaded 6-in petite Ponar grab sampler was attached to a nylon rope and deployed while wading (where depth allowed) or from a boat (fig. 7.3).

Figure 7.3. Lower Elwha Klallam Tribe fisheries technicians sampling sediments in the Elwha River, Washington, estuary using a petite Ponar grab sampler. (Photograph taken by Matt Beirne, Lower Elwha Klallam Tribe, April 17, 2006.)

The grab sampler surface area and volume were 0.023 m^2 and 2.4 L, respectively. Where the substrates were not conducive to acquiring an adequate sample volume, the grab sampler had to be manually forced into the substrate. Sediment grabs were weighed, photographed, and characterized by color and texture.

Sediment grab samples were sorted using a series of four metal sieves with successive openings of 3.55, 2.0, 1, and 0.5 mm (500 µm). A garden hose with an adjustable nozzle was used to separate the inorganic sediment fraction gently from the organic material, detritus, and macroinvertebrates. The material remaining in the bucket was then sieved through a custom 0.5 mm stainless steel sieve fashioned from a plastic tray with the bottom replaced by a welded sheet of 500 µm mesh. The primarily organic material that was retained in the sieves was gently separated using water sprayed into clear plastic tubs 32.5 cm × 75 cm for subsequent sorting.

Immediately after sieving, the remaining material was sorted using forceps, typically within 24 hours of sample collection. A squirt bottle with 70 percent ethanol was used to agitate any remaining organisms from the organic material after all macroinvertebrates were removed. Sample processing time typically averaged 2 hours per sample, including sieving and sorting. Sorted samples were preserved in 10 percent formalin, which served as a fixative for soft-bodied organisms. After 1 week, specimens were filtered, rinsed, and transferred to 70 percent ethanol. Most insects from the orders Ephemeroptera, Plecoptera, and Trichoptera were identified to genus. All other taxa were identified to family level where possible, or the next identifiable taxonomic level (for example, class, order). Arthur Frost, a local expert in macroinvertebrate identification, completed all taxonomic identification.

Sediment grab samples also were used to provide general characterizations of epibenthic/ benthic habitat types in the estuary. Eight classes of sediment descriptions were used to characterize sediment (table 7.2). The greatest homogeneity was in WESC and was exclusively comprised of silty material typically overlayed with a detritus layer of leaf litter and some marine derived algae. The isolated estuary remnant of the west estuary (Dudley Pond) showed the greatest variability ranging from fine silt to gravel and was typically associated with a green, gelatinous organic fraction, likely derived from the decomposition of the heavy green algal mats that form on the surface of this water body (Duda and others, 2011, chapter 1, fig. 1.12, this report). The east estuary tended to lack the detrital/algal epibenthic component detected in the west estuary and was variable in texture from fine silt to sandy gravel. It was predominantly (66 percent) comprised of silt, with the remainder ranging from silty sand to silty-sandy gravel. The west estuary, which is surrounded by dense, early successional willow/alder vegetation, seems to receive considerably greater detrital inputs and may not receive the same intensity or frequency of tidal flushing as in the east estuary habitats. The east estuary is significantly larger than WESC and appears to have a broader zone of emergent vegetation and shrub transitional habitat, coupled with greater tidal surges from the west and flushing from the east (Bosco Creek). This likely contributes to the small amount of detritus in the epibenthic environment.

Table 7.2. Classification of sediment type from benthic macroinvertebrate samples collected in the Elwha River estuary, Washington, May–September 2007.

[All values are number of grab samples. Abbreviation: WESC, west estuary channel]

Sediment category	East estuary	WESC	Dudley Pond
Silt	26	12	1
Silty-sand	2	0	0
Sandy-silt	6	0	1
Sandy	2	0	0
Silty sand-gravel	1	0	0
Sandy gravel	1	0	3
Silty organic	0	3	5
Organic gelatinous	0	0	2

Benthic Macroinvertebrate Assemblages of the Elwha River Estuary

During all months, we collected 40 macroinvertebrate taxa (5,943 individuals) in the east estuary, 29 taxa (3,343 individuals) in WESC and 25 taxa (1,840 individuals) in Dudley Pond in the benthic grab samples collected in 2007. These taxa belonged to 29 major taxonomic groups (Order or greater) in three phyla (Arthropoda, Nematoda, Mollusca). Of the 53 unique taxa (including a single "unknown"), 16 were detected in all three locations, 16 were detected in only the east estuary, 6 were only in Dudley pond, and 6 were only in WESC. Most taxa were identified to family (22) or genus (17), with the remainder identified to order or suborder (6), class (6), or phyla (1; nematodes). A rarefaction species accumulation curve for all sites, using the Chao1 estimator (Chao, 1984) in the program Primer (version 6; Clarke and Gorley, 2006), indicated that we were still detecting new taxa as we added more samples, a result typical of abundant and diverse groups of organisms (fig. 7.4). However, the species accumulations were different among the three locations, as the curves appeared to level off in the east estuary, but were still climbing in Dudley Pond and WESC. The taxa diversity

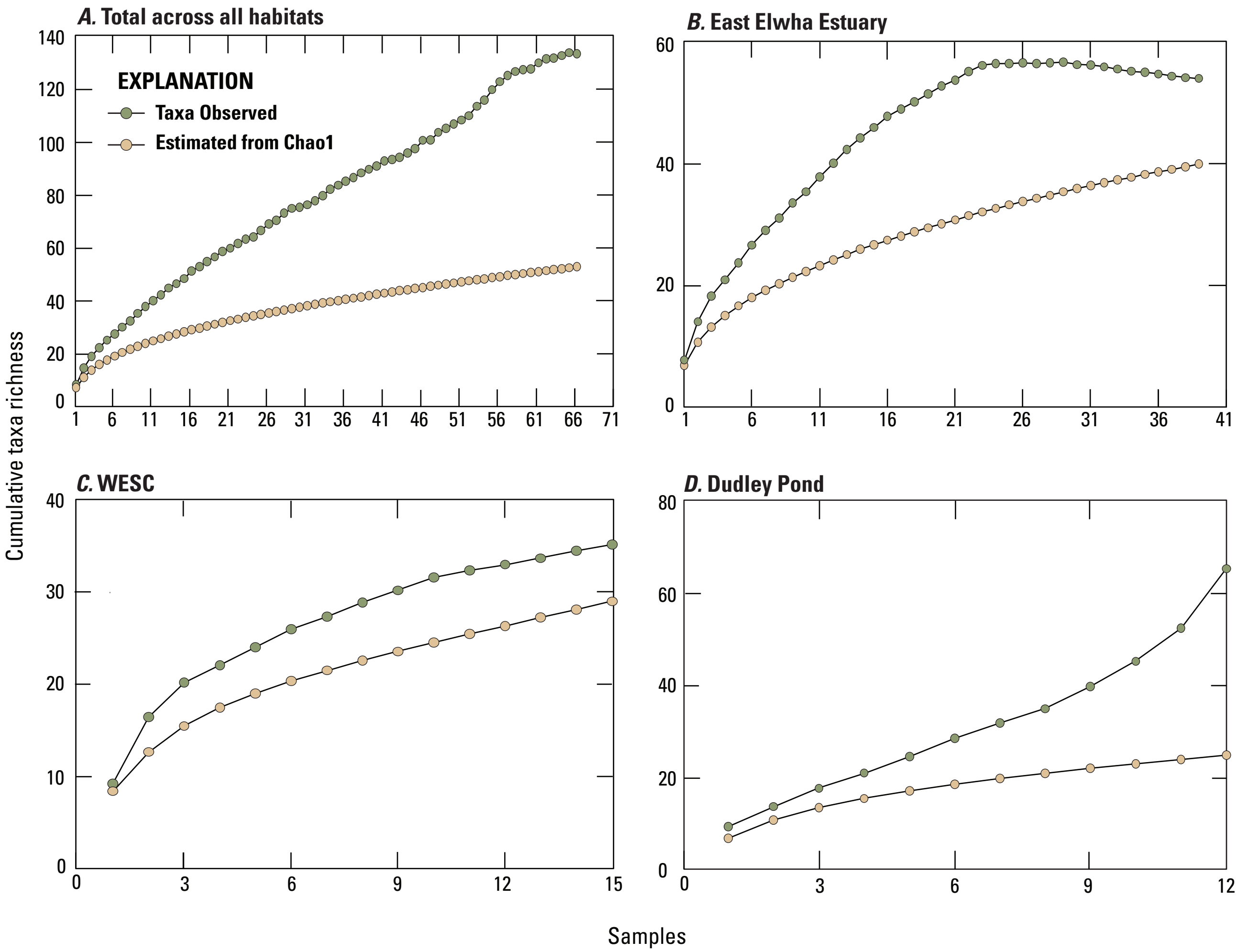


Figure 7.4. Species accumulation curves for benthic invertebrate samples , showing the observed and estimated taxa richness in the Elwha River estuary, Washington in 2007. (Estimated values from rarefaction using the Chao1 estimator.)

that was estimated with the Shannon diversity index (H′) indicated that diversity was low and did not vary among sites or months (fig. 7.5). When averaging across sampling sites and locations, amphipods and dipterans were the dominant taxa representing 50 and 21 percent of identified specimens, respectively (table 7.3). The proportion of dipterans and amphipods was similar in May and September; however, in July the proportion of dipterans in samples increased, whereas amphipods decreased. Spatially, samples collected from WESC seemed depauperate in amphipods when compared to Dudley Pond and the east estuary (table 7.3). This may be due to the paucity of sandy sediments in WESC relative to Dudley Pond and the east estuary. Grab samples indicated that amphipods and isopods seemed to prefer sandy substrates to silty or muddy substrates. In September, the total abundance of macroinvertebrates of WESC was only 8.1 percent and 12.9 percent of that detected in Dudley Pond and the east estuary sites, respectively. One possible explanation for this was the hydrologic isolation of WESC and lower flows of the Elwha River may have resulted in higher water temperatures and lower dissolved oxygen levels than conditions at other sites. Water quality data (from a Conductivity, Temperature, Depth (CTD) logger; details in Magirl and others, 2011, chapter 4, this report) collected at the time of sampling, seem to corroborate this, although some differences in sampling dates in September limits our ability to infer this from the data.

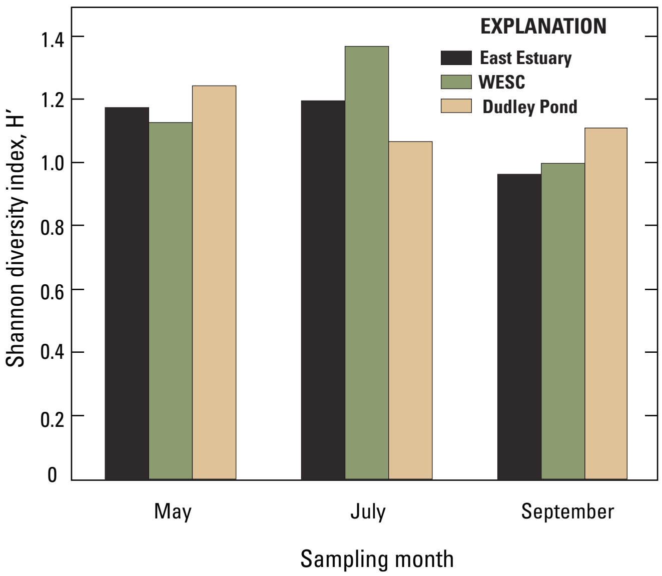


Figure 7.5. Shannon diversity index scores of the macroinvertebrate community from sediments in the East Estuary, WESC, and Dudley Pond sampling locations of the Elwha River estuary, Washington, in May, July, and September 2007.

Table 7.3. Average number per sample (standard deviation) of amphipods and dipterans, two major components of the macroinvertebrate community inhabiting sediments in the Elwha River estuary, Washington, May, July, and September 2007.

[Values in parenthesis are standard deviation. **Abbreviations:** n, number; WESC, west estuary channel]

Location	Taxon	May	July	September
East estuary (n = 13)	Amphipods	45 (72)	2 (5)	220 (440)
	Dipterans	25 (22)	52 (26)	34 (38)
WESC (n = 5)	Amphipods	0	2 (4)	3 (4)
	Dipterans	89 (65)	107 (69)	28 (27)
Dudley Pond (n = 4)	Amphipods	14 (14)	187 (317)	180 (135)
	Dipterans	20 (11)	45 (18)	11 (8)
Average percentage of total assemblage				
All sites combined	Amphipods	26 (31)	11 (24)	28 (37)
	Dipterans	46 (32)	64 (31)	41 (35)

The density of dipterans was greater in WESC than other areas of the estuary in May and July, but was not different from the east estuary in September (table 7.4). The greater densities of dipterans in WESC may be attributed partly to the significant epibenthic detrital layer in contrast to Dudley Pond and east estuary, which may provide greater cover or habitat heterogeneity for macroinvertebrate fauna. More amphipods were detected in Dudley Pond than at the other sites and Dudley Pond was the only site with significant densities of isopods.

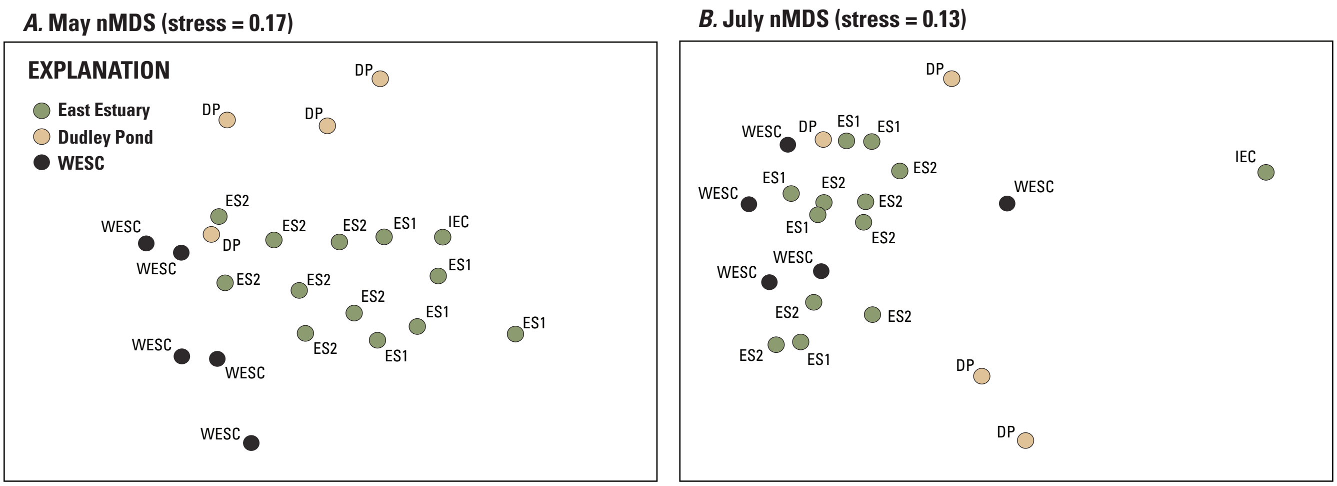
A suite of complementary nonparametric multivariate statistics was used to analyze the assemblage structure of benthic macroinvertebrates across months and sites using Primer software. The data were placed into a rectangular data matrix of taxa (rows) by samples (columns). These data were square root transformed to reduce the effects of numerically dominant taxa and matrix wide non-detections (Clarke and Warrick, 2001; McCune and Grace, 2002). Treating each sampling month separately for sediment samples, a triangular resemblance matrix was created, with each cell of the matrix giving the pair-wise similarity between sites based upon the Bray-Curtis distance. Next, non-metric multidimensional scaling (nMDS) was used to graphically analyze whether the assemblage structure of individual sites were grouped by east estuary, west estuary, and Dudley Pond locations. Pairs of sites with similar macroinvertebrate assemblages (in terms of taxa present and their abundances, exclusive of joint-absences) occur closer together in nMDS plots than dissimilar pairs of sites, which occur farther apart. These analyses were done using the full data set (a mixture of taxonomic classifications) and taxa aggregated by orders. The results were similar between these two data sets and results are shown for the full data set unless otherwise indicated.

Based on the graphical analysis of nMDS plots, benthic invertebrates in estuarine sediments differed among the three estuary locations in May and July, but not in September (fig. 7.6). We followed up the graphical analysis with an Analysis of Similarities (ANOSIM), a non-parametric analog to Analysis of Variance (ANVOA), which calculates a ratio of rank similarities (R) that takes a value between 1 and 0. At R = 1, within-group sites are more similar to each other than any sites from other groups, whereas when R approaches 0 similarities among sites do not differ among groups. We used a permutation test with 999 iterations to develop a null distribution to test whether the observed R value was statistically significant.

Table 7.4. Average density of common macroinvertebrate taxa from benthic samples collected in the Elwha River estuary, Washington, May, July, and September 2007.

[Average density is number per square meter. Values in parenthesis are standard deviation. Samples were collected using a petite Ponar grab sampler. **Abbreviations:** WESC, west estuary channel; n, number]

Taxon	WESC (n = 5 sites)			Dudley Pond (n = 4 sites)			East estuary (n = 13 sites)		
	May	July	September	May	July	September	May	July	September
Acarina	26 (58)	43 (75)	9 (19)	43 (61)	11 (22)	11 (21)	20 (42)	76 (102)	80 (116)
Amphipoda	0	104 (161)	147 (164)	606 (588)	8,083 (13,711)	7,802 (5,840)	1,951 (3,106)	93 (215)	9,506 (19,040)
Diptera	3,860 (2,816)	4,622 (2,970)	1,229 (1,170)	887 (488)	1,948 (777)	487 (368)	1,102 (948)	2,261 (1,117)	1,495 (1,665)
Isopoda	9 (19)	9 (19)	0	1,840 (2,141)	1,309 (1,491)	5,702 (7,298)	0	0	3 (12)
Odonata	86 (118)	43 (97)	95 (166)	0	0	32 (65)	3 (12)	3 (12)	23 (52)
Oligochaeta	1,930 (3,084)	1,350 (2,082)	95 (213)	357 (450)	86 (146)	2,943 (5,316)	113 (99)	1,129 (1,900)	632 (1,154)
Ostracoda	9 (19)	182 (224)	35 (56)	1,742 (2,393)	1,407 (2,813)	3,105 (5,923)	173 (395)	113 (197)	782 (1,887)
Trichoptera	17 (39)	9 (19)	0	11 (22)	0	0	13 (27)	86 (126)	0



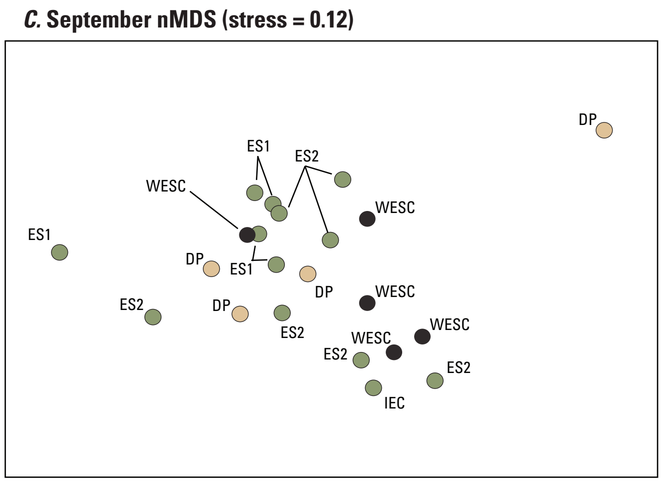


Figure 7.6. Nonmetric multidimensional scaling plots (nMDS) showing assemblage structure of sediment dwelling macroinvertebrates of the Elwha River estuary, Washington, during (*A*) May, (*B*) July, and (*C*) September 2007. (Unitless nMDS plots based on Bray-Curtis similarity of square root transformed data.)

The ANOSIM analysis confirmed the patterns from the nMDS plots. Differences among estuary locations were significantly different in May (R = 0.50, P = 0.001) and July (R = 0.27, P = 0.03) but were not significantly different in September (R = 0.07, P = 0.26). Although statistically significant, the moderate (May) to low (July) R values indicate that the assemblages in the different locations of the estuary showed a fair amount of overlap.

To determine which specific taxa were driving similarities within and differences among estuary locations, we used the SIMPER analysis in Primer. This test decomposes the Bray-Curtis dissimilarity between each pair of sites in the triangular resemblance matrix by estimating the per-taxon contribution to the dissimilarity. The average contribution then is estimated across all comparisons within and among groups; taxa with a large average contribution and a small average variance generally are good discriminating taxa driving differences in assemblage structure between pairs of locations.

In May, the differences in assemblage structure between the east estuary and WESC sites were caused by differences in abundance patterns of Dipterans (27.2 percent of the average dissimilarity), Amphipoda (25.6 percent), and Oligochaeta (23.2 percent). There were more *Corophium* sp. amphipods in the east estuary and high abundances of Oligochaetes, Tanypodinae, Tanytarsini, and Chironomini diperans in WESC. Differences in assemblage structure between the east estuary and Dudley Pond in May were caused by Ostracoda (26.6 percent of the average dissimilarity), Isopoda (23.8 percent), and Amphipoda (20.0 percent), with higher abundances of Ostracoda and Sphaeromatidae (Isopoda) in Dudley Pond and higher *Corophium* sp. abundances in the east estuary. Finally, abundances of Ostracoda (20.1 percent of the average dissimilarity), Diptera (18.4 percent), Isopoda (17.2 percent), and Oligochaeta (16.8 percent) were most responsible for the dissimilarity between WESC and Dudley pond. Again, the higher abundances of Ostracoda and Sphaeromatidae in Dudley Pond coupled with the higher abundances of Oligochaeta, Tanytarsini, Chironomini, and Tanypodinae in WESC were responsible for the differences between the locations.

In July, differences between the east estuary and WESC were again caused by different abundance patterns of Diptera (27.8 percent of the average dissimilarity), Oligochaeta (25.6 percent), and Amphipoda (8.6 percent) and Ostracoda (9.4 percent). Top taxa contributing from these groups were Tanytarsini, Tanypodinae, Chironomini, *Corophium* sp., and Chironomidae pupa.

Terrestrial Macroinvertebrates of the Elwha River Estuary

Sampling and Analysis Methods for Terrestrial Macroinvertebrates

Flight-intercept traps (hereafter fallout traps) were used to survey terrestrial insects, in the east estuary. Terrestrial insect sampling was not conducted in WESC or Dudley Pond. Traps were set during May, June, July, and September 2007. These periods generally span the season of highest invertebrate density and the time of maximum juvenile salmon use of the estuary. Five traps were placed in each of three main habitat types: littoral, shrub, and forest canopy (fig. 7.1). Littoral habitats occurred in shoreline areas with an open canopy and emergent marsh vegetation (for example, *Juncus* sp.; *Carex* sp.). Shrub habitats were areas dominated by woody shrubs such as black hawthorn (*Crataegus douglasii*), Nootka rose (*Rosa nutkana*), and twinberry (*Lonicera involucrata*). Forest habitats were dominated by red alder (*Alnus rubra*), big leaf maple (*Acer macrophyllum*), salmonberry (*Rubus spectabilis*), and red elderberry (*Sambucus racemosa*). Traps were set, on average, within 1, 6, and 50 m of the estuarine shoreline for the littoral, shrub, and forest canopy zones, respectively. Because we were interested in associating terrestrial macroinvertebrate diversity and abundance patterns with food availability for juvenile salmonids, our analysis was limited to samples associated with the littoral and shrub zones.

Each trap was constructed from a shallow 32 × 75 cm plastic tub filled with about 5 cm of water and several drops of dish soap to break up the surface tension and facilitate retention of fallen insects. Because the traps were placed on the ground, we attempted to minimize the potential for inundation from tidal fluctuations raising the water level above the lip of the tub, topping the traps, and swamping any captured insects. Around the perimeter of the trap, PVC pipes were driven into the ground; PVC eyelets were attached to the tub and secured through the PVC pipes. This helped allow the tubs to rise and fall with the tide, while they remained in the sampling location (fig. 7.7). After 3 days, the contents of each trap were passed through a 500 micron sieve and preserved in 70 percent ethanol.

Figure 7.7. Two flight-intercept (fallout) traps placed in littoral habitat of the Elwha River estuary, Washington. Each trap is attached with rings to PVC stakes, allowing the trap to rise and fall during tidal fluctuations. (Photograph taken by Matthew M. Beirne, Lower Elwha Klallam Tribe, April 26, 2006.)

Terrestrial Macroinvertebrate Assemblages of the Elwha River Estuary

Across all months, 106 (17 orders) and 73 (14 orders) taxa in the shrub (n = 2,960 individuals) and littoral (2,333) fallout traps, respectively, were collected. Forty-eight taxa were detected in both habitats, 25 taxa were detected only in littoral habitat, and 58 taxa were detected only in shrub habitats. Most taxa were identified to family (88) and genus (25), and the rest were identified to Order (13), superfamily (2), class (2), or phyla (1; nematodes). As with the sampling of benthic invertebrates in the sediment, the rarefaction species accumulation curve lacked a clear asymptote, indicating that this sampling effort probably underrepresented the diversity of the insect communities in these habitats (fig. 7.8). The rarefaction curves were of a similar shape between littoral and shrub habitats. Shannon diversity was higher in the shrub samples than in the littoral samples across all months (fig. 7.9). A large amount of literature exists on patterns of invertebrate diversity and many factors could be responsible for the greater insect diversity, including a greater vegetative diversity (Siemann and others, 1998), biomass (Haddad and others, 2001), and structural complexity (Lawton, 1983) in the shrub transitional zone.

The same suite of nonparametric statistics described previously for sediment samples was used to determine

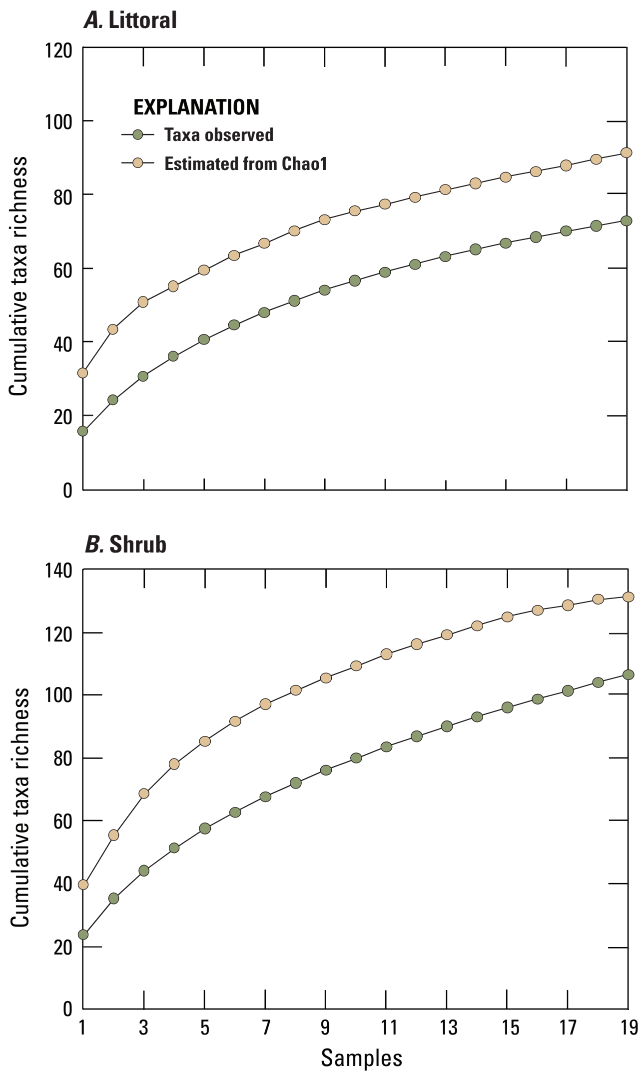


Figure 7.8. Species accumulation curves for adult invertebrate samples in littoral and shrub habitats and both habitats combined, showing observed and estimated taxa richness in the east estuary of the Elwha River, Washington, in 2007. (Estimated values are from rarefaction using the Chao1 estimator.)

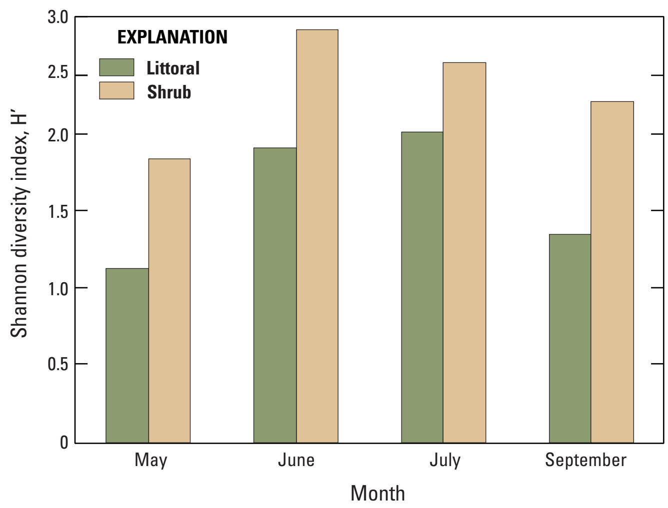


Figure 7.9. Shannon diversity index scores of the macroinvertebrate community from flight-intercept (fallout) traps deployed in shrub and littoral habitat sampling locations of the east estuary of the Elwha River, Washington, in May, June, July, and September 2007.

the assemblage structure of the terrestrial insect community. As with the benthic macroinvertebrate samples, these tests were performed on the full data set and a data set aggregated to order. Results were similar for nMDS and nearly identical for ANOSIM results, so only the results from the full data set are presented here unless otherwise noted. After square root transforming the taxa-by-site matrix for all months combined, an nMDS was run using the Bray-Curtis similarity measure. Unlike the multivariate results for sediment samples (which did not show clear separation among habitats between months), the terrestrial macroinvertebrates were different across space and time. The nMDS plot showed clear separation between the shrub and littoral insect communities, as well as differences among months. In the shrub and littoral communities, samples from September grouped separately from the May, June, and July samples, which were largely grouped together (fig. 7.10). A two-way ANOSIM analysis confirmed the clear separation shown in the nMDS plots for both location (R = 0.93, P = 0.001) and month (R = 0.63, P = 0.001). A pairwise test showed that June and July were most similar (R = 0.34, P = 0.001) and May and September (R = 0.93, P = 0.001) were most different in their terrestrial macroinvertebrate assemblage structure (across all habitat groups).

A SIMPER analysis of the data was completed to find which groups of invertebrates were responsible for differentiating littoral and shrub habitat. There was a higher abundance of Collembola (21.9 percent of the average dissimilarity), Hymenoptera (11.3 percent), Acarina (10.9 percent), and Araneae (9.6 percent) in the shrub samples and a higher abundance of Dipterans (12.7 percent) and Odonata (7.4 percent) in the littoral samples.

