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Classification of Regional Patterns of Environmental Drivers and Benthic Habitats in Pacific Northwest Estuaries

Office of Research and Development
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Western Ecology Division

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EXECUTIVE SUMMARY

Increased anthropogenic nutrient loading and the subsequent eutrophication of coastal ecosystems is a growing ecological and economic problem both in the United States and globally. Eutrophication can result in a range of ecological impacts including hypoxic conditions, fish kills, loss of submerged aquatic vegetation (SAV), degraded benthic conditions, harmful algal blooms, and detrimental increases in benthic macroalgae. The nature and severity of the impacts vary with the level of nutrient loading as well as with the estuary type and regional drivers.

One tool to help address this problem is the development of classification schemes to allow researchers and managers to extrapolate results from a limited number of well-studied estuaries to the larger domain of estuaries within the same class. Several estuarine classification schemes have been developed based on different approaches and endpoints. However, the ecological reality of these classification schemes for the Pacific Northwest (PNW) is not clear, in part, because of the limited baseline information available to evaluate the schemes. Additionally, the available information gives "mixed messages" as to whether eutrophication is occurring in the coastal PNW estuaries. Dissolved oxygen levels are generally high and chlorophyll *a* is moderate to low, indicating a non-eutrophic condition. However, nutrient loading is high and within the range of eutrophic estuaries on the East Coast. Additionally, benthic macroalgal blooms, which have been used as a diagnostic indicator of eutrophication in other parts of the world, are a seasonal event in the PNW.

To help determine the extent of eutrophication, the Pacific Coastal Ecology Branch (PCEB) of the Western Ecology Division (WED) initiated a classification study of PNW estuaries. The PNW was defined as including the coastal estuaries from Cape Mendocino in Northern California (40.440ºN) to Cape Flattery in Washington (48.383ºN). Puget Sound was not included in the present study. Two different approaches were used. The first component of the research focused on a landscape analysis using existing data for the entire PNW. An inventory of all of the PNW estuaries was generated based on the occurrence of estuarine habitat as defined by the National Wetland Inventory (NWI), and all of the associated watersheds were delineated. The PNW contains a surprisingly large number of estuaries, a total of 103 based on the present NWI analysis. However, most of these estuaries are small, <1 km^2, and only 13 of them are >10 km^2. These PNW estuaries appear to break out into seven general types based on geomorphology, oceanic exchange, and riverine influence: coastal lagoons, blind estuaries, tidally restricted coastal creeks, tidal coastal creeks, marine harbors/coves, drowned river mouth estuaries, and bar-built estuaries. The first three of these have restricted connection with the ocean on at least a periodic basis, and are potentially vulnerable to anthropogenic nutrient additions and watershed alterations. Conversely, the marine harbors/coves have extensive flushing and are presumably less sensitive to either nutrient enrichment or watershed alterations. The drowned river mouth estuaries are sub-divided into tide-dominated, moderately river-dominated, and highly river-dominated systems based on the extent of estuarine area or volume weighted freshwater flow. Because of potentially higher exchange, the river-dominated estuaries may be less sensitive to nutrient enrichment.

Geomorphology by itself did not cleanly separate groups with similar vulnerabilities among the remaining 31 tide-dominated river mouth estuaries, river-dominated river mouth estuaries, and bar-built estuaries. One approach taken to identify groups of estuaries was to evaluate spatial and temporal salinity patterns based on historical data as well as a "normalized freshwater inflow" index we developed. This index ranks estuaries by the relative amount of freshwater entering the systems, and was used as a quantitative approach to separating tide- versus river-dominated estuaries. Another approach was to conduct multivariate analyses to examine the biotic and watershed similarities among the remaining 31 estuaries. The estuaries were clustered based on the patterns of wetland habitats from the National Wetland Inventory (NWI) and independently on land cover patterns in the associated watersheds. Several different classifications were produced as well as a "crosswalk" between the two classifications to determine which estuaries were grouped together in both analyses. Six pairs of estuaries were similarly classified based on both wetland and land cover patterns, and these estuary pairs potentially could form a framework for developing ecologically relevant nutrient criteria for the PNW.

The second component of the research was to develop a classification scheme based on nitrogen loading and sources, particularly as it related to the relative importance of oceanic- versus riverine-derived nitrogen. Seasonal coastal upwelling from approximately April to October is the major source of nitrogen and phosphorous to the near-coastal region. Previous research by PCEB showed that high nutrient oceanic water advected into the Yaquina Estuary, Oregon was the major nutrient source in the lower estuary during the dry season (May to October) and riverine nutrients (i.e., terrestrially derived) were the dominant nutrient source only in the upper Yaquina Estuary. In contrast, riverine inputs dominated through most of the estuary during the wet season (November to April); however, there is little utilization of the wet season riverine inputs due to low solar irradiance and short residence times associated with high river inflow. Overlaying the distribution of *Zostera marina* L., the native seagrass species, over the spatial pattern of nitrogen sources identified oceanic inputs as the major source for the bulk of the SAV population in the Yaquina Estuary during the dry season.

To determine the generality of this pattern observed in the Yaquina Estuary, field studies were initiated in seven target estuaries spanning a range of sizes (2.0 km^2 to 54.9 km^2) - Alsea, Coos, Nestucca, Salmon River, Tillamook, Umpqua River, and Yaquina. We focused the field studies on the dry season (May to October) because this is the primary period of biological nutrient utilization. One objective was to use the nitrogen source model developed for the Yaquina Estuary and site-specific water quality and stable isotope patterns in benthic macroalgae to delineate the ocean- and river-dominated segments in a suite of estuary types. Results from this component demonstrated that advection of high nutrient ocean water was the major nitrogen source in the lower estuary for the seven target estuaries.

The next objective was to determine if the SAV populations and populations of other key estuarine resources were primarily exposed to ocean- or river-derived nutrients. Five biotic endpoints were evaluated, three primary producers, and two secondary consumers. The major focus was on the perennial, rooted aquatic seagrass *Z. marina*, which was evaluated both by field surveys and by aerial photography. In addition to *Z. marina*, probabilistic field surveys evaluated the abundance and distributions of two additional benthic primary producers, the nonindigenous seagrass *Zostera japonica* Aschers. & Graebn. and benthic green macroalgae. The secondary consumers evaluated were two burrowing shrimp, *Neotrypaea californiensis* and *Upogebia pugettensis*. One pattern that emerged from comparing across estuaries was that macroalgal blooms occurred in all the estuaries, suggesting that it is a natural phenomenon, while stable isotope data suggest the presence of these blooms is not an indication of eutrophication. Another pattern was that the intertidal occurrence of the non-native *Z. japonica* exceeded that of the native *Z. marina* in several of the target estuaries.

Overlaying the spatial distributions of these species on the spatial patterns of dry season nutrient sources showed that the bulk of the *Z. marina*, benthic macroalgae and both burrowing shrimp populations occurred in the oceanic segments of all seven estuaries. Thus, for two of the primary producers, oceanic nitrogen is the dominant source during the principal growing season. Likewise for the burrowing shrimp, the bulk of the populations would primarily be exposed to oceanic nitrogen during the dry season. During the wet season, terrestrially derived nitrogen is the major nutrient source for these estuaries. However, this is a period of low water column chlorophyll *a* and *Z. marina* and macroalgal production suggesting that this terrestrially derived nitrogen is not primarily driving production by these species.

The exception to the pattern of the primary producers occurring primarily in the lower estuary was the non-native seagrass, *Z. japonica*, which was relatively abundant in both the oceanic and riverine segments. Thus, this non-native seagrass would have a higher exposure to terrestrially derived nitrogen than the native primary producers during the summer growing season. One possibility that has yet to be explored is whether low levels of nutrient enrichment stimulate the growth and establishment of this non-native seagrass.

The field surveys also evaluated the lower depth margin of *Z. marina* across the seven estuaries and along estuarine gradients. The lower depth limit to which *Z. marina* grows decreases in the upper estuary segments, which correlates with a general decrease in water clarity in the upper segment of the estuaries. With further development, the lower depth limit of *Z. marina* could potentially be used as an integrative indicator for assessing decreases in water clarity, as could occur from nutrient stimulation of phytoplankton. Even though our research indicates that the current levels of benthic macroalgae are natural, increases in macroalgae coverage or biomass in the riverine segments could also be used as an indicator of nutrient enrichment especially if such studies were coupled with measurements of stable isotopes ($\delta^{15}N$) to identify the nitrogen sources associated with the blooms.

Results from our studies and a growing body of literature suggest several key points for the management of nutrient enrichment in the PNW. First, environmental drivers such as coastal upwelling strongly indicate that regional approaches to classification will be necessary to generate ecologically relevant groupings. Second, PNW coastal estuaries are, in general, not showing indications of cultural eutrophication. Additionally, the majority of the populations of four of the five biotic resources we evaluated occurred in the oceanic segment of the estuaries, indicating a lower vulnerability to terrestrially derived nitrogen. However, anecdotal observations of phytoplankton blooms in the upper estuary segments of a few estuaries suggest that the riverine segments of estuaries may be experiencing localized nutrient enrichment. Third, the development of national estuarine nutrient criteria that do not take into account the naturally high nutrient concentrations resulting from upwelling are likely to result in numerous false non-attainments of criteria in PNW coastal estuaries. Similarly, the development of Total Daily Maximum Loads (TMDLs) for nutrients also needs to consider the effects of oceanic nutrient loadings.

CHAPTER 1: INTRODUCTION AND FRAMEWORK FOR THE PACIFIC NORTHWEST REGIONAL CLASSIFICATION PROJECT

Henry Lee II and Walter Nelson

1.0 Scope

As part of the U.S. EPA's Aquatic Stressors Framework (U.S. EPA, 2002), a variety of approaches to the classification of estuarine systems with respect to observed or predicted responses to nutrient enrichment have been examined. This study explores approaches to classification at a regional scale using distributions of submerged aquatic vegetation (SAV), wetland classes, and other estuarine biological resources as classifying variables for Pacific Northwest (PNW) estuaries. We also explore whether spatial and temporal patterns of salinity and nutrient dynamics among PNW estuaries allow us to group them with respect to their vulnerability to nutrient enrichment. This document is a revised version of a previous internal EPA report (Lee et al., 2006), and includes new analyses and interpretations based on updated wetland and salinity data. A companion study (Dettmann and Kurtz, 2006) explored the use of empirical nutrient-SAV load-response models for estuaries of the New England region as well as the response of phytoplankton biomass to nutrient concentrations in a suite of estuaries in the eastern U.S. Together, these two studies provide assessments of a series of approaches to estuarine classification that may be of value in setting national and regional nutrient criteria for estuaries.

1.1 Problem Statement

Increased anthropogenic nutrient loading and the subsequent eutrophication of coastal ecosystems is a growing ecological and economic problem both in the U.S. and globally (e.g., Nixon, 1995; NRC, 2000; Cloern, 2001; Bricker et al., 2003; Scavia and Bricker, 2006). Eutrophication can result in a range of ecological impacts including hypoxic conditions, fish kills, loss of SAV, degraded benthic conditions, harmful algal blooms, and detrimental increases in benthic macroalgae. The nature and severity of the impacts vary with the level of nutrient loading as well as the estuary type and regional drivers (e.g., Cloern, 2001; Bricker et al., 2003). Ideally, each estuary would be evaluated independently as to the nature and extent of these impacts and any mitigation/regulatory actions would be tailored to each estuary or watershed. The reality is, however, that there are insufficient resources to conduct detailed scientific studies on each estuary or to develop estuary-specific water quality criteria or management plans for every coastal water body. Hence, approaches to reduce this complexity are needed.

One general approach to prediction across a range of water bodies is to use water quality models, such as Basins (http://www.epa.gov/waterscience/basins/basinsv3.htm) or Sparrow (http://water.usgs.gov/nawqa/sparrow/). While these models are a powerful approach to making predictions in the better studied water bodies, they require extensive data input as well as expertise, reducing their general utility. A different, and complementary, approach is to classify estuaries that have similar responses to nutrient enrichment. If successful, such classifications allow extrapolation from one estuary to others within the same class, thereby reducing the amount of site specific data needed to make environmental decisions. Although lacking the same level of site specific predictive ability as the complex water quality models, classification is potentially a more practical approach to reducing the complexity inherent in extrapolating across a range of estuaries, especially for those regions with limited data. As pointed out by the National Research Council, "a widely accepted estuarine classification scheme is a prerequisite for a systematic approach to extending lessons learned and successful management options from one estuary to another" (NRC, 2000, page 163).

A plethora of classification approaches have been applied to estuarine systems, with 26 different classification frameworks identified in the review by Kurtz et al. (2006). Two classical approaches are geomorphic classification, which classifies estuaries as drowned river valleys, bar built, fjords, and tectonically formed estuaries (e.g., Pritchard, 1955, 1967), and the hydrodynamic approach, which classifies estuaries by their circulation patterns and stratification (e.g., Hansen and Rattray, 1966). Several recent efforts focused on grouping systems based on physical attributes which may be relevant to expression of nutrient impacts, such as the EPA's classification framework for coastal systems (U.S. EPA, 2004a) or the ASSESTS approach to ranking the eutrophication status of coastal waters (Bricker et al., 2003). To build upon these efforts, we initiated a classification research program focused on PNW estuaries. As described below, this program was designed around both the potential management uses of an estuarine classification scheme and the key environmental drivers in the PNW.

1.2 Regulatory Framework for Estuarine Classifications

Given the large number of approaches to estuarine classification, one of the early questions we asked was how an estuarine classification scheme could be used in a management context, specifically in relation to the development of water quality criteria or the management of excess nutrients. That is, we viewed the initial question not as "How to classify?", but rather "Why classify?" We identified five major approaches relating to the regulation of excess nutrient, as summarized below. Using this framework and an assessment of the data needs and availability in the PNW, we focused our research on management issues #2, #4, and #5 listed below.

1) Classification by Current Ecological Condition: A fundamental management need is assessing the current condition of coastal water bodies, which is a type of classification when multiple water bodies are compared. Such a comparison has been used to classify near-coastal water bodies as having "good", "fair", or "poor" ecological condition at regional and national scales in the National Coastal Condition Report (U.S. EPA, 2004b, 2006). Estuaries can also be compared in terms of their attainment of designated-use criteria. Classifying estuaries by their existing ecological condition requires: 1) a suite of ecologically relevant indicator metrics that can be practically measured over a range of different water body types and 2) field surveys that measure the metrics at the appropriate spatial and temporal scales.

2) Classification by Nutrient Loading and Sources: Estuaries can be classified by nutrient loadings from non-point, point, and natural sources. Comparison of estuaries by the relative magnitude of different sources is critical in developing effective management strategies to reduce nutrient loadings (Driscoll et al., 2003) or, more fundamentally, evaluating the role of anthropogenic versus natural loadings. Specifically, classification by loadings can help prioritize remediation/enforcement efforts or develop Total Daily Maximum Loads (TMDLs). The data needed to generate loading/source classifications include measurements of non-point, point, and natural loadings across a suite of water body types with loading models used to estimate missing values. These types of data are similar to those needed for a TMDL; the

difference is that data to classify estuaries need to be taken over a suite of estuaries though less extensive data collection per estuary can be used to classify estuaries compared to what would be needed for regulatory actions. In addition to data on estuary nutrient concentrations, classification by loading will usually require data on land cover in the associated watershed. Accordingly, "Landscape conditions (e.g., % cover of land uses)" is listed as one of the recommended core water quality indicators in EPA guidance for Sections 303(d), 305(b) and 314 of the Clean Water Act (http://www.epa.gov/owow/tmdl/2006IRG).

3) Classification to Derive/Validate Water Quality Criteria: Comparison of ecological condition in a suite of estuaries along a nutrient gradient is one approach to deriving stressor-response relationships (e.g., Latimer and Kelly, 2003). While more of a "natural experiment" than a classification per se, the change in ecological condition in response to increased exposure can identify nutrient thresholds. Such cross-estuary comparisons can also help to identify the most sensitive endpoints and diagnostic indicators of nutrient stress. Perhaps the greatest limitation of such natural experiments is the potential, and often unknown, confounding factors when comparing across estuaries. Nonetheless, such cross-system comparisons are a powerful tool in generating or validating water quality criteria under realistic conditions. The necessary requirement for this approach is a suite of "reasonably" similar estuaries that primarily differ in their nutrient concentrations or loadings. Finding such a nutrient gradient can be challenging, as in some regions finding a true "reference" estuary may be difficult while in other regions finding highly impacted estuaries may prove challenging.

4) Classification by the Resources at Risk: A fundamental but often overlooked type of classification is to categorize estuaries by the ecological resources at risk. The States establish designated uses for water bodies as part of state water quality standards, and nutrient or other protective criteria are determined to be able to meet these specific designated uses. Given the need to protect designated uses, an evaluation of the resources at risk is an important early step in the development of a water quality criterion, risk assessment, or mitigation action. An obvious example is the application of nutrient criterion for SAV to classes of estuaries devoid of SAV, which may be under- or overprotective of the actual resources within such systems. More subtlety, the distribution of the resource within the estuary may be a key factor in the exposure of the resource to anthropogenic nutrients or other stressors. Classifying estuaries by their resources at risk requires: 1) identification of the regionally high priority ecological resources; 2) identification of how these high priority ecological resources are distributed within and across estuaries; and 3) evaluation of the overlap of the resource(s) with anthropogenically derived nutrients or other stressors.

5) Classification by Estuarine Vulnerability: An approach related to classification by resources at risk is to categorize estuaries by their inherent susceptibility to nutrient enrichment (e.g., Bricker et al., 2003; U.S. EPA, 2004a). Identification of groups of estuaries likely to display adverse impacts at low to moderate nutrient concentrations/loadings versus those that have a higher assimilative capacity can assist managers in prioritizing monitoring, remediation, or enforcement actions. To the extent that vulnerability is related to regional drivers, such as climate or ocean conditions, a classification based on vulnerability can serve as the framework for developing defensible regional water quality criteria. Predicting

vulnerability is a complex process requiring consideration of natural and anthropogenic nutrient concentrations/loadings, sensitivity of the specific endpoint(s), spatial/temporal overlap of the endpoint(s) with excess nutrient concentrations, and mitigating or enhancing factors. In many cases, it will be necessary to draw on data (e.g., dose-response relationships) from other estuaries or from similar species.

1.3 Overview of Eutrophic Conditions and Nutrient Dynamics in the Pacific Northwest

Compared to the East and Gulf Coasts, relatively little baseline information on nutrient enrichment is available for PNW estuaries. In the most comprehensive national review of eutrophication, Bricker et al. (2007; also see Bricker et al., 2003) evaluated 138 estuaries across the U.S., which included 17 coastal PNW estuaries exclusive of Puget Sound. Nine of these 17 estuaries were considered to have insufficient data for classification. Of the remaining eight estuaries, two were classified as having low eutrophic conditions, six were considered to have moderate low eutrophic conditions, and none were classified as having high eutrophic conditions. Several Pacific estuaries were classified as having high eutrophic conditions in central and southern California and in Puget Sound, but these are all in different biogeographic ecoregions than the PNW estuaries (Spalding et al., 2007).

Water quality data from U.S. EPA's Coastal Environmental and Monitoring Assessment Program (EMAP) 1999 and 2000 surveys can also be used to evaluate water quality on the Pacific Coast. The 1999 survey sampled the "small" estuaries of California, Oregon, and Washington while the 2000 survey sampled Puget Sound, San Francisco Estuary, and main stem of the Columbia River Estuary. In 200 water quality samples from the 1999 survey, the lowest dissolved oxygen (DO) value was 3.8 mg l^{-1} and less than 4% of the area of the small coastal estuaries had DO concentrations less than 5 mg l^{-1} (Nelson et al., 2005b), the level indicative of biological stress (Bricker et al., 2003). Even when the more urbanized estuaries from the 2000 survey were included, only two sites out of 371 stations had DO values <2 mg l^{-1} (U.S. EPA, 2004b), the level indicative of hypoxia (Bricker et al., 2003). Both of these low values occurred in Hood Canal in southern Puget Sound, which is a deep fjord type estuary with limited recirculation (http://www.hoodcanal.washington.edu). Likewise, chlorophyll *a* concentration on the Pacific Coast was rated as good, and only two sites in Puget Sound and one in California exceeded the threshold of 20 µg l^{-1} for a "poor" rating (U.S. EPA, 2004b). The generally high DO values and low to moderate levels of chlorophyll *a* suggested that eutrophication was not a wide-spread problem on the Pacific Coast, in particular in the PNW coastal estuaries.

Even though the available evidence did not suggest that PNW estuaries were displaying symptoms of eutrophication, there are indications of high nutrient conditions. Water quality on the Pacific Coast was rated as "fair" because of high dissolved inorganic phosphorous (DIP) and poor water clarity (U.S. EPA, 2004b). Nitrogen loading normalized to estuarine area for the Yaquina Estuary, Oregon is 25 mmole N m^{-2} d^{-1} (Brown and Ozretich, 2009), which is as great as or greater than loading in a number of eutrophic systems (Nixon et al., 2001). Though phytoplankton blooms do not appear to be a wide-spread phenomenon, we have observed phytoplankton blooms in the upper regions of a few estuaries as well as the import of high chlorophyll *a* concentrations (> 40 µg l^{-1}) from the ocean (Brown and Ozretich, 2009). Finally, benthic macroalgae, an indicator of eutrophication (CENR, 2003; Bricker et al., 1999), is abundant in several PNW estuaries (e.g., Thom, 1984; Kentula and DeWitt, 2003). Thus, the emerging pattern for the PNW is high DIP concentrations and high nitrogen loadings, high levels of benthic macroalgae yet low or moderate phytoplankton concentrations. Similar cases of high nutrient loading with low chlorophyll *a* levels have also been observed in other Pacific Coast estuaries (Cloern, 2001), and certain estuaries appear less sensitive to nutrient loading due to "modulating filters", such as strong tidal forcing, high turbidity, and dominance of benthic suspension feeders.

One key factor affecting PNW estuaries is the seasonal upwelling that occurs during the spring-summer (Hickey and Banas, 2003). Coastal upwelling is such a strong regional driver that the PNW coastal estuaries can be considered "extensions of the coastal ocean during the growing season" (Hickey and Banas, 2003). Another key factor is the highly seasonal pattern in precipitation, with dramatically reduced rainfall during the summer (Emmett et al., 2000). This pattern of increased coastal nutrient concentrations and reduced runoff potentially increases the relative contribution of nutrients from oceanic sources during the summer. Field and modeling research (Brown et al., 2007; Brown and Ozretich, 2009) showed that nutrient-rich ocean water advected into the estuary was the major source of nutrients and phytoplankton for the lower portion of the Yaquina Estuary during the spring and summer. Overlaying the distribution of seagrasses on the pattern of nitrogen sources within the estuary identified oceanic-derived nitrogen as the major source for the major portion of the SAV population in the Yaquina during the primary growing season. During the low river flow conditions of summer, riverine nutrients (i.e., terrestrially derived) were the dominant nitrogen source only in the upper Yaquina, which contains relatively little SAV. Consequently, the SAV in this estuary does not appear to be highly vulnerable to anthropogenic nitrogen increases during the primary growing season because the bulk of the population only has minor exposure to watershed-derived nutrients.

These observations highlight the major differences in the nutrient dynamics of PNW estuaries compared to East and Gulf Coast estuaries largely resulting from summertime upwelling and reduced summertime river flow in the PNW. These differences could have substantial effects on regulatory strategies. For example, what is the ecological relevance and defensibility of water quality criteria in systems dominated by natural nutrient sources that periodically exceed the criteria? Another issue is how to develop criteria or implement TMDLs in estuaries with seasonally, and even tidally, variable fluxes of nutrients. One difficulty in addressing these issues is that research on the effects of upwelling on nutrient and phytoplankton dynamics in coastal estuaries has been limited to a relatively few PNW estuaries, primarily the Columbia River Estuary (e.g., Hamilton, 1984), Coos Estuary in Oregon (Fry et al., 2001; Roegner and Shanks, 2001; Roegner et al., 2002), Willapa Estuary in Washington (e.g., Hickey et al., 2002; Hickey and Banas, 2003; Newton and Horner, 2003), and Yaquina Estuary in Oregon (Brown et al., 2007; Brown and Ozretich, 2009). In particular, the pattern of the SAV being primarily exposed to ocean-derived nutrients has only been studied in the Yaquina Estuary, and without studies in different types of systems it is not clear whether this is a general pattern across PNW estuaries.

1.4 Biotic Response Measures in PNW Estuaries

In addition to the water quality patterns mentioned above, PNW estuaries are characterized by several dominant benthic macrophytes and secondary consumers. The dominant intertidal macrophytes in the PNW are the perennial, rooted aquatic seagrasses *Zostera marina* L. and

Z. japonica Aschers. & Graebn. and the benthic green macroalgae guild, where a guild is a set of species with similar ecological function. Two dominant benthic species in many PNW estuaries are the burrowing shrimp *Neotrypaea californiensis* and *Upogebia pugettensis*. Each of these primary producers and secondary consumers strongly affect other components of estuarine ecosystems, and can be considered ecological engineering species (sensu Jones et al., 1997). Specific ecological functions and interactions of these species/guilds are summarized below and in Table 1-1. Because of their importance in nutrient dynamics and food webs, these five taxa were chosen as practical biotic response measures, or endpoints, in our surveys to relate distributions of estuarine resources to nutrient sources.

Z. marina is the dominant seagrass species in PNW estuaries, forming dense beds in the intertidal and shallow subtidal zones (e.g., Kentula and McIntire, 1986; Thom, 1990; Thom et al., 2001). Seagrasses promote estuarine productivity and diversity by serving as nurseries and foraging areas for recreationally and commercially important fish and shellfish, providing bird habitat, increasing the density of benthic invertebrates, and contributing to estuarine primary production (e.g., Bayer, 1979; Hemminga and Duarte, 2000; Jackson et al., 2001; Ferraro and Cole, 2007). A second seagrass species, the nonindigenous *Z. japonica*, was targeted because it appears to be spreading in the PNW (e.g., Young et al., 2008). The ecological role of this non-native species is not well known, but appears to have both positive and negative ecological impacts (Posey, 1988; Larned, 2003; Wonham, 2003; Ruesink et al., 2006; Kaldy, 2006), and thus can be considered both a stressor and an ecological resource.

Another important primary producer assemblage in PNW estuaries is benthic macroalgae, which primarily consists of green ulvoid species. Benthic macroalgae were targeted in our field surveys because blooms of *Ulva* and *Enteromorpha* have been associated with eutrophication in other parts of the world (e.g., Valiela et al., 1997; Hauxwell et al., 2003) and have been used as an indicator of eutrophication (Bricker et al., 1999; CENR, 2003). However, benthic macroalgal blooms are an annual event in many PNW estuaries and are likely to be an important component of primary production (e.g., Phillips, 1984; Thom, 1984; Kentula and McIntire, 1986; Kentula and DeWitt, 2003; Thom et al., 2003; Bulthuis and Shull, 2006). Thus, as with *Z. japonica*, benthic macroalgae can be considered both a resource and a stressor.

In addition to the primary producers, the ghost shrimp, *N. californiensis*, and the mud shrimp, *U. pugettensis*, were selected as biotic indicators. Both species are harvested commercially and recreationally though they also have a negative effect on oyster aquaculture (Feldman et al., 2000). Both of these deep burrowing ecological engineers modify sediment properties by turning over large quantities of sediment and by irrigating large amounts of water through the sediment (DeWitt et al., 2004). In response to this intense bioturbation, sediment nitrogen fluxes are increased and shrimp-dominated tide flats are second only to the ocean as a source of dissolved inorganic nitrogen (DIN) during the summer in the Yaquina Estuary (DeWitt et al., 2004). While potentially stimulating phytoplankton by adding nutrients to the water column, burrowing shrimp simultaneously reduce phytoplankton concentrations by filtering large quantities of overlying water, perhaps as much as the entire volume of the Yaquina Estuary daily (Griffen et al., 2004). The intense bioturbation by *N. californiensis* also reduces the ability of *Z. japonica* to spread into unvegetated areas (Dumbauld and Wyllie-Echeverria, 2003).

Table 1-1. Ecological interactions of five taxa evaluated in the field surveys. The effects listed here are to illustrate the general types of interspecific and trophic interactions expected in Pacific Northwest estuaries and are not a complete listing of all biotic interactions.

TAXON	EFFECTS ON WATER COLUMN NUTRIENTS	EFFECTS ON PHYTOPLANKTON	EFFECTS ON *Z. MARINA*	EFFECTS ON BENTHIC INFAUNA	EFFECTS ON JUVENILE FISHES & CRABS	EFFECTS ON HIGHER TROPHIC LEVELS	EFFECTS OF NUTRIENT ENRICHMENT ON TAXON
Z. marina	Decrease by uptake and potential export from estuary via rafting	Compete for water column nutrient. Minor effect in PNW estuaries?	NA	Structure enhances species richness and density over bare sediment. Provides total organic carbon (TOC) to sediment.	Nursery	Habitat refuge and foraging for fish and crabs. Consumed by brant geese & American widgeon	Increase in turbidity and increase in epiphytes reduce lower depth limit of *Z. marina*.
Z. japonica	Decrease by uptake and potential export from estuary via rafting	Compete for water column nutrient. Minor effect in PNW estuaries?	Competitor in some estuaries	Structure enhances species richness and density over bare sediment. Provides TOC to sediment. Both to a lesser extent than *Z. marina*?	Nursery, though to lesser extent than *Z. marina*? Potential "habitat sink" for juvenile crabs during spring tides?	Habitat refuge and foraging for fish and crabs? Consumed by brant geese & American widgeon	Increase in turbidity and increase in epiphytes reduce lower depth limit? Stimulate growth at lower enrichment level?
Benthic macroalgae	Decrease by uptake (release during decay)	Compete for water column nutrient. Minor effect in PNW estuaries?	Impact at high biomass	Provides TOC to sediment. Detrimental at high levels due to smothering, shading, and H_2S formation	Minimal?	Consumed by some amphipods	Increase area covered and biomass, with impacts on SAV and benthos
Neotrypaea	Increase benthic flux	Minimal	Inhibits colonization	Reduces certain benthic guilds	Minimal	Prey	Decrease if low dissolved oxygen
Upogebia	Increase benthic flux	Decrease by feeding	Neutral or may facilitate colonization	Enhanced relative to bare sediment	Minimal	Prey	Stimulate due to higher phytoplankton at moderate levels? Decrease if low dissolved oxygen

1.5 Design and Objectives of the Pacific Northwest Regional Study

Based on our understanding of the nutrient dynamics and distribution of estuarine resources within the PNW, we focused our research on three of the previously identified management needs: 1) classification by nutrient loading and sources; 2) classification by the resources at risk; and 3) classification by estuarine vulnerability. Research to address these needs was conducted at two spatial scales. The first was at the landscape scale of entire estuaries and associated watersheds within the PNW, which is discussed in Chapter 2. The six primary objectives at the landscape scale were:

1) Identify and delineate all estuaries and associated watersheds within the PNW.

2) Classify estuaries by geomorphology and extent of river versus ocean influence.

3) Classify estuaries by spatial and temporal salinity patterns

4) Classify estuaries by wetland patterns.

5) Classify coastal watersheds by land cover patterns.

6) Integrate wetland and watershed classifications to identify functionally similar estuaries.

A total of 103 estuaries were identified within the PNW based on the most recent NWI data, with their sizes spanning more than five orders of magnitude. A variety of classification schema were generated from these landscape-scale attributes based on the concept that different types of classifications are useful for addressing different scientific or management issues. A "crosswalk" of the wetland and land use classifications was then generated to identify functionally similar estuaries.

The second spatial scale was at the level of the individual estuary where we conducted both field and aerial surveys. We identified seven target estuaries in Oregon spanning a range of size (3.1 to 54.9 km^2), geomorphology, and perceived ocean versus riverine influence. The seven target estuaries were the Alsea, Coos, Nestucca, Salmon River, Tillamook, Umpqua River, and Yaquina. One of the goals at the estuary scale was to determine patterns and drivers of water quality across a range of estuary types. The specific objectives of the water quality surveys and modeling were to:

1) Determine the within-estuary and among-estuary spatial patterns of water quality based on nutrients, chlorophyll *a*, and dissolved oxygen (Chapter 4).

2) Assess the among-estuary patterns of spatial and temporal variation in salinity (Chapter 4).