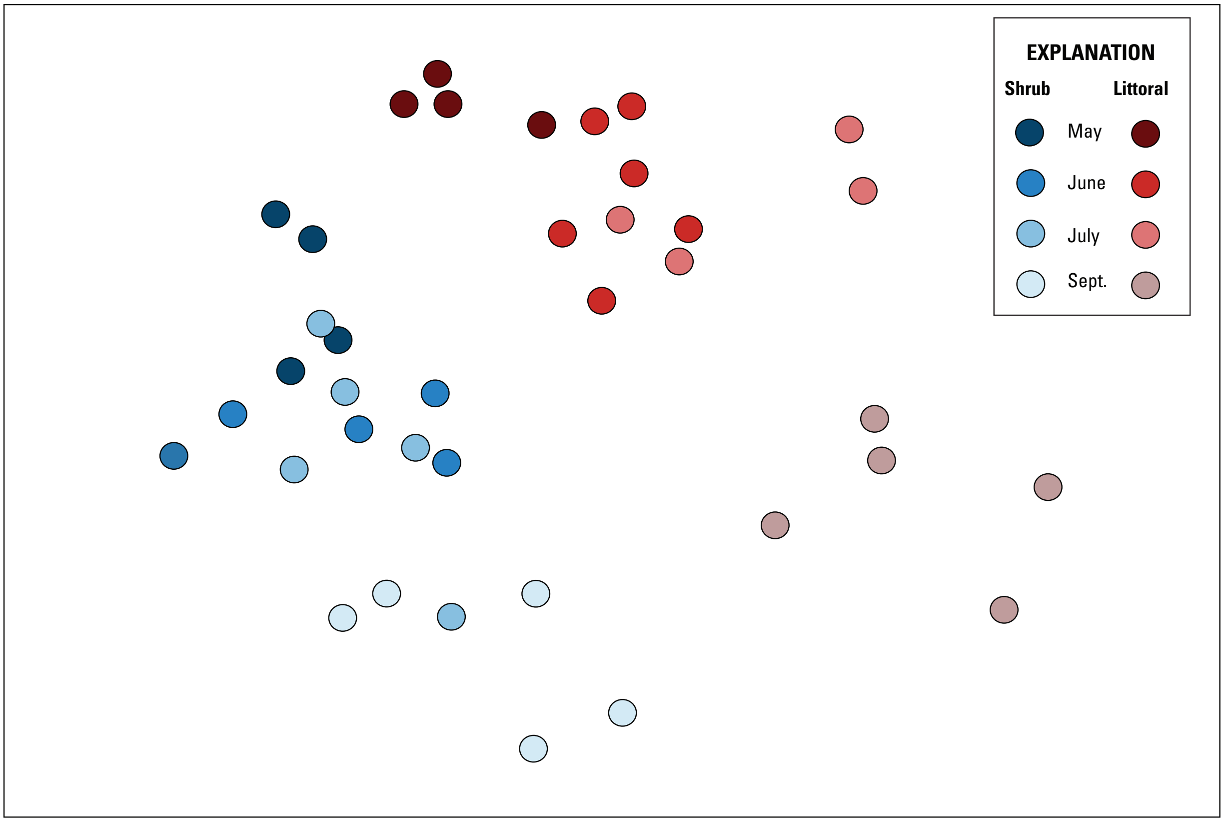


Figure 7.10. Nonmetric multidimensional scaling plots (nMDS) showing assemblage structure of adult invertebrates captured with flight-intercept (fallout) traps placed in (*A*) shrub and (*B*) littoral habitats of the east estuary of the Elwha River, Washington, in 2007. (Unitless nMDS plots based on Bray-Curtis similarity of square root transformed data.)

Juvenile Salmon in the Elwha River Estuary

The restoration of prime spawning and rearing habitat in the Elwha River upstream of two dams that have been in place for nearly a century is expected to benefit salmon as populations recolonize a watershed that once produced large runs (Wunderlich and others, 1994; Pess and others, 2008; Ward and others, 2008; Winter and Crain, 2008). A critical component of salmon population recovery in the Elwha River following dam removal will be from the maintenance or increase in life history and genetic diversity, which will in turn depend in part upon habitat use and performance within the Elwha River estuary (Bottom and others, 2005a). The capacity of the Elwha River estuary to support increased population sizes and life-history diversity of salmon populations will be determined by physical and biological factors, such as food sources, sediment deposition, temperature, salinity, and other habitat features.

The emigration of juvenile Chinook salmon from natal spawning grounds is a critical phase of their life history. Date of spawning, competition, food availability, local genetic adaptation, and variable life history strategies in populations determines the timing and duration of the emigration period, which also varies among species in the same river (Groot and Margolis, 1991). As described by Myers and others (1998), the high potential for interaction among these factors, coupled with spatial and temporal variability driven by natural and anthropogenic factors, creates complexity in determining factors responsible for trends in salmon population size, or phenotypically driven shifts in life history expression. In addition to these basic ecological questions of interest to fisheries biologists, the timing and abundances of emigrating juvenile salmon, along with estimates of run size, serves fisheries managers tasked with conservation of salmon populations. Finally, estimating vital rates associated with juvenile emigration has been used to track the effects of restoration actions geared toward salmon conservation (Bottom and others, 2005a).

Estimates of the species-specific timing and relative abundance patterns of anadromous juvenile Pacific salmonids during their migration from the Elwha River were made using various techniques throughout the lower river, the estuary, and adjacent nearshore areas (fig. 7.1).

Migration of Juvenile Salmon from the Lower Elwha River

Estimating In-River Migrants Using a Rotary Screw Trap

The Lower Elwha Klallam Tribe has been operating a 2.4 m rotary screw trap at river kilometer 0.5 since 2005 to characterize the emigration of juvenile salmonids from the Elwha River (fig. 7.11). The trap is manufactured from a perforated metal cone mounted between two pontoons. Two tapered flights are wrapped around the inside of the cone, causing the trap to rotate with the flow of the river. The trap is suspended between two pontoons and this barge-like structure is anchored to shore using steel cables. As downstream flow progresses through the trap, smolts traveling downstream are swept into the wide end of the cone and are forced into a trap box at the narrow end of the cone, where they can be identified, measured, and returned to the river (Volkhardt and others, 2007). The trap is monitored daily to process captured smolts. Occasionally, wood debris or other factors will cause the trap to stop working and no data are collected. For these instances, the daily catch total is estimated based on the total hours of operation during the day in question. If entire days are missed (a rare occurrence), then catch is estimated for missed days using a non-linear (LOESS) regression (G.R. Pess and others, National Oceanic and Atmospheric Administration, written commun., 2010).

Operation of the screw trap in the Elwha River typically is from February or March until mid-June when the Washington Department of Fish and Wildlife (WDFW) releases juvenile Chinook salmon from the Elwha River hatchery. Because of the size of the release (about 3 million fingerlings per year) and the proximity of the release point to the trap, operation of the rotary screw trap ceases upon hatchery release. The data collected from the rotary screw trap provide emigration timing, species composition, length, and genetic samples for part of the emigration period, mostly the wild-origin component of the population. Population estimates are determined by calculating the efficiency of the trap, which is estimated by releasing multiple groups of 1,000 dye-marked (Bismark brown) juvenile chum salmon above the trap and then counting the number of the marked fish caught in the trap downstream. This total then can be used to convert the catch into an estimate of total population size (Volkhardt and others, 2007; G.R. Pess and others, National Oceanic and Atmospheric Administration, written commun., 2010). The estimated trap efficiency was 3.5 percent in 2006 and 4.7 percent in 2007.

Figure 7.11. A 2.5 meter rotary screw trap operated at river kilometer 0.5 by the Lower Elwha Klallam Tribe fisheries office for counting juvenile salmon emigrating from the Elwha River, Washington. (Photograph taken by Michael L. McHenry, Lower Elwha Klallam Tribe, May 26, 2010.)

Results from the Elwha River Screw Trap, 2006–07

An estimated 119,357 (± 3,443, 95 percent confidence interval) and 14,309 (± 3,440, 95 percent confidence interval) age-0 Chinook salmon migrated past the Elwha River screw trap in 2006 and 2007, respectively (fig. 7.12). This estimate is based on 3,376 (2006) and 266 (2007) captures at the screw trap and does not include hatchery fish or "stream type" age-1 Chinook salmon smolts. However, these numbers should be used with caution. The large difference in smolt numbers between 2006 and 2007 may be explained, in part, by the large flood that occurred on November 6, 2006 (peak flow of 346 m^3/s [12,219 ft^3/s] reported by Curran and others, [2009]), which may have scoured redds and affected recruitment of the 2007 brood year. In 2006, the trapping did not begin until mid-March, but trapping in previous years indicated that migration timing typically starts earlier than this date in the Elwha River. Additionally, trapping efficiency is variable from year-to-year. Finally, a road closure in spring 2007 precluded deployment of the large screw trap. A smaller 1.5 m trap was used for the 2007 outmigration estimate. Rather than provide estimates of the emigrating population size, the data are provided to show general trends in run timing and juvenile abundance in 2006 and 2007, when estuary use and diet analyses were conducted.

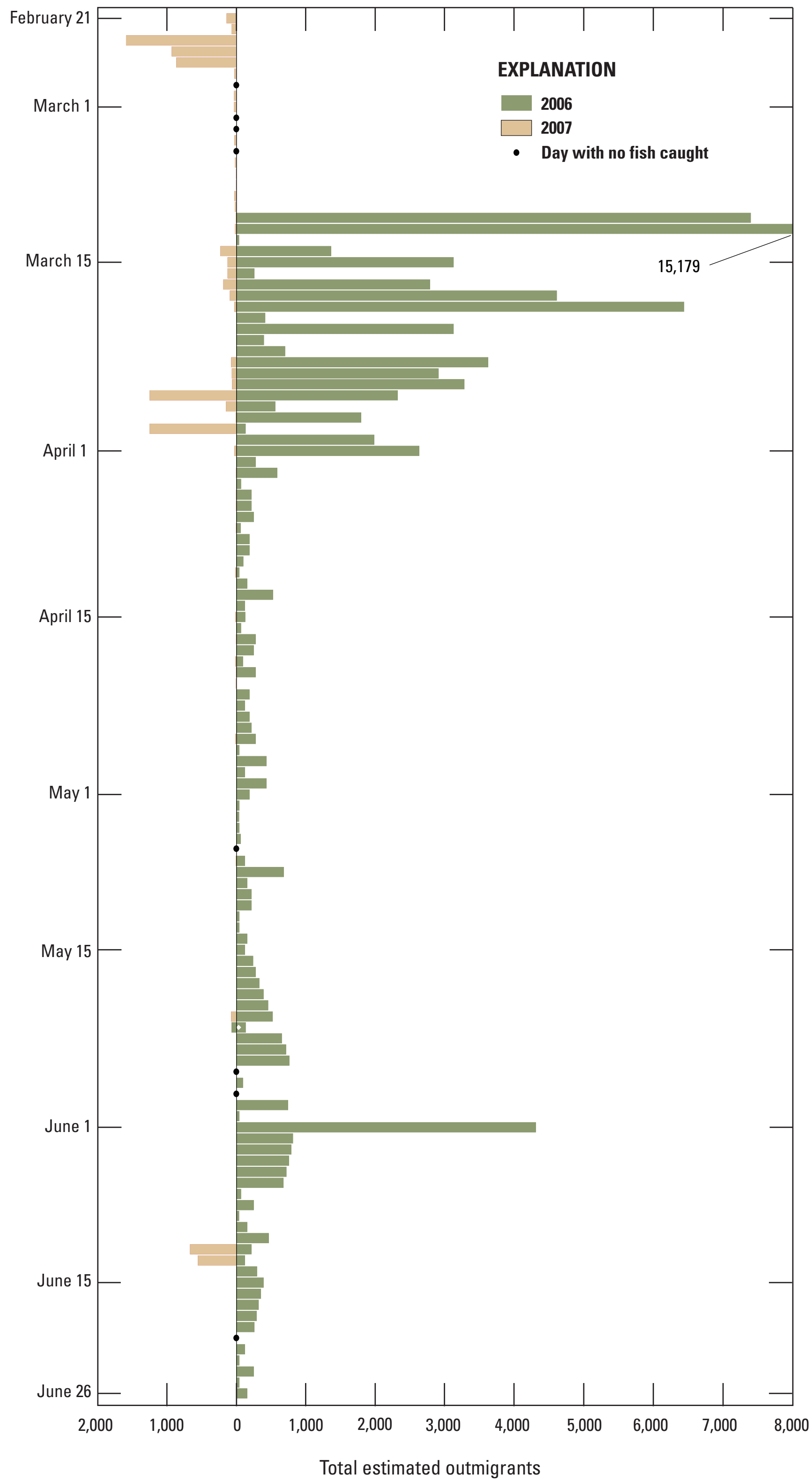


Figure 7.12. Run timing and estimated daily emigration numbers of juvenile Chinook salmon (age-0) based on rotary screw trap data collected by the Lower Elwha Klallam Tribe in 2006 and 2007, Elwha River, Washington. Data from G.R. Pess and others, National Oceanic and Atmospheric Administration, written commun., 2010.

Migration of Juvenile Salmon to the Elwha River Estuary

Detection of Estuary Migrants from Beach Seining

Beach seining techniques were used to sample fish populations in the Elwha River estuary on the east side of the river (fig. 7.13). Due to the shallow nature of much of the estuary and the presence of woody debris obstructing proper deployment of the net, access to suitable seining sites was limited to three primary sites in ES1 (1 site) and ES2 (2 sites) and one secondary site (IEC) (fig. 7.1). Seining started in mid-March and ended in December (2006) or September (2007). Generally, samples were collected every 2 weeks during daylight hours. Samples were collected on 13 occasions each year during 2006 and 2007 (table 7.5). The dimensions of the beach seine were 38 m long × 2 m deep, with a 2 m × 2 m bag in the center of the net. Mesh size was 3.18, 6.35, and 31.75 mm, for the bag, center panel, and wings, respectively.

Figure 7.13. Lower Elwha Klallam Tribe fisheries technicians and students from Peninsula College deploying a beach seine used to sample fish assemblages in the Elwha River estuary, Washington. (Photograph taken by Patrick Shafroth, U.S. Geological Survey, July 20, 2006.)

Table 7.5. Monthly sampling effort (number of occasions and seine hauls) used to estimate the fish assemblage structure of the east Elwha River estuary, Washington, 2006 and 2007.

[Locations are shown in figure 7.1. **Abbreviations**: ES1, east estuary 1; ES2, east estuary 2; IEC, intraestuarine channel]

Month	2006				2007			
	Sampling occasions	ES1 seine hauls	ES2 seine hauls	IEC seine hauls	Sampling occasions	ES1 seine hauls	ES2 seine hauls	IEC seine hauls
March	1	2	1	0	2	4	4	2
April	2	3	3	0	2	4	4	4
May	2	4	3	0	2	4	7	3
June	1	2	2	0	2	3	4	1
July	1	2	2	0	2	2	6	2
August	2	3	4	0	0	0	0	0
September	2	4	4	0	2	2	2	1
October	1	2	2	0	0	0	0	0
November	0	0	0	0	0	0	0	0
December	1	0	2	0	0	0	0	0

Results from Beach Seining in Elwha River Estuary, 2006–07

Sixteen species of fish were identified in the east estuary during 2006–2008 (table 7.6), including five species of salmonids (Chinook, coho, chum, steelhead, cutthroat trout) and one species of charr (bull trout, *Salvenlinus confluentus* captured in 2008). The catch per unit effort (CPUE) was calculated by dividing the number of fish caught by the number of seine hauls. Data from 2006 and 2007, but not 2008, are presented.

The most abundant fish taxa detected, adjusted for CPUE and summed across all sampling occasions, was threespined stickleback (*Gasterosteus aculeatus*; CPUE, 306 in 2006 and 1,823 in 2007), Pacific staghorn sculpin (*Leptocottus armatus*; CPUE, 180 and 134), unidentified sculpin spp. (*Cottidae* sp.; CPUE, 150 and 136), and starry flounder (*Platichthys stellatus*; CPUE, 99 and 93). The most abundant salmonid species detected were Chinook (CPUE, 136 in 2006 and 73 in 2007), coho (CPUE, 70 and 32), and chum (CPUE, 24 and 20) (table 7.6).

The patterns of seasonal estuary use were similar between 2006 and 2007 for all three salmonid species (fig. 7.14). Juvenile Chinook were caught in the estuary from March through October, with peak abundances in June. Coho showed a similar pattern, with a peak in June, but numbers dropped off more steeply during summer compared with Chinook, to a level where they were rarely detected. Chum were present in relatively lower numbers across all months, with an earlier peak in March or April and were not detected after June. The number of Chinook likely was influenced by hatchery releases from the State of Washington Elwha River Fish Hatchery about 3.2 km upstream. These fish were not externally marked prior to release (however, they do receive thermal marks on their otoliths); therefore, estimates of the proportion of hatchery- to wild-origin could not be made. A sub sample (n=115) of fish collected over the entire season in 2006, however, showed a ratio of 75:40 for wild- versus hatchery-origin juvenile Chinook salmon (see *Juvenile Growth by Habitat, 2006 Brood Year*).

Coastal storms that occurred in winter of 2006 resulted in flooding that significantly changed the morphology of the Elwha River mouth and reduced access to the east estuary in spring and summer of 2007. A large flood on November 6, 2006, coupled with high tidal conditions (+2.81 m, MLLW) and significant storm surges, resulted in a sediment bar forming, which appeared to preclude fish passage to the east estuary (ES1 and ES2) during 2007, except during high tides and high river flow events. As noted by Shaffer and others (2009), this sediment bar likely disrupted fish movements into the east estuary. Their results showed significant differences in abundances of juvenile salmon between sites in this study in the east estuary and their significantly smaller site in WESC. Results also indicate that the entire fish assemblage of the east estuary was different between 2006 and 2007, probably due to the formation of the sediment bar. The change in CPUE values between 2006 and 2007 were negative for Chinook (-46.3 percent), coho (-54.6 percent), and chum (-14.9 percent salmon (table 7.6).

Table 7.6. Total counts and catch per unit effort of the fish assemblage during beach seining in the eastern Elwha River estuary, Washington, 2006 and 2007.

[**Abbreviations:** CPUE, catch per unit effort; –, not applicable]

	2006		2007		CPUE change (percent)
	Total catch	CPUE	Total catch	CPUE	
Salmonids					
Chinook	266	136	129	73	-46.3
Coho[1]	127	70	37	32	-54.6
Chum	34	24	38	20	-14.9
Steelhead[2]	26	24	44	30	27.1
Cutthroat Trout	1	0.5	2	0.8	60.0
Sculpins					
Pacific Staghorn Sculpin	350	180	282	134	-25.7
Prickly Sculpin	0	0	22	12	–
Cottid spp.	274	150	324	136	-9.5
Flatfish					
English Sole	1	0.5	0	0	-100
Starry Flounder	191	99	190	93	-6.0
Flatfish spp.	37	18	0	0	-100
Forage Fish					
Surf Smelt	3	1.5	0	0	-100
Osmerid spp.	15	12	0	0	-100
Other					
Shiner Perch	2	1	0	0	-100
Threespine Stickleback	591	306	2,675	1,823	496.7

[1]Total does not include two coho "jacks."

[2]Most were hatchery fish from the Lower Elwha Klallam Tribe hatchery that were released into the estuary.

Other pelagic fish, such as English sole (*Parophrys vetulus*), surf smelt (*Hypomesus pretiosus*), other osmerids, and shiner perch (*Cymatogaster aggregata*) were detected in 2006 but not in 2007.

Mean lengths of juvenile salmon in the Elwha River estuary generally increased through the sampling season (fig. 7.15). An outlier was a steep increase in average length of coho salmon in April of 2006 attributed to two coho "jacks" (adult fish who stay in the ocean for less than 1 year and return to the river earlier than their cohort) caught in the east estuary.

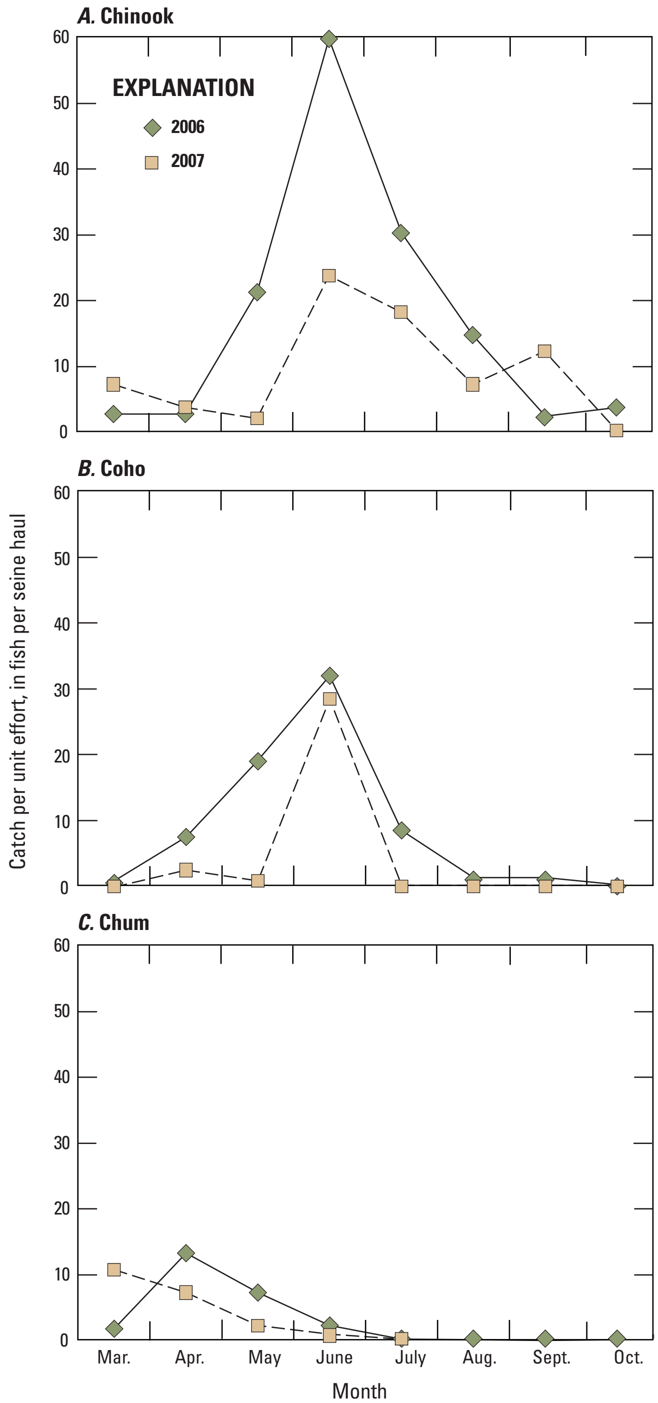


Figure 7.14. Catch per unit effort (number of fish captured on occasion *i* / number of seine hauls on occasion *i*) of juvenile (*A*) Chinook, (*B*) coho, and (*C*) chum salmon detected in the east estuary of the Elwha River estuary, Washington, in 2006 and 2007.

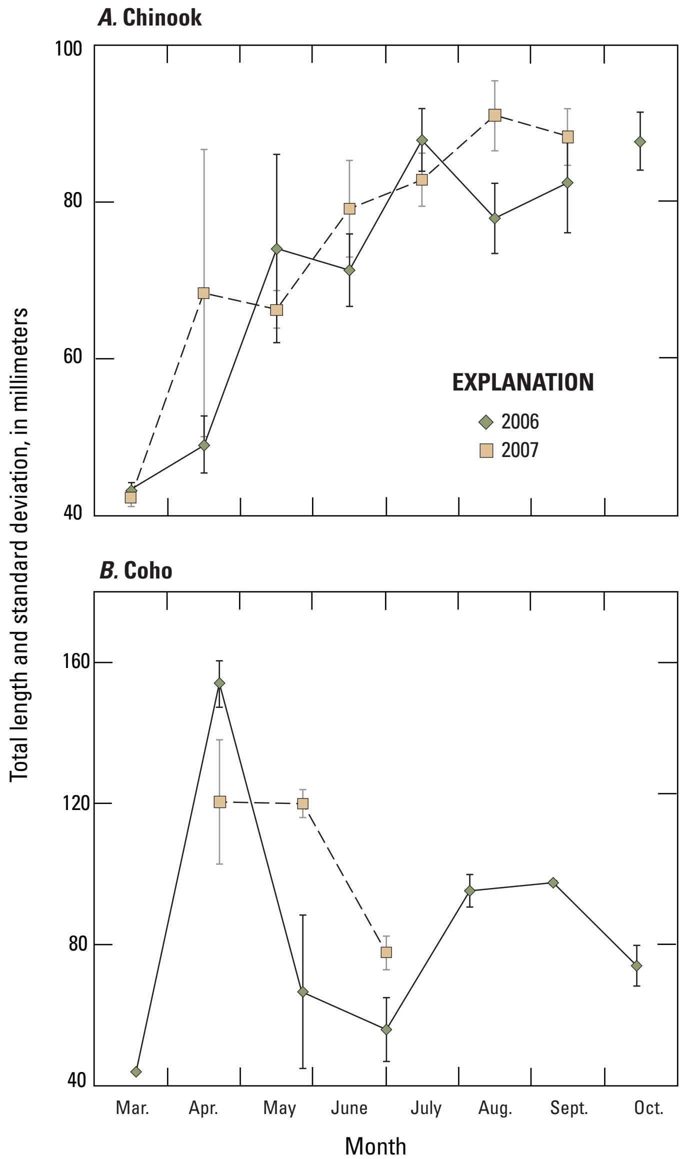


Figure 7.15. Average total length of juvenile (*A*) Chinook and (*B*) coho salmon captured during beach seining in the east estuary of the Elwha River, Washington, during 2006 and 2007. Coho results for April 2006 include two "jacks."

Estuary and Nearshore Habitat Use by Juvenile Chinook Salmon Inferred from Otolith Analysis

Salmonid otolith microstructure can identify different juvenile life history strategies and events (for example see, Neilson and others, 1985; Volk and others, 1995). Previous work on more extensive and complex estuarine habitats in Puget Sound has shown that juvenile Chinook display distinct developmental/life history "checks" and patterns of growth on the otolith microstructure (Beamer and Larsen, 2004; Lind-Null and others, 2008; Lind-Null and Larsen, 2009, 2010). These distinct checks (a disruption in the microstructure growth pattern) usually corresponded with a transition between habitat types followed by distinct growth patterns, allowing us to quantify differential habitat use. Checks on juvenile salmonid otoliths can be induced (1) naturally, when fish migrate from one habitat zone to another (for example, freshwater to saltwater); (2) naturally, during other life history events (for example, hatching or emergence); and (3) artificially, by thermal marking or other manipulation (such as ponding checks when hatchery fish are moved between rearing vessels).

Otolith Microstructure Methods

Processing and Analysis of Juveniles

In the laboratory, fish fork length (mm) and weight (g) were measured, the sagittal otoliths were extracted from the left and right side of each fish, otoliths were measured by length and weight, and the left otolith was embedded in resin, which was then baked to harden. Each resin block was mounted onto a slide and then ground and polished on both sides to a thin cross-section, which revealed the otolith microstructure for analysis. If the left otolith was composed of vaterite, a crystalline transparent morph of calcium carbonate, which lacks clearly defined growth increments, the right sagittal otolith was substituted for analysis (Neilson, 1992).

Next, images of the ground otoliths were imported into Image Pro Plus image analysis software (ver. 5.1.2) using a CCD color video camera and a Zeiss compound microscope. Two main analyses were of interest: (1) the identification of hatchery fish through thermal marks on the otolith, and (2) the interpretation of otolith microstructure for estimates of residence and growth. On each saved image, a consistent radial axis was drawn and then increments were marked along the axis using standard techniques (Stevenson and Campana, 1992). The distance between marks provides increment width, an indirect measurement of daily growth. Checks indicative of habitat transition also were identified. The number of marks and their average width provided estimates of residence time and growth in designated habitat zones.

River, Estuary, and Nearshore Otolith Microstructural Patterns of Juveniles

Samples were collected from the lower Elwha River, the tidally influenced region of the estuary near the mouth of the river (ES1, ES2, IEC, WESC; locations shown in fig. 7.1), and a few other sites in the river just upstream of the river mouth. Samples also were collected from nearshore areas accessible by beach seine in the Strait of Juan de Fuca east and west of the Elwha River mouth (provided by colleagues; Kurt Fresh, National Oceanic and Atmospheric Administration Fisheries and Shaffer and others, 2009). Based on work in other Puget Sound river systems (Beamer and Larsen, 2004; Lind-Null and others, 2008; Lind-Null and Larsen, 2009, 2010) we divided our sampling *a priori* into freshwater (FW), forested riverine tidal (FRT), emergent forested transition (EFT), estuarine emergent marsh (EEM), and nearshore (NS) habitat areas. These areas are defined as FW, main stem and side channels, above tidal influence, in the lower Elwha River; FRT, main stem and side channels/sloughs within tidal influence, generally occurring in the uppermost estuary; EFT, a transition between riparian forest and salt marsh, characterized by cattails and scrub/shrub habitat (ES2, IEC); EEM, true salt marsh (lowermost estuary), cyclically inundated by the tide and characterized by salt tolerant plants (ES1, WESC); and NS, saltwater, shallow sub-tidal and intertidal areas.

We processed 266 juvenile Chinook salmon, collected between March and October 2006 and between May and September 2007. We attempted to collect equivalent numbers of fish in each habitat zone, equally divided between two periods (before and after the release of hatchery fish on June 15th). About one-half of the samples were collected prior to the June release of hatchery fish.

Approximately 3 million juvenile Chinook are released from the Elwha River Hatchery operated by the WDFW after they are thermally marked and raised to a certain size. Thermal marking occurs at one of two rearing facilities (Sol Duc and Hurd Creek). Each facility produces a unique mark by exposing the fish to short-term temperature manipulations during early embryonic development, creating easily recognized patterns of dark, distinct increments on their otoliths. These patterns stand in stark contrast to those produced in otoliths from wild fish (compare fig. 7.16 with fig. 7.17; see also Volk and others, 1999). We familiarized ourselves with the unique thermal marking patterns (fig. 7.16) on the otolith microstructure of Elwha River Hatchery fish. This was done for the 2006 and 2007 outmigrant years by processing 25 reference otoliths collected from the two facilities in 2006 and from images available at the North Pacific Anadromous Fish Commission website (for 2006 and 2007). Eighteen sub-yearlings collected in 2006 from

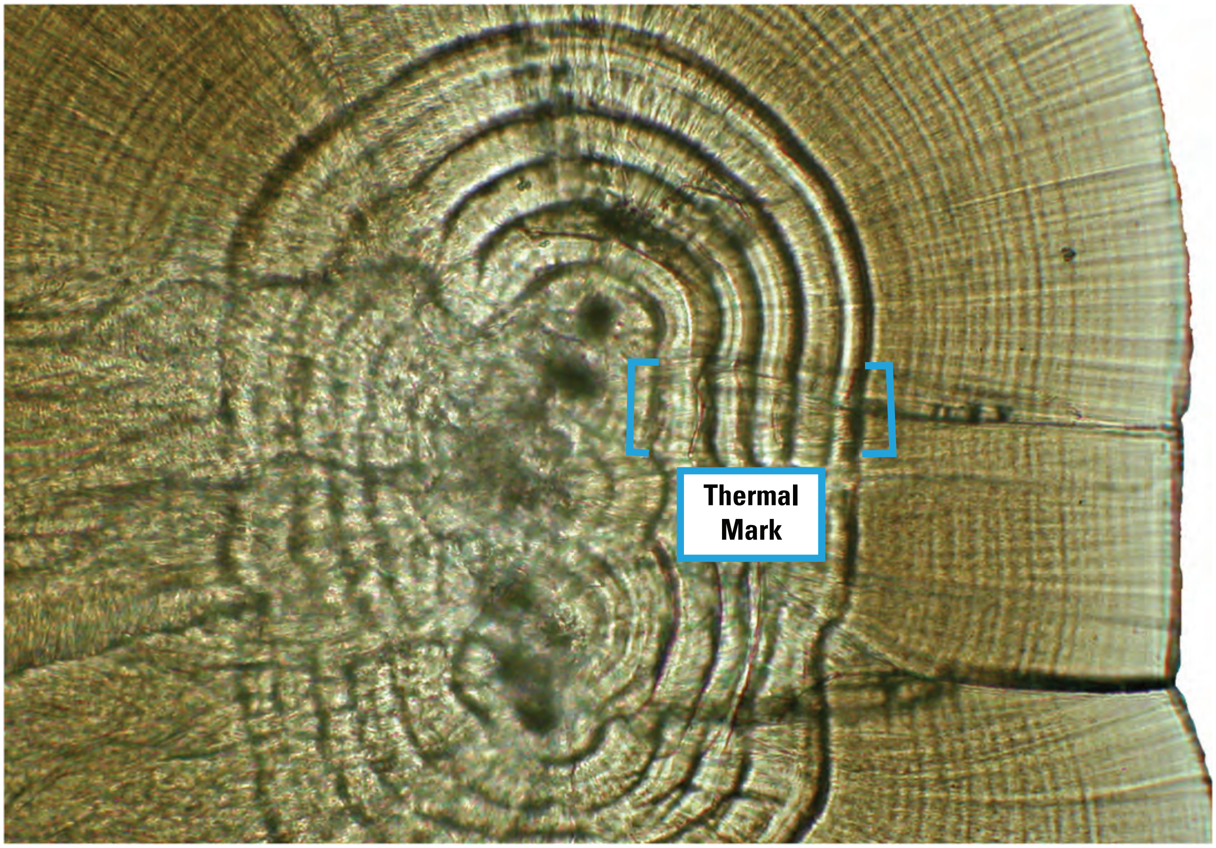


Figure 7.16. Thermally marked juvenile Chinook salmon otolith (ear stone) collected prior to release from the Washington Department of Fish and Wildlife Elwha River hatchery, Washington. Brackets highlight the clear and unique marking patterns present due to a thermal marking process whereby water temperatures are fluctuated during embryonic development to slow and speed growth. This results in a specific pattern that is representative of a particular hatchery, or even brood year, and can be used for positive stock identification. (Photograph taken by Karl Stenberg, U.S. Geological Survey, January 24, 2008.)

the Dungeness River Hatchery also were processed to determine if any mixing occurred between the non-thermally marked Dungeness River Hatchery juveniles and wild-origin Elwha River juveniles in the Elwha River nearshore habitat. Differences in microstructural patterns are more subtle when comparing unmarked hatchery otoliths to wild fish otoliths. Daily growth increments formed in otoliths from hatchery Chinook salmon, immediately after the onset of exogenous feeding, are usually wider and more uniform in width than increments of wild-origin fish. Additionally, hatchery Chinook salmon frequently produce a release/ponding check when they are released from the hatchery or transported to holding ponds (Zhang and others, 1995). Later in the season, Chinook salmon from other river systems could be captured near the Elwha; however, the goals of this study are met simply by the ability to separate Elwha and Dungeness individuals from other stocks.

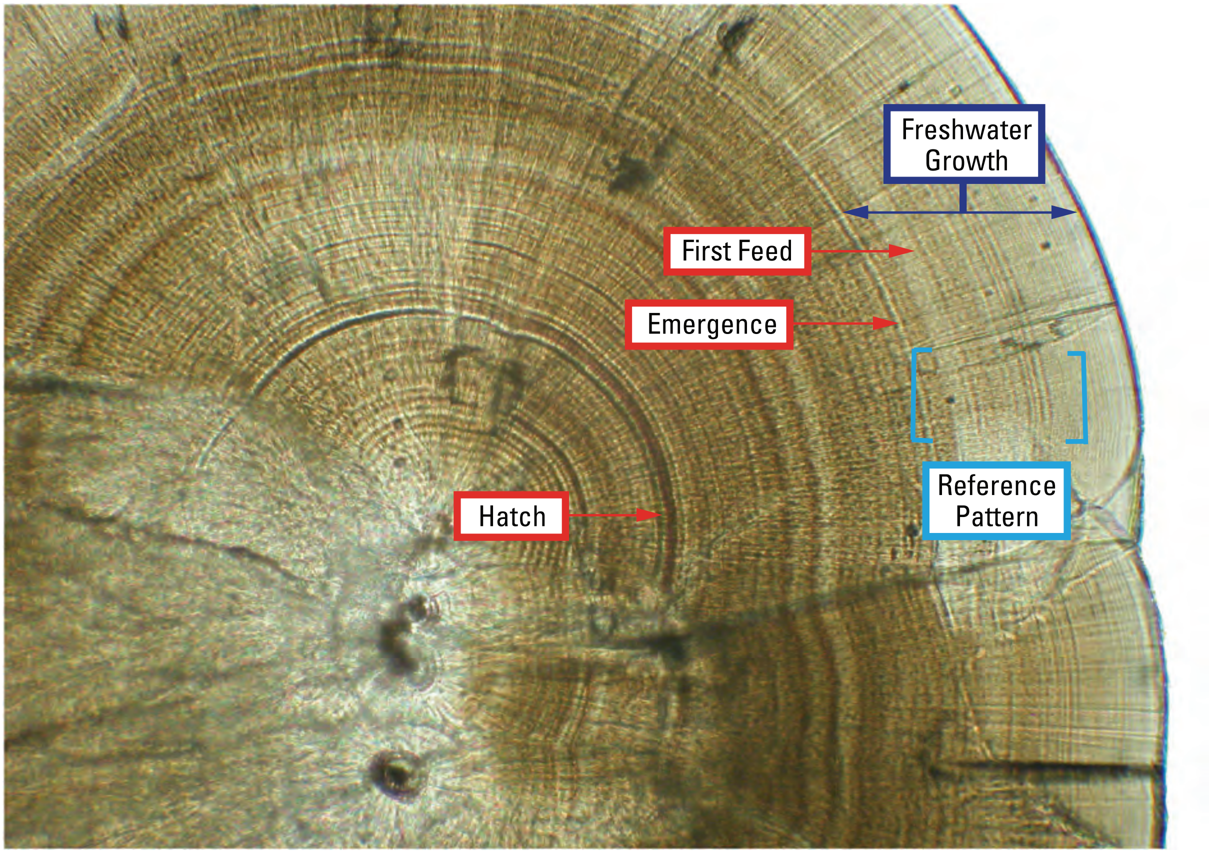


Figure 7.17. Otolith (ear stone) displaying characteristic growth patterns of wild-origin Chinook salmon captured in freshwater or upper estuary habitats of the Elwha River, Washington. The otolith microstructure depicts points of development (hatch, emergence from the gravel, first feeding) and a freshwater microstructural pattern used as a reference when analyzing otoliths from fish caught in other habitats downstream. (Photograph taken by Karl Stenberg, U.S. Geological Survey, January 24, 2008.)

Processing and Analysis of Adults

The overall objective was to collect otoliths from wild Elwha adult returns to analyze the otolith microstructure from the juvenile portion of the adult otoliths for life history, growth, and habitat utilization. The 2008 and 2009 adult returns were sub-sampled to correspond to juvenile Chinook outmigrants previously collected from the 2005 and 2006 brood years (2006 and 2007 collections). Forty-four and 53 otolith samples, respectively, were collected from September to mid-October in 2008 and 2009, from spawned-out adult Chinook salmon in the lower Elwha River (mouth to river kilometer 4.8). Elwha Klallam Tribal biologists collected samples during the height of the fall Chinook spawning season in the Elwha River.

All 97 samples from the 2008 and 2009 collections were initially aged (table 7.7). The otoliths were placed in deionized water and viewed with a dissection microscope. With the aid of transmitted and reflected light, the number of annuli present were visually determined. Both sagittal otoliths were used to determine the age of each fish. The left sagittal otolith was then processed (embedded and ground, sulcus-side down) and visually analyzed to determine hatchery or wild origin and on occasion to obtain a more accurate age determination. The continued analysis of the successful wild-origin spawners for life-history information and habitat utilization from the juvenile portion of their otolith microstructure was not possible due to small sample size (see results, below).

Table 7.7. Origin, age, gender, and size (fork length) of adult Chinook salmon collected by Lower Elwha Klallam Tribal biologists in the lower Elwha River, Washington, 2008 and 2009.

[Origin was determined by otolith analysis, which reveals whether a fish was born in a hatchery or the wild. **Marked:** Thermal otolith mark. **Origin:** Elwha River juvelile Chinook salmon were thermally marked and reared at two different hatcheries, the Sol Duc and the Hurd Creek Hatcheries. **Abbreviations:** mm, millimeter; Y, yes; N, no; N/A, not applicable; cwt, coded-wire tag]

Date sampled	Fish identifier	River kilometer	Age (years)	Sex	Fork length (mm)	Marked	Origin
			2008				
September 16	16	0.0–1.0	3	M	670	Y	Hurd Creek (cwt)
September 17	17	1.6–4.8	4	F	850	Y	Sol Duc
	18	1.6–4.8	4	F	940	N	Wild
	19	1.6–4.8	4	M	1,040	N	Wild
	20	1.6–4.8	4	F	920	N	Wild
	21	1.6–4.8	4	F	760	Y	Hurd Creek
	22	1.6–4.8	4	F	920	Y	Sol Duc
	23	1.6–4.8	4	M	900	N	Wild
	24	1.6–4.8	3	M	790	Y	Sol Duc
	25	1.6–4.8	4	F	930	Y	Hurd Creek
	26	1.6–4.8	3	M	850	Y	Hurd Creek
September 18	27	0.0–3.2	4	M	1,020	N	Wild
	28	0.0–3.2	4	M	1,030	N	Wild
	29	0.0–3.2	4	F	890	N	Wild
	30	0.0–3.2	3	M	800	Y	Sol Duc
	31	0.0–3.2	3	F	750	Y	Sol Duc
	32	0.0–3.2	3	M	750	Y	Sol Duc
	33	0.0–3.2	3	M	790	Y	Sol Duc
	34	0.0–3.2	5	M	1,070	N	Unknown
	35	0.0–3.2	3	M	780	Y	Sol Duc
	36	0.0–3.2	3	F	700	Y	Sol Duc
September 22	37	0.0–1.0	3	M	860	Y	Sol Duc
	38	0.0–1.0	4	F	780	Y	Sol Duc

Table 7.7. Origin, age, gender, and size (fork length) of adult Chinook salmon collected by Lower Elwha Klallam Tribal biologists in the lower Elwha River, Washington, 2008 and 2009.—Continued

[Origin was determined by otolith analysis, which reveals whether a fish was born in a hatchery or the wild. **Marked:** Thermal otolith mark. **Origin:** Elwha River juvelile Chinook salmon were thermally marked and reared at two different hatcheries, the Sol Duc and the Hurd Creek Hatcheries. **Abbreviations:** mm, millimeter; Y, yes; N, no; N/A, not applicable; cwt, coded-wire tag]

Date sampled	Fish identifier	River kilometer	Age (years)	Sex	Fork length (mm)	Marked	Origin
			2008—Continued				
September 29	39	0.3–3.2	4	M	940	Y	Sol Duc
	40	0.3–3.2	3	F	640	Y	Sol Duc
	41	0.3–3.2	4	F	790	Y	Sol Duc
	42	0.3–3.2	3	M	790	Y	Sol Duc
	44	0.3–3.2	4	M	810	Y	Sol Duc
	45	0.3–3.2	3	F	690	Y	Sol Duc
	46	0.3–3.2	3	M	840	Y	Sol Duc
September 30	47	3.2–3.7	4	M	960	N	Wild
	48	3.2–3.7	4	F	750	Y	Hurd Creek
	49	3.2–3.7	3	F	650	Y	Sol Duc
	50	3.2–3.7	3	M	700	Y	Sol Duc
	51	3.2–3.7	3	F	700	Y	Sol Duc
	52	3.2–3.7	3	M	670	Y	Sol Duc
	53	3.2–3.7	3	M	640	Y	Sol Duc
	54	3.2–3.7	3	F	650	Y	Sol Duc
	55	3.2–3.7	4	M	840	N	Wild
October 1	56	2.9–3.2	3	M	690	Y	Sol Duc
	57	2.9–3.2	4	F	910	N	Wild
	58	2.9–3.2	4	F	840	Y	Sol Duc
	59	2.9–3.2	5	M	1,050	N	Unknown
October 6	60	0.0–4.8	3	F	690	Y	Sol Duc
			2009				
September 22	61	2.9–3.2	4	N/A	N/A	Y	Sol Duc
	62	2.9–3.2	4	N/A	N/A	Y	Sol Duc
	63	2.9–3.2	4	N/A	N/A	Y	Sol Duc
	64	2.9–3.2	5	N/A	N/A	N	Wild
	65	2.9–3.2	4	N/A	N/A	Y	Sol Duc
	66	2.9–3.2	4	N/A	N/A	Y	Sol Duc
	67	2.9–3.2	4	N/A	N/A	Y	Sol Duc
	68	2.9–3.2	4	F	760	Y	Sol Duc
	69	2.9–3.2	4	F	750	Y	Sol Duc
September 23	70	2.4–2.7	4	F	750	Y	Sol Duc
	71	2.4–2.7	4	F	790	Y	Sol Duc
	72	2.4–2.7	4	F	710	Y	Sol Duc
	73	2.4–2.7	4	F	680	Y	Sol Duc
	74	2.4–2.7	4	M	780	Y	Sol Duc
	75	2.4–2.7	4	M	770	Y	Sol Duc
	76	2.4–2.7	4	F	730	Y	Sol Duc
	77	2.4–2.7	4	F	660	Y	Sol Duc

Table 7.7. Origin, age, gender, and size (fork length) of adult Chinook salmon collected by Lower Elwha Klallam Tribal biologists in the lower Elwha River, Washington, 2008 and 2009.—Continued

[Origin was determined by otolith analysis, which reveals whether a fish was born in a hatchery or the wild. **Marked:** Thermal otolith mark. **Origin:** Elwha River juvelile Chinook salmon were thermally marked and reared at two different hatcheries, the Sol Duc and the Hurd Creek Hatcheries. **Abbreviations:** mm, millimeter; Y, yes; N, no; N/A, not applicable; cwt, coded-wire tag]

Date sampled	Fish identifier	River kilometer	Age (years)	Sex	Fork length (mm)	Marked	Origin
			2008—Continued				
September 24	78	0.5–1.0	4	M	870	Y	Sol Duc
	79	0.5–1.0	4	M	800	Y	Sol Duc
	80	0.5–1.0	4	M	820	Y	Sol Duc
	81	0.5–1.0	4	M	870	Y	Sol Duc
September 30	82	2.2–2.4	4	M	830	Y	Sol Duc
	83	2.2–2.4	4	M	740	Y	Sol Duc
October 1	84	2.2–2.4	4	F	860	Y	Sol Duc
	85	2.2–2.4	4	F	790	Y	Sol Duc
	86	2.2–2.4	4	F	860	Y	Sol Duc
	87	2.2–2.4	4	M	890	Y	Sol Duc
October 5	88	2.4–2.7	4	M	840	Y	Sol Duc
	89	2.4–2.7	4	M	680	Y	Sol Duc
	90	2.4–2.7	4	M	780	Y	Sol Duc
	91	2.4–2.7	4	F	840	Y	Sol Duc
	92	2.4–2.7	4	F	780	Y	Sol Duc
	93	2.4–2.7	4	M	730	Y	Sol Duc
	94	2.4–2.7	4	F	810	Y	Sol Duc
	95	2.4–2.7	4	M	860	Y	Sol Duc
	96	2.4–2.7	4	M	810	N	Wild
	97	2.4–2.7	4	F	810	Y	Sol Duc
October 1	98	2.4–2.7	4	F	850	Y	Sol Duc
October 13	99	0.0–0.8	4	F	850	Y	Sol Duc
	100	0.0–0.8	4	M	870	Y	Sol Duc
	631	0.0–0.8	4	F	870	Y	Sol Duc
	632	0.0–0.8	4	F	870	Y	Sol Duc
	633	0.0–0.8	4	F	830	Y	Sol Duc
	634	0.0–0.8	4	F	880	Y	Sol Duc
	635	0.0–0.8	4	M	950	Y	Sol Duc
	636	0.0–0.8	4	M	N/A	Y	Sol Duc
	637	0.0–0.8	4	F	830	Y	Sol Duc
	638	0.0–0.8	4	F	900	Y	Sol Duc
October 15	654	0.0–0.8	4	F	950	Y	Sol Duc
	655	0.0–0.8	4	F	830	Y	Sol Duc
	656	0.0–0.8	4	F	N/A	Y	Sol Duc
October 13	657	0.0–0.8	4	F	880	Y	Sol Duc
October 13	658	0.0–0.8	4	M	1,020	Y	Sol Duc

Otolith Microstructure Results

Juvenile Growth by Habitat, 2006 and 2007 Brood Years

We separated 115 (2006) and 142 (2007) emigrating Chinook juveniles into marked hatchery origin (fig. 7.16) and unmarked wild or natural origin (figs. 7.17 and 7.18) based on otolith microstructure patterns. Of the 115 Chinook from 2006, 41 were of hatchery origin (40 Elwha River Hatchery and 1 Dungeness River Hatchery), 3 were stream type (age-1) outmigrants, and 71 were wild Elwha River (age-0). Of the 142 Chinook from 2007, 98 were of hatchery origin (90 from the Elwha River and 8 from the Dungeness River) and 44 were wild Elwha River (age-0). Of the 97 fish collected in the nearshore, 65 were from the Elwha River Hatchery and 8 were from the Dungeness River Hatchery. We analyzed 107 wild-origin juvenile Chinook otoliths (8 samples were unavailable due to poor otolith quality or processing damage). Three life history checks (hatch, emergence, and first feed) were visually and quantitatively located on these samples (figs. 7.17 and 7.18). The increments became more consistent and identifiable across the radial axis beyond the checks associated with emergence and first feed. Reference samples of juveniles collected from freshwater sites were used to characterize a microstructural reference pattern beyond emergence and first feed unique to the FW habitat zone. Samples collected in the estuary (FRT, EFT, and EEM) and NS were analyzed visually and quantitatively for checks and increased growth (inferred from larger increment widths) beyond the microstructural reference pattern observed on the samples caught in FW habitat.

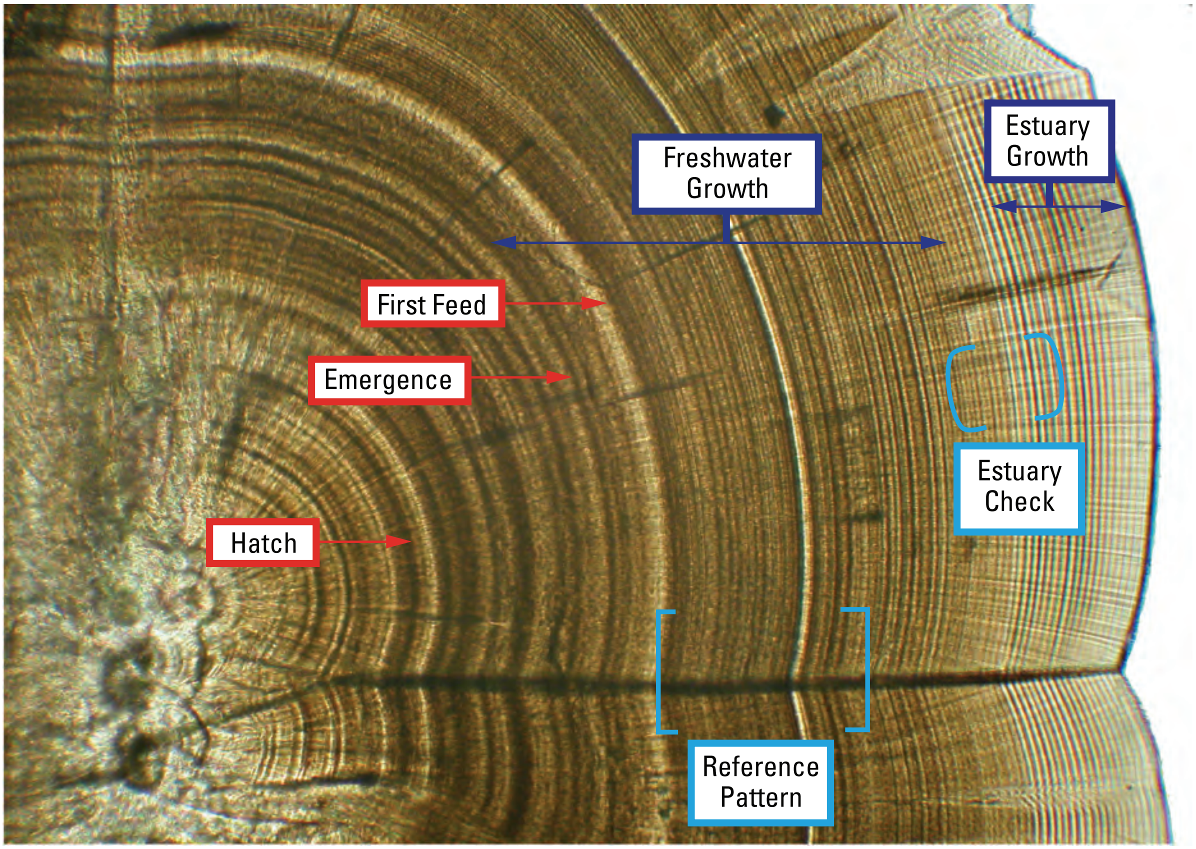


Figure 7.18. Otolith (ear stone) displaying characteristic microstructure growth patterns of wild-origin Chinook salmon captured in the lower estuary (2006 and 2007) or nearshore (2006) habitats of the Elwha River, Washington. The otolith microstructure depicts points of development (hatch, emergence from the gravel, first feeding), a freshwater reference pattern, an estuarine check where increments are transitioning between freshwater and estuary residence, and increased increment widths beyond the check that are indicative of higher growth potential in some estuarine habitats. (Photograph taken by Karl Stenberg, U.S. Geological Survey, January 24, 2008.)

The results for wild juvenile Chinook salmon revealed two distinct otolith microstructural patterns in 2006 and three distinct patterns in 2007 based on the criteria mentioned previously. The first pattern represented fish resident or captured in FW and FRT (uppermost estuary) habitats for both sampling years (fig. 7.17), because no distinct check or increased growth were apparent on the otoliths of fish caught in FW and FRT habitats. The second pattern (fig. 7.18), characterized by the presence of an estuarine check followed by increased daily growth, represented fish captured in EFT, EEM (lowermost estuary), and NS habitat types in 2006 and in EFT and EEM habitat types only in 2007. The 2007 collection further revealed a third check associated with entrance into the NS habitat type. The estuarine check was comprised of a series of increments with spacing representative of FW and estuarine growth, indicative of a transition period for the fish. The increments beyond the estuarine check were consistently wider and had higher contrast than increments associated with residence in the FW or FRT habitats prior to the check. An estuarine check alone or the check followed by a number of increments representing increased growth was not displayed on the otolith microstructure of fish collected in EFT, EEM, and NS habitats in 2006 until mid May (5 of 10), even though samples were collected in March and April (5 and 3 samples, respectively). The other five samples in May were split between no representation of estuarine residence and presence of an estuarine check at the edge of the otolith. Most of the remaining samples collected from June through September 2006 (20 of 23) displayed an estuarine check followed by an increase in growth. Two of the three samples not displaying increased growth, one in July and one in August, displayed no sign of estuary residence, and the third sample, from August, displayed the estuarine check at the otolith edge.

No samples were collected in the estuary before May 2007, so it is not known what the otolith pattern of early migrants was in 2007. However, 42 of 44 samples from 2007 displayed the estuarine check, with an increase in growth following the check. The absence of increased growth associated with fish captured in the lower estuary (EFT and EEM) before mid-May and the four caught thereafter could be due to lower food availability, which is consistent with stomach content results or lower temperatures affecting growth rates of these early emigrating fish. A more likely scenario emerges, however, given the placement of the estuarine check near the edge of the otolith. Fish collected in March and April may have been caught immediately upon their entrance into the lower estuary habitat and therefore these fish did not have a chance to experience the higher growth potential of the estuary. Some of the fish collected in the middle of the season also were fresh arrivals, whereas those collected later in the season were more likely to have had enough residence time to display increased growth.

A similar scenario of collecting fish shortly after their arrival may also explain the few individuals displaying a check indicative of entrance into NS habitat. A NS check was identified in a single individual in 2006. Although otoliths were collected from EFT and EEM habitats, no unique patterns of growth or checks associated with transition and growth were determined for these two habitat types.

It is not known whether a lack of unique patterns among the five habitat zones is due to small sample size, limitations of the technique, or minimal differentiation among habitat zones. This differentiation, seen in other river systems of Puget Sound (for example, Skagit River), often is caused by habitat specific differences in food availability, water temperature, water chemistry, salinity, spatial complexity, or a combination of these variables. The Elwha River does not have a large and heterogeneous estuary, as opposed to other Puget Sound systems (see Duda and others, 2011, chapter 1, this report), which could have contributed to the minimal differentiation observed on the otolith microstructure.

Once the different patterns of growth associated with transition from freshwater (FW and FRT capture sites) to lower estuary (EFT and EEM sites) and nearshore habitats were identified, the mean increment width (MIW) was used as an indirect measure of growth within these broad habitat categories. The number of daily increments assigned in habitats also was used to determine habitat-specific residence time.

The region of the otolith corresponding to freshwater growth was similar in MIW regardless of the habitat where individual fish were caught, with an average of 3.20 microns (fig. 7.19). The region of the otoliths assigned as lower estuary and nearshore growth showed an increase in MIW by 27 percent (4.06 microns). The average size of juvenile Chinook salmon increased as they migrated successively among freshwater, estuary, and nearshore environments (fig. 7.19*B*). The average estuarine residence time determined from Chinook captured in the estuary (a minimum estimate, as the fish were sacrificed) was 16 days (range = 4 to 41 days) (fig. 7.19*B*). Not surprisingly, the average estuarine residence time determined from Chinook salmon captured in the NS habitat showed a longer average residence of 25 days (range = 5 to 50 days), as this number is a more complete estimate because the individuals were captured after they left the estuary.

Based on a preliminary analyses, mostly wild-origin fish were using the lower estuary (EFT and EEM), whereas mostly hatchery fish were using the NS. Therefore, competition between hatchery and wild fish may be minimized in the estuary because most hatchery juveniles seem to bypass this habitat in favor of the Strait of Juan de Fuca. However, additional study would be required to confirm this hypothesis.

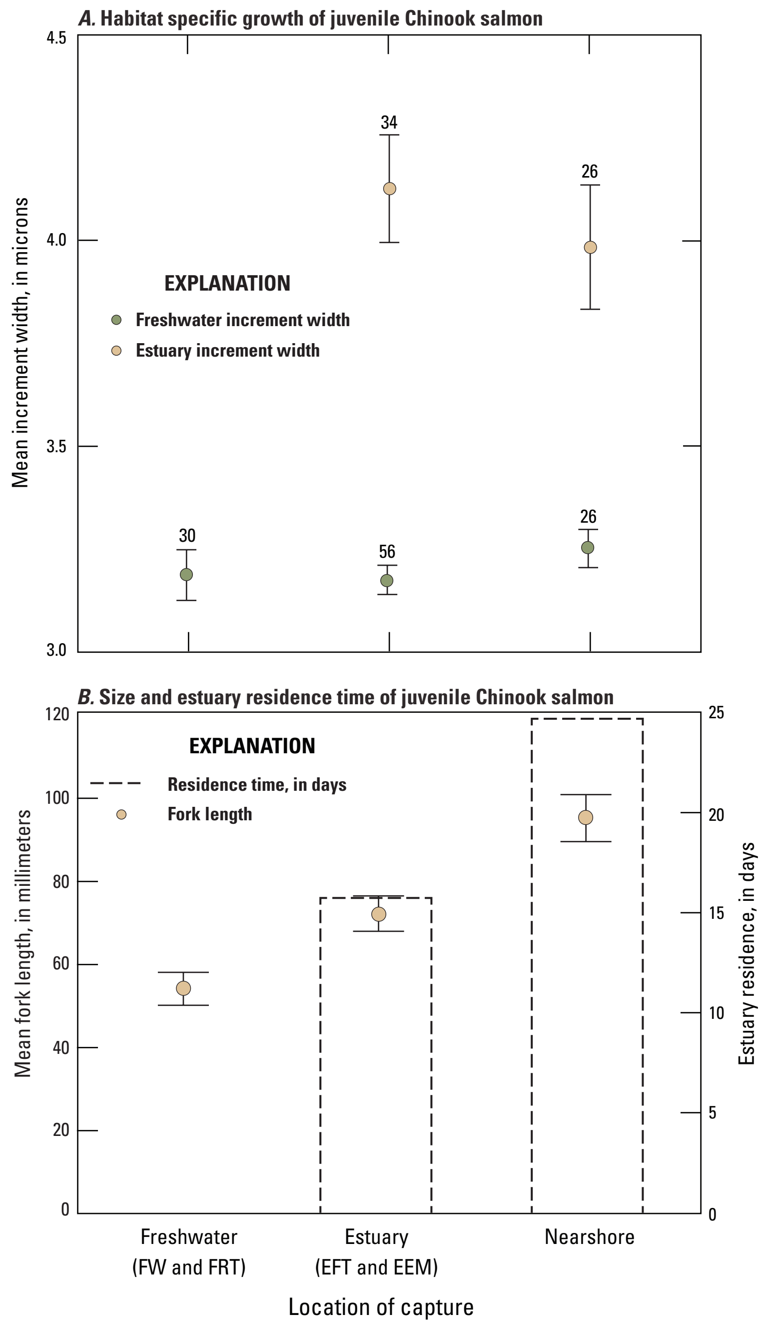


Figure 7.19. Estimates of (*A*) average daily growth in freshwater and "estuary" habitats for wild-origin juvenile Chinook salmon captured in the Elwha River, Washington, its estuary, and nearshore habitats from March to October 2006 and May to September 2007. Estimates include possible nearshore growth on samples collected in 2006. Values are based on mean increment width (in microns) measured from the microstructure portions on the otolith assigned to each habitat. (*B*) The average fork length of wild-origin juvenile Chinook salmon captured in each habitat and the average estuarine residence time for juvenile Chinook salmon captured in the Elwha River estuary and nearshore habitats. Residence from fish captured in the estuary is a minimum estimate because fish were sacrificed before completing their residence in the estuary. Fish captured in the nearshore provide a more complete estimate of estuarine residence.

An interesting observation was made with three FW and four FRT Chinook salmon collected on July 12, 2006. These Chinook salmon displayed fork lengths and otolith lengths/radial distances that were much smaller than all other fish collected around this time, looking more similar to samples caught in April than to other samples caught in July. Upon examination of "developmental checks" (hatch and emergence relative to capture), we surmised that these fish were progeny of late-spawning parents. Additional sampling would be required to determine whether these results suggest a unique life-history strategy.

Juvenile Dispersal Patterns in Elwha Estuary and Strait of Juan de Fuca, 2007 Brood Year

Because the aforementioned sediment bar formation in the east estuary reduced the catch of wild-origin fish in 2007, an otolith analysis was used to determine origin of 97 juveniles sampled from June 12 to September 11, 2007, in littoral drift cells (drift cells are discreet nearshore areas that receive local sources of sediment) adjacent to the Elwha in the Strait of Juan de Fuca (n = 98) by WDFW (Shaffer and others, 2009) and NOAA (K. Fresh, unpublished data, 2007) as well as from May 17 to August 28, 2007, in the Elwha River estuary (n = 45). The goal was to assess the dispersal of wild- and hatchery-origin (Elwha River hatchery or Dungeness River hatchery) juvenile Chinook salmon in nearshore waters to the east and west of the Elwha River. Samples were collected by beach seine from 16 locations (ordered from west to east): Pysht River, Crescent Bay, Freshwater Bay, east of the Elwha River mouth, Port Angeles Harbor, and Dungeness River. Elwha River estuary samples were collected from the east (ES1, ES2, IEC) and WESC. A random sample of individuals from each sampling occasion was collected, when available, for otolith extraction. Samples were processed as described previously and were compared to reference collections to determine hatchery or wild origin.

Most individuals collected from the Strait of Juan de Fuca were to the east of the Elwha River, although effort was concentrated in these areas. Across all sampling locations in the Strait of Juan de Fuca, most samples were Elwha River hatchery-origin (65) and wild-origin (44) fish. Only eight individuals of Dungeness River hatchery origin were detected (fig. 7.20). Of the Elwha River estuary fish, the largest numbers were collected in WESC (Shaffer and others, 2009), mostly because the sediment bar that was deposited following winter flooding restricted access to the east estuary. Throughout the estuary, 20 wild-origin and 25 hatchery-origin fish were identified.

For management and recovery purposes, Chinook salmon populations are apportioned into evolutionary significant units (Waples, 1995). The Puget Sound Chinook evolutionarily significant unit (ESU) ranges from the Puget Lowlands west to and including the Elwha River. The Strait of Juan de Fuca west of the Elwha River and the coastal rivers comprise the Washington Coast ESU. Populations in these two geographically adjacent ESUs have distinct genetic and life history characteristics (Myers and others, 1998), although Elwha River Chinook salmon appear to be a transition between the two units. Based on coded-wire tag (CWT) recoveries from commercial harvest, Washington Coast ESU Chinook tend to favor a more northern ocean migration, to southeast Alaskan waters, compared to Puget Sound ESU populations. Ocean migration of Elwha River Chinook is more typical of other Puget Sound ESU populations, with most CWT tags occurring in British Columbia and the rest occurring in southeast Alaska, coastal Washington, and Puget Sound (Puget Sound Indian Tribes and Washington Department of Fish and Wildlife, 2010). It is not known what proportion of the eastward migrating Elwha River Chinook juveniles become resident in Puget Sound, migrate north to the Pacific Ocean through the Strait of Georgia, or return westward to the Strait of Juan de Fuca towards the Pacific Ocean.