3) Based on water quality patterns, models, and isotope ratios in macroalgae, divide each of the estuaries into segments that are primarily dominated by ocean-derived nutrients versus riverine-derived nutrients (Chapter 5).

Another goal at the estuary scale was to quantify the within- and among-estuary distributions and abundances of each of the five biotic endpoints. Specific objectives of this research were to:

1) Determine *Z. marina*'s intertidal/shallow subtidal bathymetric distribution using aerial survey data (Chapter 6).

2) Using the aerial surveys of *Z. marina* distributions, assess the potential exposure to terrestrially derived nutrients by quantifying what proportions of the population occurred in the ocean- and river-dominated segments of the target estuaries (Chapter 6).

3) Based on field probabilistic surveys, assess the potential exposure of each of the five biotic endpoints to terrestrially derived nutrients by quantifying what proportions of the populations occurred in the ocean- and river-dominated segments of the target estuaries (Chapter 7).

4) Classify the target estuaries based on similarities in the relative abundances of the five biotic endpoints (Chapter 7).

5) Determine the lower depth limit of *Z. marina* to determine its relationship to ambient water clarity and estuary type (Chapter 8).

The final objective was to summarize the key patterns and processes that differentiate PNW estuaries from those on the East and Gulf coasts as well as those that result in different vulnerabilities to nutrient enrichment among PNW estuaries (Chapter 9). We suggest that this summary forms the nucleus of an emerging "Pacific Northwest Paradigm". This paradigm can, and should, be refined as additional data are collected and as we gain better insights into the oceanic, estuarine, hydrologic, climatic, watershed, and ecological processes affecting nutrient dynamics and biotic distributions.

CHAPTER 2:
REGIONAL CLASSIFICATION OF PACIFIC NORTHWEST ESTUARIES BY WETLAND AND LAND COVER PATTERNS

Henry Lee II, Deborah A. Reusser (USGS), Patti Haggerty (Indus Corp.),
Cheryl A. Brown, and Patrick J. Clinton

Key Findings

- **A census of the coastal waterbodies on the Pacific Coast identified 230 estuaries (exclusive of Puget Sound) of which 103 occurred in the Pacific Northwest (PNW). The majority of the PNW estuaries are small, with 73 of the 103 estuaries less than 1 km^2, and most have extensive intertidal habitat.**

- **PNW estuaries provide a suite of estuarine services and functions that vary by estuary size, with small- and moderate-sized estuaries ($\leq$ 10 km^2) disproportionately important to Coho smolts**.

- **PNW estuaries were classified based on geomorphology and extent of ocean exchange. Eight estuaries were classified as coastal lagoons, 6 as blind estuaries, 53 as tidally restricted coastal creeks, 2 as tidal coastal creeks, 3 as marine harbors/coves, 3 as bar-built estuaries, and 28 as drowned river mouth estuaries.**

- **The drowned river mouth estuaries were further divided into river- or tidal-dominated based on an "area-normalized freshwater inflow" metric, with 15 classified as "highly river-dominated", 7 as "moderately river-dominated", and 6 as tide-dominated estuaries. River-dominated estuaries have greater flushing and, hence, are less vulnerable to nutrient enrichment.**

- **The 31 major river mouth and bar-built estuaries were then classified using a clustering approach based on either wetland or watershed land cover patterns.**

- **A "crosswalk" was conducted to group estuaries that clustered together in both the wetland and watershed analysis. This approach identified groups of estuaries that presumably have similar nutrient dynamics and vulnerabilities.**

- **Most PNW estuarine watersheds display relatively low levels of alteration based on development, agriculture, population density, and impervious surfaces.**

2.0 Introduction

The overall goal of the regional classification was to evaluate the similarities and differences among Pacific Coast estuaries based on estuarine wetland distributions, land cover patterns in the associated coastal watersheds, and spatial and temporal patterns of salinity distributions. Wetland and watershed data were synthesized for all the coastal estuaries of the Pacific Coast from the Tijuana Estuary in Southern California (32.5574ºN) north to Cape Flattery in Washington (48.383ºN). The current effort focuses on the outer coastal estuaries of the Pacific Northwest (PNW), which is defined as the coastal segment from Cape Mendocino in Northern California (40.440ºN) to Cape Flattery in Washington. The Strait of Juan de Fuca and Puget Sound were not included in this analysis.

The first step in the analysis was a comprehensive delineation of all the coastal estuaries and watersheds to assure a one-to-one relationship between each waterbody and its associated watershed. Based on this inventory of PNW estuaries and watersheds, we addressed the following objectives:

1) Identify the broad resource types and general ecological services provided by PNW estuaries.

2) Group estuaries by their similarities in wetland patterns and estuarine landscape attributes, including development of estuarine-scale metrics of key physical/climatic drivers.

3) Group watersheds by their similarities in land cover patterns and other watershed attributes.

4) Conduct a matrix match ("crosswalk") of the classifications by wetlands and watersheds to identify estuaries overlapping in the two analyses. Use these groups to identify functionally similar estuaries and to help develop a research framework to evaluate the proposed classification schema.

5) Develop a baseline of estuarine and watershed landscape data for the PNW.

2.1 Estuarine, Wetland, and Landscape Data Sources and Methods

Achieving the five objectives of the regional classification required a synthesis of multiple types of estuarine and watershed data and GIS analyses. Figure 2-1 provides an overview of this synthesis to achieve objectives 1-4. In terms of the fifth objective, the summarized wetland and watershed data are provided in Tables 2-2, 2-3, and 2-4.

2.1.1 Estuary Definition and Inventory

A national standard, the National Wetland Inventory (NWI) (http://www.fws.gov/nwi/; U.S. Fish Wild. Ser., 2002), was used as the criterion for defining estuaries. NWI classifies aquatic habitat types using a hierarchical set of attributes based on Cowardin et al. (1979), and includes marine, estuarine, riverine, palustrine, and lacustrine areas with further subdivisions for tide height, substrate type, and the presence of broad classes of wetland plants (e.g., emergent vs. aquatic bed). These geospatial data were obtained from NWI digital databases and on-site digitization of paper maps to fill in data gaps not available in digital format. The NWI codes found in PNW estuaries are summarized in Table 2-1.

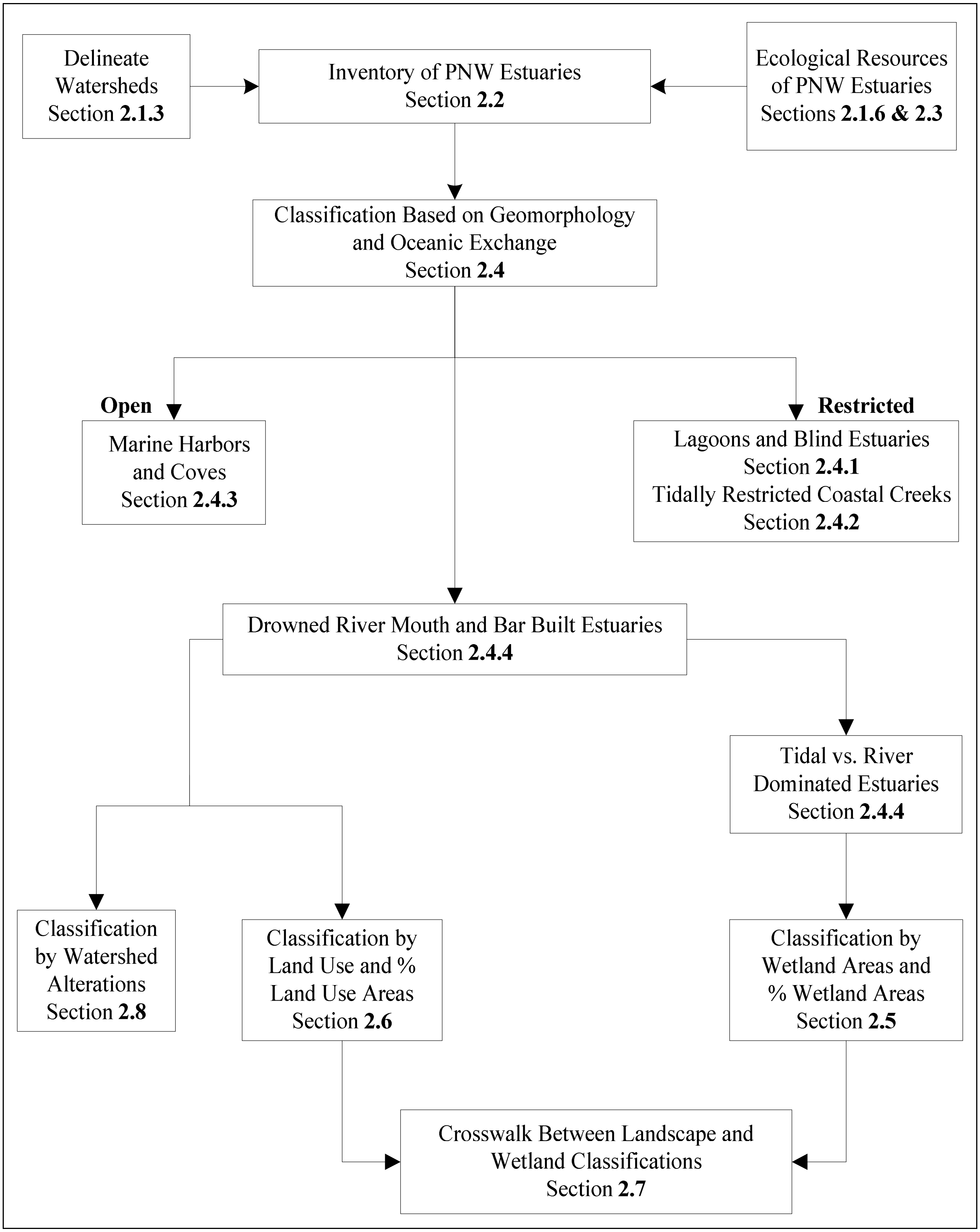

Figure 2-1. Overview of the generation of the PNW estuary inventory and the wetland and landscape data synthesis and analysis.

We defined estuaries as coastal water bodies that had a NWI estuarine polygon and that discharged directly into the ocean. Rivers, creeks, tributaries, and other water bodies within an estuary that do not discharge directly into the ocean were considered subestuaries. While only the estuarine polygons were used to identify estuaries, the sum of the marine, estuarine, and tidal riverine polygons was used to calculate the total estuarine area so as to capture the entire habitat area likely to contain euryhaline flora and fauna as well as to minimize the effects of any misclassifications of the salinity classes. The list of estuaries in the PNW was updated from those in the earlier version (Lee et al., 2006) based on the 2007 NWI revisions in the Pacific Northwest. In addition, any coastal water body that had an estuarine polygon in the 2001 NOAA coverage (see Section 2.1.3) but did not have an estuarine polygon in NWI was identified and delineated. These additional systems are listed for completeness but are not included in the wetland or watershed analyses. Semi-enclosed harbors or bays with only marine polygons and coastal streams with only tidal riverine polygons were delineated but were not classified as estuaries and were not included in the analyses.

Based on these criteria, 103 PNW estuaries were identified (Figures 2-2 to 2-5). Sixteen of these estuaries were in California, 24 in Washington, 62 in Oregon, and the Columbia River Estuary which is divided between Washington and Oregon. The geomorphological class of each of these estuaries is given in Table 2-2. The classification of estuaries as drowned river mouth, bar-built, and blind comes from various sources (e.g., Bottom et al., 1979; Seliskar and Gallagher, 1983; Cortright et al., 1987; Rumrill, 1998; Emmett et al., 2000) as well as our own analysis. The classification of estuaries as coastal lagoons, tidally restricted coastal creek, and marine harbor/coves is based on the arguments made in Section 2.4 as is our approach to quantitatively separating tide- versus river-dominated estuaries.

2.1.2 Wetland Habitats Based on NWI

The NWI provides a regional-scale dataset to classify estuaries by wetland type. An advantage of using wetlands to classify estuaries is that a biotic/habitat endpoint integrates a wide range of environmental conditions, such as salinity patterns, flushing, and nutrient loading. Thus, estuaries with similar NWI wetland patterns presumably will display similar responses to nutrient enrichment.

While the NWI provides a dataset to evaluate estuarine habitats at both local and regional scales, there are limitations. The NWI classifies habitats at a coarse resolution, such as “aquatic bed” or “unconsolidated shore”, and is thus unable to differentiate between benthic assemblages such as burrowing shrimp beds versus sand flats without shrimp. A second limitation is that some of the NWI data were generated in the late-1970s and early-1980s, and thus are historical snapshots of estuarine conditions. However the NWI is continuing to update its habitat maps, including an update of the Oregon estuaries in 2007. The list of estuaries and their sizes are based on these updated data so that some estuarine areas will differ slightly from the values in the earlier version of this document (Lee et al., 2006). A third limitation is that there was minimal field validation and some of the classifications may be incorrect. For example, our observations in Beaver Creek (Oregon) suggest that one potential error is the classification of tidal riverine wetlands as estuarine emergent wetlands in smaller coastal systems.

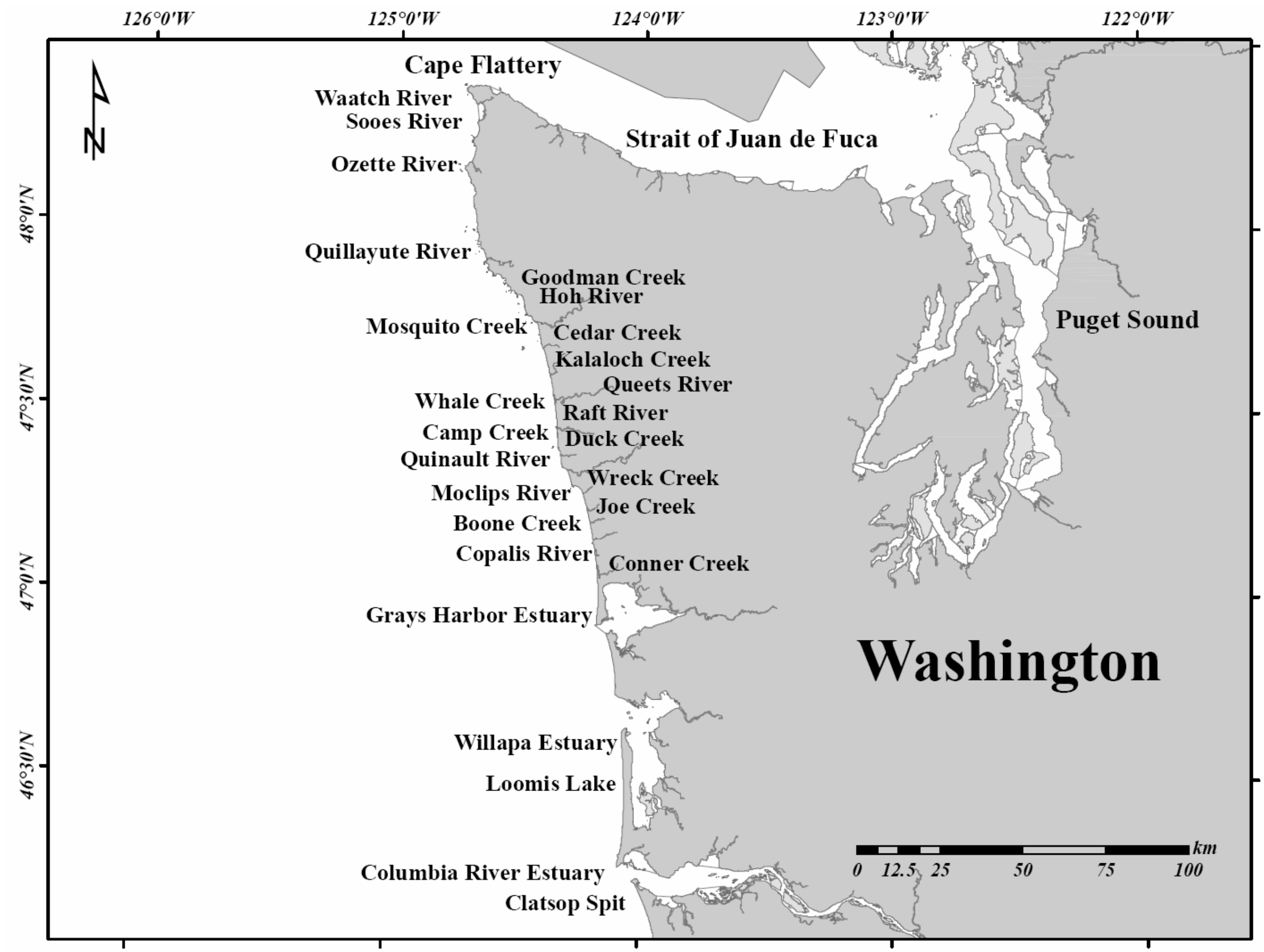

Figure 2-2. Estuaries along the Washington coast from Cape Flattery to the Columbia River.

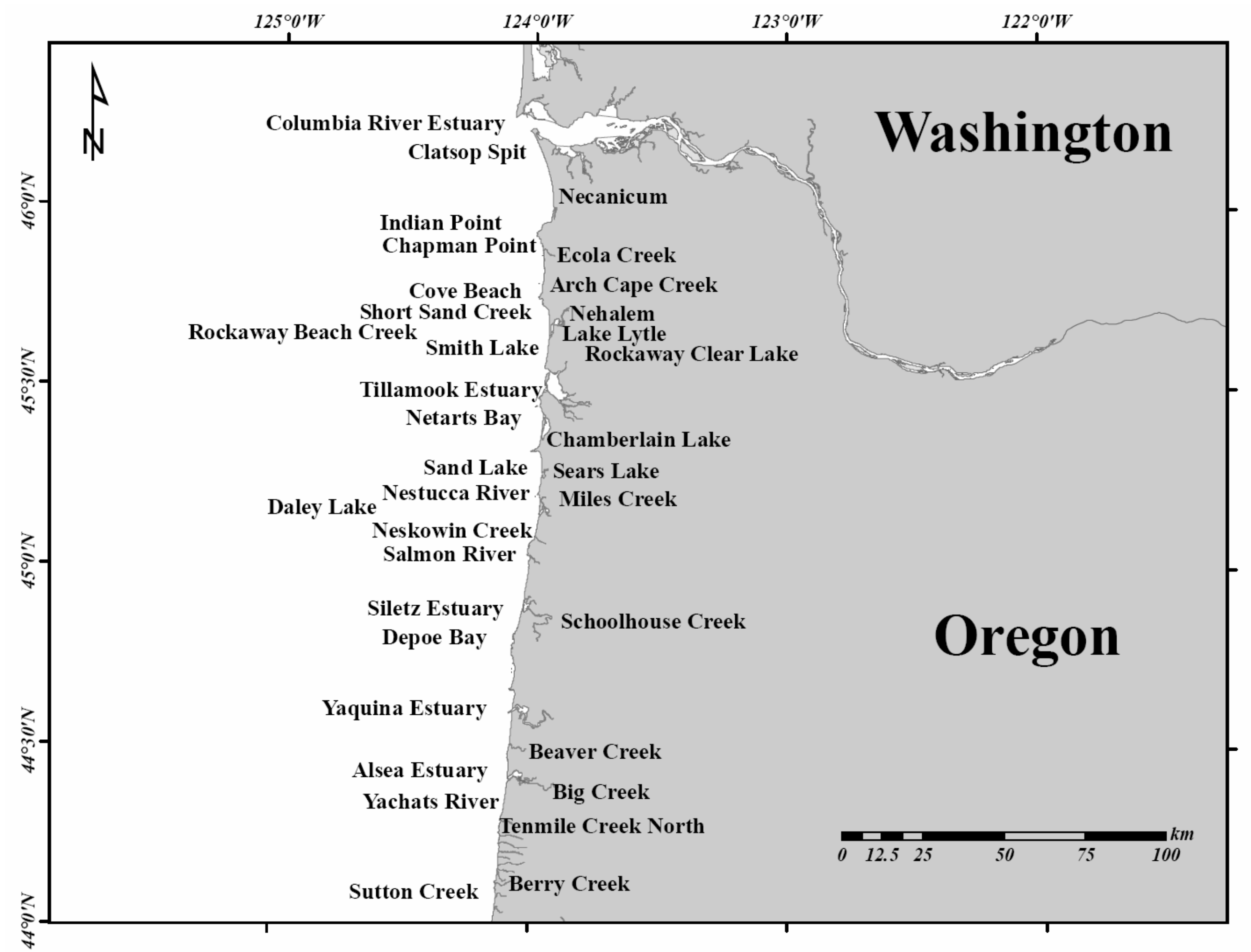

Figure 2-3. Estuaries in northern Oregon. The Alsea, Nestucca, Salmon River, Tillamook, and Yaquina estuaries were five of the seven target estuaries in the field surveys.

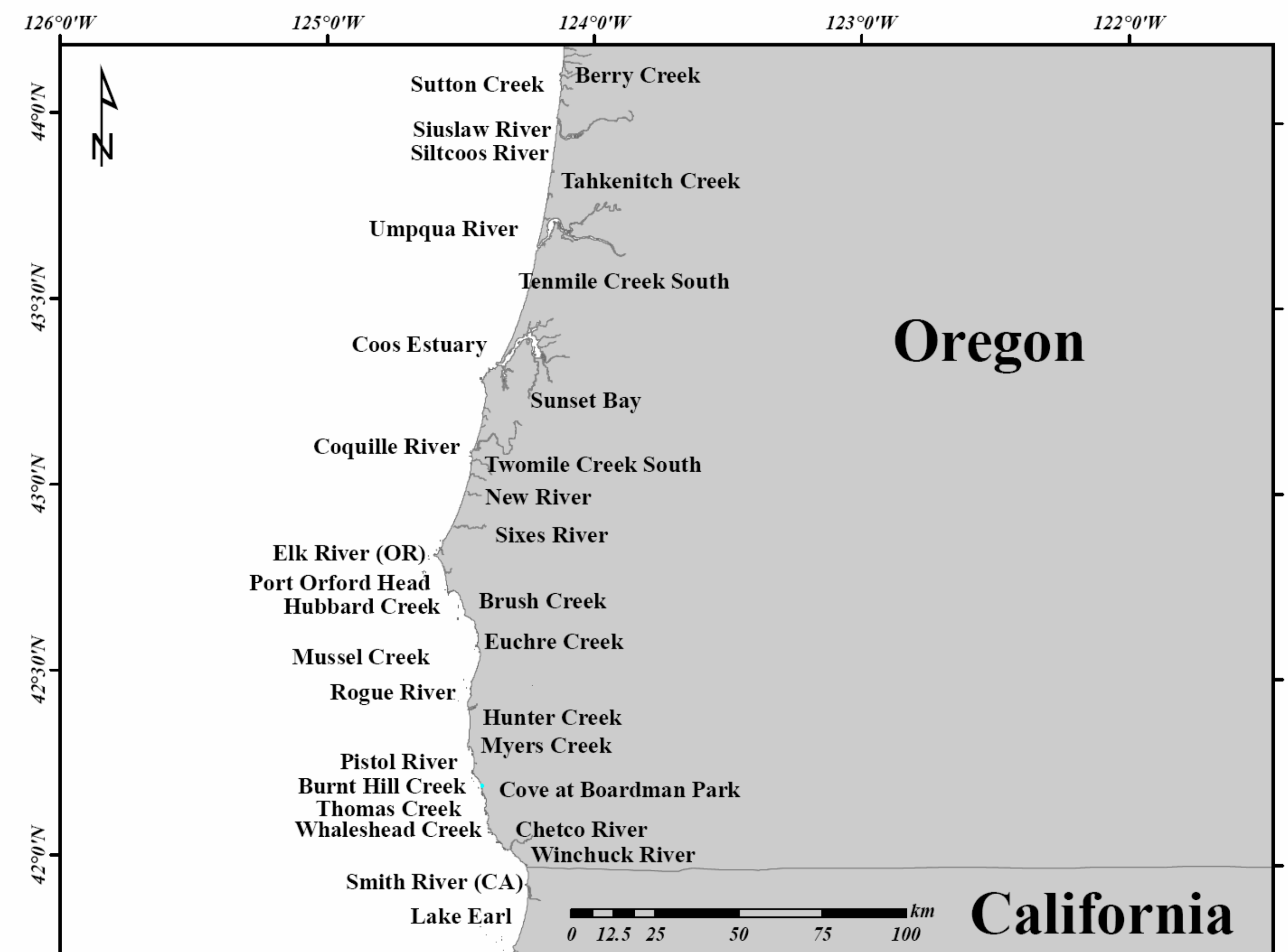

Figure 2-4. Estuaries in southern Oregon. The Umpqua River and Coos estuaries were two of the target estuaries in the field surveys.

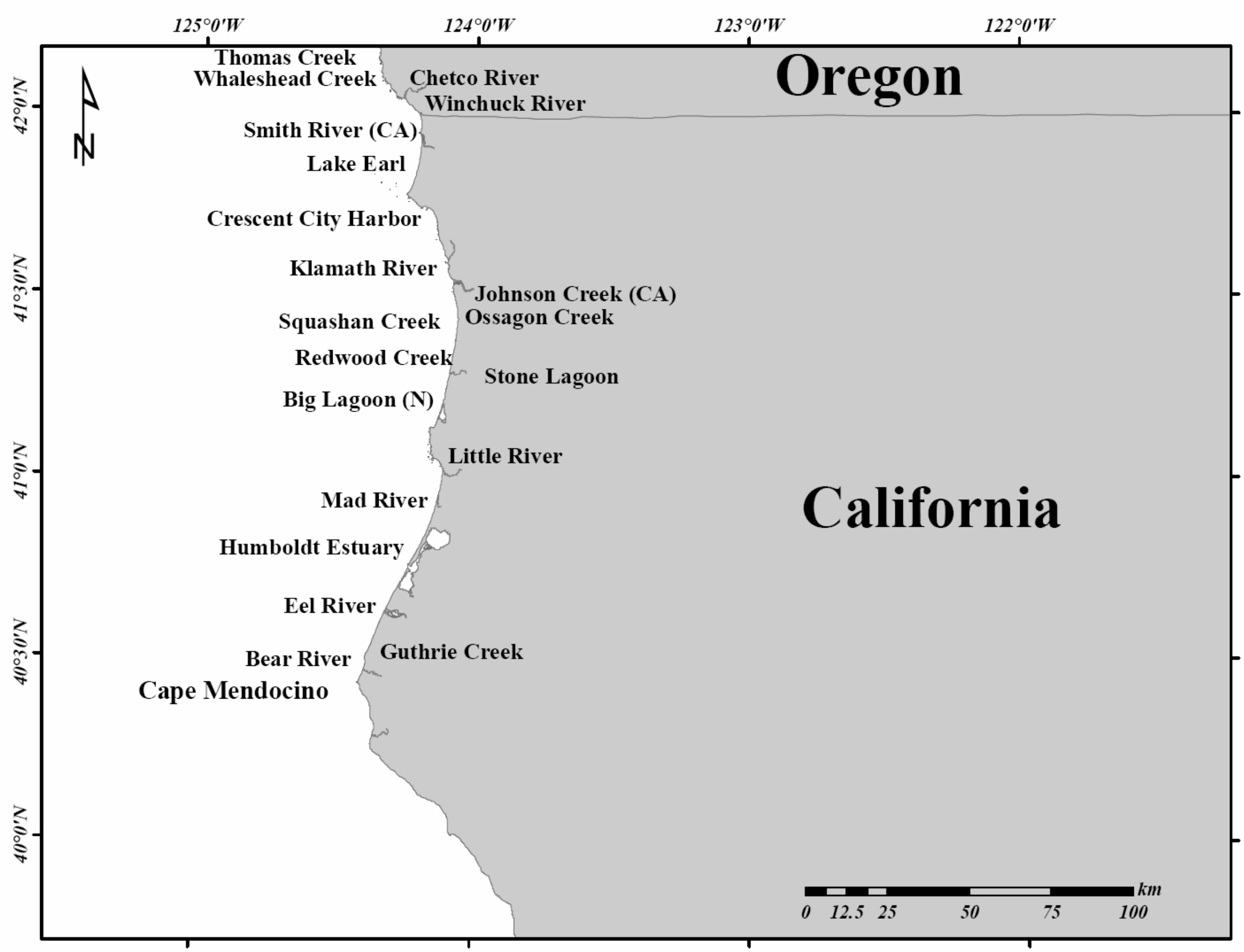

Figure 2-5. Estuaries in Northern California, north of Cape Mendocino.

One procedural challenge was that some of the older maps either had obsolete or incorrect NWI codes, though we were able to translate these into the current standards. Another challenge was the inconsistent resolution in the use of special habitat modifiers for salinity and sediment type among estuaries. For example, codes to indicate the specific salinity class (e.g., mesohaline) were included in a few estuaries but were lacking in many others. When conducting statistical clustering using the original NWI codes habitats with a higher level of resolution would be considered different from the exact same habitat that did not contain the special modifiers.

We took a number of steps to reduce the effects of these limitations on the classifications. As mentioned, we updated obsolete or incorrect codes; the updated codes are referred to as "corrected NWI codes." To minimize the effect of misclassification of riverine tidal wetlands as estuarine wetlands, we defined total estuarine area as the sum of marine, estuarine, and tidal riverine polygons. Thus, the total estuarine area would be the same regardless of the specific salinity designation within this range. Including the riverine tidal portion is ecologically justifiable as inclusion of the low salinity habitat captures the full habitat range for euryhaline and oligohaline species. To control for different levels of habitat resolution, we derived "consolidated NWI codes" that merged the original and corrected NWI codes into broader habitat classes (Table 2-1). For example, both "regularly flooded emergent wetland" and "regularly flooded rooted aquatic bed" were consolidated into a "regularly flooded rooted" class. The original 118 marine, estuarine, and tidal riverine NWI codes in the PNW estuaries were merged into 48 consolidated codes (Table 2-1). These consolidated NWI codes were used in the classification analysis to assure similar levels of resolution in the clustering. Additionally, merging into broader habitat classes helped to minimize the effects of certain wetland misclassifications (e.g., a site classified as emergent vegetation versus a rooted aquatic bed). Merged NWI codes have been used in previous efforts, such as in NOAA's "Spatial Wetland Assessment for Management and Planning" program (SWAMP; Sutter, 2001) and in wetland site prioritizations (Brophy, 2003). Brophy (2003) suggested that merging NWI polygons better identifies the "high biological value of a large, hydrologically interconnected wetland."

With these steps to standardize the data, the NWI should provide sufficient accuracy and resolution to address regional scale wetland patterns. As stated in Oregon's "Wetlands Inventory User's Guide" (Oregon Division of State Lands, no year), "The NWI provides excellent information for a variety of planning purposes" which include the identification of "the general location, extent, and type of wetlands on a regional basis, such as watershed or on tribal lands." Also, as pointed out in the Oregon Watershed Assessment Manual (Brophy, 2007), the revised NWI is a suitable base layer for estuarine assessments and is an alternative to hydrogeomorphic (HGM) maps. Nonetheless, the NWI does not have the resolution of detailed site-specific studies, such as conducted previously in Oregon (Cortright et al., 1987). It is beyond the scope of this regional assessment to conduct a detailed comparison of the NWI wetland classes with the previous habitat delineations in Oregon, but a cursory comparison suggests that there is reasonable agreement on broad habitat classifications (e.g., aquatic beds, subtidal). The Oregon study does, however, provide details as to specific habitat type, such as the importance of algae growing on cobble/gravel in the Chetco River, which provide insights not possible from the broader NWI classes.

Table 2-1. Consolidation of National Wetland Inventory (NWI) habitat codes found in PNW estuaries. The NWI broad habitat classes (estuarine, marine, tidal riverine) and tidal heights (subtidal, irregularly exposed, regularly flooded, irregularly flooded) are given at the top in capitals and the consolidated habitat types falling within that salinity-tide height are given in the cells underneath in bold. The original NWI codes making up the consolidated class are listed beneath the consolidated habitat. Original NWI codes that are obsolete or incorrect are given in italics with our interpretation of the corrected code in parenthesis. The "rooted" classes in this analysis are used to capture the NWI "EM" codes (emergent = "erect, rooted, herbaceous hydrophytes") and any other codes that are identified as rooted or persistent. The "aquatic bed" classes are used to capture all "AB" codes (aquatic bed = "habitats dominated by plants that grow principally on or below the surface of the water for most of the growing season in most years") other than those identified as rooted or persistent. In PNW estuaries, areas classified as aquatic beds may be covered with SAV, macroalgae, or a combination of both. Because of the uncertainty of the presence of vegetation with the "unconsolidated shore" codes, they are classified as unvegetated though they may contain up to 30% vegetation. Discussion of NWI codes can be found in Cowardin et al. (1979) and Smith (1991) while an online translator is available at http://www.fws.gov/wetlands/Data/webatx/atx.html.