Analysis of Adult Chinook Salmon Otoliths

Forty-four Chinook adult returns collected in 2008 were segregated into wild and hatchery origin based on their otolith microstructure (table 7.7). Of these fish, 32 were of hatchery origin, 10 were of wild origin, and 2 were of unknown origin. Therefore, 76 percent of the known-origin Chinook analyzed were hatchery-reared based on the presence of a thermal mark. Twenty-two Chinook were aged age-3, 20 were age-4, and 2 were age-5. Two of the age-3 fish were Hurd Creek marked, 20 were Sol Duc marked, and no fish were wild-origin Elwha age-3 fish. Of the age-4 fish, 3 were marked at Hurd Creek, 7 were marked at Sol Duc, and 10 were wild-orgin Elwha fish. The two age-5 individuals did not have thermal marks, however, it is not known whether they were of hatchery origin because the 2003 brood year pre-dates the Elwha River thermal marking program.

In 2009, all age-3, age-4, and age-5 adult hatchery Chinook would be thermally marked. Of the 53 Chinook adult returns collected in 2009, 51 (96 percent) were of hatchery origin, and two adults were wild-orgin Elwha fish. Fifty-two Chinook were age- 4 and 1 was age-5. No age-3 fish were present in the sub-sample. Fifty-one of the age-4 fish were marked at Sol Duc, and one was wild-orgin Elwha fish. The one age-5 fish was of wild origin.

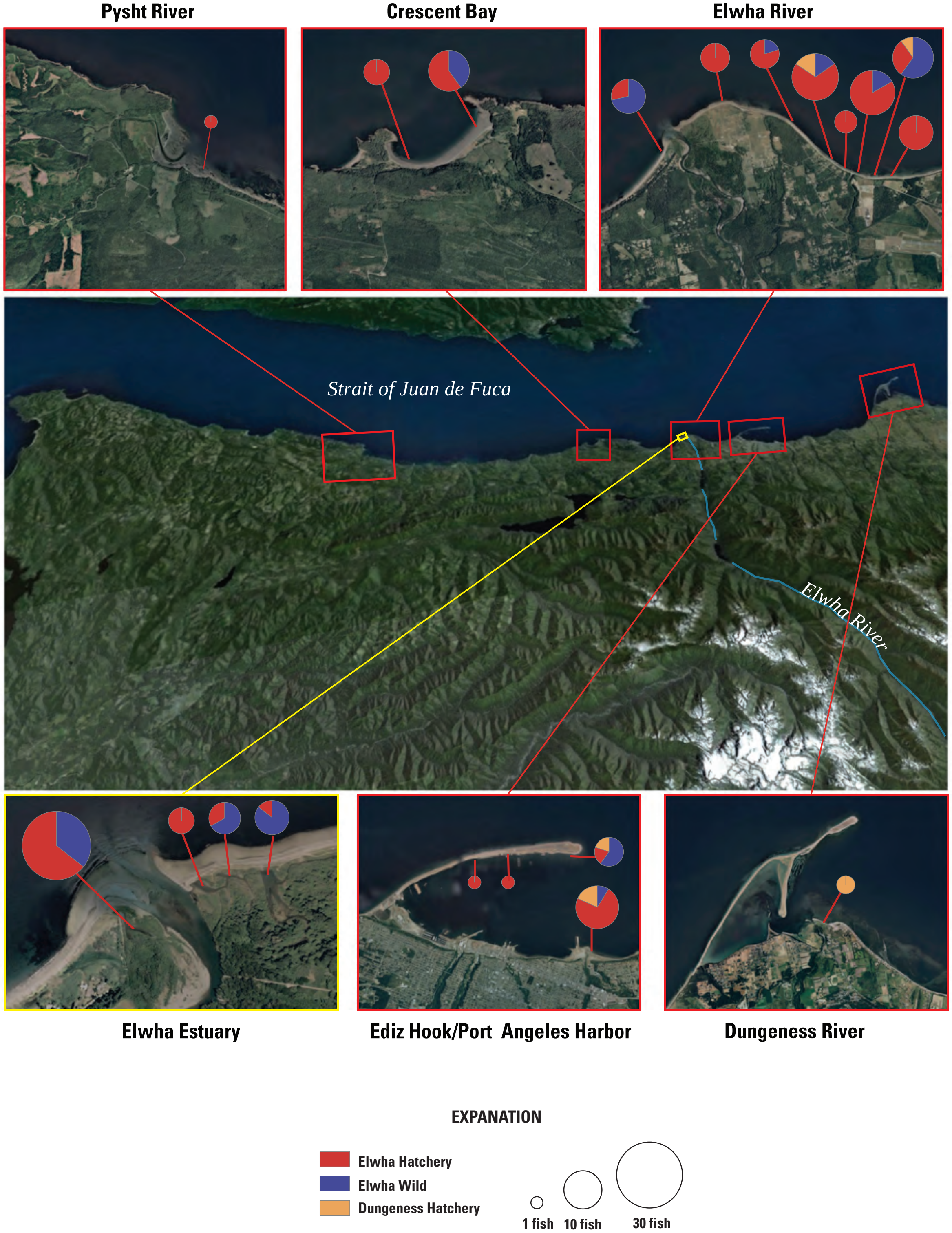


Figure 7.20. Aerial photographs and pie charts showing results of otolith analysis identifying origin of juvenile Chinook salmon captured during beach seining by Washington Department of Fish and Wildlife (Shaffer and others, 2009) and (K. Fresh, National Oceanic and Atmospheric Administration, unpub. data, 2007). (Origins were from Washington Department of Fish and Wildlife, Elwha, or Dungeness hatcheries and Elwha River wild-origin.) Pie charts depict the proportion of individuals from each location.

Combined returns from both collection years, showed only one wild adult (2009, age-4) corresponding to our 2 years of juvenile out-migrant sampling (2006–07), obviously too few samples to proceed with any life history or estuarine utilization analyses. Our preliminary work from 2008 and 2009 revealed that a significant part of the adult Chinook salmon naturally spawning in the Hunt Road Complex (outside of the index area that WDFW uses for spawner surveys in the lower Elwha River) were of hatchery origin. Combining both collection years, 87 percent of the adult Chinook otoliths examined were of hatchery origin (excluding two unknown origin age-5 fish from the 2008 collection).

Diet and Feeding Strategies of Elwha River Juvenile Salmon

Collection of Stomach Contents for Diet Analysis

Few studies of juvenile salmonid diet in estuarine environments of the central Strait of Juan de Fuca have been completed and no known studies are published for the Elwha River. The relatively static nature of structural features of the Elwha River estuary over recent decades (Warrick and others, 2009; Shafroth and others, 2011, chapter 8, this report) provides an ideal opportunity to determine the nature of juvenile salmon diets prior to the anticipated changes to these conditions following dam removal. Estuarine seining activities in 2006 and 2007 provided an opportunity to characterize the diet of juvenile salmonids in the estuary. A subsample of the juvenile salmonids detected with beach seining throughout the migration period were selected at random and then sampled for diet. The stomachs of fish sacrificed for otolith analysis were dissected. Some of these fish were collected in the Elwha River above tidal influence. For the purpose of this report, most diet analyses were conducted with juvenile Chinook salmon due to their higher relative abundance (hatchery and wild) and their value to other ongoing scientific studies in the Elwha River basin. Stomach content samples from 149 individuals were collected (99 in 2006 and 50 in 2007) for juvenile Chinook salmon, with 44 from the Elwha River and 105 from the estuary. Stomach content samples also were collected from juvenile coho, steelhead, and cutthroat trout from the Elwha River estuary, but samples were not in high enough numbers during the sampling period for comparative diet analysis.

Stomachs were flushed with gastric lavage (fig. 7.21), following standard techniques (Meehan and Miller, 1978; Cordell and others, 1999, 2001) for collection of stomach contents of juvenile salmonids. Across all sampling occasions, mean length of Chinook was 80 mm (standard deviation [SD] 21) and 88 mm (SD 16) in 2006 and 2007, respectively. Juvenile salmonids were anesthetized with tricaine methanesulfonate (MS-222) until lightly sedated. The stomach contents of the fish were then gently flushed using a 10 or 20 cc syringe filled with freshwater. Stomach contents were either strained through a 106 micron sieve or through a section of nylon attached to a PVC pipe. The stomach contents were preserved in 70 percent ethanol.

Figure 7.21. Gastric lavage of a juvenile Chinook salmon. Flushed contents of its stomach consist of various invertebrate taxa. (Photograph taken by Matt Beirne, Lower Elwha Klallam Tribe, July 13, 2006.)

Results of Diet Analysis for Elwha River Juvenile Chinook Salmon

The diets of juvenile salmonids (all species) in the Elwha River estuary were comprised of a diverse macroinvertebrate assemblage. In 2006, more than 1,100 macroinvertebrates, representing 23 higher level taxonomic groups (order or greater) were collected from stomach lavage and dissections of fish caught in the estuary and lower Elwha River. In 2007, 1,563 macroinvertebrates, representing 16 higher level taxonomic groups were collected in diet samples. When looking at Chinook salmon diets, most of higher taxonomic groups were detected in diets during both years. However, 12 macroinvertebrate orders were detected only in 2006 samples and 2 orders were unique to 2007 samples (table 7.8). When both years were combined, the average number of different taxa per stomach was similar between juvenile Chinook salmon captured in the river (mean = 4.48, SD = 2.5) and in the estuary (mean = 4.72, SD = 2.9) and the overall frequency of taxa richness per stomach was similar (fig. 7.22).

Table 7.8. Major macroinvertebrate types in stomach contents of juvenile Chinook salmon samples collected in the lower Elwha River, Washington, and its estuary complex, 2006 and 2007.

[Macroinvertebrate types are by order, unless indicated otherwise. Samples were collected in 2006 (99 individuals) and 2007 (50 individuals). Fewer unique taxa are represented in 2007 likely because samples were collected only in the estuary, whereas in 2006 samples collected in the lower Elwha River also were included]

Macroinvertebrate taxa	Percentage	
	2006	2007
Acarina (mites and ticks)	0.2	0
Amphipoda (amphipod crustaceans)	8.9	3.6
Aranaceae (spiders)	0.7	0
Chilopoda (Class; centipedes)	0.1	0
Coleoptera (beetles)	1.4	0.9
Collembola (springtails)	0.3	0.2
Diptera (true flies)	78.5	90.1
Ephemeroptera (mayflies)	4.6	0.2
Hemiptera (true bugs)	1.2	0.8
Hymenoptera (ants, wasps)	0.4	0.6
Lepidoptera (moths and butterflies)	0.2	0
Megaloptera (snakefly)	0.1	0
Malacostraca (Class; mysid shrimps)	0	0.9
Nematoda (Phylum; roundworms)	0.5	0.2
Nueroptera (brown lacewing)	0.1	0
Odonata (dragonfly and damselfly)	0.1	0.4
Oligochaeta (Class; aquatic worms)	0.1	0
Osteichthyes (Class; fishes)	0.1	0
Ostracoda (Class; seed shrimp)	0	0.3
Plecoptera (stoneflies)	0.9	0.3
Psocoptera (barkflies)	0.3	0.1
Thysanoptera (thrips)	0.5	0
Trichoptera (caddisfly)	1.0	0
Turbellaria (Class; flatworms)	0.1	0

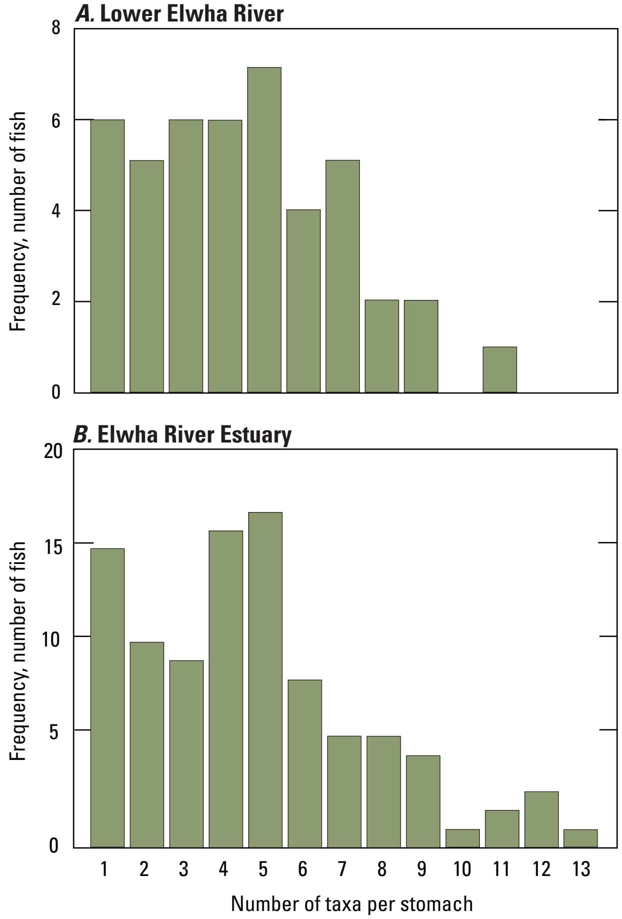


Figure 7.22. Frequency of taxa richness (number of unique taxa) in stomach contents collected from juvenile Chinook salmon captured in (*A*) the lower Elwha River above tidal influence and (*B*) the Elwha River estuary, Washington.

Of the 16 higher taxonomic groups of macroinvertebrates present in the diet, 10 were different orders of insects. Dipterans (2-winged flies), representing 11 families were the most abundant taxa, representing 80 and 88 percent of the total number of macroinvertebrates identified from Chinook salmon diet samples in 2006 and 2007, respectively. Amphipods, representing 14 and 5 percent, were the second most abundant taxonomic group in the Chinook diet in 2006 and 2007, respectively. Chironomids were the predominant family of dipterans represented in the diet of all salmonids, with Ceratopogonids, Empidids, and Sphaerocerids represented in smaller numbers (fig. 7.23). The life stage of consumed dipterans shifted from predominantly pupa in early summer to primarily adult forms in late summer, a pattern that was consistent between years (fig. 7.24). Chironomid larva were consumed at relatively low levels throughout spring and summer. Chironomid larvae provide a valuable food source for fish and other aquatic organisms due to the relatively high levels of protein and high digestibility (Armitage and others, 1995). Although caloric value varies with species, season, and life history, it usually ranges between 4.6 and 6.1 kcal g^{-1} (Armitage and others, 1995).

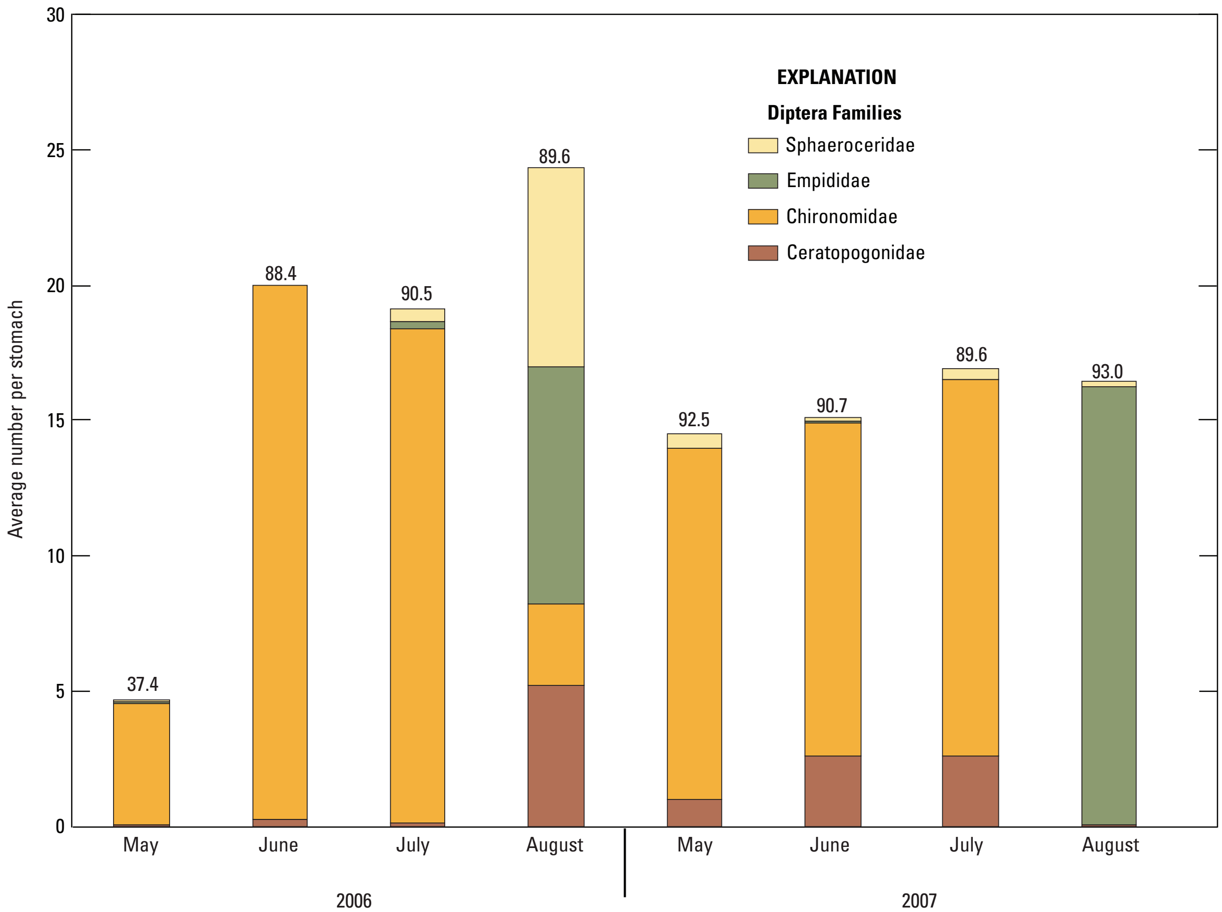


Figure 7.23. Average number of individuals per stomach of four main dipteran families in samples from juvenile Chinook salmon Elwha River estuary, Washington, during May–August 2006 and 2007.

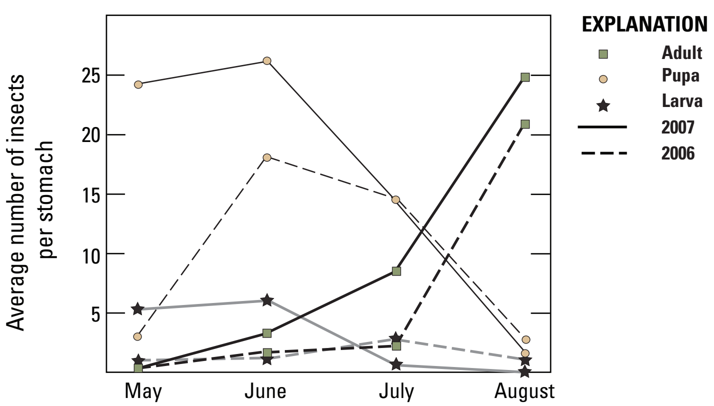


Figure 7.24. Average number of chironomids per stomach by life forms (adult, larva, pupa) for all sites in the lower Elwha River and its estuary during May–August 2006 and 2007.

Comparison of Diet Overlap Among Habitat Locations

Habitat-specific diet of juvenile Chinook salmon were examined by comparing the dietary overlap between samples collected from freshwater compared with estuary (2006) and the east and west parts of the estuary (2007). Schoener's index of diet overlap (Schoener 1970) is calculated as:

$$\alpha = 1 - 0.5\left(\sum_{i=1}^{n}\left|P_{xi} - P_{yi}\right|\right), \qquad (1)$$

where

P_{xi} is proportion of food category i in diet of juvenile Chinook salmon in habitat x,

P_{yi} is proportion of food category i in diet of juvenile Chinook salmon in habitat y,

n is number of food categories.

Values of α greater than or equal to 0.6 indicate significant dietary overlap. Standard diversity metrics, based on the orders of invertertebrates, also were calculated for comparisons. Dietary overlap between freshwater and estuary habitats was significant for March–May (α = 0.61) and June–September (α = 0.68) (table 7.9). In March–May, a higher number of orders was measured for juvenile Chinook salmon prey in freshwater samples with less diversity and evenness compared with the estuary samples. A higher proportion of dipterans was selected in freshwater (85 percent of total dietary items) than in estuary habitat (56 percent diptera and 38 percent amphipods). Interestingly, this pattern was inverse during June–September, as more dipterans were present in estuary diets (91 percent) than in freshwater diets (63 percent diptera and 23 percent mayflies, mostly of the genus *Baetis*). In 2007, a comparison of dietary overlap between the east and west estuary detected a high degree of overlap in March–May (α = 0.91) and June–September (α = 0.95).

Comparison of Prey Use versus Prey Availability in the Elwha River Estuary

As a vital requirement for animal growth, foraging is one of the primary factors in determining the fitness of an organism. Variables such as prey caloric value, handling time, and encounter rate all interact to affect food preference, with some theories suggesting that optimal foraging strategies are evolutionarily driven (Emlen, 1966). Most descriptive studies for juvenile Chinook salmon are limited to comparing the relative frequencies of prey taxa in the diet and in the environment. We examined patterns in prey use of juvenile Chinook salmon in using two approaches. The first was with modified Costello Diagrams (Costello, 1990; Amundsen and others, 1996), which graphically examines the proportion and frequency of occurrence of prey in the diet independent of prey availability. The second, Standard Forage Ratio (SFR; Manley 1974), constructs ratios of prey specific use and patterns of prey abundance in the environment.

Table 7.9. Summary statistics of diet composition (insect orders) and dietary overlap of Chinook salmon collected during two seasons (March–May and June–September) in the lower Elwha River (2006), Washington, and its estuary (2006 and 2007).

[Diet items for fish in each habitat type were tallied by order and the total number of orders present (S), the Shannon diversity (H' Prey), evenness (E Prey), and the sum of all prey individuals (N Prey) for all sampled stomachs (N) were calculated. Diet composition was compared between the lower Elwha River (multiple locations between RKms 0.5 and 0.0) and the East estuary in 2006 and between the East estuary and WESC in 2007 using Schoener's (1970) index of proportional similarity (α). **Dipt_Freq.:** frequency of dipteran occurrence. **Abbreviation:** FW, freshwater; WESC, west estuary channel]

Year	Months	α	Location	N	S Prey	H' Prey	E Prey	N Prey	Dipt_Freq. (percent)
2006	March–May	0.61	FW	17	13	0.76	0.30	306	79
			East estuary	19	9	0.95	0.43	322	94
	June–September	0.68	FW estuary	27	13	1.21	0.47	819	85
			East estuary	36	15	0.54	0.20	270	100
2007	March–May	0.91	WESC	16	4	0.30	0.22	342	75
			East estuary	12	10	0.58	0.25	151	83
	June–September	0.95	WESC	11	7	0.22	0.11	227	100
			East estuary	12	9	0.41	0.19	211	92

Modified Costello diagrams were used to interpret diet information of juvenile Chinook salmon collected in the freshwater and estuarine habitats of the Elwha River (fig. 7.25). This graphical method constructs a diagram of percentage of occurrence of each food item (unique taxon) on the ordinate and the prey-specific abundance on the abscissa. Prey-specific abundance is the proportional abundance of a particular taxon in only those fish stomachs in which it was present. Amundsen and others (1996) provided guidance for interpreting the distribution of prey items in the modified Costello diagram in terms of prey importance, feeding strategy, and niche contribution. The distribution of points along the prey-specific abundance axis indicates whether fish exhibit specialized (51–100 percent) or generalized (1–50 percent) feeding. A diagonal line from the origin to the upper right corner of the diagram indicates prey importance, as rare taxa are found in the lower left quadrant of the diagram and abundant taxa are found in the upper right quadrant of the diagram. The other diagonal, from the upper left to lower right, indicates the contribution of a prey taxon to the niche breadth of the predator. Points in the upper left quadrant indicate high selectivity among individual variability on the prey items consumed, which determines niche breadth, whereas points occurring in the lower right quadrant indicate generalized feeding on the same prey items across individuals (Amundsen and others, 1996; Vile and others, 2005).

Feeding strategies inferred from modified Costello diagrams (fig. 7.25) showed that juvenile Chinook salmon specialized on chironomids in the lower Elwha River and the eastern and western parts of the estuary. Chironomid pupa were particularly dominant in the estuary and to a lesser extent in the lower Elwha River where larval orthocladiinae and mayflies of the genus *Baetis* were also important. Most other prey species were rare and fed on in a generalized fashion by Chinook. The only other non-dipteran prey of any importance was the amphipod genus *Corophium* in the eastern Elwha estuary.

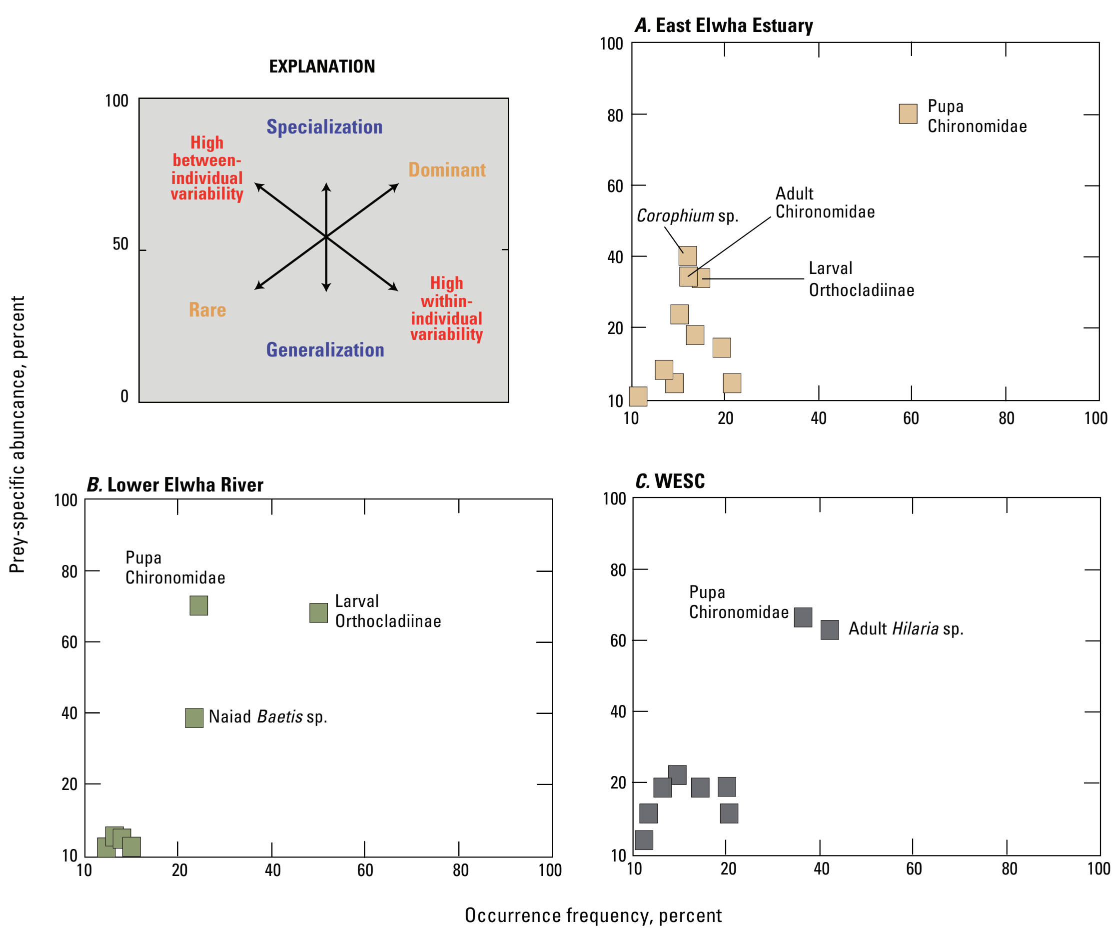


Figure 7.25. Modified Costello diagram showing feeding strategy of juvenile Chinook salmon in three habitats of the Elwha River, Washington. Diagrams modified from Costello, 1990, and Amundsen and others, 1996. Upper left panel shows three interpretive axes related to niche contribution prey importance (rare to dominant), and feeding strategy (generalization to specialization).

Prey use and prey availability patterns were estimated using SFR in the estuary where macroinvertebrate abundance in the diet as well as in the environment was sampled (fig. 7.26). Diet data from 2006 and 2007 was pooled; benthic and terrestrial (from shrub and littoral habitat) macroinvertebrate data were collected from the east and west estuary in 2007. Data were split into three seasons: early (March through May), mid (June through July) and late (August through September). Terrestrial and benthic prey availability data for the early and late seasons were collected in May and September 2007, respectively. For the mid season, benthic invertebrate samples collected in July and the average of terrestrial samples collected in June and July were used. The SFR for prey orders was calculated using the formula provided by Manley (1974):

$$P_i = \frac{\beta_i x_i}{\sum_{i=1}^{k} \beta_i x_i}, \tag{2}$$

where

β_i is proportion of prey in the diet,

x_i is proportion of prey in the environment,

i equals $1, 2...k$.

Values of P_i greater than 0.5 indicate specialization, whereas values less than 0.5 indicate that use is in proportion to availability.

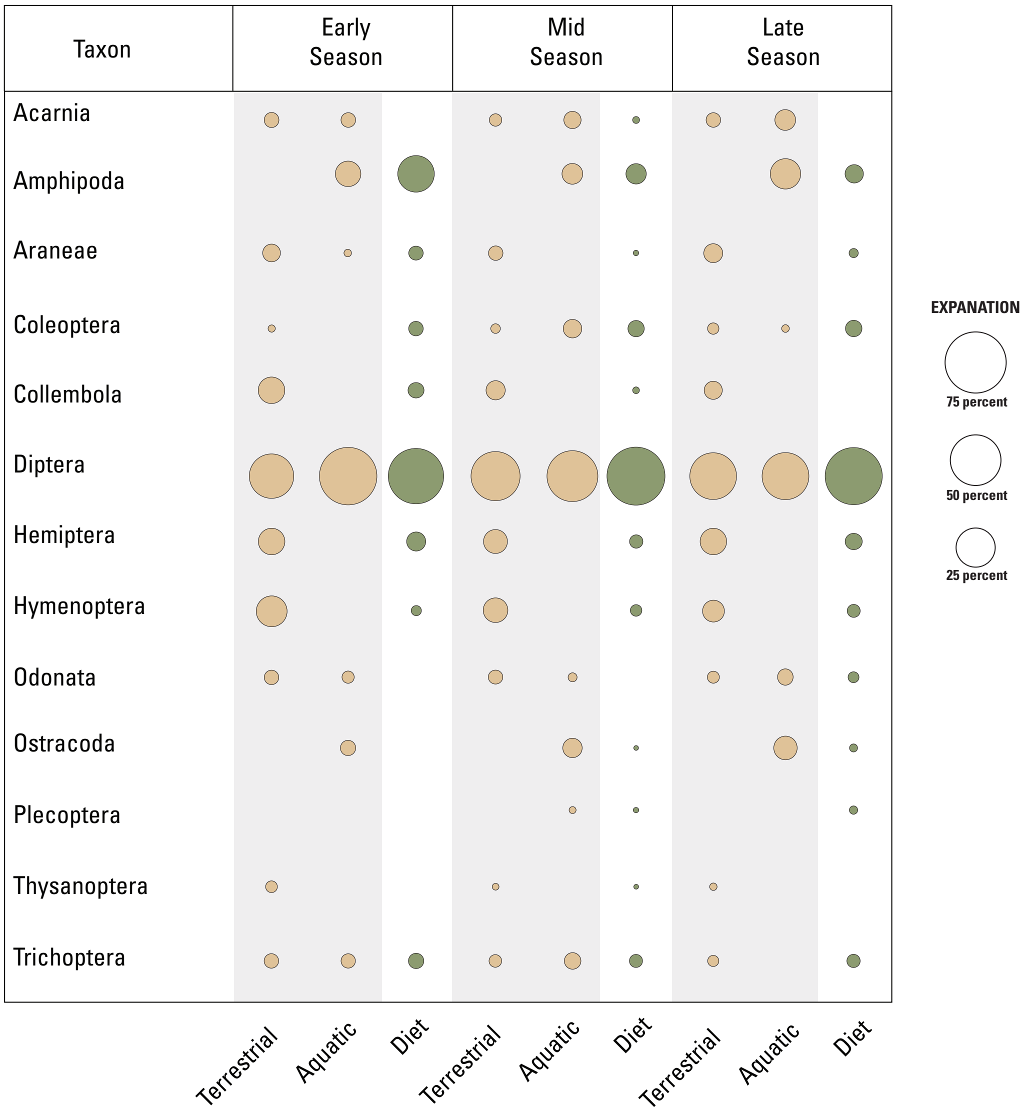


Figure 7.26. Bubble plots showing proportion of invertebrate taxa in fallout traps (terrestrial), sediment samples (aquatic), and diet of juvenile Chinook salmon captured in the lower Elwha River and the east estuary, Washington. Samples were pooled for fish collected in 2006 and 2007. Gastric lavage samples were standardized by number of invertebrates divided by the weight of each fish.