ESTUARINE - SUBTIDAL								
Unvegetated	**Unvegetated – Excavated**	**Unvegetated – Diked/ Impounded**	**Aquatic Bed**	**Rooted**				
E1UBL, E1UB2L, E1UB3L, *E1OWL (E1UBL)*	E1UBLx	E1UBLh	E1ABL	E1AB3L				
ESTUARINE – IRREGULARLY EXPOSED								
Unvegetated	**Unvegetated – Cobble-Gravel**	**Unvegetated - Streambed**	**Aquatic Bed**	**Rooted**				
E2USM, E2US2M, E2US3M, *E2FLM (E2USM)*	*E2UB1M (E2US1M)*	E2SBM	E2ABM, *E2AB/FLM (E2ABM)*, E2AB/USM	E2AB3M, E2EM1M				

ESTUARINE – REGULARLY FLOODED								
Unnvegetated	**Unvegetated - Streambed**	**Unvegetated - Spoils**	**Unvegetated - Excavated**	**Aquatic Bed**	**Rooted**	**Rooted – Diked/ Impounded**	**Rooted - Spoil**	**Rooted - Excavated**
E2USN, *E2BBN (E2USN)*, *E2FLN (E2USN)*, E2US2N, E2US3N, *E2FL3N (E2US3N)*, *E2FL6N (E2USN)*	E2SBN	E2USNS	E2USNx	E2AB1N, E2EM/AB1N, E2AB1/US3N, E2US3/AB1N, E2ABN, *E2AB/FLN (E2ABN)*, E2AB/USN, E2AB2/US3N, E2US/ABN	E2EMN, E2EM/ABN, E2EM1N, *E2EM1/FLN (E2EM1N)*, E2EM1/US3N, E2EM1/ABN, E2EM/USN, E2EM1/USN, E2EM5N	E2EM5Nh	E2EMNs	E2EMNx
ESTUARINE – REGULARLY FLOODED (CONT.)								
Shrub-Scrub	**Rocky**							
E2SSN	E2RSN							

ESTUARINE – IRREGULARLY FLOODED								
Unnvegetated	**Unvegetated – Cobble-Gravel**	**Rocky**	**Rooted**	**Rooted – Diked/ Impounded**	**Rooted - Spoils**	**Shrub-Scrub**	**Forested**	
E2US2P, *E2BBP (E2USP)*, *E2FLP (E2USP)*, E2USP	E2US1P	*E2RSPR (E2RSPr)*	*E2EM1/FLP (E2EM1P)*, E2EM1P, E2EM5P, E2EMP, *E2EM1R (E2EM1P)*	*E2EM5Ph* (E2EMPh)	E2EMPs	E2SSP, *E2US/SSP (E2SSP)*, *E2SS/EM1P (E2SSP)*, E2SS1P	E2FOP, E2FO1P, E2FO5/EM1P, E2FO4/1P, E2FO4P, *E2FO4/EM1P (E2FO4P)*	
MARINE – SUBTIDAL & REGULARLY FLOODED, & IRREGULARLY FLOODED								
Irregularly Flooded - Unvegetated	**Irregularly Flooded – Aquatic Bed**	**Irregularly Flooded – Rocky**	**Regularly Flooded - Unvegetated**	**Regularly Flooded – Rocky – Vegetated & Unvegetated**	**Subtidal - Rooted**	**Subtidal - Unvegatated**		
M2USP, M2US2P	M2AB1/USN	M2RS2P, M2RSNr, *M2RSPR (M2RSPr)*	M2USN, M2US2N	M2RSN, M2RS/ABN, M2AB/RSN	M1AB3	M1UBL		

RIVERINE – TIDAL – PERMANENTLY & REGULARLY & SEMIPERMANENTLY FLOODED								
Permanently Flooded Tidal - Unvegetated - Subtidal	**Permanently Flooded Tidal – Unvegetated – Subtidal – Excavated**	**Permanently Flooded Tidal - Aquatic Bed**	**Regularly Flooded – Unvegetated - Shore**	**Semipermanently Flooded Tidal – Unvegetated - Subtidal**	**Semipermanently Flooded Tidal – Unvegetated - Shore**			
R1OWV R1UBV	R1UBVx	R1ABV	R1USN, *R1FLN (R1USN)*	R1UBT	R1UST, *R1FLT (R1UST)*			
RIVERINE – TIDAL – SEASONALLY & TEMPORARILY & IRREGULARLY FLOODED								
Seasonally Flooded Tidal - Unvegetated - Shore	**Seasonally Flooded Tidal - Unvegetated – Shore - Spoil**	**Seasonally Flooded Tidal - Rooted**	**Seasonally Flooded Tidal - Streambed**	**Temporarily Flooded Tidal – Unvegetated - Shore**	**Irregularly Flooded – Unvegetated - Shore**			
R1USR, *R1FLR (R1USR), R1SS/FLR (R1USR)*	*R1USRS (R1USRs)*	R1EMR	*R1SS/USR (R1SB/USR)*	R1USS	*R1FLY (R1USP)*			

2.1.3 Watershed Delineation

The watershed associated with each coastal estuary on the Pacific Coast was delineated, so that there was a unique one-to-one relationship between the estuary and its watershed. The estuarine watershed geospatial layer was derived and augmented from the watershed layer originally generated for NOAA's Coastal Assessment Framework (ftp://sposerver.nos.noaa.gov/datasets/CADS/GIS_Files/ShapeFiles/caf/). This layer was not sufficiently detailed to represent the smaller estuaries on the Pacific Coast, requiring further delineation of a number of coastal watersheds in California, Oregon, and Washington. Watershed boundaries subtending these estuary sites were determined from several sources. In Washington, Oregon, and northwest California, the sixth field hydrologic unit (HUC) sub-watershed geospatial layer created by the U.S. Forest Service from 1:24,000 scale USGS maps, digital elevation models, and other data sources (http://www.reo.gov/gis/projects/watersheds/REOHUCv1_3.htm) were used as primary references. A watershed layer refined to the seventh field HUC boundary lines (http://www.fsl.orst.edu/clams/cfsl0233.html) was also used for most of coastal Oregon north of the Rogue River. The digital basin layer CALWATER was used as the primary source for major basin delineations in central and southern California. CALWATER is the "California Interagency Watershed Map" produced in 1999 and updated in 2004, and represents the State of California's working definition of watershed boundaries (http://gis.ca.gov/meta.epl?oid=22175).

In all states, final refinements to the drainage boundaries were based on review of the hydrologic drainage patterns derived from digital elevation data (10-meter horizontal resolution in Oregon and 30-meter horizontal resolution in Washington and California) and from USGS 1:24,000 scale quadrangle maps. In addition, photographs and digital imagery for coastal features were examined from the California Coastal Records Project (http://www.californiacoastline.org), State of Washington database of shoreline photos (http://apps.ecy.wa.gov/shorephotos/), Terraserver (http://terraserver-usa.com/), and other on-line imagery sources. Digital boundaries for extraordinary sites – the Columbia River Basin, the interior portion of the San Francisco watershed, and the perimeter of the Tijuana River watershed – were located from additional sources and incorporated as being the "best available data". Boundary lines and water bodies were plotted and reviewed for accuracy of coding and fidelity to the original sources. In some cases, adjustments were made to the attribute coding of water bodies to reflect judgments that these were part of an estuarine system.

The entire watershed was delineated for each estuary (see Table 2-2). By delineating the entire watershed, these drainage areas are equivalent to the sum of NOAA's Estuarine Drainage Area (EDA, portion of watershed that empties directly into the estuary and is affected by tides) and Fluvial Drainage Area (FDA, component of an estuary's watershed upstream of the EDA boundary; http://www.csc.noaa.gov/crs/lca/gloss.html). Entire watershed areas were analyzed rather than the area represented by EDAs to capture the entire landscape contributing to the nutrient loading of the estuary. For the Columbia, the tidal portion of the Columbia River watershed was delineated up to the Bonneville Dam using the NOAA land cover data while the entire Columbia Basin, including into Canada, was delineated by the Interior Columbia Basin Ecosystem Project Management Project (ICBEMP, 1996; http://www.icbemp.gov). The NOAA land cover data truncated the northeastern end of the Klamath watershed which was filled in using the National Land Cover Data (NLCD; http://www.epa.gov/mrlc/nlcd-2001.html) data.

Because of the difficulty in delineating watersheds for the smallest estuaries (<0.1 km^2), the estimates for these systems may overestimate the actual area draining into these systems. We also identified the coastal areas not containing an estuarine polygon that drain directly into the ocean, which are referred to as Coastal Drainage Areas (CDAs) by NOAA. The watershed associated with each CDA was delineated, but these results are not reported here. A number of watershed areas calculated here differ from those previously reported by NOAA's Coastal Assessment Framework because multiple estuaries contained within a single NOAA EDA were split out to create a one-to-one relationship between each estuary and its watershed.

2.1.4 Land Cover Patterns, Impervious Surfaces, and Population Density

The land cover pattern of each coastal watershed was determined using the 1992 National Land Cover Data (NLCD, http://www.mrlc.gov) and both the 1995 and 2001 land cover data from NOAA's Coastal Change Analysis Program (C-CAP, http://www.csc.noaa.gov/crs/lca/ccap.html). These data were derived from Landsat satellite imagery and produced at a 30-meter spatial resolution with an overall target accuracy requirement of 85%. The present analysis primarily utilized the 2001 NOAA data since they are the most recent, though the earlier NLCD data were used to fill in gaps such as portions of the Klamath watershed. The 2001 NOAA data are based on 22 land cover classes (http://www.csc.noaa.gov/crs/lca/oldscheme.html), which are not exactly the same as used in the NLCD. The percent impervious surface was calculated for each of the estuarine watersheds from the 2001 NLCD land cover data (http://www.epa.gov/mrlc/nlcd-2001.html). This dataset estimated the percent impervious surfaces on a scale of 0 to 100% by 30 meter cells over the entire contiguous United States, the highest resolution of impervious surfaces available at a regional scale. The earlier version of this report (Lee et al., 2006) calculated impervious surfaces using the default impervious coefficients in Attila for different land cover classes (U.S. EPA, 2004c).

The population within each coastal watershed was calculated using both the 1990 and 2000 census data. This analysis was conducted at the census block scale, the smallest unit used by the Census Bureau. Population was prorated by area for census blocks that were transected by a watershed boundary. Population density (# per km^2) was calculated for each watershed using the total delineated watershed area and the percent population change from 1990 to 2000 was calculated.

2.1.5 Classification Strategy and Methods

We used a "hybrid" classification approach that combined qualitative analysis with more formal statistical methods. The qualitative analysis was used to initially separate out major types of estuaries primarily based on nutrient forcing functions and geomorphology especially as it related to oceanic exchange. An advantage of this type of analysis is that it allows the identification of broad groups of estuaries based on factors that are recognized as critical drivers but for which there is either a lack of quantitative data and/or a lack of suitable metrics to incorporate into a statistical analysis. A qualitative analysis also allows us to incorporate temporally variable or intermittent attributes, such as whether an estuary periodically closes off at the mouth ("blind" or intermittent estuary).

Table 2-2. Area and geomorphology of PNW estuaries and watersheds. This table lists all coastal water bodies with a NWI estuarine polygon, with the areas (km^2) for the marine, estuarine, and tidal riverine NWI habitats. "Total Estuary" is the sum of the area of the three NWI habitat classes, and is the area used for estuaries in our analyses. The total watershed area (km^2) from the 2001 NOAA land cover analysis is given for the associated watersheds. The watershed area for the Columbia River is up to the Bonneville Dam. The four italicized water bodies are those that have an estuarine emergent wetland or estuarine aquatic bed polygon in the 2001 NOAA land cover analysis (see Table 2-3) but do not have a NWI estuarine polygon. These additional "estuaries" are included for completeness but are not included in the analyses. Alternative geomorphological classifications are given in parentheses. * = new estuary not included in the previous analysis (Lee et al., 2006).

ESTUARY	LATITUDE	WATERSHED AREA (km^2)	AREA (km^2)				ESTUARY TYPE
			MARINE	ESTUARINE	TIDAL RIVERINE	TOTAL ESTUARY	
Waatch River	48.344	38.4	0.0	0.93	0.24	1.16	Tide-dominated drowned river mouth
Hobuck Creek	*48.336*	*2.3*	*0.0*	*0.0*	*0.0*	*0*	*Tidally restricted coastal creek? (Coastal lagoon?)*
Sooes River	48.324	107.9	0.0	0.50	0.07	0.56	Highly river-dominated drowned river mouth
Ozette River	48.181	232.1	0.0	0.03	0.0	0.03	Tidally restricted coastal creek
Quillayute River	47.908	1,625.0	0.0	0.50	0.46	0.96	Highly river-dominated drowned river mouth
Goodman Creek	47.823	81.7	0.0	0.03	0.0	0.03	Tidal coastal creek
Mosquito Creek	47.798	43.8	0.0	0.01	0.0	0.01	Tidally restricted coastal creek
Hoh River	47.749	773.4	0.0	0.11	0.73	0.84	Highly river-dominated drowned river mouth
Cedar Creek	47.711	26.9	0.0	0.02	0.0	0.02	Tidally restricted coastal creek
Kalaloch Creek	47.607	45.4	0.0	0.02	0.0	0.02	Tidally restricted coastal creek
Queets River	47.544	1,166.3	0.0	0.60	0.84	1.43	Highly river-dominated drowned river mouth
Whale Creek	47.490	32.1	0.0	0.01	0.0	0.01	Tidally restricted coastal creek
Raft River	47.463	204.0	0.0	0.17	0.07	0.24	Tidally restricted coastal creek (Blind – Drowned river mouth)
Camp Creek	47.398	22.7	0.0	0.03	0.0	0.03	Tidally restricted coastal creek
Duck Creek	47.387	18.4	0.0	0.01	0.0	0.01	Tidally restricted coastal creek?
Quinault River	47.349	1,133.7	0.0	0.42	0.24	0.66	Highly river-dominated drowned river mouth

ESTUARY	LATITUDE	WATERSHED AREA (km^2)	AREA (km^2)				ESTUARY TYPE
			MARINE	ESTUARINE	TIDAL RIVERINE	TOTAL ESTUARY	
Wreck Creek	47.284	17.9	0.0	0.01	0.0	0.01	Tidally restricted coastal creek
Moclips River	47.248	84.2	0.0	0.08	0.02	0.10	Tidally restricted coastal creek (Tidal coastal creek)
Joe Creek	47.206	60.8	0.0	0.05	0.0	0.05	Tidally restricted coastal creek
Boone Creek	47.159	20.3	0.0	0.01	0.01	0.01	Tidally restricted coastal creek
Copalis River	47.126	105.3	0.0	0.78	0.08	0.86	Moderately river-dominated drowned river mouth
Conner Creek	47.091	36.9	0.0	0.17	0.0	0.17	Tidally restricted coastal creek?
Grays Harbor	46.950	6,981.3	0.0	254.39	8.34	262.73	Tide-dominated drowned river mouth (Bar built)
Willapa	46.373	2,774.5	0.0	389.74	1.12	390.86	Tide-dominated drowned river mouth (Bar built)
Loomis Lake Creek	46.490	5.6	0.0	0.01	0.0	0.01	Tidally restricted coastal creek
Columbia River	46.263	14,520.8 (to Bonneville Dam)	0.0	411.53	257.45	668.98	Highly river-dominated drowned river mouth
Clatsop Spit*	46.277	0.7	0.0	0.08	0.0	0.08	Tidally restricted coastal creek? (Tidal coastal lagoon?)
Necanicum	46.011	216.8	0.0	1.57	0.06	1.63	Moderately river-dominated drowned river mouth (Bar built)
Indian Creek	45.931	7.0	0.0	0.004	0.0	0.004	Tidally restricted coastal creek (Tidal coastal creek)
Chapman Point	45.915	0.7	0.0	0.001	0.0	0.001	Tidally restricted coastal creek?
Ecola Creek	45.899	54.7	0.0	0.06	0.002	0.06	Tidally restricted coastal creek
Arch Cape Creek*	45.803	11.6	0.0	0.001	0.0	0.001	Tidally restricted coastal creek? (Tidal coastal lagoon?)
Cove Beach*	45.794	0.4	0.0	0.005	0.0	0.005	Tidally restricted coastal creek?
Short Sand Creek*	45.760	14.9	0.0	0.002	0.001	0.003	Tidally restricted coastal creek
Nehalem	45.658	2,215.2	0.0	10.43	1.23	11.65	Highly river-dominated drowned river mouth
Lake Lytle	45.636	7.4	0.0	0.0004	0.0034	0.004	Coastal lagoon