The SFR ratios indicate that the two main prey taxa, Amphipods and Dipterans, generally were not positively selected for, but rather were selected because they were the most abundant prey available (table 7.10). An exception was selectivity for Amphipods in the mid season (SFR = 0.5), although the calculated value was marginally positive. These results are in contrast to the interpretation of feeding strategy based on Costello diagrams, although the importance of Dipterans and Amphipods was apparent using both methods. Those cases that did show highly positive selectivity generally were those represented by few fish and therefore probably are not reliable representations of juvenile Chinook salmon preference. Coleopterans were positively selected for in all seasons and Trichoptera were positively selected for in the early and late seasons.

Other studies in estuarine environments in the Pacific Northwest also have determined that Dipterans, particularly chironomids, and Amphipods are important prey sources. Shreffler and others (1992) diet study of the Puyallup River estuary in south Puget Sound also determined that chironomids were the most abundant prey consumed by fall Chinook salmon, with Amphipods (particularly of the genus Corophium), Plecopterans, and Cladocerans as secondary prey. Grey and others (2002) also determined that Dipterans, Amphipods, and Trichopterans were of importance in control and restored sites of the Salmon River estuary in Oregon. Roegner and others (2004), working in various habitats of the Columbia River estuary, also found that chironomids and Amphipods were the dominant prey source for juvenile Chinook salmon.

A more comprehensive analysis, for example bioenergetic modeling (see Beauchamp, 2009; Cordell and others, 2011), would give a better estimate of how food availability, consumption, and environmental factors are playing a role in the growth efficiencies of juvenile Chinook salmon in the Elwha River and how those factors relate to the ongoing restoration efforts.

Table 7.10. Average Standard Forage Ratio values for juvenile Chinook salmon diet for individuals captured in the Elwha River estuary, Washington, in three seasons during 2006 and 2007.

[Standard Forage Ratio from Manley (1974). Values in **bold** indicate prey that were positively selected for by juvenile Chinook salmon in each season. Environmental values of prey availability based on data collected in 2007. Values could not be computed during a given season when a prey taxon was not present in the diet (a) or in the environment (b). **Abbreviations:** N, number of individuals captured; n, (standard deviation) number of fish with taxon present; na, not applicable]

Macoinvertebrate taxon	Early season (March–May) N = 20	Mid season (June–July) N = 58	Late season (August–September) N = 20
Acarina	a	0.24 (0.15, n = 2)	a
Amphipoda	0.39 (0.40, n = 13)	**0.50** (0.33, n = 20)	0.004 (0.001, n = 6)
Araneae	0.25 (0.12, n = 2)	**0.66** (0.08, n = 1)	0.10 (na, n = 1)
Coleoptera	**0.97** (0.01, n = 2)	**0.54** (0.24, n = 12)	**0.83** (0.11, n = 5)
Collembola	0.38 (0.35, n = 3)	0.32 (0.40, n = 2)	a
Diptera	0.35 (0.38, n = 18)	0.42 (0.42, n = 57)	0.42 (0.45, n = 20)
Hemiptera	0.42 (0.35, n = 5)	0.49 (0.31, n = 7)	0.22 (0.17, n = 5)
Hymenoptera	0.44 (na, n = 1)	0.40 (0.30, n = 6)	0.19 (0.18, n = 3)
Odonata	a	a	**0.77** (na, n = 1)
Ostracoda	a	0.28 (na, n = 1)	0.01 (na, n = 1)
Plecoptera	a, b	**1.0** (na, n = 1)	b
Thysanoptera	a	**0.97** (na, n = 1)	a
Trichoptera	**0.82** (0.08, n = 3)	0.30 (0.33, n = 6)	**0.95** (0.07, n = 3)

Summary

The capacity of the Elwha River estuary to support increased population sizes and life-history diversity of salmon populations will be determined by physical and biological factors, such as food sources, sediment deposition, temperature, salinity, and other habitat features. Our results showed that juvenile salmon use the Elwha River estuary for extended periods and show increased growth when they feed in the estuary. Chinook, coho, and chum salmon were the most common salmonids detected during 2 years of beach seining in the estuary. Fish assemblages in the Elwha River estuary varied between 2 years of sampling (2006 and 2007) likely due to storm-driven physical changes that precluded access to outmigrating juvenile salmon as well as marine fish movements in the estuary.

The differences in fish assemblage structure in the Elwha River estuary suggest that even modest volumes of sediment may create barriers that preclude fish access to critical estuarine habitat. This has management implications in the context of Elwha River restoration that may warrant a more active approach to maintaining fish access during periods of heavy sediment deposition following dam removal. Adaptive management planning, already underway for the lower main stem, would be well informed by taking into consideration the potential sediment impacts on fish access to estuarine habitat. Although significantly elevated turbidity levels are anticipated in the months to years following dam removal, the magnitude of these levels is difficult to anticipate, particularly in the Elwha River estuary. The extent to which elevated post-dam removal turbidity levels in the estuary may adversely affect juvenile salmonid life history and feeding efficiency also is uncertain. Simenstad and others (1982) postulated that naturally elevated turbidity in estuaries during periods of typical juvenile salmonid residence might reduce predator efficiency and benefit their survival. The concentration threshold at which the benefits of elevated turbidity are replaced by adverse life history effects is uncertain and will be a potential subject of study during the dam removal and restoration process.

The effects of dam removal in the Elwha River estuary, like that of the entire river, is likely to be complex, with ecological processes and functions changing across multiple spatial and temporal scales. In the near term, high sediment loads due to erosion of delta deposits in the two reservoirs could have a negative impact on many habitats of the river and estuary, which could change the carrying capacity for species across trophic levels. Conversely, in the long term, as salmon recolonize the watershed and populations rebuild, greater life-history diversity of salmonids will be a key component of successful recolonization and the maintenance of self-sustaining populations (Schindler and others, 2010).

Future studies of the Elwha River estuary, during and following dam removal, should examine the effects of physical changes due to sediment. Macroinvertebrate and fish communities are likely to respond if sediment deposition has large effects on physical factors such as substrate composition and turbidity. Therefore, collection of complementary data to that presented herein would allow examination of potential affects to these communities. Additional data collection, such as sediment traps and turbidity sensors, would help explain the level of changes experienced by biological communities in the estuary. Repeating many of the data collection efforts described in this chapter will be essential in documenting the effects of dam removal and salmon recolonization on the Elwha River estuary.

Acknowledgments

This project involved the coordination with and assistance from researchers from several institutions, including state and Federal agencies. We acknowledge and are grateful for the assistance of Anne Shaffer (Washington Department of Fish and Wildlife [WDFW]) who assisted with seining activities and collected valuable complementary data, and provided specimens for otolith analysis. Kurt Fresh, Anna Kagley, and Joshua Chamberlin (National Oceanic and Atmospheric Administration) also assisted by contributing samples for otolith analysis. Jeff Grimm (WDFW) provided reference otolith samples. Scott Williams and staff of the Dungeness hatchery provided assistance with obtaining hatchery samples. We would like to thank the following individuals for assisting with field work: Ernest “Sonny” Sampson, Larry Ward, Mel Elofson, Mitch Boyd, Raymond Moses (Lower Elwha Klallam Tribe); Rebecca Paradis, Bryan Hara, David Kirner, Jimmy McDougall, Brian Matne, and Michelle Surrette (Peninsula College); Dorothy Chase, Christopher Konrad, Cecile Mathieu, Steve Rubin, Guy Gelfenbaum, and Jon Warrick (U.S. Geological Survey). Database management and mapping were provided by Randall McCoy (Lower Elwha Klallam Tribe) and Theresa Olson (USGS). Special thanks are due to Arthur Frost for completing all of the macroinvertebrate taxonomic identification. This chapter benefitted from discussions with Rich Sheibley (USGS), Jeff Cordell (University of Washington), and Tom Friesen (Oregon Department of Fish and Wildlife). Funding was provided by the USGS Multi-disciplinary Coastal Habitats in Puget Sound (MD-CHIPS) program and from a grant to the Lower Elwha Klallam Tribe from the Environmental Protection Agency. Use of trade names is for convenience of the reader and does not constitute endorsement by the U.S. government.

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Chapter 8

Vegetation of the Elwha River Estuary

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Abstract

The Elwha River estuary supports one of the most diverse coastal wetland complexes yet described in the Salish Sea region, in terms of vegetation types and plant species richness. Using a combination of aerial imagery and vegetation plot sampling, we identified 6 primary vegetation types and 121 plant species in a 39.7 ha area. Most of the estuary is dominated by woody vegetation types, with mixed riparian forest being the most abundant (20 ha), followed by riparian shrub (6.3 ha) and willow-alder forest (3.9 ha). The shrub-emergent marsh transition vegetation type was fourth most abundant (2.2 ha), followed by minor amounts of dunegrass (1.75 ha) and emergent marsh (0.2 ha). This chapter documents the abundance, distribution, and floristics of these six vegetation types, including plant species richness, life form, species origin (native or introduced), and species wetland indicator status. These data will serve as a baseline to which future changes can be compared, following the impending removal of Glines Canyon and Elwha Dams upstream on the Elwha River. Dam removals may alter many of the processes, materials, and biotic interactions that influence the estuary plant communities, including hydrology, salinity, sediment and wood transport, nutrients, and plant-microbe interactions.

Introduction

Unique wetland complexes occur where river mouths meet the sea. The vegetation of these areas ranges from riparian habitats that differ little from habitats associated with the river upstream to salt marshes that have a distribution limited to tidally influenced areas with elevated water salinity. The processes that drive the distribution and dynamics of plant communities in river estuary wetland complexes are also varied, ranging from fluvial-dominated processes such as flooding and channel migration to marine processes such as tidally driven water-level fluctuations. Patterns of seawater intrusion influence salinity levels in estuarine water and soils and can be a strong determinant of plant distributions (Mitsch and Gosselink, 1993; Keddy, 2000). At the scale of an entire river-mouth wetland complex, these conditions and processes can produce multiple plant communities with high beta diversity and high species richness (Keddy, 2000).

River mouth wetland complexes of the Salish Sea region (which extends from Olympia, Washington, to Canada's Desolation Sound) are relatively uncommon and small in area (Collins and Sheikh, 2005; Todd and others, 2006). Many of these systems have been modified by human activities such as channelization, large wood removal, and levee and dike construction, which typically reduce wetland extent and alter key processes such as tidal inundation (Thom and others, 2002; Todd and others, 2006; Brennan, 2007). By one estimate, Puget Sound tidal wetlands have been reduced to about 19 percent of their historical area, from approximately 29,500 ha to approximately 5,650 ha (Collins and Sheikh, 2005).

The Elwha River delta and estuary are relatively small and have been subject to alteration by the construction and operation of two large dams upstream that have significantly reduced sediment and wood transport through the estuary. Dike and levee construction also have constrained channel migration (Kloehn and others, 2008; Draut and others, 2011; Warrick and others, 2011, chapter 4, this report) and reduced the tidally influenced area (Duda and others, 2011, chapter 1, this report; Magirl and others, 2011, chapter 4, this report).

Removal of the two dams upstream on the Elwha River, anticipated to begin in 2011, will release and transport coarse woody debris and much of the approximately 19 million m^3 of sediment stored behind the dams (Bountry and others, 2010; Czuba and others, 2011, chapter 2, this report). This influx of sediment might directly influence estuarine plant communities through deposition of fine sediments, for example, or might indirectly affect these areas by inducing higher rates of channel change (Shafroth and others, 2002; Hood, 2010) and by altering the transport of sediment in the Strait of Juan de Fuca and adjacent beaches and beach berms (Warrick and others, 2011, chapter 3, this report).

This study is part of a U.S. Geological Survey (USGS) and Lower Elwha Klallam Tribe multi-disciplinary effort to characterize the ecosystems of the Elwha River estuary before dam removal. Our objectives were to identify the principal vegetation types within the estuary, estimate their areal extent, and characterize their structure and species composition, including species richness. We further elucidate floristic patterns by comparing the distribution of life forms, native versus introduced species, and the wetland indicator status among the primary vegetation types in the Elwha River mouth/estuary wetland complex.

Methods

Patch Delineation and Vegetation Plot Selection

In August 2007, polygons were delineated representing distinct vegetation patches visible on 1 m resolution, 2006 National Agriculture Imagery Program (NAIP) imagery within the study area. The study area was then ground-truthed to refine the polygon boundaries and to classify each polygon as one of six vegetation types: mixed riparian forest (labeled "Riparian forest" in figures), willow-alder forest, riparian shrub, shrub-emergent marsh transition (shrub-marsh transition), emergent marsh, and dunegrass (fig. 8.1, table 8.1). Figure 8.1 shows the polygons and plot locations, overlayed on a more recent aerial photograph (April 2008), which better depicts current conditions. Table 8.1 also provides the classification of these vegetation types according to Cowardin and others (1979). Next, random x-y coordinates were generated to identify plot locations. At least three plots were sampled in each of our original six vegetation types. For two of the more extensive vegetation types (riparian forest and riparian shrub) five plots were sampled. A total of 22 plots were sampled.

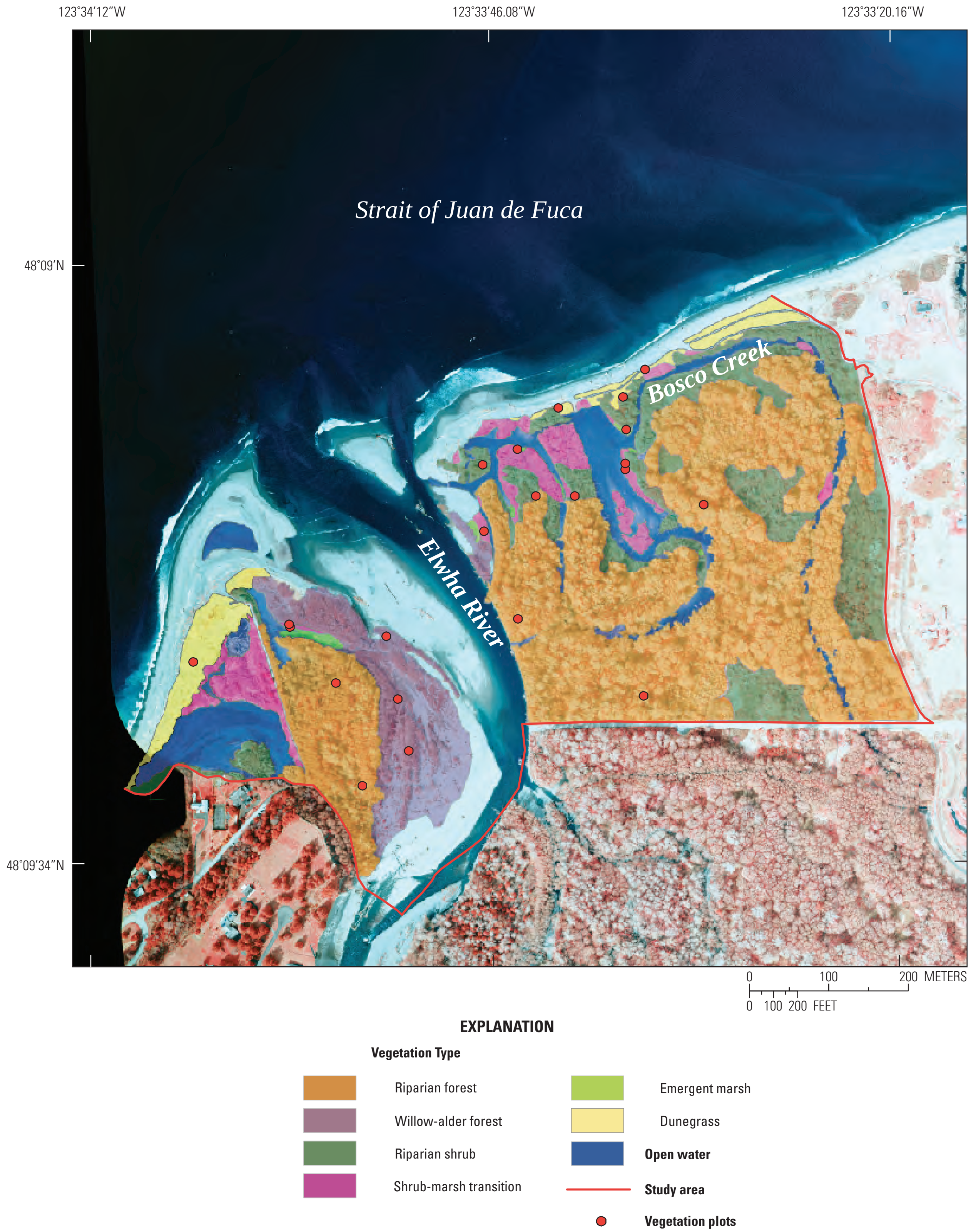

Figure 8.1. Study area showing six vegetation patch types, gravel bars, open water, and vegetation plot locations, Elwha River estuary, Washington.

Table 8.1. Vegetation types named in this study in relation to the Cowardin and others (1979) classification used by the U.S. Fish and Wildlife National Wetlands Inventory (NWI).

Elwha vegetation type (this study)	Cowardin classification	Comments
Mixed riparian forest	PFO1A, Palustrine, Forested, Broad-leaved deciduous, Temporarily flooded	Many of these areas appear to be classified as non-wetland by the NWI. Areas classified as wetland that overlap this vegetation type in our study typically fall into the PFO1A category.
Willow-alder forest	E2USP, Estuarine, Intertidal, Unconsolidated shore, Regularly flooded	These early successional areas conform to areas currently mapped as estuarine, intertidal by the NWI, but due to vegetation encroachment would likely now be classified as an estuarine or palustrine forested, depending on the amount of tidal influence.
Riparian shrub; Shrub-emergent marsh transition	PSS1C, PSS1Ch, PSS1R, Palustrine, Scrub shrub, Broad-leaved deciduous, Seasonally flooded, Seasonally-flooded-impounded or Seasonal-tidal flooded or PF01A (see above)	Much of the area classified as riparian shrub is not classified by the NWI. Classified areas that do overlap with these vegetation types typically fall into the palustrine, scrub-shrub, broad-leaved deciduous categories with variable flooding regimes. Some areas are also classified as palustrine, forested, broad- leaved deciduous.
Emergent marsh	PEM1A, Palustrine, Emergent, Persistent, Temporarily flooded	These areas are often found in the ecotone from open water (estuarine or palustrine) to drier areas (riparian shrub and mixed riparian forest)
Dunegrass	M2USN, Marine, Intertidal, Unconsolidated shore, Regularly flooded	Dunegrass dominated berms adjacent to the shore either fall into this category or are considered “upland” and not mapped by the NWI.

Vegetation Plot Sampling

At each plot location, a 100 m^2 plot was established. Each vascular plant present in a plot was assigned to 1 of 10 cover classes: trace, 0–1 percent, 1–2 percent, 2–5 percent, 5–10 percent, 10–25 percent, 25–50 percent, 50–75 percent, 75–95 percent, and 95–100 percent. Because individual plants and vegetation strata commonly overlapped, total plant cover values within a plot commonly exceeded 100 percent. In most cases, plot dimensions were 10 × 10 m, but where the vegetation patch was narrow, plot dimensions were sometimes 4 × 25 m or 5 × 20 m. With these data, the total number of species across all of our plots were tallied (estuary-scale species richness), as were the total number of species per vegetation type (community-scale species richness), and the average number of species per vegetation type (plot-scale species richness).

In August 2008, the mixed riparian forest and willow-alder plots were revisited, and tree cover by species and the diameter of all trees (stems greater than 2.5 cm at breast height) were measured in 0.1 ha (for mixed riparian forest) or 0.01 ha (for willow-alder forest) plots. Tree stem density and basal area were calculated with the diameter data.

Floristic Categories

Each species was identified by its life form (or habit): graminoid, forb, shrub, or tree, and whether it was native or introduced was noted (per the U.S. Department of Agriculture [USDA] Plants database, http://plants.usda.gov/; and the University of Washington Herbarium, Washington Flora Checklist, http://biology.burke.washington.edu/herbarium/waflora/checklist.php). If the USDA Plants database indicated that a species can have multiple growth forms (for example, a tree or shrub form), the species was categorized as having the larger of the two forms (for example, a tree rather than a shrub). Species richness patterns are summarized by life form, species origin (native or introduced), and wetland indicator status (for those species for which this information was available per the USDA Plants database; http://plants.usda.gov/). Wetland indicator status is a five-point scale that estimates the probability that a plant occurs in wetlands: Obligate wetland species = 1; Facultative wetland species = 2; Facultative = 3; Facultative upland species = 4; Upland = 5. The relative cover of plants by life form and native versus introduced status are also summarized, and wetland indicator status weighted by the relative cover of each species is examined.

Vegetation Plot Classification

To assess the extent to which the plant community data from the plots was consistent with the subjective classification of the six vegetation types (based on the aerial photography and ground-truthing, above), a nonmetric multidimensional scaling (nMDS) ordination of the vegetation plot data was run using Primer, version 6.0 (Clarke and Gorley, 2006). The ordination was derived from a Bray-Curtis similarity matrix generated from square-root transformed percent cover (mid-point of cover class, by species) values for each species in each plot.

Vegetation Plot Elevation Surveys

To explore the relation between plot elevation and vegetation type, the elevation (above sea level) of each plot was estimated using two methods. In September 2009, 18 of the plot locations were surveyed with a Magellan ProMark 3 Differential Global Positioning System operating in Real-Time Kinematic mode (RTK-DGPS), mounted on a survey pole, receiving corrections from a base station on a permanent survey monument. The estimated systematic and random error (combined) in the vertical and horizontal dimensions was ±10 cm. Points could not be captured with the RTK-DGPS system for four of the plots due to canopy density. For these, plot elevations were estimated from a digital elevation model produced from a lidar flight in April 2009, which had an estimated absolute vertical error of ±30 cm (Entrix, Inc., 2009).

Elwha Estuary Vegetation Sampling Results

Vegetation Plot Classification

The six vegetation types that were identified to initially stratify the vegetation of the Elwha estuary (fig. 8.1) corresponded well to plot groupings revealed by the non-metric multi-dimensional scaling ordination (fig. 8.2). The final stress value for the two-dimensional analysis was 0.16, a value sufficiently low (that is, less than 0.2) to indicate a useful summary of multivariate pattern (Clarke and Warwick, 2001). Given the consistency between the classifications based on plot data and field mapping, these six vegetation types were retained for both the estuary-scale and community- and plot-scale descriptions. Here, the vegetation types are briefly described, and then compared with respect to various floristic metrics.

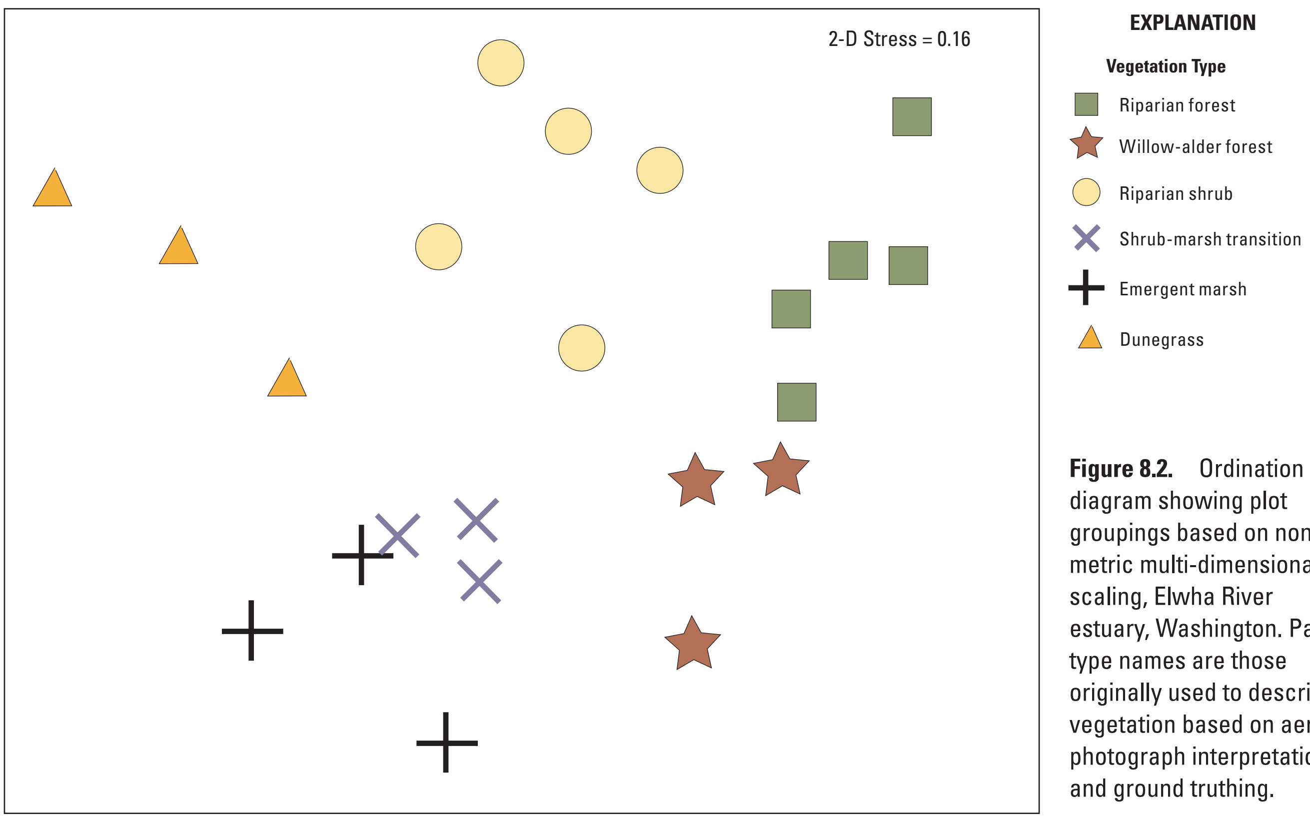

Figure 8.2. Ordination diagram showing plot groupings based on non-metric multi-dimensional scaling, Elwha River estuary, Washington. Patch type names are those originally used to describe vegetation based on aerial photograph interpretation and ground truthing.

Mixed Riparian Forest

The mixed riparian forest vegetation type is the most extensive in the study area, occupying approximately 20 ha in April 2008 (fig. 8.1). In the Elwha River estuary, mixed riparian forests can be dominated by various deciduous species, such as *Acer macrophyllum* (bigleaf maple), *Populus balsamifera* ssp. *trichocarpa* (black cottonwood), *Alnus rubra* (red alder) in the overstory, with *Sambucus racemosa* (red elderberry), *Rubus spectabilis* (salmonberry), and *Oemleria cerasiformis* (Indian plum) common in the understory (fig. 8.3). One conifer stand was sampled with *Pseudotsuga menziesii* (Douglas-fir) in the overstory. *Thuja plicata* (western redcedar), *Abies grandis* (grand fir) and *Picea sitchensis* (Sitka spruce) are other conifer species that occur within the study area. Some of these mixed riparian forest patches are relatively old (probably more than100 years) and often contain large trees (up to 163.5 cm in diameter within our sample plots). Mean ± standard deviation of tree stem density, basal area, and total percent cover of vascular plants were 676±125 stems/ha; 33.6±20.4 m^2/ha, and total percent cover was 192.4±36.7, respectively.

Figure 8.3. Mixed riparian forest vegetation type in the Elwha River estuary, Washington. (Photograph taken by Thomas O. Bates, formerly with ASRC Management Services, July 9, 2006.)

Willow-Alder Forest

Young willow-alder forests are early successional communities that occupy recently deposited gravel bars, typically close to the active river channel (fig 8.4). River channel migration over the past seven decades eroded many of the willow-alder forests on the east side of the channel but also deposited new gravel bars on the west side where several patches of this vegetation type existed during our study (Draut and others, 2008; fig. 8.1), occupying approximately 3.9 ha in April 2008 (fig. 8.1). *Alnus rubra* had the highest mean cover (61.5 percent) across the three plots sampled in this patch type. *Salix sitchensis* (Sitka willow; 23.3 percent mean cover) dominated one of the three plots, and *S. hookeriana* (Hooker's willow; 6.3 percent mean cover) was subdominant in one of the plots. Mean ± standard deviation of tree stem density in willow-alder stands was 2,767±2,450 stems/ha, mean ± standard deviation of basal area was 24.0±19.6 m^2/ha, total plant cover was 114.8±43.3 percent.

Figure 8.4. Willow-alder forest vegetation type growing on a gravel bar in the Elwha River estuary, Washington. (Photograph taken by Patrick B. Shafroth, U.S. Geological Survey, July 16, 2006.)

Riparian Shrub

Riparian shrub communities are characterized by high shrub cover and can occur in different contexts within the estuary. For example, along the margins of the estuarine water bodies and Bosco Creek, riparian shrub communities typically occurred either adjacent to the water or a short distance from the water if a strip of emergent vegetation was present. In places, they also occurred along a topographic gradient from water's edge to the riparian forest, between the shrub-emergent marsh transition community and the mixed riparian forest community. Finally, riparian shrub patches can be interspersed with mixed riparian forests (fig. 8.1). The riparian shrub vegetation type occupied approximately 6.3 ha in April 2008. Several shrubs were relatively abundant in the sampled plots: *Rosa nutkana* (Nutka rose), *Rosa pisocarpa* (clustered wild rose), *Crataegus douglasii* (black hawthorne; fig. 8.5), *Lonicera involucrata* (black twinberry), *Malus fusca* (Pacific crab apple), *Oemleria cerasiformis* (Indian-plum), *Rubus spectabilis* (salmonberry), and *Symphoricarpos albus* (common snowberry). Total plant cover in riparian shrub plots was 128.0±40.8 percent (mean ± standard deviation).

Figure 8.5. Riparian shrub vegetation type within the Elwha River estuary, Washington, study area. (Photograph taken by Tracy L. Fuentes, U.S. Geological Survey, August 18, 2007.)

Shrub-Emergent Marsh Transition

Shrub-emergent marsh transition vegetation usually occurs between the narrow bands of emergent marsh vegetation that are subject to regular and relatively large, tidally driven water-level fluctuations, and patches of riparian shrub or riparian forest that typically are inundated by river flooding (fig. 8.6). The shrub-emergent marsh transition zone is apparently inundated by tidal waters, but to relatively shallow depths for relatively short durations. This vegetation type occupied approximately 2.2 ha in April 2008. Vegetation is characterized by high total cover, predominantly herbaceous. The forb *Argentina egedii* (Pacific silverweed) dominated vegetative cover in the three plots sampled in this patch type (average of approximately 45 percent). Woody plant cover is comprised largely of scattered, relatively small *Salix sitchensis* and *Alnus rubra* individuals. Several emergent marsh species are also present in these plots, including *Carex* spp. (sedges), *Juncus* spp. (rushes), *Eleocharis palustris* (creeping spike-rush), and *Typha latifolia* (cattail). Total plant cover in shrub-emergent marsh transition plots was 161.0±58.4 percent (mean ± standard deviation).

Figure 8.6. Shrub-emergent marsh transition vegetation type within the Elwha River estuary, Washington, study area. (Photograph taken by Tracy L. Fuentes, U.S. Geological Survey, August 17, 2007.)