ESTUARY	LATITUDE	WATERSHED AREA (km^2)	AREA (km^2)				ESTUARY TYPE
			MARINE	ESTUARINE	TIDAL RIVERINE	TOTAL ESTUARY	
Rockaway Beach Creek*	45.613	2.2	0.0	0.0004	0.0	0.0004	Tidally restricted coastal creek
Rockaway Clear Lake*	45.605	1.6	0.0	0.0005	0.0	0.0005	Tidally restricted coastal creek
Smith Lake	45.596	9.2	0.0	0.009	0.005	0.015	Coastal lagoon (Tidally restricted coastal creek)
Tillamook	45.513	1,455.3	0.0	36.98	0.51	37.48	Tide-dominated drowned river mouth
Netarts Bay	45.402	46.4	0.0	10.43	0.001	10.43	Bar built
Chamberlain Lake	45.308	0.5	0.0	0.04	0.0	0.04	Coastal lagoon?
Sand Lake	45.276	51.5	0.0	4.28	0.0	4.28	Bar built
Sears Lake	45.247	1.5	0.0	0.002	0.0	0.002	Coastal lagoon
Miles Creek	45.231	2.7	0.0	0.06	0.0	0.06	Tidally restricted coastal creek
Nestucca	45.182	826.3	0.0	4.65	0.35	5.00	Highly river-dominated drowned river mouth
Daley Lake*	45.124	5.1	0.0	0.001	0.0	0.001	Tidally restricted coastal creek (Tidal coastal lagoon)
Neskowin Creek	45.100	53.3	0.0	0.009	0.005	0.014	Tidally restricted coastal creek
Salmon	45.046	192.6	0.0	3.08	0.04	3.11	Moderately river-dominated drowned river mouth (Bar built)
Siletz	44.903	954.8	0.0	7.48	1.38	8.86	Moderately river-dominated drowned river mouth
Schoolhouse Creek	44.873	2.9	0.0	0.01	0.0	0.01	Coastal lagoon (Tidally restricted coastal creek)
Depoe Bay	44.808	13.4	0.0	0.04	0.0	0.04	Marine harbor/cove (Drowned river mouth)
Yaquina	44.620	650.5	0.0	18.97	0.99	19.96	Tide-dominated drowned river mouth
Beaver Creek	44.524	87.2	0.0	0.53	0.02	0.55	Tidally restricted coastal creek
Alsea	44.422	1,221.6	0.0	12.49	0.0	12.49	Moderately river-dominated drowned river mouth
Big Creek	44.370	21.8	0.0	0.09	0.0	0.09	Tidally restricted coastal creek
Yachats	44.309	112.8	0.0	0.11	0.0	0.11	Tidally restricted coastal creek
Tenmile Creek North	44.225	60.6	0.0	0.04	0.0	0.04	Tidally restricted coastal creek

ESTUARY	LATITUDE	WATERSHED AREA (km^2)	AREA (km^2)				ESTUARY TYPE
			MARINE	ESTUARINE	TIDAL RIVERINE	TOTAL ESTUARY	
Berry Creek	44.094	9.4	0.0	0.02	0.0	0.02	Tidally restricted coastal creek (Tidal coastal creek)
Sutton Creek	44.060	38.5	0.0	0.15	0.0	0.15	Tidally restricted coastal creek
Siuslaw River	44.017	2,008.7	0.0	15.59	0.0	15.59	Moderately river-dominated drowned river mouth
Siltcoos River	43.873	185.3	0.0	0.33	0.04	0.36	Tidally restricted coastal creek
Tahkenitch Creek	43.815	93.1	0.0	0.07	0.19	0.26	Tidally restricted coastal creek
Umpqua River	43.669	12,146.2	0.0	27.73	6.05	33.78	Highly river-dominated drowned river mouth
Tenmile Creek South	43.561	222.9	0.0	0.04	0.46	0.50	Tidally restricted coastal creek
Coos	43.429	1,575.5	0.0	54.20	0.70	54.90	Tide-dominated drowned river mouth
Sunset Bay	43.335	14.7	0.0	0.12	0.0	0.12	Marine harbor/cove
Twomile Creek North	*43.236*	*8.9*	*0.0*	*0.0*	*0.0008*	*0.0008*	*Tidal coastal creek (no NWI estuarine polygon)*
Coquille River	43.123	2,729.8	0.0	5.08	1.81	6.89	Highly river-dominated drowned river mouth
Twomile Creek South	43.044	40.7	0.0	0.11	0.0	0.11	Tidally restricted coastal creek
New River	43.001	329.0	0.0	1.63	0.04	1.67	Blind – Drowned river mouth
Sixes River	42.853	347.5	0.0	0.31	0.08	0.39	Blind – Drowned river mouth
Elk River	42.793	236.4	0.0	0.51	0.16	0.66	Blind – Drowned river mouth
Port Orford Head	42.746	0.3	0.0	0.01	0.0	0.01	Tidally restricted coastal creek (Tidal creek)
Hubbard Creek*	42.734	17.5	0.0	0.01	0.0	0.01	Tidally restricted coastal creek
Brush Creek	42.685	28.5	0.0	0.02	0.0004	0.02	Tidally restricted coastal creek
Mussel Creek	42.616	26.6	0.0	0.02	0.0	0.02	Tidally restricted coastal creek
Euchre Creek	42.564	96.6	0.0	0.11	0.001	0.12	Tidally restricted coastal creek
Gregg Creek*	42.546	6.7	0.0	0.02	0.001	0.02	Tidally restricted coastal creek
Rogue River	42.422	13,500.9	0.0	1.32	1.44	2.77	Highly river-dominated drowned river mouth
Hunter Creek	42.386	115.2	0.0	0.07	0.02	0.1	Tidally restricted coastal creek

ESTUARY	LATITUDE	WATERSHED AREA (km^2)	AREA (km^2)				ESTUARY TYPE
			MARINE	ESTUARINE	TIDAL RIVERINE	TOTAL ESTUARY	
Myers Creek	42.307	14.7	0.0	0.02	0.0	0.02	Tidally restricted coastal creek (Tidal coastal creek)
Pistol River	42.28	271.9	0.0	0.44	0.11	0.55	Blind – Drowned river mouth
Burnt Hill Creek*	42.232	7.0	0.0	0.004	0.0	0.004	Tidal coastal creek?
Cove at Boardman Park*	42.216	0.4	0.0	0.005	0.0	0.005	Tidally restricted coastal creek?
Thomas Creek*	42.166	7.0	0.0	0.01	0.0	0.01	Tidally restricted coastal creek
Whaleshead Creek*	42.144	0.0	0.0	0.02	0.0	0.02	Tidally restricted coastal creek?
Chetco River	42.045	911.5	0.0	0.40	0.32	0.72	Highly river-dominated drowned river mouth
Winchuck River	42.005	184.6	0.0	0.09	0.03	0.12	Blind – Drowned river mouth
Smith River	41.945	1,942.6	0.0	2.04	0.34	2.38	Highly river-dominated drowned river mouth
Lake Earl	41.826	73.1	0.0	9.01	0.0	9.01	Coastal lagoon
Crescent City Harbor	41.744	29.0	1.61	0.01	0.0	1.63	Marine harbor/cove
Klamath River	41.547	40,580.9	0.0	1.09	1.18	2.27	Highly river-dominated drowned river mouth
Johnson Creek	41.463	1.4	0.0	0.01	0.0	0.01	Tidally restricted coastal creek
Ossagon Creek	41.442	2.5	0.0	0.01	0.0	0.01	Tidally restricted coastal creek (Coastal lagoon?)
Squashan Creek	41.389	0.9	0.0	0.01	0.0	0.01	Tidally restricted coastal creek
Redwood Creek	41.292	731.7	0.0	0.22	0.09	0.30	Blind – Drowned river mouth
Freshwater Lagoon	*41.269*	*5.5*	*0.0*	*0.0*	*0.0*	*0.0*	*Coastal lagoon (permanently blocked)*
Stone Lagoon	41.244	19.8	0.0	2.30	0.0	2.30	Coastal lagoon
Dry Lagoon	*41.224*	*3.2*	*0.0*	*0.0*	*0.0*	*0.0*	*Blind – Tidally restricted coastal creek*
Big Lagoon	41.174	136.4	0.0	5.03	0.05	5.09	Coastal lagoon
Little River	41.027	117.3	0.0	0.06	0.0	0.06	Tidally restricted coastal creek
Mad River	40.942	1,286.8	0.17	1.16	0.0	1.33	Highly river-dominated drowned river mouth

ESTUARY	LATITUDE	WATERSHED AREA (km^2)	AREA (km^2)				ESTUARY TYPE
			MARINE	ESTUARINE	TIDAL RIVERINE	TOTAL ESTUARY	
Humboldt	40.75	472.1	0.07	71.42	0.0	71.49	Bar built (Coastal lagoon)
Eel River	40.641	9,535.9	0.0	10.18	5.43	15.61	Highly river-dominated drowned river mouth
Guthrie Creek	40.542	22.6	0.0	0.005	0.0	0.005	Tidally restricted coastal creek
Bear River*	40.476	214.5	0.004	0.12	0.05	0.18	Tidally restricted coastal creek (Blind – Drowned river mouth)

After the initial identification of major estuarine types, hierarchical clustering based on group averages (see Clarke and Warwick, 2001; McCune and Grace, 2002) was used to classify the estuaries or watersheds based on similarities in wetland habitat or watershed land cover patterns, respectively. The Bray-Curtis metric was used as the measure of similarity in these analyses. Clustering was conducted using both the absolute area of the NWI or land cover classes as well as the relative proportions of the classes using untransformed data. When clustering variables measured in different units (e.g., population, % impervious surfaces), the data were normalized by subtracting the mean from each value and dividing by the standard deviation to generate a standard score, with Euclidean distance used as the similarity metric. All clustering was conducted with Primer6 (Clarke and Gorley, 2006; http://www.primer-e.com/).

Two criteria were used to define estuary groups in the clustering analysis. The primary approach was whether the cluster was significantly different from a random reordering of the data within the branch in the dendrogram using the SIMPROF test in Primer6 (Clarke and Gorley, 2006). By testing for differences within each branch independently, significant differences can occur at different similarity levels in separate branches of the dendrogram, so that it is not necessary to choose a single similarity level. Clusters of estuaries that were not significantly different were not further divided regardless of the level of similarity. A value of $p=0.05$ was used as the default significance level though there is no inherent reason that this value is the ecologically "correct" level to identify groups of estuaries with similar functional attributes. Accordingly, we evaluated a range of significance values ($p = 0.10, 0.20, 0.40, 0.60$) in the SIMPROF tests. As the significance level is relaxed (i.e., p increases), a greater number of significant clusters will be generated and within-cluster similarity will increase though there will be fewer estuaries or watersheds per cluster. This sequential approach provides flexibility in developing a management framework, allowing managers to balance the extent of variability within estuary groups versus the practical issues of increasing the number of groups. A secondary criterion to defining groups was to combine estuaries within the branch if similarity was high (>75% with Bray-Curtis or <25% of the maximum dissimilarity with Euclidean distance) even if the branch showed a significant difference with SIMPROF. While this introduces a degree of subjectivity, it eliminates the problem of generating numerous classes that would likely require similar management strategies.

While it would be desirable to have a single classification that captures all aspects of current impacts and future vulnerability, the reality is that multiple classification systems and approaches are needed to address different scientific and management issues. For example, a classification based on resource availability (e.g., extent of wetlands) might group an estuary differently than one based on loading (e.g., land cover pattern). Our approach was to evaluate several classifications schema both in this chapter and in Chapter 7 based on different biotic and landscape attributes related to nutrient dynamics or estuarine vulnerability. It should be recognized, however, that any classification system should be considered a hypothesis until it is evaluated with independent datasets demonstrating similar estuarine responses, nutrient dynamics, or vulnerability to increased loadings.

2.1.6 Ecological Resources in Pacific Northwest Estuaries

The ultimate purpose of this classification exercise is to derive the insights required for the efficient management of ecological resources in PNW estuaries. Different types of resources, for example salmonids and submerged aquatic vegetation, are likely to respond differently to nutrient enrichment, thus requiring different management strategies. Additionally, the type and extent of ecosystem services varies across estuaries, which may suggest different management prioritizations among estuaries. Thus, as part of this effort, a preliminary evaluation of the general types of ecological services provided by each of the estuaries was summarized in Table 2-3 and further discussed in Section 2.3.

One class of ecological services relates to the occurrence of estuarine intertidal and subtidal wetlands, which was evaluated using both the emergent wetland and aquatic bed classes in NWI and the 2001 NOAA land cover analysis. The irregularly flooded estuarine forest and shrub classes were not included since these semi-terrestrial wetland types are less vulnerable to estuarine water quality. Three other important PNW resources are oyster aquaculture, native salmon runs, and native and migratory birds. Oyster aquaculture was evaluated from known facilities as well as from state permits. The presence of current or historic salmon runs was based on reports in the literature, while Lawson et al. (2004) provided the predicted number of Coho smolts historically present for Oregon estuaries based on a watershed model. Two measures of an estuary's importance to birds were whether the estuary had been classified as an Important Bird Area (IBA) by the Audubon Society (http://www.audubon.org/bird/iba/index.html) and whether it is recognized as an important shorebird site in the U.S. Shorebird Conservation Plan for the Northern Pacific Coast (Drut and Buchanan, 2000; http://www.fws.gov/shorebirdplan/RegionalShorebird/RegionalPlans.htm). The final function considered, but not quantified, was recreational support. Recreational use of PNW estuaries is primarily limited to boating, fishing, crabbing, and clamming, with relatively little direct water contact.

2.2 Inventory of Estuaries on the Pacific Coast

The starting point of our comprehensive classification of estuaries was to conduct an inventory of the estuaries on the Pacific Coast. We found that the Pacific Coast contains a surprisingly large number of estuaries – 230 based on the criterion that the waterbody contains an estuarine NWI polygon and discharges directly into the ocean. This number does not include subestuaries within larger water bodies nor does it include the Strait of Juan de Fuca or Puget Sound. In addition, there are several harbors and coastal streams that do not contain NWI estuarine polygons but which fall within the size range of the coastal estuaries. Although not counted as estuaries, these additional water bodies may provide some of the same ecological functions as similar sized estuaries. As a result of the 2007 NWI update, the number of PNW estuaries increased from 216 in the earlier report (Lee et al., 2006) to 230 in the present analysis. The number estuaries may continue to change as NWI adds or deletes estuarine polygons from the smallest coastal waterbodies.