Emergent Marsh

In the Elwha River estuary, patches of emergent marsh vegetation are typically limited to narrow bands (less than 5 m) along the margins of the estuarine ponds (fig. 8.7). They occupy the smallest area of any of the vegetation types—approximately 0.2 ha in April 2008 (fig. 8.1). Sediment particle size is usually fine and plants are subject to substantial tidally driven water-level fluctuations (Magirl and others, 2011, chapter 4, this report). Individual patches are often relatively species poor, with high dominance by one or two species. Abundant species in the three plots sampled within this vegetation type included *Juncus arcticus* (arctic rush), *Argentina egedii*, *Carex obnupta* (slough sedge), *Rumex salicifolius* (willow dock), *Eleocharis palustris*, and *Typha latifolia*. Total plant cover was 118.7±30.6 in the emergent marsh plots (mean ± standard deviation).

Figure 8.7. Emergent marsh vegetation bordering open water in the Elwha River estuary, Washington. (Photograph taken by Patrick B. Shafroth, U.S. Geological Survey, July 20, 2006.)

Dunegrass

Dunegrass communities occur at relatively high topographic positions on or adjacent to beach berms. They occupied approximately 1.75 ha in April 2008 (fig. 8.1). These communities are strongly dominated by *Leymus mollis* (American dunegrass), which averaged more than 75 percent cover in the three plots sampled (fig. 8.8). Forb diversity was high in the dunegrass communities (see below), and some forbs had relatively high cover, such as *Ambrosia chamissonis* (silver bur ragweed), *Lathyrus japonicus* (beach pea), *Rumex salicifolius*, *Senecio sylvaticus* (woodland ragwort), and *Solidago canadensis* (Canada goldenrod). Total plant cover in the three plots sampled was 162.4 ± 89.8 percent (mean ± standard deviation).

Figure 8.8. Dunegrass vegetation type in the Elwha River estuary, Washington. (Photograph taken by Tracy L. Fuentes, U.S. Geological Survey, August 17, 2007.)

Floristics and Species Richness

A total of 121 unique vascular plant taxa in 46 families were identified within our 22 sample plots (table 8.2). Six of these taxa were identifiable only to the genus level, but were distinct from all other plants we identified. Given this, these taxa are considered "species" in the context of calculating richness values hereafter. The riparian shrub and shrub-emergent marsh transition vegetation types had the highest total richness (52 species), while the emergent marsh vegetation type had the lowest total richness (31 species; fig. 8.9). Plot-scale species richness differed in similar ways as total richness across the six vegetation types (fig. 8.9). The shrub-emergent marsh transition and mixed riparian forest vegetation types contained the most unique species (not found in any other vegetation type; 12 and 13 species, respectively); the willow-alder forest vegetation type contained only one unique species (fig. 8.9). The greater number of plots in the mixed riparian forest and riparian shrub vegetation types could bias the comparisons of total species richness and number of unique species.

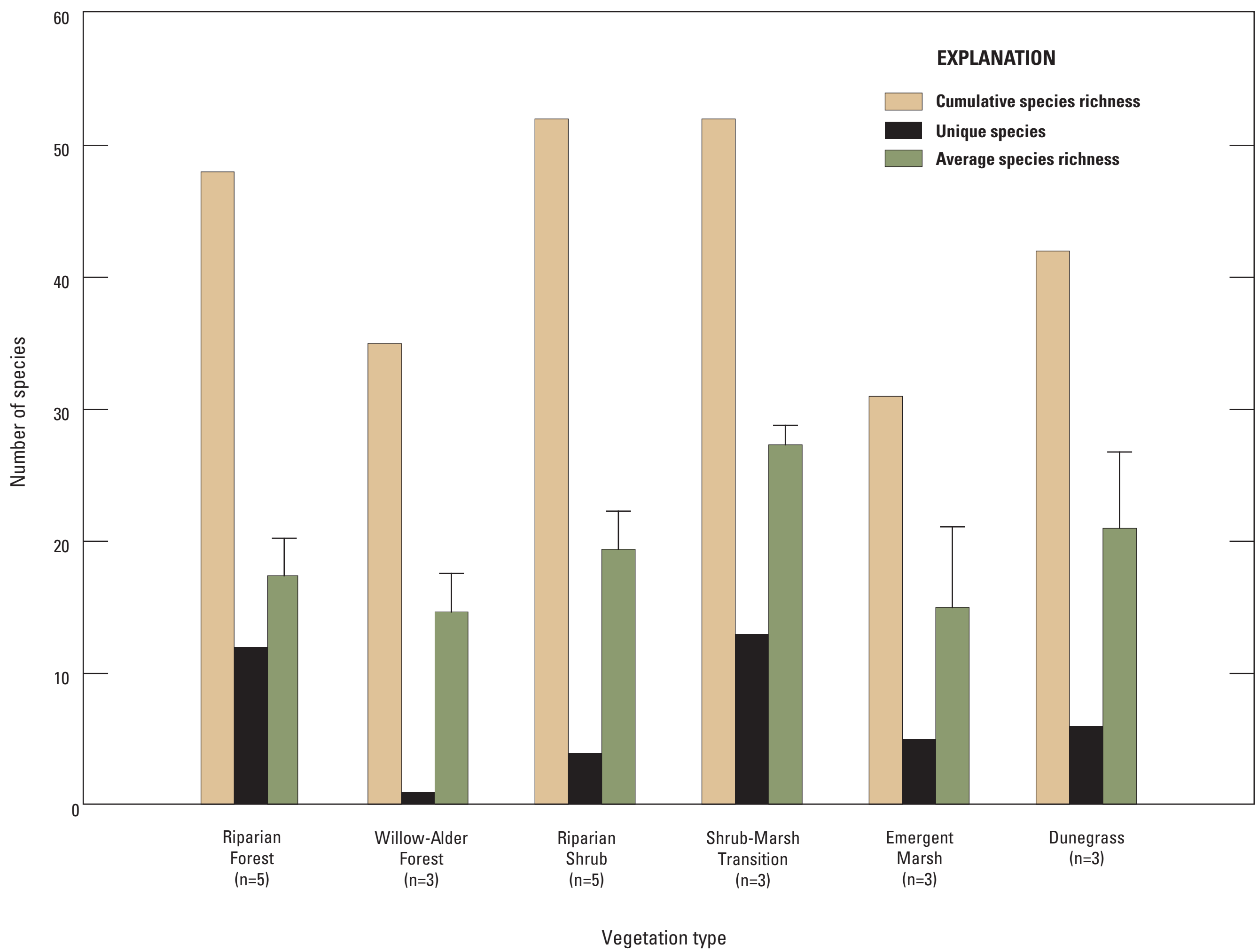

Figure 8.9. Species richness in the six vegetation types described for the Elwha River estuary, Washington. Light brown bars indicate the total number of species among all plots within a given vegetation type (cumulative species richness). Black bars indicate the number of species that are only in a given vegetation type (unique species). Green bars indicate the mean + standard error (black whisker) number of species (average species richness) per plot, by vegetation type. Number of plots sampled per vegetation type is indicated in parentheses beneath the x-axis labels. The area sampled is equal to the number of plots sampled times 100 square meters

Table 8.2. Characteristics of 121 unique plant taxa identified in 22 vegetation plots sampled in the Elwha River estuary, Washington, August 2007.

[Nomenclature for scientific names and plant family is based on the U.S. Department of Agriculture (USDA) Plants database (http://plants.usda.gov/). An "X" in a row beneath one of the six vegetation community column headings indicates that the plant was present in at least one plot sampled in that community. **Life form:** T, tree; F, forb; S, shrub; V, vine; SS, subshrub; G, graminoid. **Origin:** N, native plant; I, introduced plant; N/A, information not available **Noxious weeds:** Plants designated as noxious weeds by the Washington State Noxious Weed Board (2011) are labeled either B, (class B noxious weeds are nonnative species whose distribution is limited to parts of Washington State) or C, (Class C noxious weeds that already are widespread in Washington or are of special interest to the State's agricultural industry). **Wetland indicator status:** Based on information from USDA Plants database. Wetland indicator status is a five-point scale that estimates the probability that a plant occurs in wetlands compared to non-wetlands (OBL, obligate wetland species; FACW, faculative wetland species; FAC, FAC+, FAC-, faculative; FACU, faculative upland species; UPL, upland]

Scientific name (genus and species)	Family	Mixed riparian forest	Willow alder forest	Riparian shrub	Shrub-marsh transition	Emergent marsh	Dunegrass	Life form	Origin	Noxious weeds	Wetland indicator status
Acer macrophyllum	Aceraceae	X	X	X				T	N		FACU
Alisma plantago-aquatica	Alismataceae					X		F	I		OBL
Daucus carota	Apiaceae				X			F	I	B	N/A
Oenanthe sarmentosa	Apiaceae			X		X		F	N		OBL
Hedera helix	Araliaceae			X				S/V	I	C	N/A
Achillea millefolium	Asteraceae		X	X	X		X	F	N		FACU
Adenocaulon bicolor	Asteraceae	X						F	N		N/A
Ambrosia chamissonis	Asteraceae			X			X	F	N		N/A
Anaphalis margaritacea	Asteraceae				X			F	N		N/A
Cirsium arvense	Asteraceae				X		X	F	I	C	FACU+
Cirsium vulgare	Asteraceae				X	X	X	F	I	C	FACU
Erigeron sp.	Asteraceae				X			F	N/A		N/A
Hypochaeris radicata	Asteraceae		X	X	X		X	F	I	C	FACU
Lapsana communis	Asteraceae	X	X					F	I		FAC
Leucanthemum vulgare	Asteraceae	X	X	X	X		X	F	I	C	N/A
Mycelis muralis	Asteraceae	X	X					F	I		N/A
Senecio jacobaea	Asteraceae					X	X	F	I	B	FACU
Senecio sylvaticus	Asteraceae			X			X	F	I		N/A
Solidago canadensis	Asteraceae			X	X		X	F	N		FACU
Sonchus oleraceus	Asteraceae						X	F	I		UPL
Taraxacum officinale	Asteraceae		X	X				F	I		FACU
Alnus rubra	Betulaceae	X	X	X	X			T	N		FAC
Myosotis scorpioides	Boraginaceae				X	X		F	I		FACW
Cakile maritima	Brassicaceae						X	F	I		FACU
Lepidium campestre	Brassicaceae				X		X	F	I		N/A
Lepidium virginicum	Brassicaceae						X	F	N		FACU
Lonicera ciliosa	Caprifoliaceae	X						S	N		FAC+
Lonicera involucrata	Caprifoliaceae		X	X	X	X	X	S	N		FAC+
Sambucus racemosa	Caprifoliaceae	X						T/S	N		FACU
Symphoricarpos albus	Caprifoliaceae	X	X	X				SS/S	N		FACU
Stellaria crispa	Caryophyllaceae	X						F	N		FAC+
Chenopodium sp.	Chenopodiaceae						X	F	N/A		N/A
Convolvulus sp.	Convolvulaceae		X				X	F/V	N/A	B	N/A
Cornus stolonifera	Cornaceae	X		X				T/S	N		FACW
Thuja plicata	Cupressaceae			X				T	N		FAC
Carex deweyana var. *deweyana*	Cyperaceae	X						G	N		FACU
Carex lenticularis	Cyperaceae				X	X		G	N		FACW+

Table 8.2. Characteristics of 121 unique plant taxa identified in 22 vegetation plots sampled in the Elwha River estuary, Washington, August 2007.—Continued

Scientific name (genus and species)	Family	Mixed riparian forest	Willow alder forest	Riparian shrub	Shrub-marsh transition	Emergent marsh	Dunegrass	Life form	Origin	Noxious weeds	Wetland indicator status
Carex lyngbyei	Cyperaceae				X			G	N		OBL
Carex obnupta	Cyperaceae			X	X	X		G	N		OBL
Carex stipata var. *stipata*	Cyperaceae				X	X		G	N		OBL
Eleocharis palustris	Cyperaceae				X	X		G	N		OBL
Pteridium aquilinum	Dennstaedtiaceae	X						F	N		FACU
Dipsacus fullonum	Dipsacaceae				X			F	I		FAC
Athyrium filix-femina	Dryopteridaceae	X		X	X			F	N		FAC
Polystichum munitum	Dryopteridaceae	X		X				F	N		FACU
Equisetum arvense	Equisetaceae	X		X	X	X		F	N		FAC
Equisetum hyemale var. *affine*	Equisetaceae	X						F	N		FACW
Arbutus menziesii	Ericaceae						X	T	N		N/A
Gaultheria shallon	Ericaceae	X						SS/S	N		FACU
Vaccinium parvifolium	Ericaceae	X						S	N		N/A
Cytisus scoparius	Fabaceae	X			X			S	I	C	N/A
Lathyrus japonicus	Fabaceae				X		X	F/V	N		FACU-
Lathyrus sylvestris	Fabaceae				X			F/V	I		N/A
Lotus corniculatus	Fabaceae				X	X	X	F	I		FAC
Trifolium repens	Fabaceae		X		X		X	F	I		FAC
Vicia hirsuta	Fabaceae			X			X	F/V	I		N/A
Vicia nigricans ssp. *gigantea*	Fabaceae		X	X			X	F/V	N		N/A
Geranium robertianum	Geraniaceae	X	X					F	I	B	N/A
Ribes divaricatum	Grossulariaceae	X	X	X				S	N		FAC
Myriophyllum aquaticum	Haloragaceae					X		F	I	B	OBL
Iris pseudacorus	Iridaceae					X		F	I		OBL
Juncus acuminatus	Juncaceae				X			G	N		OBL
Juncus arcticus ssp. *littoralis*	Juncaceae					X		G	N		FACW+
Juncus articulatus	Juncaceae				X			G	N		OBL
Juncus effusus	Juncaceae				X			G	N		FACW
Mentha arvensis	Lamiaceae			X	X		X	F	N		FACW-
Mentha xpiperita [aquatica+spicata]	Lamiaceae				X	X		F	N		FACW+
Prunella vulgaris	Lamiaceae			X	X	X		F	N		FACU+
Maianthemum dilatatum	Liliaceae		X	X				F	N		FAC
Chamerion angustifolium ssp. *angustifolium*	Onagraceae				X			F	N		FACU+
Circaea alpina	Onagraceae	X						F	N		FAC+
Epilobium ciliatum ssp. *watsonii*	Onagraceae					X	X	F	N		FACW-
Epipactis gigantea	Orchidaceae	X		X				F	N		OBL
Piperia elongata	Orchidaceae	X		X				F	N		N/A
Pseudotsuga menziesii	Pinaceae	X		X				T	N		FACU
Plantago lanceolata	Plantaginaceae			X	X	X		F	I		FAC
Plantago major	Plantaginaceae		X				X	F	I		FACU+
Plantago maritima	Plantaginaceae				X			F	N		FACW+
Agrostis capillaris	Poaceae	X	X	X	X	X	X	G	I		FAC

Table 8.2. Characteristics of 121 unique plant taxa identified in 22 vegetation plots sampled in the Elwha River estuary, Washington, August 2007.—Continued

Scientific name (genus and species)	Family	Mixed riparian forest	Willow alder forest	Riparian shrub	Shrub-marsh transition	Emergent marsh	Dunegrass	Life form	Origin	Noxious weeds	Wetland indicator status
Agrostis stolonifera	Poaceae	X	X	X	X			G	I		FAC
Bromus sitchensis	Poaceae	X	X					G	N		N/A
Dactylis glomerata	Poaceae	X	X		X		X	G	I		FACU
Elymus repens	Poaceae		X	X				G	I		FAC-
Festuca rubra	Poaceae			X	X			G	I		FAC+
Holcus lanatus	Poaceae	X	X	X	X		X	G	I		FAC
Leymus mollis	Poaceae			X			X	G	N		N/A
Lolium perenne ssp. *multiflorum*	Poaceae						X	G	I		FACU
Phalaris arundinacea	Poaceae			X	X	X	X	G	N	C	FACW
Phleum pratense	Poaceae				X			G	I		FAC-
Poa palustris	Poaceae			X	X			G	N		FAC
Schedonorus pratensis	Poaceae	X		X	X		X	G	I		FACU+
Polygonum sp.	Polygonaceae		X			X		F	N/A		N/A
Rumex crispus	Polygonaceae				X	X		F	I		FAC+
Rumex salicifolius var. *salicifolius*	Polygonaceae		X		X	X	X	F	N		FACW
Polypodium glycyrrhiza	Polypodiaceae	X		X				F	N		N/A
Claytonia sibirica	Portulacaceae	X						F	N		FAC
Potamogeton sp.	Potamogetonaceae					X		F	N/A		OBL
Ranunculus repens	Ranunculaceae	X	X	X			X	F	I		FACW
Argentina egedii ssp. *egedii*	Rosaceae		X	X	X	X	X	F	N		FAC-
Crataegus douglasii	Rosaceae			X				T/S	N		FAC
Crataegus monogyna	Rosaceae	X						T	I		FACU+
Holodiscus discolor	Rosaceae	X						S	N		N/A
Malus fusca	Rosaceae			X				T/S	N		FACW
Oemleria cerasiformis	Rosaceae	X	X	X				T/S	N		FACU
Rosa nutkana var. *nutkana*	Rosaceae			X			X	SS	N		FAC
Rosa pisocarpa	Rosaceae			X			X	SS	N		FAC
Rubus armeniacus	Rosaceae			X			X	SS	I	C	FACU
Rubus parviflorus	Rosaceae	X	X					SS	N		FAC-
Rubus spectabilis	Rosaceae	X	X	X	X	X		SS/V	N		FAC+
Rubus ursinus	Rosaceae	X		X			X	SS	N		FACU
Galium boreale	Rubiaceae		X	X	X	X		F	N		FAC
Galium sp.	Rubiaceae	X						F	N/A		N/A
Populus balsamifera ssp. *trichocarpa*	Salicaceae	X		X				T	N		FAC
Salix hookeriana	Salicaceae		X					T/S	N		FACW-
Salix lasiandra ssp. *lasiandra*	Salicaceae				X			T/S	N		FACW+
Salix sitchensis	Salicaceae	X	X	X	X	X	X	T/S	N		FACW
Tolmiea menziesii	Saxifragaceae	X	X					F	N		FAC
Digitalis purpurea	Scrophulariaceae	X	X					F	I		FACU
Solanum dulcamara	Solanaceae						X	SS/F/V	I		FAC+
Typha latifolia	Typhaceae				X	X		F	N		OBL
Urtica dioica	Urticaceae	X						F	N		FAC+

Life Form

Of the 121 unique taxa in our sample plots, there were 15 tree, 15 shrub, 67 forb, and 24 graminoid species. Within all of the vegetation types, species richness of forbs was higher than the other life forms. Forb richness was especially high in the shrub-emergent marsh transition and dunegrass plots. Species richness of shrubs was relatively high in the riparian shrub plots and somewhat high in the mixed riparian forest plots. Graminoid richness was relatively high in the shrub-emergent marsh plots and in the emergent marsh plots. Finally, tree species richness was notably low in the emergent marsh and dunegrass plots (fig. 8.10*A*). Relative cover of trees was high in mixed riparian forest and willow-alder forest plots (greater than 60 percent); relative cover of shrubs was highest in riparian shrub (39 percent) and mixed riparian forest (26 percent) plots; relative cover of forbs was highest in the shrub-emergent marsh transition (65 percent) and emergent marsh (42 percent) plots; relative cover of graminoids was highest in the dunegrass (64 percent) and emergent marsh (53 percent) plots (fig. 8.10*B*).

Native Compared with Introduced Species

Of the 121 unique taxa, 113 had information indicating whether they are native or introduced. Of these 113 species, 71 (63 percent) are native and 42 (37 percent) are introduced. Thirteen of the introduced species are State-listed noxious weeds (Washington State Noxious Weed Control Board, 2011). The numbers of introduced species were highest in the shrub-emergent marsh transition and dunegrass vegetation types (20 and 21 species, respectively) and lowest in the emergent marsh and riparian forest vegetation types (10 and 12 species, respectively; fig. 8.11). Differences among vegetation types were more pronounced when cover of native versus introduced species was considered. The relative cover of introduced species was highest in the shrub-emergent marsh transition and dunegrass vegetation types (33 percent and 18 percent, respectively) and lowest in the emergent marsh, riparian forest, and riparian shrub vegetation types (1.9, 3.4, and 3.6 percent, respectively; fig. 8.11).

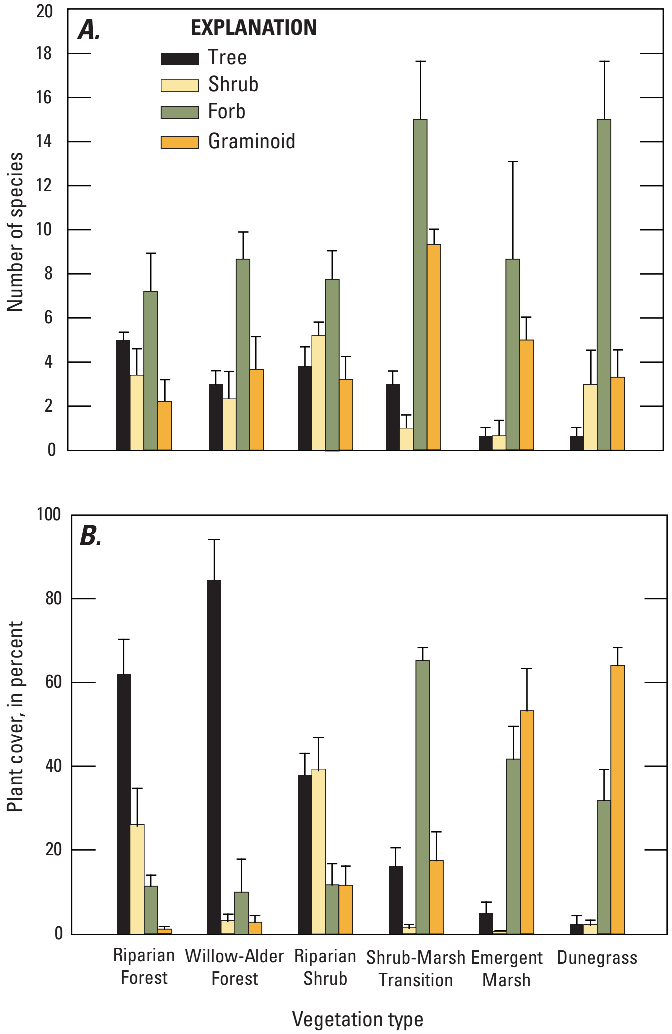

Figure 8.10. Life form composition of vegetation, Elwha River estuary, Washington. (*A*) Contribution of different life forms to species richness in the six Elwha River estuary vegetation types described in this study (plot means + standard error [black whisker]). *(B)* Relative plant cover associated with different life forms in the six Elwha River estuary vegetation types described in this study (plot means + standard error [black whisker]).

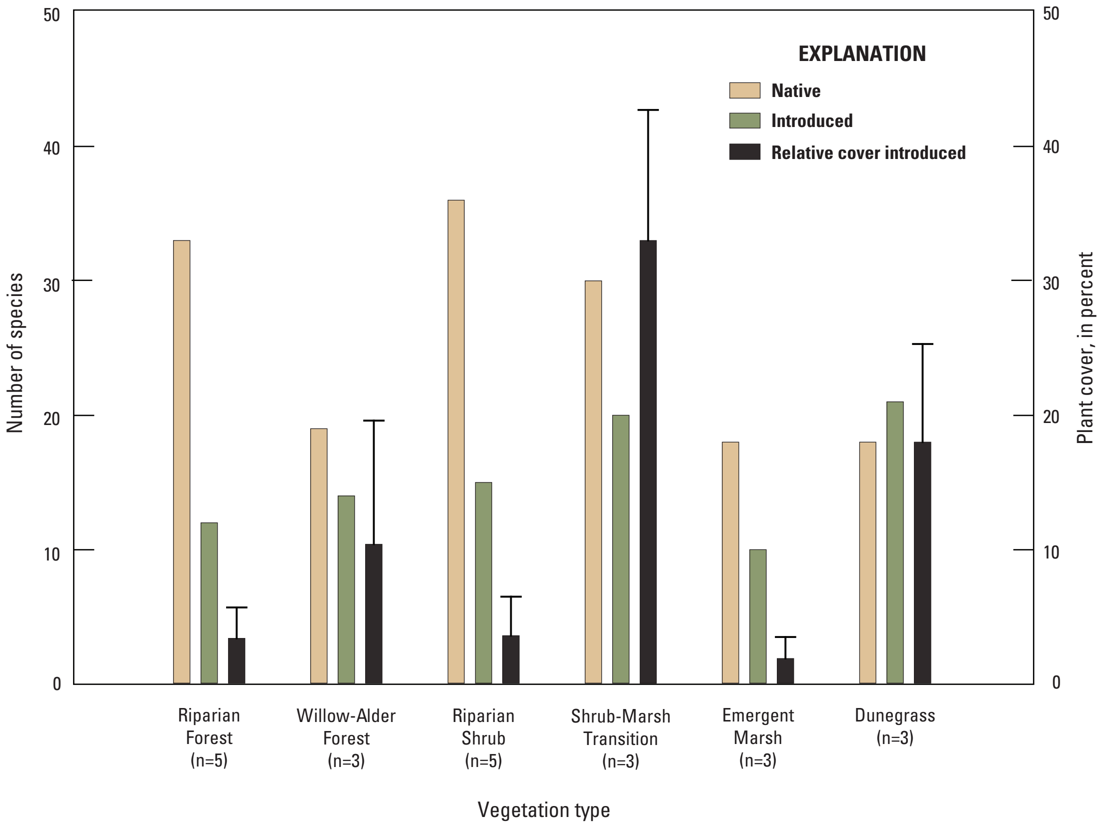

Figure 8.11. Number of native and introduced species and relative plant cover of introduced species in the six vegetation types described for the Elwha River estuary, Washington. Numbers of native and introduced species are cumulative across all plots in a particular vegetation type. Relative plant cover of introduced species is the mean + standard error (black whisker).

Wetland Indicator Status

Wetland indicator status information was available for 94 of the 121 unique taxa. The average wetland indicator value associated with species present in plots was highest (least likely to occur in wetlands) for the dunegrass vegetation type, followed by the mixed riparian forest and riparian shrub vegetation types (fig. 8.12). As would be expected, the emergent marsh vegetation type had the lowest value, and the shrub-marsh transition vegetation types had the second lowest value (fig. 8.12). The pattern of wetland indicator values weighted by relative cover of each species was similar (fig. 8.12).

Vegetation Types Along a Topographic Gradient

Consistent with the wetland indicator status results, average plot elevation above NAVD 88 was highest for dunegrass and mixed riparian forest (fig. 8.13). Willow-alder forest and riparian shrub plots were intermediate and shrub-emergent marsh transition and marsh plots were at the lowest elevations (fig. 8.13).

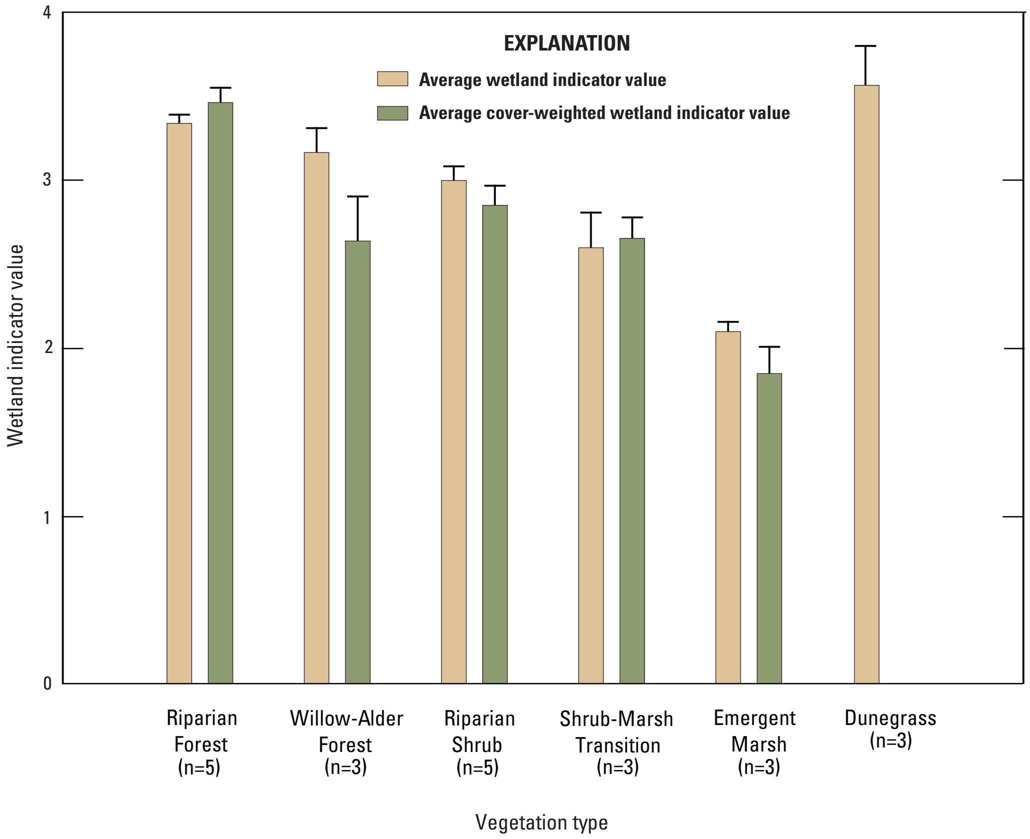

Figure 8.12. Wetland indicator values of plants in the six vegetation types described for the Elwha River estuary, Washington. Lower values indicate a greater tendency to occur in wetlands. Light brown bars indicate the average (of all plots within a vegetation type) of the average wetland indicator value of all plants for which a value was available in each plot. Green bars indicate the average (of all plots within a vegetation type) of the average wetland indicator value of plants in a plot, by vegetation type, weighted by the relative cover of those plants. Black whiskers indicate one standard error. Plots from the cover-weighted calculations were excluded if plants without a known wetland indicator value comprised greater than 20 percent of the plot's relative cover, which included all of the dunegrass plots.

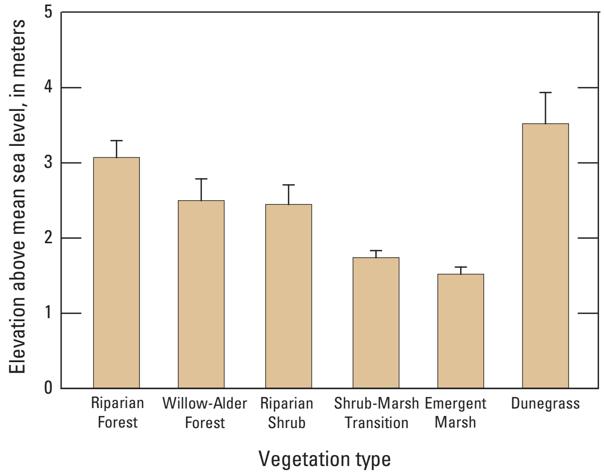

Figure 8.13. Average plus standard error (black whisker) elevation of plots within the six vegetation types described for the Elwha River estuary, Washington. See the "Methods" section for a detailed description of topographic survey data collection.

Elwha Estuary Vegetation—Discussion of Results

Species Richness and Vegetation Type Diversity

Despite its small size, the Elwha River estuary is quite diverse in terms of vegetation types present and species richness. 121 unique plant taxa in 6 vegetation types were documented in 2,200 m^2 total (twenty-two 100 m^2 plots) within 39.7 ha of land area within the Elwha estuary study area (fig. 8-1). This is the greatest plant species diversity yet documented, both in terms of total species richness and species richness within a particular vegetation type, within the admittedly sparse Pacific Northwest coastal wetland literature (MacDonald and Barbour, 1974; Brennan, 2007). Thom and others (2002) documented between 8 and 14 plant species per year in the 15 ha Elk River estuary (Washington Coast) over an 11-year period following removal of a dike in seventy-two 1 m^2 plots. Tanner and others (2002) documented 61 unique plant taxa in the 23.7 ha restoration project area associated with the Spencer Island dike breaching in the Snohomish River estuary (central Puget Sound). Plant species richness varied by year, beginning with 37 species prior to dike removal in 1994, then dropping to a low of 14 species in 1995, and increasing to 38 by 1998. Most species observed in 1994 were within forested wetland plant communities (35 species), whereas by 1998 most species occurred within emergent marsh plant communities. Plots varied in size and shape, so calculations of sampled area were not possible. Hutchinson (1988) sampled 17 intertidal delta marshes in the Strait of Georgia and the Puget Sound, not including those of the Strait of Juan de Fuca, and documented a total of only 80 plant species, two of them macroalgae. Of the sampled deltas, the Cowichan (32 species), Campbell (30), Fraser (28), and Skokomish (28) deltas had the greatest plant species richness. However, Hutchinson sampled only 1,810 m^2 (905 1×2 m plots). The relatively high plant diversity present in the Elwha River estuary is striking, given that it historically and presently represents less than 1 percent of Salish Sea coastal wetlands (Collins and Sheikh, 2005).