The naming convention used in this report is to refer to a waterbody as an "estuary" if it consists of multiple named embayments, rivers, or creeks that constitute major geographic features. This is to avoid confusion whether the name refers to the entire waterbody or a single component. For example, "San Francisco Estuary" is used to refer to the combination of San Francisco Bay and the Delta, including San Pablo Bay, Suisan Bay, and other components of the Delta. The use of "San Francisco Bay" is restricted to identifying the bay proper. Similarly, we refer to the "Yaquina Estuary" or "Yaquina Bay Estuary" rather than "Yaquina Bay" because the estuary is

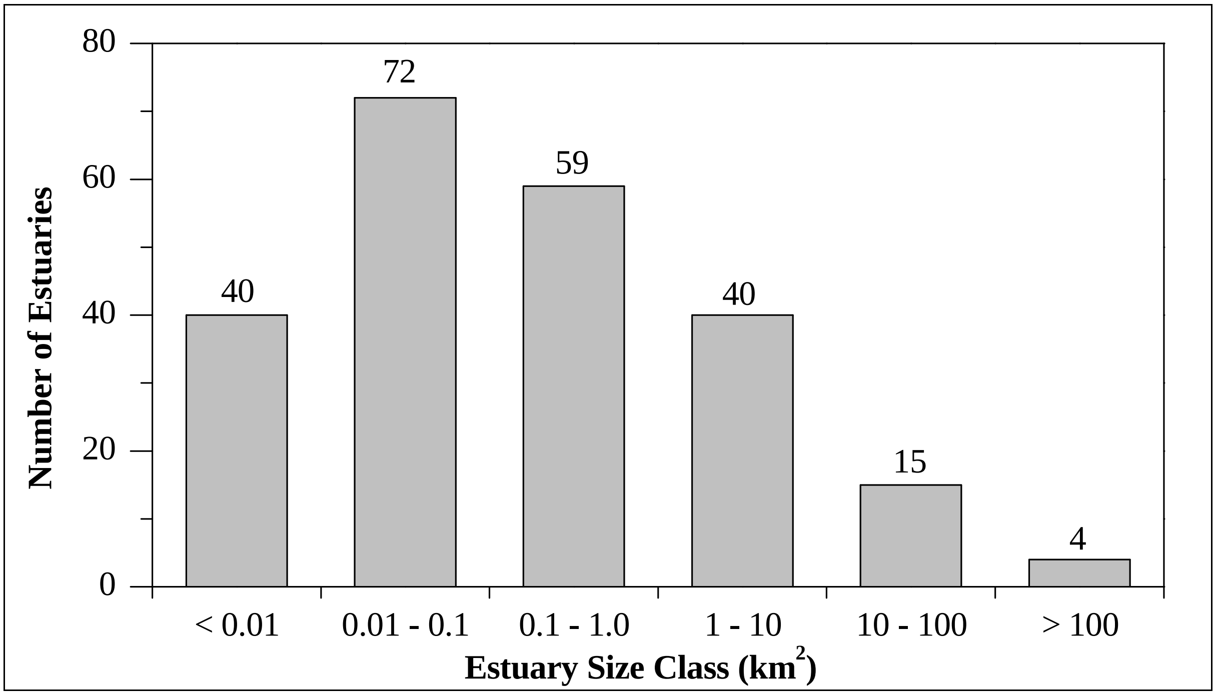

Figure 2-6. Size distribution of all 230 coastal estuaries in California, Oregon, and Washington (exclusive of Puget Sound) identified by the presence of a NWI estuarine polygon.

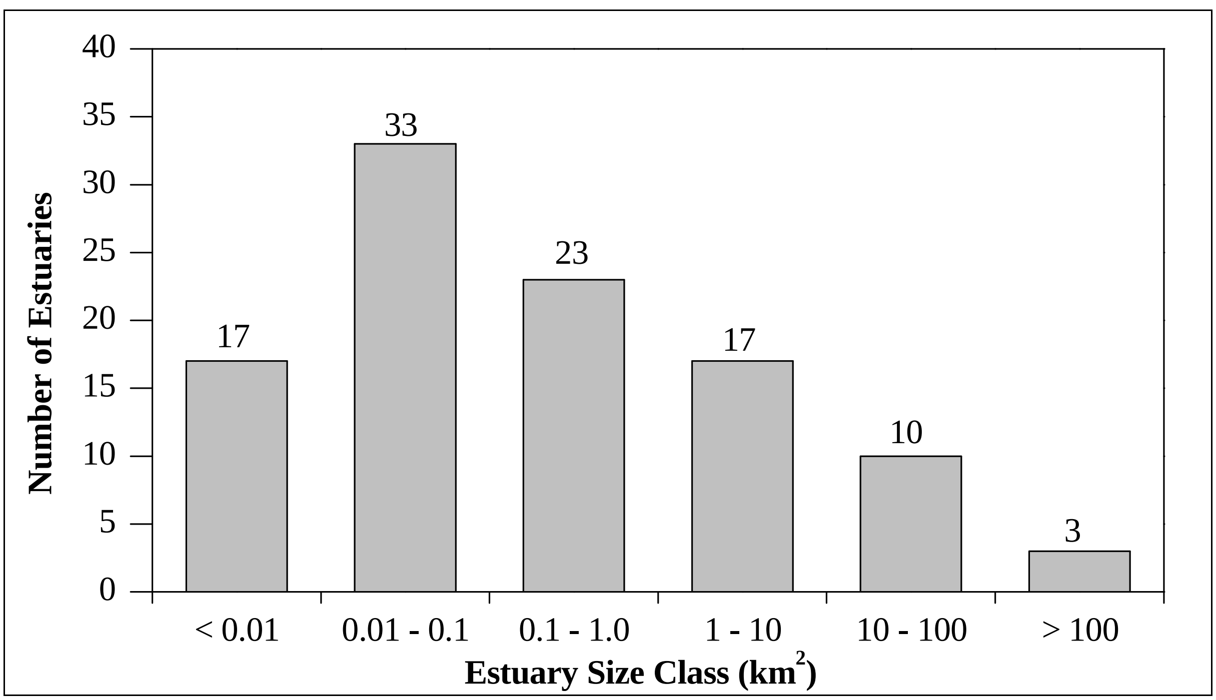

Figure 2-7. Size distribution of the 103 PNW coastal estuaries identified by the presence of a NWI estuarine polygon.

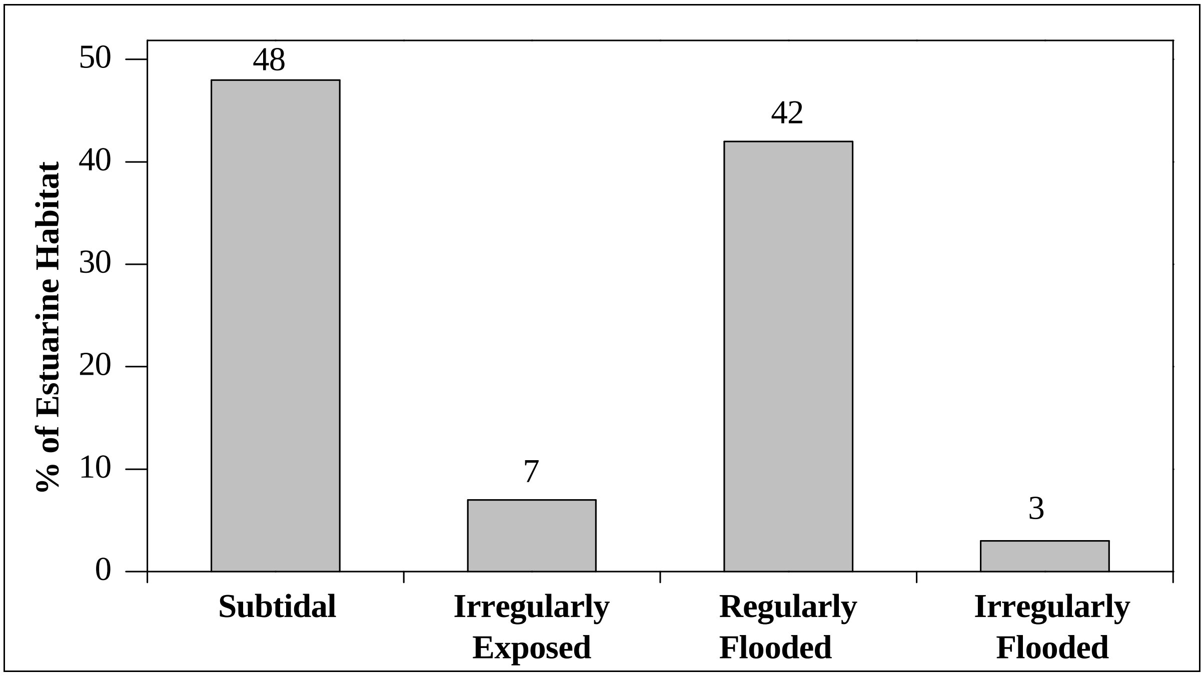

Figure 2-8. Distribution of estuarine subtidal and intertidal habitats in the 103 PNW estuaries. Intertidal habitat is defined as the sum of the irregularly exposed, regularly flooded, and irregularly flooded NWI estuarine and marine classes.

composed of both Yaquina Bay proper and lower portions of the Yaquina River. Names of waterbodies composed of a single named component, such as Beaver Creek, are not modified.

Estuaries on the Pacific Coast vary greatly in size. The smallest is Liddell Creek in Southern California (0.00017 km^2) and the largest is the San Francisco Estuary (3346 km^2). Most of the Pacific Coast estuaries are small, with 171 (74%) having an estuarine area <1 km^2 (Figure 2-6). Although there are numerous small estuaries, the bulk of the estuarine habitat is concentrated within a few systems. The San Francisco Estuary contains almost two thirds of the total estuarine area on the Pacific Coast, exclusive of Puget Sound, with the four largest estuaries (San Francisco, Columbia River, Willapa Estuary, and Grays Harbor) accounting for 88% of the estuarine area.

A similar analysis for the PNW identified 103 coastal estuaries (see Figures 2-2 to 2-5) of which Burnt Hill Creek and Rockaway Beach Creek are the smallest (0.0004 km^2; Table 2-2). The largest is the Columbia River Estuary with an area of 669 km^2, of which 257 km^2 is tidal riverine habitat much of which is fresh. Most PNW estuaries are small, with 59 (66%) of the estuaries <1 km^2 (Figure 2-7). Only the Columbia River, Willapa, and the Grays Harbor estuaries are larger than 100 km^2. On an area basis, these three largest estuaries account for 79% of the total PNW estuarine area, exclusive of Puget Sound. In addition to these estuaries defined by NWI, the NOAA 2001 land cover survey identified four small water bodies that they classified as containing estuarine habitat but which did not contain a NWI estuarine polygon. These water bodies are listed in Table 2-2 but were not included in the numerical tallies of estuaries or the statistical analyses.

One prominent characteristic of PNW estuaries is their extensive intertidal area. The intertidal area for the 103 estuaries was calculated as the sum of the NWI estuarine polygons defined as "irregularly exposed", "regularly flooded" and "irregularly flooded", which approximates tidal heights from mean lower low water (MLLW) to mean higher high water (MHHW). Averaging the NWI estuarine polygons across all of the PNW estuaries, intertidal habitat constituted 52% of the estuarine area (Figure 2-8). There is no apparent pattern in the extent of intertidal habitat as a function of estuary size, with the percent intertidal in the NWI estuarine habitats ranging from about 39% to 57% across estuarine size classes.

2.3 Ecological Services of Small Pacific Northwest Estuaries

Estuaries smaller than about 1 km^2 or even less than 10 km^2 have generally been excluded from previous estuarine classification efforts. For example, the National Estuarine Eutrophication Assessment (Bricker et al., 1999) included all 13 estuaries in the PNW >10 km^2 but only 4 of the 17 estuaries between 1 and 10 km^2 and none <1 km^2. However, the large number of small PNW estuaries (Figure 2-7) suggests that it was important to include them in the regional analysis for several reasons. These smaller estuaries are important for native salmon, a critical regional resource and management issue (Lackey, 2004; Lackey et al., 2006a, b). Forty-six of the 73 estuaries <1 km^2 have reported salmon runs (Table 2-3). An analysis of the predicted historical number of Coho smolts produced per watershed along the Oregon coast (Lawson et al., 2004) indicated that the small estuaries produced a proportionally greater number of smolts than the larger estuaries when normalized to estuarine area (Figure 2-9). Estuaries <1 km^2 were contributing about 8% to total Coho runs while estuaries <10 km^2 were contributing about 24%. These smaller coastal watersheds and estuaries may also serve as a future refuge for wild salmon with the increasing development and alteration of the larger estuaries and watersheds. Lackey et al. (2006c) suggest that the coastal rivers in California, Oregon, Washington, and southern British Columbia offer greater potential for preserving wild salmon runs than do the Sacramento-San Joaquin, Columbia, and lower Fraser Rivers, places where the most expensive salmon recovery efforts are now focused. In addition to salmon, small estuaries in Northern California (e.g., Smith River and Stone Lagoon) provide critical habitat for the endangered tidewater goby (*Eucyclogobius newberryi*) (Federal Register, 2002; Ahnelt et al., 2004), and many of the smaller estuaries provide habitat for resident and migratory birds (Table 2-3), including bald eagles and brown pelicans.

In terms of other ecosystem services, several of the coastal lagoons <10 km^2 contain extensive emergent wetlands (e.g., Big Lagoon, Lake Earl) while many of the drowned river estuaries and tidal creeks between about 0.5 km^2 and 10 km^2 also contain considerable wetlands (Table 2-3). The wetlands in these small estuaries are likely important for coastal and migratory birds and to a limited number of brackish water fishes such as the tidewater goby. Among the smallest estuaries (<0.1 km^2) only 7 of the 50 contain any emergent wetlands, none contain any aquatic beds (Table 2-3), and these smallest estuaries are little utilized by recreational crab or clam species. Thus, the ecosystem services provided by these smallest estuaries appear to be related to supporting wild salmon runs, recreation, and perhaps as bird habitat. These smallest estuaries and their associated rivers and creeks may also provide economic benefits in terms of increased property value, though we made no attempt to estimate this.

Table 2-3. Examples of ecological resources and services in PNW estuaries. Estuarine emergent wetlands and aquatic beds from the NWI are the sums of classes with “EM” or “AB” codes, respectively. The estuarine emergent wetland and aquatic beds from NOAA are areas for these two land cover classes. “Estuary Area” is the sum of the NWI estuarine, marine, and tidal riverine areas (see Table 2-2). “Oyster culture” indicates whether there is commercial oyster aquaculture or whether there are permits to allow oyster aquaculture. “Salmon present” indicates whether salmon are currently or have historically been reported from the estuary or from the streams and rivers flowing into the estuary. “N?” is used to indicate estuaries where we suspect salmon are absent based on estuary size or other landscape attributes while “?” indicates that we are unaware of any reports of salmon from that estuary. The “# smolts” is the historical potential number of Coho smolts predicted to have occurred within Oregon estuaries (Lawson et al., 2004). The five italicized water bodies are those that have an estuarine emergent wetland or estuarine aquatic bed polygon in the 2001 NOAA land cover analysis (see Table 2-2) but do not have a NWI estuarine polygon. SCP = listed as important site in the Shorebird Conservation Plan with sites marked with an asterisk (*) supporting ≥ 4000 birds. IBA = Important Bird Areas. NA = Estuary not identifiable in NOAA dataset.