The relatively high species richness probably is a function of intense searching in a larger sample area (Stohlgren and others, 1997), as well as the presence of a greater diversity of vegetation types compared with other Pacific Northwest coastal wetland studies. For example, "mixed riparian forest," "willow-alder forest," and "riparian shrub" vegetation types may not have been sampled in some of the other studies. Based on cluster analyses, Hutchinson (1988) identified five classes of deltas and seven plant communities. All seven of Hutchinson's plant communities would be classified as emergent marsh under our sampling scheme; all are graminoid-dominated or graminoid/forb co-dominated. No other types were sampled. Of his seven, two are possibly present within the Elwha estuary: *Scirpus americanus-Scirpus maritimus* and *Juncus balticus-A. egedii*. Burg and others (1980) identified 12 plant associations and 24 plant species characteristic of those associations in the salt marshes of the Nisqually River estuary in southern Puget Sound. In their results, they did not include a full plant species list, so we cannot compare floristic diversity between our study and theirs. However, new vegetation studies of the Nisqually estuary are currently in progress, and the preliminary plant list includes 51 species (U.S. Geological Survey, 2009). Of the 12 associations they identified, 2 of them also could be within our study area: pure stands of *Carex lyngbyei* and *C. lynbyei-Festuca rubra*. These are the most productive plant associations that Burg and others (1980) sampled. The *C. lyngbyei* association was the most productive per unit dry weight (1,390 g dry weight/m^2, contributing 6.7 percent total marsh production), followed by *C. lynbyei-Festuca rubra* (1,086 g dry weight/m^2, contributing 10.5 percent total marsh production). Both species are present within the Elwha estuary and were encountered in vegetation patches classified as emergent marsh or shrub-emergent marsh transition. The other plant associations they identified are much more characteristic of low tidal salt marshes, which are not present within our study area.

A somewhat unusual aspect of the Elwha estuary vegetation is the lack of a well-developed salt marsh community, given the historic extent of salt marshes in the Salish Sea region and their representation in Pacific Northwest coastal wetland literature. Nor does an Elwha River salt marsh community appear to have been present historically, which Collins and Sheik (2005) attributed to the high wave energy environment on the Strait of Juan de Fuca coast. Characteristic salt marsh taxa, including *Salicornia virginica* (pickleweed), *Distichlis spicata* (saltgrass), and *Triglochin maritima* (seaside arrowgrass) are entirely absent from our sample plots in the Elwha estuary (table 8.2).

Introduced Species

Although the literature on estuarine vegetation vulnerability to invasion is not well developed, riparian systems have been shown in a number of studies to harbor a greater number of introduced species than adjacent upland ecosystems (DeFerrari and Naiman, 1994; Stohlgren and others, 1998; Brown and Peet, 2003), largely due to high levels of disturbance, rapid seed dispersal potential, and high resource availability. The number of introduced species and the proportion of the local flora in our study exceed that reported in several other studies of riparian ecosystems from around the world (DeFerrari and Naiman, 1994; Planty-Tabacchi and others, 1996; Hood and Naiman, 2000; Brown and Peet, 2003). For example, DeFerrari and Naiman (1994) reported species richness and cover of introduced (exotic) plants in riparian communities along the Dungeness and Hoh Rivers on the Olympic Peninsula, Washington, and found 40 and 30 species, respectively, comprising 28 percent of the flora in both systems. These lower values were found despite the fact that they sampled approximately three times as many plots (plot size was 50 m^2). Planty-Tabacchi and others (1996) reported that 30 percent of 851 total species found along three drainages in western Oregon (Willamette River, MacKenzie River, Lookout Creek) were introduced. Thus, our finding of 37 percent of the Elwha estuary flora as introduced is higher than commonly reported and is of potential concern to resource managers (Woodward and others, 2008). Physical habitat changes expected following dam removals on the Elwha could increase physical disturbance and perhaps facilitate the spread of some of these taxa, as is projected further upstream in the Elwha watershed (Woodward and others, 2011).

Dam Removal Context

Removal of the two dams on the Elwha River is expected to result in the release and transport of sediment and large woody debris trapped behind the reservoirs to sediment-starved river reaches downstream (Konrad, 2009; Czuba and others, 2011, chapter 2, this report). Both of these inputs could lead to higher rates of sediment deposition, increased channel dynamics (Draut and others, 2011; Warrick and others, 2011, chapter 3, this report) and associated vegetation change (Shafroth and others, 2002; Kloehn and others, 2008; Naiman and others, 2010). Maximum depth of sediment deposition in the channel bed in the lower Elwha River near the estuary is expected to be less than 1 m (Konrad, 2009; Czuba and others, 2011, chapter 2, this report). We are not aware of any predictions of flood plain aggradation depth, though it might be expected to be relatively high in reaches with relatively low gradients, and broad flood plain and island complexes, such as those in the first few kilometers upstream of the estuary (Konrad, 2009; Draut and others, 2011). Flood plain aggradation rates and depths will depend largely on the interaction of overbank flooding (including flood magnitude and duration), suspended sediment concentration, and flood plain roughness (including the density and structure of riparian vegetation).

Most research in the Pacific Northwest has focused on characterizing tidal wetlands (Burg and others, 1980; Hutchinson, 1988) or their responses to restored tidal influence from dike removals (Thom and others, 2002; Tanner and others, 2002). A few studies from the Pacific Northwest have examined interactions between sediment, wood, and estuary vegetation. The addition of sediment on subsided intertidal areas accelerated marsh vegetation colonization in Coos Bay's south slough, Oregon (Cornu and Sadro, 2002). Hood (2010) described feedbacks between sediment deposition-induced channel meandering and marsh island vegetation development in the Skagit River delta, Washington. Hutchinson (1988) characterized morphology, physical environment, and vegetation, and concluded that flow volume and discharge regime of contributing rivers, along with the exposure of the delta fronts, accounted for much of the variation in delta vegetation across the region. Large woody debris appears to strongly influence the distribution of some species, such as *Myrica gale* (sweetgale), a nitrogen-fixing shrub, in the Skagit River estuary (Hood, 2007). More generally, the influence of large woody debris on channel and vegetation dynamics in Pacific Northwest river systems is well documented (for example, Abbe and Montgomery, 2003).

Pacific Northwest tidal wetlands appear to respond rapidly to changes to key physical conditions, with elevation above mean sea level and tidal influence strongly driving changes in vegetation patterns. Bucknam and others (1992) documented shifts in wetland types at two sites on Bainbridge Island, Washington (Restoration Point and Winslow). At Restoration Point, a tidal flat was abruptly uplifted about 1,700 years ago, rapidly converting it to freshwater swamps and meadows, with no intermediate, brackish stages. Conversely, a freshwater wetland at Winslow was inferred to have been only slightly elevated above mean sea level before being converted to more saline vegetation types as a result of subsidence. Rapid vegetation change also occurred following dike breaching projects at the Elk River estuary on Washington's southwest coast (Thom and others, 2002) and Spencer Island in the Snohomish River estuary, Puget Sound (Tanner and others, 2002). Within about 5 years of dike breaching and associated reconnection to tidal inundation, most of the Elk River estuary changed from a *Phalaris arundinacea*-dominated wet pasture to a low-elevation salt marsh community dominated by *Distichlis spicata* (salt grass) and *Salicornia virginica* (pickleweed). High coastal marsh, dominated by *Deschampsia caespitosa* (hairgrass) and *D. spicata*, was also present but much less abundant (Thom and others, 2002). In contrast, *P. arundinacea*-dominated communities on Spencer Island either converted to mudflats or to a different freshwater vegetation type (Tanner and others, 2002). Experimental additions of sediment to an Oregon estuary resulted in different marsh vegetation composition and development rates on surfaces that differed in elevation above sea level (Cornu and Sadro, 2002). Thus, rapid changes to the physical environment associated with dam removals could prompt rapid vegetation changes in the Elwha River estuary.

Summary

Six primary vegetation types were identified in the Elwha River estuary: mixed riparian forest, willow-alder forest, riparian shrub, shrub-emergent marsh transition, emergent marsh, and dunegrass. These were identifiable on high resolution aerial imagery and were confirmed by the composition of plants in vegetation plots and grouping of plots determined by a non-metric multi-dimensional scaling ordination.

Overall species richness was quite high in the Elwha River estuary; we identified 121 unique taxa in 22, 100 square meter plots. Between 30 and 52 plant taxa occurred within each vegetation type, typically including several taxa unique to each type. Native species predominated, but approximately 37 percent of taxa were introduced. Introduced species were most common and abundant in the shrub-emergent marsh transition and dunegrass vegetation types. Plants in lower elevation plots were most likely to be categorized as obligately or facultatively occurring in wetlands.

Future analyses will focus on relating the distribution of patch types and patterns of species richness with key physical processes in the Elwha River estuary, such as hydrologic and sediment dynamics, and salinity regimes. Changes to these processes and associated disturbance regimes and environmental gradients following dam removals could change parts of the Elwha estuary vegetation. Quantitative understanding of how physical processes and environmental gradients interact with species dispersal, establishment, and survival would greatly improve our understanding of how Elwha River estuary vegetation may respond to dam removals, and might ultimately be linked to physical process models to predict biological responses over time.

Acknowledgments

This work could not have been completed without key contributions by numerous collaborators. In particular, we thank Ian Miller for conducting some of the RTK-DGPS topographic surveying, Rebecca Paradis for tireless assistance with vegetation plot sampling, Daniel Bennett for help with various field tasks, Randall McCoy for multi-faceted assistance with aerial photography, Jeff Duda for assistance with the ordination analysis, Jim Bromberg for assistance with finalizing figures, Tammy Fancher for assistance with preparation of figure 1, and Fred Sanford for inspiration. Thanks to Andrea Woodward and Rebecca Brown for helpful discussions about this work.

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Chapter
9

Summary and Anticipated Responses to Elwha River Dam Removal

By Guy Gelfenbaum, Jeffrey J. Duda, and Jonathan A. Warrick

Abstract

Starting in September 2011, the removal of two large dams on the Elwha River will begin an unprecedented river restoration project because of the size of the dams, the volume of sediment released, the pristine watershed upstream of the dam sites, and the potential for renewing salmon populations. Ecosystem studies of the Elwha watershed indicate that the effects of almost 100 years of damming are measurable and of consequence. These effects include smaller spawning salmon populations, massive sediment retention behind the dams, coarsening of the riverbed downstream of the dams, low nutrient concentrations in the river waters, and coastal erosion that has accelerated markedly with time. During and after the removal of these dams, the Elwha River and its ecosystems will be altered by a renewal of sediment discharge downstream of the dams and a reintroduction of salmon spawning upstream of the dams. This chapter summarizes the pre-dam and current state of the river and its coastal ecosystems, and describes the likely outcomes of river restoration on the Elwha River ecosystems.

Introduction

The removal of two large dams on the Elwha River presents an opportunity to restore natural fluvial processes to a mostly pristine watershed and rebuild iconic salmon runs. After removal of the dams, salmon populations are expected to increase dramatically over their present numbers (U.S. Department of the Interior, 1995a; Ward and others, 2008), restoring several important runs to this Pacific Northwest river that once produced large numbers of fish (U.S. Department of the Interior and others, 1994; Wunderlich and others, 1994). The removal of the Elwha and Glines Canyon Dams also presents an opportunity to restore beaches along the Strait of Juan de Fuca with sediment that has been trapped for nearly a century by dam-impounding reservoirs (fig. 9.1). Coastal erosion of as much as 22 m over the past 16 years east of the river mouth is encroaching on the Lower Elwha Klallam Tribal reservation (Warrick and others, 2009), and on vital wetlands and estuaries that provide critical habitat for rearing juvenile salmon.

Although expectations are high for returning salmon populations to the river, restoring river function, and restoring beach sediment supply, a number of uncertainties are associated with the restoration of the Elwha River ecosystem. The rebuilding of salmon populations, for example, is of particular interest. The spatial and temporal patterns of reemerging life-history diversity and competition among natural- and hatchery-origin fish are complex and not fully understood. However, these processes ultimately will affect salmon recolonization, future population status (Brenkman and others, 2008; Pess and others, 2008), and will be a large determinant of ecosystem restoration success. Similarly, the supply of sediment to the lower river and beaches downstream of the dams is uncertain. It is not known how much of the sediment trapped behind the dams will be eroded and transported downstream. Of the sediment that is transported downstream, it is not known how much will be transported all the way to the Strait of Juan de Fuca, and if that amount will be enough to slow or stop the coastal erosion.

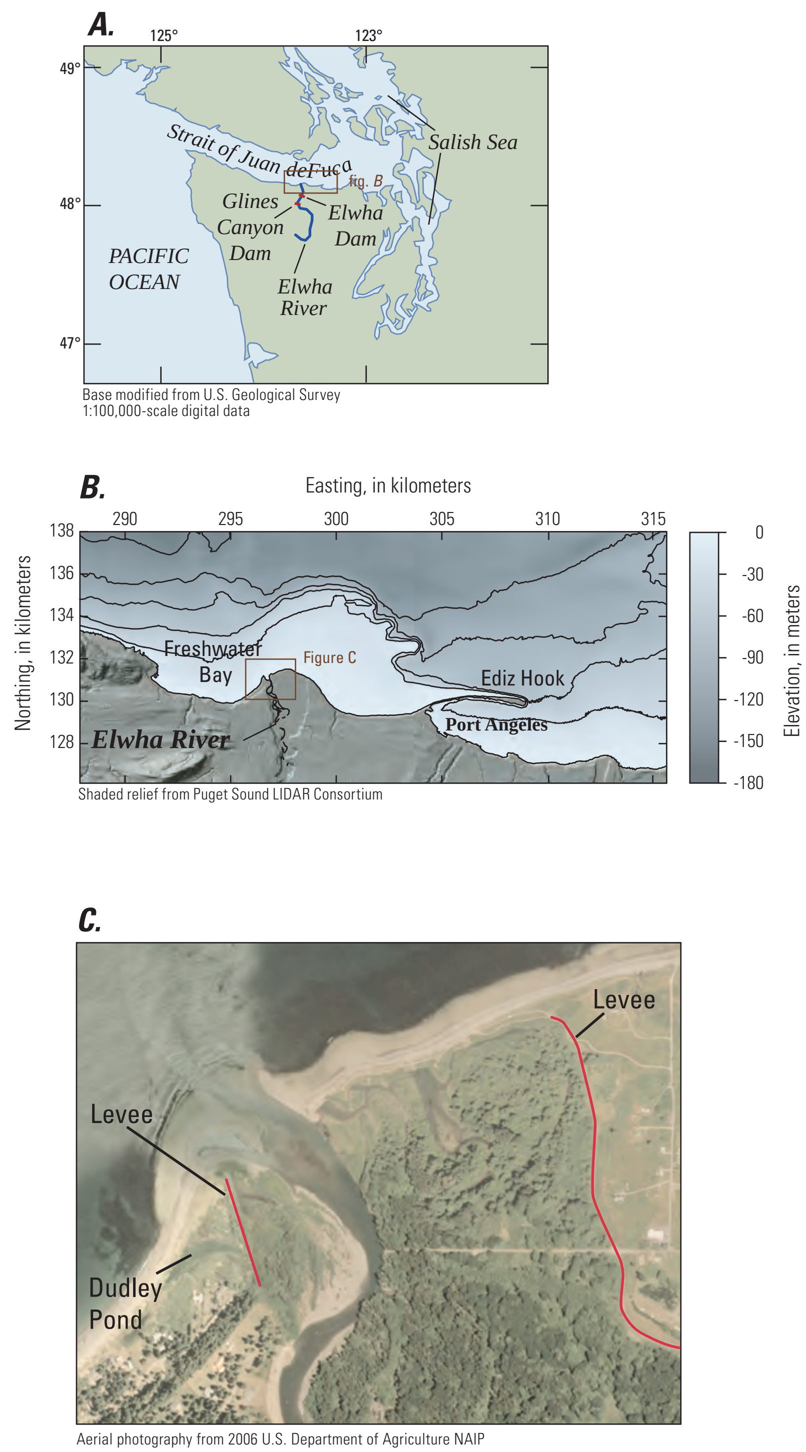

Figure 9.1. The coastal setting of the Elwha River, Washington.

The return of fish populations and restoration of sediment supply will cause numerous changes to the Elwha River ecosystem. For example, large amounts of silt and sand, along with gravel and cobbles may be deposited in the bed of the lower river, which would alter the grain-size distribution in the riverbed, the structure of pools and riffles, and the suitability of spawning habitat. The return of large numbers of fish to the lower river and estuary may alter nutrient concentrations, thereby changing the chemistry, and ultimately the productivity, of these systems. The fining of bed sediments may alter the suitability of the ecosystem for kelp species if large amounts of fine sediment are delivered offshore. The response of ecosystem processes, structure, habitat, and the biological resources they support will be complex, and are largely unpredictable. The scientific knowledge of many of these ecosystem linkages is limited and the results of this large-scale restoration project are uncertain, which highlights the need for ongoing multidisciplinary scientific research.

The scale of the Elwha River Restoration Project is unprecedented. The Elwha and Glines Canyon Dams will be the largest dams removed in the United States, and to our knowledge, the world. Moreover, the removal of these dams will cause the largest controlled release of sediment into a river and adjacent marine waters. This project represents one of the best remaining opportunities in the conterminous United States to restore salmon in large portions of a watershed that are protected as wilderness. The Elwha River dam-removal and ecosystem-restoration project also is an opportunity for increasing scientific understanding (Gelfenbaum and others, 2006). Billions of dollars are spent on ecosystem restoration around the nation, sometimes with highly uncertain outcomes. With the large numbers of dams around the nation reaching the end of their constructed lifespan or economic productivity, the impetus for dam removal is accelerating and becoming much less controversial than in the past (Hart and others, 2002; Heinz Center, 2002). Moreover, relicensing efforts for existing dams have a critical need for the best available sediment- and river-response data to assess the costs and benefits of dam removal compared with continued operation. Meaningful monitoring efforts are needed to document ecosystem recovery and to assess the benefits of restoration expenditures (Duda and others, 2008).

As noted in the previous chapters of this report, as well as numerous other reports documenting scientific investigations of the Elwha River basin, the Elwha River ecosystem has been affected by the Elwha and Glines Canyon Dams, and removal of those dams will affect it further. However, it is also likely that dam removal and ecosystem restoration may not simply restore the ecosystem to its pre-dam state, but may instead result in something new. The concept of “alternative stable states” proposes that there are suites of abiotic and biotic conditions in an ecosystem that lead to particular assemblages of uniquely adapted species. As the system is perturbed with small changes, the density or species composition of the system may temporarily change, but will drift back into the equilibrium state when the perturbation ceases. Larger perturbations that exceed a given threshold, on the other hand, can lead to a phase shift in the system that gives rise to a new stable state (Holling, 1973; Bender and others, 1984; Beisner and others, 2003). Due in part to the steepness of the basin, and the proximity between the upper reaches of the river and the coast, the dam removal project will create short duration, high intensity changes and long term, low intensity ecosystem changes. It may be that the high sediment load during and following dam removal will be large enough to change the existing aquatic and terrestrial flood plain ecosystems into a new stable state. This is certainly the case for the former reservoir sections that will change from lentic (lake-like) to lotic (river-like) water bodies. In the years following dam removal, as more coarse-grained sediments arrive from the former reservoirs, this lower intensity stressor may or may not change the ecosystem. It is not known if multiple phase shifts will arise in the Elwha River after dam removal, or if the ecosystem will settle into a new equilibrium following the high intensity changes shortly after dam removal.

Additionally, the large numbers of salmon anticipated to return to the river and the large volume of sediment expected to be mobilized after dam removal, will create a system response with large “signals” relative to the “noise” of natural background changes. For these reasons, a comprehensive research and monitoring program such as that described in this report will have a high likelihood of successfully documenting ecological changes caused by dam removal.

A comprehensive description of the ecosystem response to dam removal will require the monitoring of multiple interrelated variables across a varied geographic domain that will depend on the sediment grain size mobilized (Woodward and others, 2008). For example, ecosystem responses will vary across the different sections of the river (upstream of, between, and downstream of the dams), the reservoirs, beaches, and the adjacent nearshore zone of the Strait of Juan de Fuca. The response in each of these domains will depend on the grain size of the sediment that accumulates there, which may include very fine-grained (silt and clay) to very coarse-grained (cobble and gravel) sediments. Sediment accumulations may ultimately affect each of the relevant ecosystem components, including:

(1) processes such as sediment transport, nutrient dynamics, and spawning; (2) habitat structure defined by substrate type, channel morphology, flow velocity, and water depth; and (3) biological function such as species assemblages into communities of, for example, invertebrates or riparian vegetation. Ultimately, the effects on these ecosystem processes, habitats, and biological functions will influence the economic, cultural, and societal benefits that these ecosystem services provide. Not all components of the ecosystem will respond on the same time scale. The response will vary with time, from pre-dam removal, during removal (1–3 years), post-removal (3–7 years), and long-term (7–20 years). Some responses will occur quickly, and others will take much longer.

One of the primary purposes of this report is to document the physical and biological conditions of the lower Elwha River and coastal ecosystems in anticipation of the large-scale changes that may be caused by dam removal. This summary chapter (1) provides a summary of the important findings or characteristics of the lower Elwha River and nearshore zone as described in the previous chapters of this report, (2) describes linkages among the components to form a system perspective, and (3) describes predictions of the lower river and coastal physical and biological responses to dam removal. Although scientists and managers may strive for quantitative predictions of the responses to dam removal, at this stage, many predictions will be qualitative and conceptual. The remainder of this summary chapter is organized by geographic domain, starting with the lower river and estuary, then moving to the beaches and nearshore. Within each geographic domain, the important physical and biological characteristics of the ecosystem, as they exist prior to dam removal, and how they might change during and after dam removal will be described.

Lower Elwha River and Estuary

The 7.8-km-long lower Elwha River, between Elwha Dam and the Elwha River estuary, is characterized by various morphologies, including a narrow bedrock gorge, reaches with multiple channels and large vegetated islands, and reaches with a single meandering channel. In the upper 1.3 km of the reach, the channel is confined to a narrow bedrock gorge, limiting channel migration and sediment storage. The last 6.5 km before the river meets the sea is characterized by an ‘anabranching’ morphology that is the result of a unique combination of river gradient, flow conditions, large woody debris, and sediment supply, which are functional attributes associated with extensive channel switching, or avulsions. Detailed analysis of channel position over the last century from historical aerial photographs documented that the lower river channel has moved tens to hundreds of meters by gradual channel migration and by more abrupt avulsions (Draut and others, 2008, 2011). Avulsions are thought to occur when obstacles, such as piles of woody debris, temporarily block flow, as may have happened in winter floods of 1979 and 1980, forcing the river to cut a new path downstream. Repeated avulsions result in the formation of numerous interconnected channels, backwaters, and side channels, which provide important biological functions, a dynamic lotic ecosystem, and shifting riparian and estuarine habitats for many plant and animal species (Beechie and others, 2006; Tooth and others, 2008).

The hydrology of the lower river is typical for moderately sized watersheds in the Pacific Northwest, with the largest peak flows in winter due to heavy rainfall and smaller peak flows caused by spring snowmelt. The lowest flows of the year occur from August through October. Because the dams were built primarily for power generation and were typically operated as run-of-river, the dams have had little effect on the duration and magnitude of peak flows (Duda and others, 2011a, chapter 1, this report). Dam operations have had moderate effect on some aspects of the hydrograph, such as low flows and the rates at which flows increased and decreased, and they have had considerable effects on the supply of sediment and of large woody debris.

Sediment supply to the lower Elwha River from upstream has been reduced by the two dams, except for small quantities of fine-grained suspended sediment that bypasses the dams during floods. However, the lower river has compensated somewhat for the loss of upstream sediment supply by recruiting sediment from within its flood plain. Over the last century about 19 million cubic meters of sediment has been trapped behind Elwha and Glines Canyon Dams (U.S. Department of the Interior, 1995b; Bountry and others, 2010; Czuba and others, 2011, chapter 2, this report). Deltas formed at the inlets of Lake Mills and Lake Aldwell (combined) consist of about 50 percent coarse sand and gravel, and about 50 percent fine-grained silt and clay. This is the largest known reservoir sediment volume known to be associated with any dam removal project.

Reductions in the upstream sediment supply have resulted in bed coarsening and armoring downstream of the dams (Pohl, 2004; Kloehn and others, 2008, Morley and others, 2008; Draut and others, 2011). Additionally, Kloehn and others (2008) determined that the proportion of the flood plain older than 75 years, as determined by vegetation mapping and historical aerial photographs, increased with time downstream of the dams over the last half century, whereas the proportion of younger surfaces decreased with time. The conclusion drawn by Kloehn and others (2008) is that the reduced sediment supply has increased the stability of the river flood plains downstream of the dams. Kloehn and others (2008), however, did not collect channel stability data directly and an alternative explanation for the aging vegetation population could be changes in logging practices. Draut and others (2011) refined the characterization of the lower river, having determined that the upper reaches of the lower river were more stable because of bed armoring caused by lack of sediment supply. However, mobility in the lower reaches of the river was similar to the unregulated river upstream of the dams. Despite the lack of sediment supply from upstream of the dams, the lowermost river recruits enough sediment from bank erosion to maintain near-natural mobility of the channel in the lower reaches before the estuary (Draut and others, 2011). The response of the Elwha riverbed generally is consistent with other rivers with reduced sediment supply downstream of dams (Williams and Wolman, 1984; Collier and others, 1996; Pizzuto, 2002; Grant and others, 2003).

The lower Elwha River merges into a geomorphically diverse estuarine complex. The estuary includes the river mouth, several coastal ponds on either side of the river mouth, and several small side channels that connect these water bodies (fig. 9.1*C*). Within the estuary, river water, seawater, and groundwater mix to provide brackish conditions and habitat for young salmonids and other fish and wildlife species (Magirl and others, 2011, chapter 4, this report; Duda and others, 2011b, chapter 7, this report). The zone of saltwater-freshwater mixing varies as a function of tidal stage, river discharge, and storm surges, as well as channel morphology.

The estuary has a broad suite of vegetation types, including (1) mixed riparian forest, (2) young willow/alder forests, (3) riparian shrub, (4) shrub–emergent marsh transition, (5) emergent marsh, and (6) dunegrass. It contains a diverse assemblage of plant species, as Shafroth and others (2011, chapter 8, this report) found more than 120 plant species during surveys of each of the vegetation types. The riparian shrub and shrub–emergent marsh transition vegetation types had the highest total richness (52 species), although the emergent marsh vegetation type had the lowest total richness (31 species). About one-third of plant species currently around the estuary are non-native.

The estuary is tidally influenced and brackish, although water level and salinity patterns across this system vary in space and time. Numerous channels, which play an important role in estuarine circulation, regulate the exchange of water between the river mouth and the coastal ponds. This is determined by the differences in the water levels, salinity, and temperature between the eastern and western areas of the estuary across all seasons. Dudley Pond is contained behind a flood-control levee on the west side of the river mouth. Lacking a direct surface-water connection to the river, the pond is less saline, warms to higher temperatures, and has smaller variability in water levels compared to the lentic waters of the eastern estuary that contain numerous surface water connections (Magirl and others, 2011, chapter 4, this report; fig. 4.3). The morphology of the interestuarine channels does change with time as sediment is moved in the river mouth by river discharge events and coastal waves. High flows in the river also alter the water conditions in the estuary by raising water levels and flushing out saline water for as long as several days (see example in Magirl and others, 2011, this report, fig. 4.7).

The removal of Elwha and Glines Canyon Dams will not instantaneously result in resupply of sediment to the lower river and coast. Instead, the dam removal process itself will take several years, with complete removal anticipated in about 2.5 years after deconstruction commences (Duda and others, 2011a, chapter 1, this report). While the dams are being removed, it is anticipated that some fine-grained sediment will be remobilized from reservoir deposits, especially during winter and spring floods. Most of the fine sediment will be transported through the lower river and enter the nearshore region (Randle and others, 2006), to be dispersed by tidal currents and waves (fig. 9.2). A much smaller amount of the fine sediment will be deposited in the flood plain of the lower river, although fine-sediment deposition, where it occurs, likely will influence the hydrology, vegetation, and habitats of the river flood plain.

The rate of transport and timing of arrival of coarse-grained bedload to the lower Elwha River will depend on the number, magnitude, and duration of floods in the years during and after dam removal (Randle and others, 2006; Konrad, 2009). Coarse-sediment release may begin during the second winter season after the start of deconstruction, although large quantities of bedload probably will not pass the Glines Canyon Dam site until the winter of the final year of the project (2013–14). This enhanced bedload transport will continue down the middle reach of the river (between the two dams) for another 2–5 years after complete removal of Glines Canyon Dam (Randle and others, 2006; Czuba and others, 2011, chapter 2, this report).

As sediment is transported downstream of the reservoir sites, alluvial sections of the Elwha River will accumulate sediment and aggrade (fig. 9.3). Sand and gravel is expected to fill pools, and may cause more active channel migration and channel avulsions. Because of aggradation and fining of material in the lower river, the channel could see increased braiding, at least temporarily (Draut and others, 2011). This likely would result in a straighter channel alignment (steeper slope), which is consistent with a braided channel planform (Randle and Bountry, 2008). In the decades after dam removal, Kloehn and others (2008) suggest the rate of channel migration may increase in the river sections below the dams. Draut and others (2011) determination of local fine sediment supply along the flood plain of the lower river indicates that the rate of channel migration may remain stable over the long term.

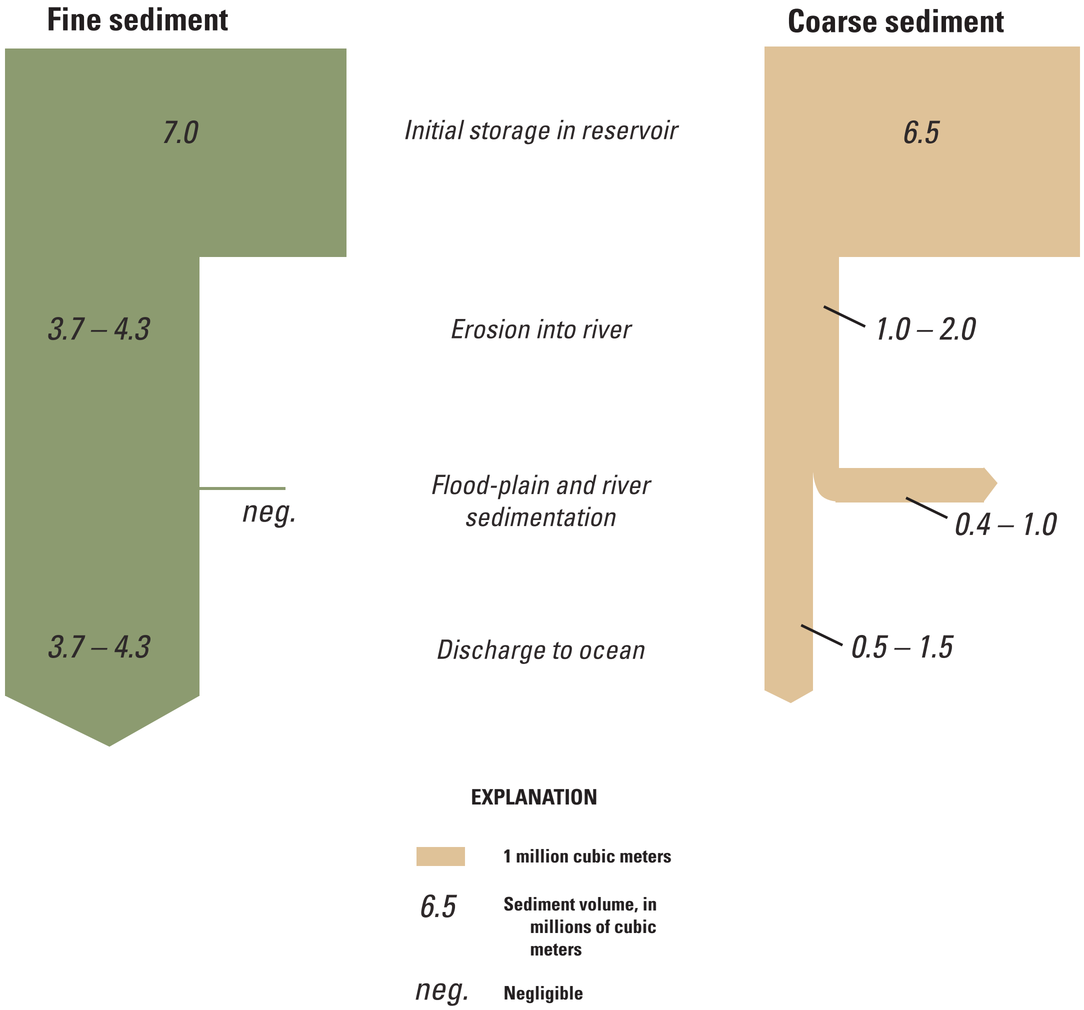

Figure 9.2. Conceptual sediment budget for fine and coarse sediment stored in the two Elwha River reservoirs during the first 3 years after dam removal, Elwha River, Washington. (After numerical modeling simulations by Randle and others, 1996). The range of values presented in this figure show the variation in simulated output of four contrasting hydrologic scenarios. Fine and coarse sediment separated by the grain-size threshold of 0.063 millimeters.

In the lower river, the initial increases in sediment load likely will disrupt aquatic and riparian ecology. Vegetation changes related to dam removal will depend largely on how much an influx of large woody debris and sediment alters the distribution and character of geomorphic surfaces, in particular estuarine shorelines (fig. 9.4). Another factor relates to the spatial and temporal nature of tidal connections and whether or not the connections change as the river mouth reconfigures after dam removal. These connections affect water level fluctuations and salinity levels, which likely would influence the “emergent marsh” and “shrub-emergent marsh transition” vegetation types. Additionally, any changes that appreciably alter the hydrologic connections between the river and estuary or the sedimentation and turbidity in estuarine habitats could affect the timing and habitat use patterns of migrating juvenile salmonids, as well as the resident fish assemblage of the estuary (Shaffer and others, 2009; Duda and others, 2011b, chapter 7, this report).

As sediment deposition and incorporation of large-woody debris and organic matter occur, the quantity and quality of fish habitat is expected to increase downstream of the dams (Woodward and others, 2008). The major ecological change in the lower river and estuary likely will result from the return of large numbers of fish. If recolonization of upper areas of the watershed causes salmon to return to the river in numbers larger than current populations, as projections suggest, then the nutrient concentrations in the river and estuary may change at least during some parts of the year.

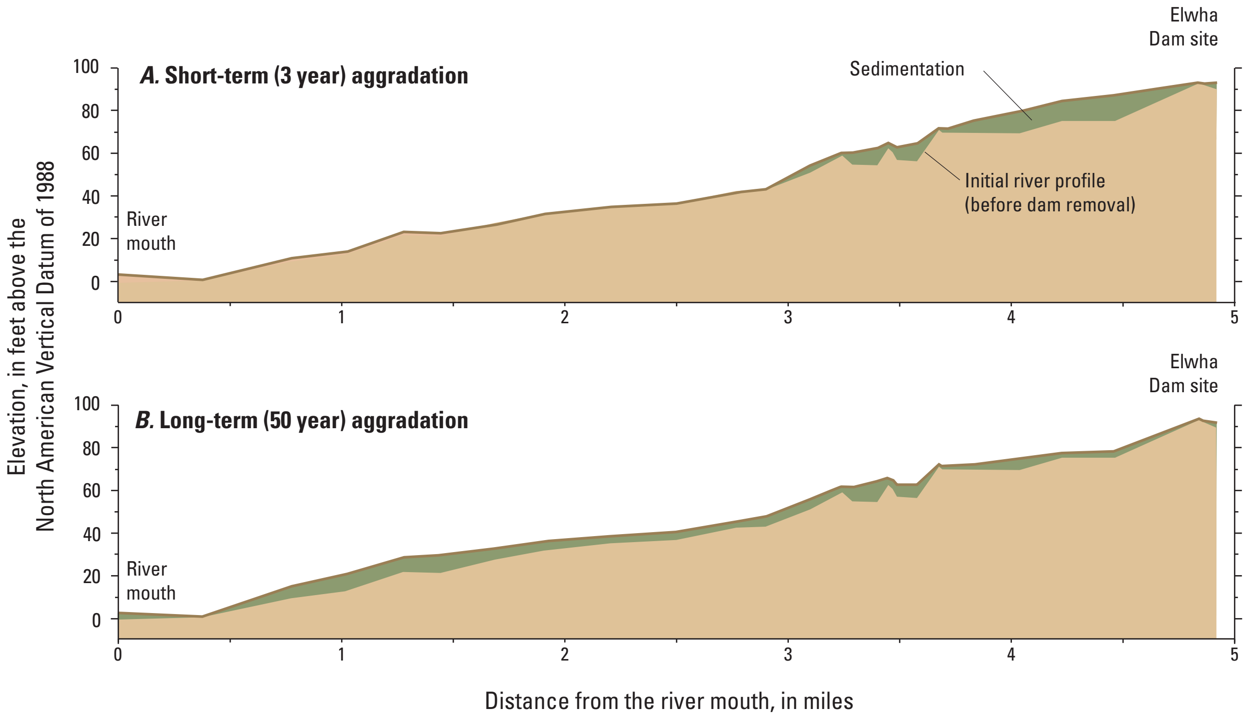

Figure 9.3. Schematic cross sections showing simulated response of the elevation to increased sediment loads after dam removal, Elwha River channel, Washington. (After Randle and others, 1996). Results shown for (*A*) short-term response following 3 years of renewed sediment supply and (*B*) long-term response after 50 years.

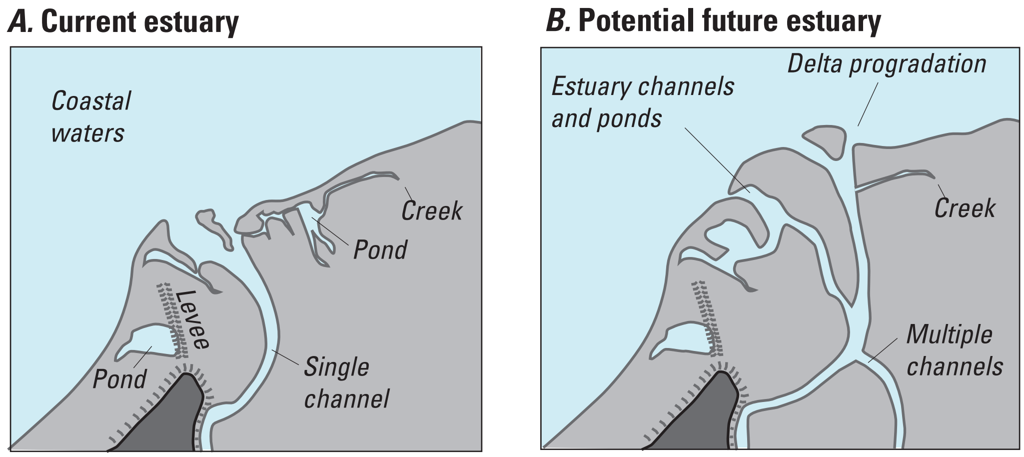

Figure 9.4. Schematic diagrams showing the (*A*) current estuary morphology and (*B*) potential morphologic changes to the estuary after several decades following dam removal and lower river sedimentation. The future changes to the estuary have not been predicted and therefore will not precisely mimic the patterns shown in (*B*); however, the future changes likely will exhibit many of the qualities shown in (*B*) such as multiple channels and delta progradation.

If phosphorous and nitrogen limit the biomass of primary producers in the Elwha River ecosystem, then increases in these nutrients may have important implications. Currently, the river is classified as oligotrophic (Munn and others, 1999; Duda and others, in press), and any increase in nutrient concentrations could significantly change the aquatic ecology of the river and estuary. Based on estimates of future salmon returns following full recovery, 1,275–10,900 kg of nitrogen and 210–1350 kg of phosphorous derived from salmon could be put into the river annually (estimates based on assumptions of Munn and others, 1999, and projected spawners presented by Ward and others, 2008). Although it is not likely that increased numbers of salmon will cause increases in dissolved water-column nutrients because of rapid flushing during the high flow season, salmon-derived nutrients may enter aquatic and riparian foodwebs through direct consumption by fish and aquatic invertebrates, or by indirect pathways like the guts of scavengers. This could provide temporal increases in the biomass and growth rates of resident biota (for example, Schuldt and Hershy, 1995; Bilby and others, 1996; Wipfli and others, 1998; Chaloner and Wipfli, 2002; Duda and others, in press).

Elwha River Beaches and Nearshore

The beaches and delta adjacent to the mouth of the Elwha River are the product of geomorphic processes that have continually formed and modified these landforms over multiple millennia. These processes include tectonic land-level changes, glaciation and related sea-level changes, erosion and sediment transport by the river, and coastal sediment transport by waves and currents (Warrick and others, 2011a, chapter 3, this report). Over the past century, the Elwha River beaches and nearshore have been modified by the presence of the two dams through a reduction in sediment delivery. Although coastal waters still receive sediment from the river, derived from erosion of channel-bank sediments deposited in the flood plains, most of the upriver sediment supply is intercepted by the reservoirs (Curran and others, 2009; Bountry and others, 2010). Several potential pathways for sediment discharged from the river exist, including along the beach, onto the nearshore submarine delta, and out into the deeper Strait of Juan de Fuca (fig. 9.5). The transport, fate, and implications of this sediment will be largely dependent on the physical processes and ecosystems within these settings (Miller and others, 2011; Warrick and others, 2011b, chapter 5, this report).

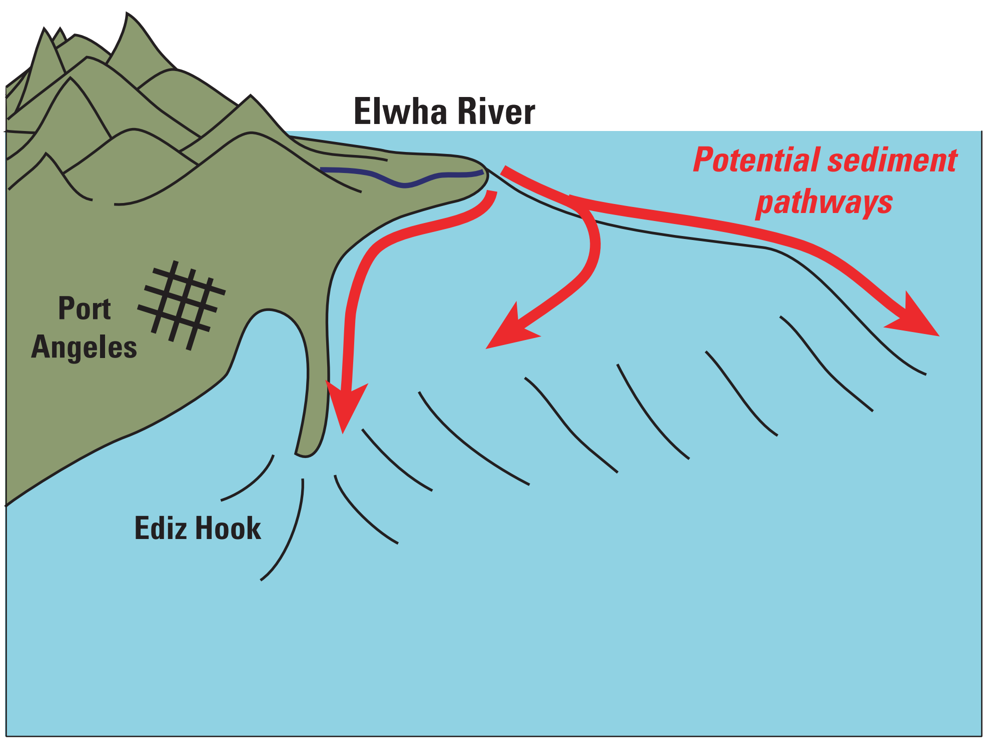

Figure 9.5. Schematic diagram showing the potential transport pathways for sediment offshore of the Elwha River mouth, Washington. Actual transport directions will be determined by sediment grain size and the strength, direction, and persistence of coastal currents and waves.

Discharge from the Elwha River enters the Strait of Juan de Fuca as a buoyant freshwater plume, influencing the water properties of the nearshore from Freshwater Bay to Ediz Hook and Port Angeles. The position of the freshwater plume is largely determined by tidal conditions. Warrick and Stevens (2011) examined the behavior of the freshwater plume and determined that about 50 percent of the time the plume is directed toward the east, and about 40 percent of the time the plume maintains a radial spread from the river mouth. The remainder of the time, less than 10 percent, the river plume is directed toward the west (fig. 9.6). These tidally influenced river plume conditions have implications for sediment dispersal and nearshore conditions during and after dam removal. As millions of cubic meters of sediment are released from the reservoirs, the fine-grained material will be transported in the river plume and arrive in the nearshore. Warrick and Stevens (2011) suggest that the nearshore area within 1 km of the Elwha River mouth will see the greatest suspended sediment and turbidity levels.

The reduced sediment supply to the Elwha River nearshore has changed the physical character of the delta and nearshore sea floor, which in turn has changed the biological assemblages dependent upon this habitat. The shoreline and submarine areas of the Elwha River delta are dominated by coarse sediment (gravel, cobbles, and boulders; Warrick and others, 2008; 2011b, chapter 5, this report). A time-series analysis of historical aerial photographs and contemporary topographic surveys of the beaches have identified significant coastal erosion since dam construction (Warrick and others, 2009). Net erosion rates have increased over time, from 0.8 m yr^{-1} from 1936 to 1990 to 1.4 m yr^{-1} during 1990–2006 and local erosion rates as much as 3.8 m yr^{-1} during 2004–07 (Warrick and others, 2009). It is uncertain whether a lack of fine sediment supply from the river also has resulted in a coarsening of the beaches over time.

These physical changes have influenced the plants and animals living in the nearshore environment. Recent scuba dive surveys (Rubin and others, 2011, chapter 6, this report) determined that a species-rich and biologically diverse community that included 10 kelp species, each with different growth forms and habitat affinities, inhabited the Elwha River nearshore. Community structure, including density, taxa richness, and habitat associations, was controlled in part by substrate composition, seafloor relief, and depth. Taxa richness (total number of kelp, invertebrate, and fish taxa) was more strongly associated with seafloor relief than with substrate type, although because relief usually was present in the form of scattered boulders perched on mixed sand, gravel, and cobble substrate, substrate also indirectly contributed to this relation. On average, 12 (59 percent) more taxa occurred where boulders were present compared to areas lacking boulders but with otherwise similar conditions.

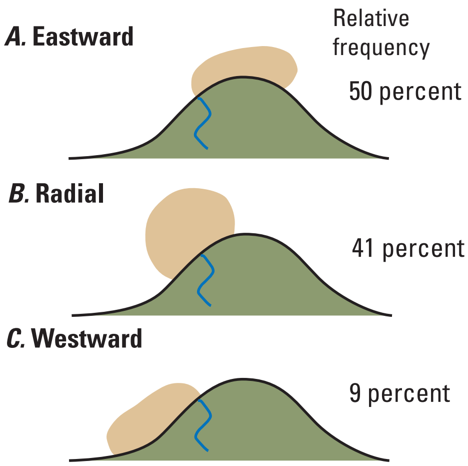

Figure 9.6. Schematic diagrams showing the behavior of the buoyant coastal plume of the Elwha River, Washington. (After Warrick and Stevens, 2011). The plume is bent toward the east (*A*) and west (*C*) during strong tidal currents, and is radial (*B*) during weak currents. Oceanographic observations suggest that the plume is bent toward the east more frequently than toward the west.

Four main species-habitat associations were identified in the Elwha River nearshore zone. The highest kelp density and taxa richness were in bedrock/boulder reefs, and were characterized by a surface canopy of bull kelp and a secondary canopy of perennial kelp 1–2 m above the seafloor. Mixed sand and gravel-cobble habitats with moderate relief had the highest density of invertebrates and a taxa richness nearly as high as in bedrock/boulder reefs. Mixed sand and gravel-cobble habitats with low relief were areas lacking in boulder cover (which generally reduced relief scores and excluded species that preferred seafloor relief) supported a moderate density of kelp (primarily annual species with blades close to the seafloor) and the lowest invertebrate density among the four habitat types. Sand habitats in localized sand bodies were limited in extent. These sandy areas had the lowest kelp density and kelp taxa richness, with a moderate density of invertebrates. Sandy substrate along the west side of Freshwater Bay supported eelgrass meadows, an important habitat for juvenile salmon and other fish species.

A significant amount of the sediment released by the dam removal project eventually will be discharged into the Strait of Juan de Fuca at the Elwha River mouth (fig. 9.2). Waves and currents that move sediment along and across the delta will dictate the fate of this sediment. The waves of the Elwha River delta region are primarily from the northwest.

This occurs for swells (10–18 second wave periods), which are derived from storms in the North Pacific Ocean, and wind waves (2–6 second wave periods), which are generated in the Strait of Juan de Fuca from westerly winds (Warrick and others, 2008; 2011b, chapter 5, this report). The oblique direction of these waves results in strong littoral transport along the delta, driving sediment transport toward the east (Miller and others, 2011). The currents offshore of the beaches are driven primarily by the tides; however, the shape of the delta influences water that must flood and ebb past the Elwha River delta due to these tides. Eddies that are 2–5 km in diameter are formed on the downstream sides of this deltaic headland during most tidal floods and ebbs. These currents influence the location of the turbid river plume, the location of the initial sediment settling, and subsequent movement of sediment (Gelfenbaum and others, 2009).

A flood on December 3, 2007, foreshadowed the potential changes that dam removal may bring to the Elwha River mouth and nearshore area. A large scale rain-on-snow storm caused rivers of the Olympic Peninsula to swell, with the peak discharge on the Elwha River recorded as 1,020 m^3/s, and an estimated 40-year recurrence interval (Draut and others, 2011; Duda and others, 2011a, chapter 1, this report). Detailed mapping of elevations of the Elwha River mouth in September 2007, and again in September 2008, revealed an accumulation of about 34,000 m^3 of sediment directly offshore of the river mouth. This sediment accumulation likely was associated with the peak discharge from the intervening December 2007 flood. It also changed salmon access to the east estuary, as revealed by observations on the ground and changes in fish community structure and density (Shaffer and others, 2009; Duda and others, 2011b, chapter 7, this report).

A calibrated hydrodynamic and sediment transport model described by Gelfenbaum and others (2009) provides some predictions of the fate of sediment during and following dam removal. Forced with waves, currents and river discharge, the model predicts the dispersal of fine-grained silt and sand across the delta. Model simulation results indicate that a larger sediment supply, such as that which is expected following dam removal, would spread in a zone to the east and west of the river mouth, largely driven by tidal currents (fig. 9.7). Because of the asymmetry in the tidal currents off the river mouth, sand will be mostly deposited east of the river mouth, whereas silt will be more evenly deposited to east and west of the river mouth. The weaker tidal currents directed to the west should be strong enough to transport silt in this direction, but probably are not strong enough to transport much sand to the west. Thin accumulations of silt and sand are expected across most of the delta, with thicker accumulations in deep water off the eastern edge of the delta and east of Ediz Hook, just inside of Port Angeles harbor.

The initial large sediment influx to the nearshore zone from released reservoir sediments will stress nearshore communities, but in the long-term, the communities should benefit from restoration of the natural, pre-dam sediment delivery regime (fig. 9.8). Separating the short-term effects from long-term benefits will be a primary challenge facing scientists in the years following dam removal. In the short term, negative effects will be greatest for species most vulnerable to deposited and suspended sediments. Kelp, for example, are vulnerable because they require relatively sediment-free surfaces for spore settlement. Their microscopic gametophyte and young sporophyte life stages also are susceptible to burial. Plants that can propagate vegetatively, on the other hand, may be resistant to sedimentation because they bypass these vulnerable juvenile life stages.

As high sediment transport and increased turbidity continue for 2–3 years during dam removal, community composition of plants and invertebrates is likely to change (Airoldi, 2003), and species richness may decline. The spatial extent, vertical thickness, frequency, duration, seasonal timing, and particle characteristics (for example, grain size and shape, mineral and chemical composition) of deposited sediment will largely determine the ecological response. Of the four main community-habitat associations described, each will have different responses to increased sedimentation. Bedrock/boulder reefs may be most vulnerable if deposited sediments bury organisms attached to the reef or prevent establishment of their propagules. On the other hand, vertical surfaces of hard substrate, such as the sides of large boulders, will not collect sediment and may serve as refugia for some reef-adapted taxa. If localized accumulations of sediment are great enough, some areas could convert from hard substrate to soft substrate, causing dramatic community shifts and decreasing habitat heterogeneity. Accumulations of very fine sediment (silt and clay) could benefit tube-building amphipods (*Ampelisca* spp.), which provide food for various other organisms (see Rubin and others, 2011, Chapter 6, sidebar 6.1, this report).

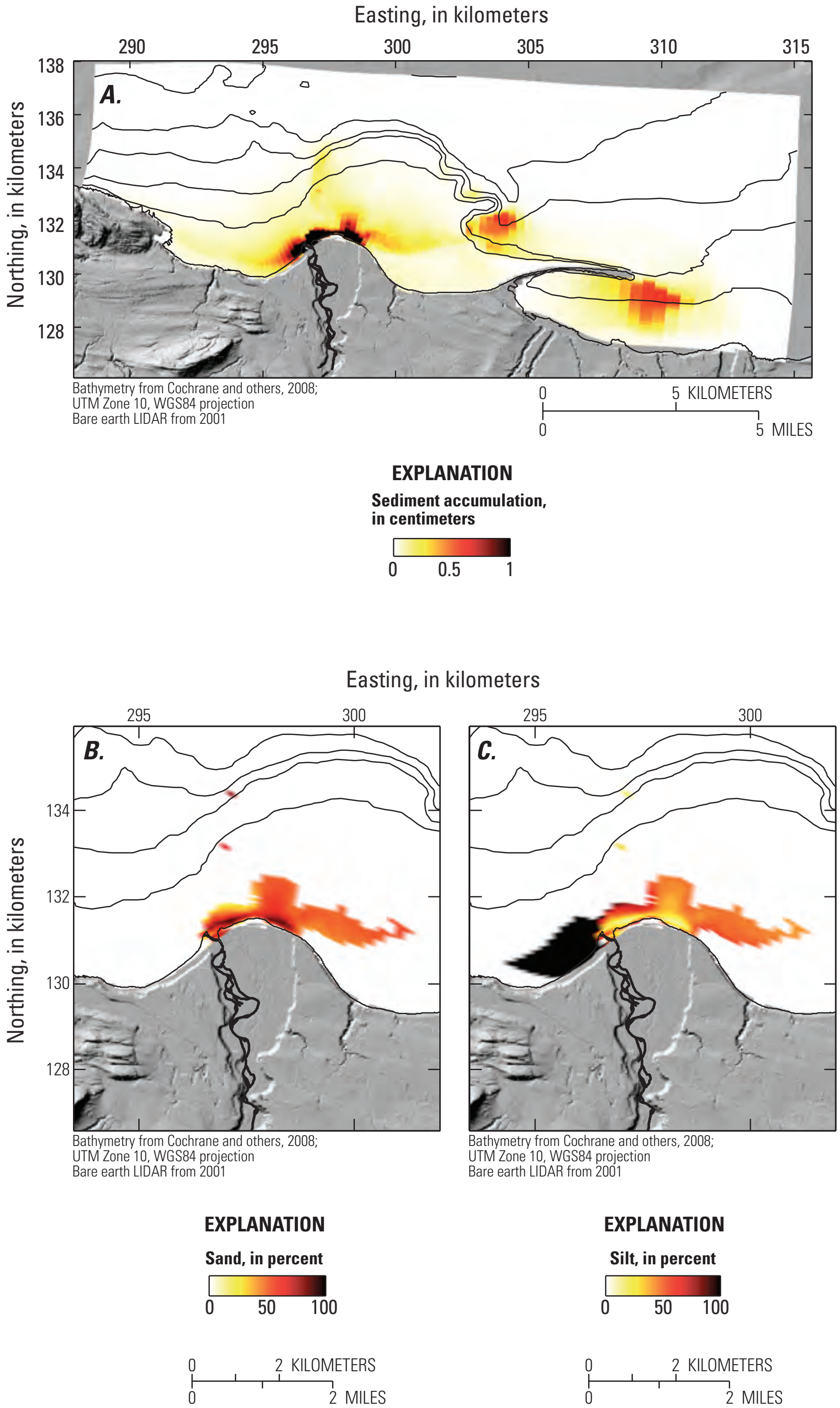

Figure 9.7. (*A*) simulated sediment accumulation offshore of the Elwha River mouth, Washington, after flood event from the river. (After Gelfenbaum and others, 2009). Results derived from a three-dimensional hydrodynamic model of water and two classes of sediment, sand, and silt. The sand (*B*) preferentially accumulates east of the river mouth, and the silt (*C*) accumulates on both sides of the river mouth.

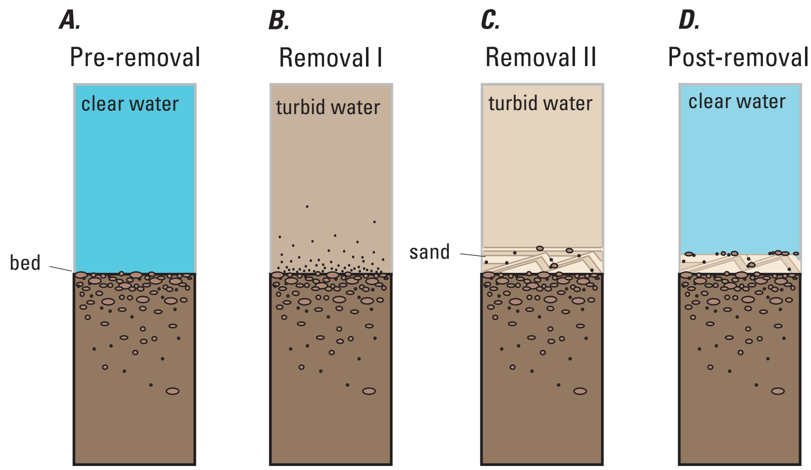

Figure 9.8. Schematic diagram showing the coastal water and seafloor near the Elwha River mouth, Washington, and hypothetical changes before, during, and after dam removal. (*A*) The pre-removal condition has relatively clear water and a gravel bed. (*B*) Sediment loading during removal initially will increase suspended-sediment concentrations and turbidity in the water column. (*C*) Sand introduced to the coastal waters likely will accumulate and change the grain size of the seafloor. (*D*) Several years following the dam removal, the coastal waters will be much less turbid, although the grain size of the seafloor likely will be sandier than observed before the dam removal.

Summary

The Elwha River dam removal and ecosystem restoration project is an unprecedented opportunity for scientific discovery. It also is inherently multidisciplinary, with changes to the physical and biological components of the ecosystem expected. The multidisciplinary team assembled as part of this project has attempted to develop baseline information about the river from multiple perspectives. From species diversity in the nearshore region to vegetation communities in the estuary, and growth, habitat use, and migratory patterns of juvenile salmon, many species will be affected by the dam removal project. Complex interactions among trophic levels and the environments upon which they depend are expected during and after dam removal. As environments change due to sediment release from the reservoirs, species will reassemble into new configurations in the estuary and nearshore environments. Similarly, the hydrology, geomorphology, and coastal oceanography will change in the river, estuary, and nearshore zone below the dams. The largest controlled release of sediment in history, characterized by high suspended sediment in the near term followed by more gradual movement of larger material over the coming decades, will also bring about changes. These changes to the physical habitat and biological characteristics of the ecosystem will need to be monitored to allow for a full evaluation of the restoration of the Elwha River.

Acknowledgments

Funding for this work came from the U.S. Geological Survey, Coastal Habitats in Puget Sound project. We are grateful to all of the authors who contributed to this report. Their dedication and passion for improving the scientific understanding of large-scale ecosystem change associated with the Elwha River Restoration Project is evident in their work. We thank the Lower Elwha Klallam Tribe for valuable discussion and insights into the Elwha River and estuary. This chapter benefited from conversations with Steve Rubin and Christopher S. Magirl.

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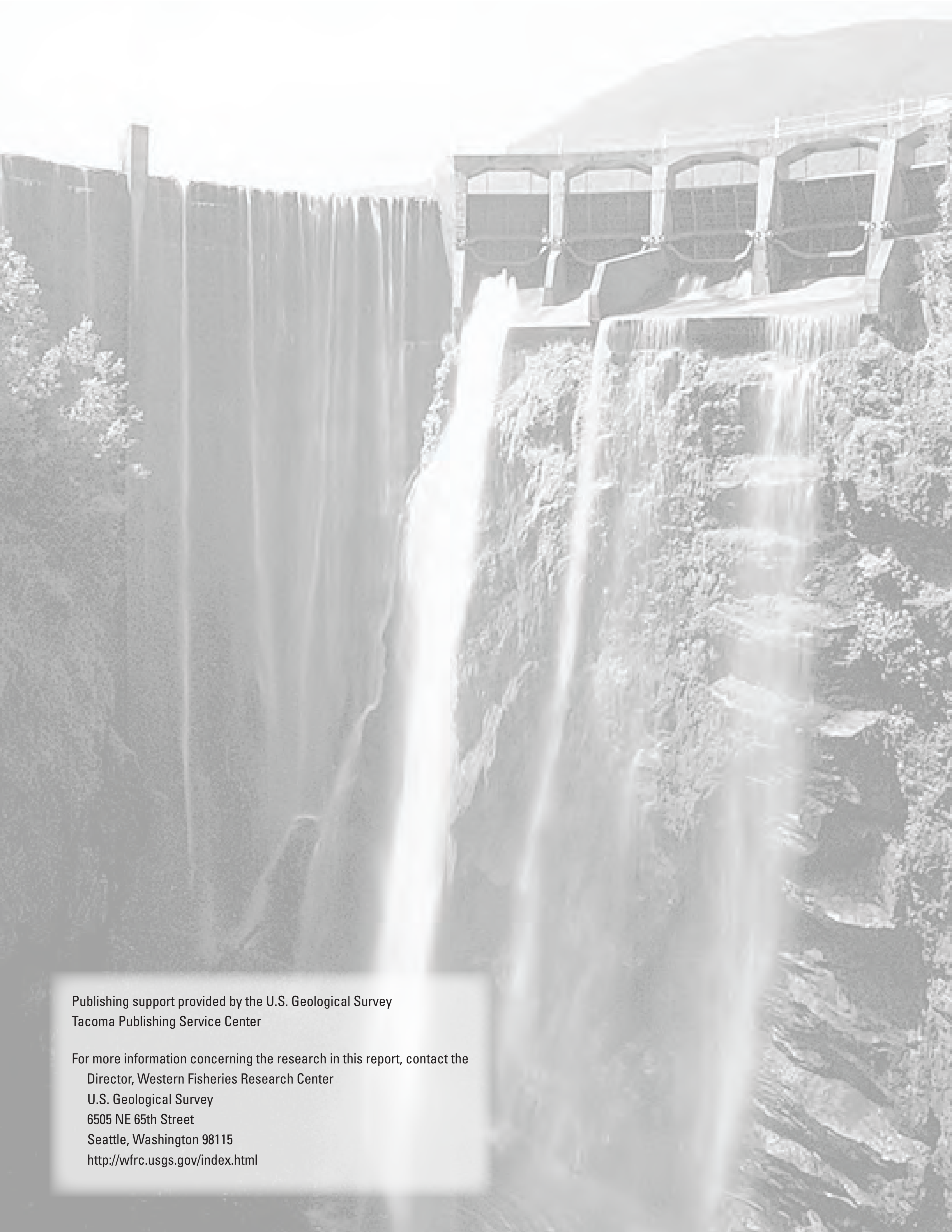

Publishing support provided by the U.S. Geological Survey
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