ESTUARY	ESTUARY AREA (km^2)	NWI ESTUARINE EMERGENT WETLAND (km^2)	NWI ESTUARINE AQUATIC BED (km^2)	NOAA ESTUARINE EMERGENT WETLAND (km^2)	NOAA ESTUARINE AQUATIC BED (km^2)	OYSTER CULTURE	SALMON PRESENT (# SMOLTS)	BIRD HABITAT
Waatch River	1.16	0.93	0.0	0.0	0.0	N	Y	
Hobuck Creek	*0.0*	*0.0*	*0.0*	*0.003*	*0.0*	*N*	Y	
Sooes River	0.56	0.17	0.0	0.01	0.0	N	Y	
Ozette River	0.03	0.0	0.0	0.0	0.002	N	Y	
Quillayute River	0.96	0.05	0.0	0.03	0.002	N	Y	
Goodman Creek	0.03	0.0	0.0	0.0	0.0	N	Y	
Mosquito Creek	0.01	0.0	0.0	0.0	0.0	N	Y	
Hoh River	0.84	0.0	0.0	0.004	0.001	N	Y	
Cedar Creek	0.02	0.0	0.0	0.0	0.0	N	Y	
Kalaloch Creek	0.02	0.0	0.0	0.002	0.0	N	Y	
Queets River	1.43	0.15	0.0	0.09	0.01	N	Y	
Whale Creek	0.01	0.0	0.0	0.0	0.0	N	Y	
Raft River	0.24	0.08	0.0	0.002	0.002	N	Y	
Camp Creek	0.03	0.0	0.0	0.002	0.0	N	Y	
Duck Creek	0.01	0.0	0.0	0.0	0.0	N	Y	

ESTUARY	ESTUARY AREA (km^2)	NWI ESTUARINE EMERGENT WETLAND (km^2)	NWI ESTUARINE AQUATIC BED (km^2)	NOAA ESTUARINE EMERGENT WETLAND (km^2)	NOAA ESTUARINE AQUATIC BED (km^2)	OYSTER CULTURE	SALMON PRESENT (# SMOLTS)	BIRD HABITAT
Quinault River	0.66	0.01	0.0	0.003	0.01	N	Y	
Wreck Creek	0.01	0.0	0.0	0.001	0.0	N	Y	
Moclips River	0.10	0.003	0.0	0.0	0.0	N	Y	
Joe Creek	0.05	0.0	0.0	0.002	0.0	N	Y	
Boone Creek	0.01	0.0	0.0	0.004	0.0	N	?	
Copalis River	0.86	0.40	0.0	0.001	0.0	N	Y	
Conner Creek	0.17	0.0	0.0	0.0	0.0	N	Y	
Grays Harbor	262.73	15.24	147.59	11.98	1.11	Y	Y	SPC*
Willapa	390.86	39.44	181.47	29.04	2.67	Y	Y	SPC*
Loomis Lake Creek	0.01	0.0	0.0	0.06	0.003	N	N?	
Columbia River	668.98	28.63	4.07	18.99	0.79	N	Y	IBA
Clatsop Spit	0.08	0.06	0.0	NA	NA	N	N	
Necanicum	1.63	0.42	0.0	0.01	0.06	N	Y (685,000)	IBA
Indian Creek	0.004	0.0	0.0	NA	NA	N	Y (100)	
Chapman Point	0.001	0.0	0.0	NA	NA	N	?	
Ecola Creek	0.06	0.02	0.0	0.0	0.001	N	Y (72,000)	
Arch Cape Creek	0.001	0.0	0.0	NA	NA	N	N?	
Cove Beach	0.005	0.002	0.0	NA	NA	N	N?	
Short Sand Creek	0.003	0.0	0.0	NA	NA	N	N?	
Nehalem	11.65	2.34	0.14	2.76	0.02	N	Y (3,330,000)	SPC
Lake Lytle	0.004	0.0	0.0	NA	NA	N	?	
Rockaway Beach Creek	0.0004	0.0	0.0	NA	NA	N	N?	
Rockaway Clear Lake	0.0005	0.0	0.0	NA	NA	N	N?	
Smith Lake	0.015	0.0	0.0	NA	NA	N	?	

ESTUARY	ESTUARY AREA (km²)	NWI ESTUARINE EMERGENT WETLAND (km²)	NWI ESTUARINE AQUATIC BED (km²)	NOAA ESTUARINE EMERGENT WETLAND (km²)	NOAA ESTUARINE AQUATIC BED (km²)	OYSTER CULTURE	SALMON PRESENT (# SMOLTS)	BIRD HABITAT
Tillamook	37.48	4.29	0.02	4.51	0.50	Y	Y (3,288,000)	SPC*/IBA
Netarts Bay	10.43	1.07	2.60	1.16	0.89	Y	Y (15,000)	SPC*/IBA
Chamberlain Lake	0.04	0.0	0.0	NA	NA	N	?	
Sand Lake	4.28	2.37	0.07	3.19	0.02	N	Y (123,000)	
Sears Lake	0.002	0.0	0.0	NA	NA	N	?	
Miles Creek	0.06	0.0	0.0	NA	NA	N	?	
Nestucca	5.00	0.88	0.09	1.14	0.36	N	Y (1,037,000)	IBA
Daley Lake	0.001	0.0	0.0	NA	NA	N	N?	
Neskowin Creek	0.014	0.0	0.0	0.002	0.0	N	Y (49,000)	
Salmon River	3.11	2.13	0.004	2.54	0.0	N	Y (168,000)	IBA
Siletz River	8.86	2.16	0.0	1.54	0.45	N	Y (1,217,000)	SPC/IBA
Schoolhouse Creek	0.01	0.0	0.0	0.0	0.0018	N	Y (2,000)	
Depoe Bay	0.04	0.0	0.0	0.002	0.0	N	Y (7,000)	
Yaquina	19.96	3.03	3.65	1.11	0.72	Y	Y (1,217,000)	SPC*/IBA
Beaver Creek	0.55	0.42	0.0	0.03	0.01	N	Y (265,000)	
Alsea	12.49	2.49	3.25	2.43	0.99	N (under consider-ation)	Y (1,628,000)	SPC/IBA
Big Creek	0.09	0.08	0.0	NA	NA	N	Y	
Yachats River	0.11	0.0	0.02	0.0	0.0	N	Y (110,000)	

ESTUARY	ESTUARY AREA (km^2)	NWI ESTUARINE EMERGENT WETLAND (km^2)	NWI ESTUARINE AQUATIC BED (km^2)	NOAA ESTUARINE EMERGENT WETLAND (km^2)	NOAA ESTUARINE AQUATIC BED (km^2)	OYSTER CULTURE	SALMON PRESENT (# SMOLTS)	BIRD HABITAT
Tenmile Creek North	0.04	0.02	0.0	0.0	0.0	N	Y (28,000)	
Berry Creek	0.02	0.0	0.0	0.0	0.0	N	Y (54,000)	
Sutton Creek	0.15	0.0	0.03	0.0	0.01	N	Y (84,000)	
Siuslaw River	15.59	3.32	1.30	1.14	0.09	Y	Y (2,674,000)	SPC/IBA
Siltcoos River	0.36	0.08	0.0	0.0	0.06	N	Y (771,000)	IBA
Tahkenitch Creek	0.26	0.0	0.0	0.001	0.07	N	Y (228,000)	IBA
Umpqua River	33.78	3.18	0.07	1.43	0.94	Y	Y (8,199,000)	IBA
Tenmile Creek South	0.50	0.01	0.0	0.0	0.01	N	Y (525,000)	
Coos	54.90	7.28	5.98	4.91	4.20	Y	Y (2,058,000)	SPC*/IBA
Sunset Bay	0.12	0.0	0.0	NA	NA	N	?	
Twomile Creek North	*0.0008*	*0.0*	*0.0*	*0.01*	*0.0*	*N*	*N?*	
Coquille River	6.89	1.07	0.06	1.38	0.17	N	Y (4,169,000)	SPC*/IBA
Twomile Creek South	0.11	0.01	0.0	0.02	0.0	N	Y (134,000)	
New River	1.67	0.48	0.0	0.27	0.02	N	Y (396,000)	SPC/IBA
Sixes River	0.39	0.03	0.0	0.05	0.0	N	Y (372,000)	
Elk River	0.66	0.05	0.0	0.01	0.0	N	Y	
Port Orford Head	0.01	0.0	0.0	NA	NA	N	?	
Hubbard Creek	0.01	0.0	0.0	NA	NA	N	?	
Brush Creek	0.02	0.0	0.0	0.0	0.0	N	Y	

ESTUARY	ESTUARY AREA (km²)	NWI ESTUARINE EMERGENT WETLAND (km²)	NWI ESTUARINE AQUATIC BED (km²)	NOAA ESTUARINE EMERGENT WETLAND (km²)	NOAA ESTUARINE AQUATIC BED (km²)	OYSTER CULTURE	SALMON PRESENT (# SMOLTS)	BIRD HABITAT
Mussel Creek	0.02	0.0	0.0	0.02	0.0	N	Y	
Euchre Creek	0.12	0.02	0.0	0.08	0.0	N	Y	
Greggs Creek	0.02	0.002	0.0	NA	NA	N	?	
Rogue River	2.77	0.06	0.0	0.25	0.10	N	Y	
Hunter Creek	0.10	0.0	0.0	0.01	0.0	N	Y	
Myers Creek	0.02	0.0	0.0	0.0	0.0	N	Y	
Pistol River	0.55	0.01	0.0	0.14	0.0	N	Y	
Burnt Hill Creek	0.004	0.0	0.0	NA	NA	N	N	
Cove at Boardman Park	0.005	0.0	0.0	NA	NA	N	N	
Thomas Creek	0.01	0.0	0.0	NA	NA	N	?	
Whaleshead Creek	0.02	0.0	0.0	NA	NA	N	?	
Chetco River	0.72	0.01	0.0	0.03	0.01	N	Y	
Winchuck River	0.12	0.004	0.0	0.05	0.0	N	Y	
Smith River	2.38	0.07	0.0	0.56	0.0	N	Y	
Lake Earl	9.01	0.0	9.01	4.31	0.0	N	Y	
Crescent City Harbor	1.63	0.0	0.0	0.0	0.0	Y (Permits)	No?	
Klamath River	2.27	0.03	0.0	0.01	0.005	N	Y	
Johnson Creek	0.01	0.0	0.0	0.0	0.0	N	?	
Ossagon Creek	0.01	0.0	0.0	0.0	0.0	N	?	
Squashan Creek	0.01	0.0	0.0	NA	NA	N	?	
Redwood Creek	0.30	0.0	0.0	0.18	0.0	N	Y	
Freshwater Lagoon	*0.0*	*0.0*	*0.0*	*0.10*	*0.0*	*N*	*No*	
Stone Lagoon	2.30	0.0	0.0	0.08	0.0	N	Y	
Dry Lagoon	*0.0*	*0.0*	*0.0*	*0.12*	*0.0*	*N*	*No?*	
Big Lagoon	5.09	0.0	5.03	0.57	0.0	N	Y	
Little River	0.06	0.0	0.0	0.14	0.0	N	Y	

ESTUARY	ESTUARY AREA (km^2)	NWI ESTUARINE EMERGENT WETLAND (km^2)	NWI ESTUARINE AQUATIC BED (km^2)	NOAA ESTUARINE EMERGENT WETLAND (km^2)	NOAA ESTUARINE AQUATIC BED (km^2)	OYSTER CULTURE	SALMON PRESENT (# SMOLTS)	BIRD HABITAT
Mad River	1.33	0.02	0.0	0.33	0.0	N	Y	
Humboldt	71.49	3.93	19.82	9.35	0.0	Y	Y	IBA
Eel River	15.61	1.68	0.06	4.95	0.002	N	Y	
Guthrie Creek	0.005	0.0	0.0	0.0	0.0	N	Y	
Bear Creek	0.18	0.0	0.0	0.003	0.0	N	?	
TOTAL	**1677.44**	**128.22**	**384.34**	**111.15**	**14.31**	**10 Y**	**67 Y**	-

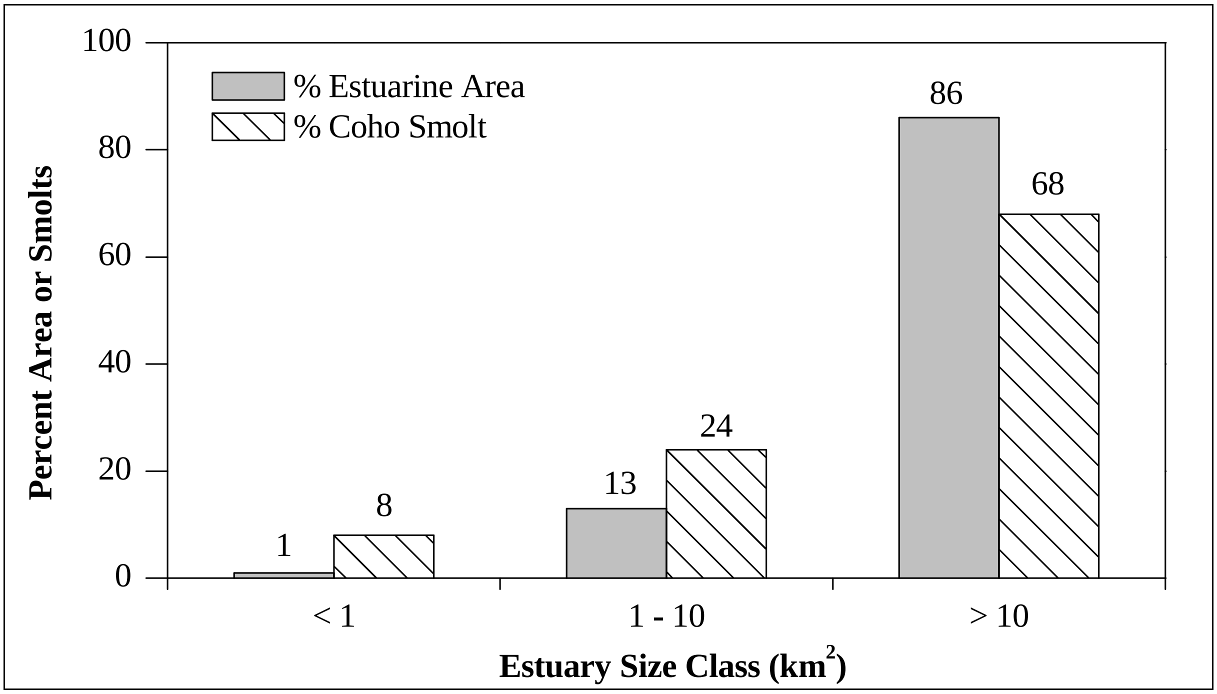

Figure 2-9. Distribution of the projected historical number of Coho smolts by estuary size classes in 29 Oregon estuaries. Coho smolt data from Lawson et al. (2004; see Table 2-3).

The final reason for the inclusion of these small water bodies is that they are likely to require different management strategies. Factors such as their smaller volumes, differences in seasonal and tidal variability in exchange and salinity, and different biotic communities are all likely to result in different exposures and vulnerabilities. Research focused on these largely ignored small systems is required to better understand how they function, but it is possible to speculate that their small size makes them more vulnerable to certain types of stressors while at the same time offering greater opportunities for cost-effective protection and/or mitigation efforts

2.4 Classification by Geomorphology and Oceanic Exchange

The 103 PNW estuaries were initially separated into broad classes based on the potential extent of oceanic exchange, a fundamental feature affecting vulnerability to terrestrial nutrient loading. Since residence times are not available for most of these waterbodies, we qualitatively estimated relative oceanic exchange from estuarine geomorphology based on the literature, aerial photographs, and personal observations. Photographic sources included the aerial photographs taken by the Pacific Coastal Ecology Branch (PCEB; see Chapter 6) as well as online sources including the Oregon Coastal Atlas (http://www.coastalatlas.net), high resolution (1:12,000) panchromatic imagery commercially available through GlobeExplorer ™, Terraserver (http://www.terraserver.com), GoogleEarth (http://www.google.com), Washington Coastal Atlas (http://www.ecy.wa.gov/programs/sea/SMA/atlas_home.html), Washington's Department of Ecology Washington Shoreline Aerial Photos (http://apps.ecy.wa.gov/shorephotos/index.html), and the California Coastal Records Project (http://www.californiacoastline.org/). Additionally, the NWI coverages were used to evaluate the overall waterbody shape and structure.

It is important to recognize that many estuaries display characteristics of several geomorphological classes so assigning an estuary to a single class is somewhat artificial. In recognition of this gradation, alternate classifications are given when appropriate. To the extent practical, an estuary's structure and dynamics should be considered as well as its classification when assessing vulnerability or developing a management strategy.

2.4.1 Coastal Lagoons and Blind Estuaries

We identified two types of moderate to large estuaries with restricted exchange. The first are coastal lagoons, defined here as coastal water bodies located behind berms with NWI estuarine polygons that are only intermittently open to the ocean. For example, some Southern California lagoons remain open only about a third of the time (Elwany et al., 1998). Coastal lagoons can breach during storms, after periods of heavy rain, or long-shore movement of dune sand (Elwany et al., 1998). Coastal lagoons are also artificially breached, especially in California, to increase circulation, improve water quality, and reduce flooding (e.g., Williams et al., 1999; Merritt Smith Consulting, 1999). We classified eight PNW water bodies as coastal lagoons (Table 2-2). The largest of these is Lake Earl while the smallest is the Sears Lake.

The second type of waterbody with restricted oceanic exchange is the blind or intermittent estuary. Mouths of blind estuaries periodically close due to the formation of an ephemeral berm. The mouth of a blind estuary may not close every year, but closure is frequent enough to be a regular characteristic of the system. Blind estuaries are more common in the southern portion of the Pacific coast, with many of the smaller southern California estuaries closing during the late summer and early fall before the rains. We separate blind estuaries from coastal lagoons based on these attributes: blind estuaries are open to the ocean more frequently than lagoons; there appears to be more among-year variation in blind estuaries whether the mouth closes; and blind estuaries tend to have a more riverine shape compared the more "pond" or "lake" shape of lagoons that often run parallel to the dunes. Such a distinction is also made with the South African "Temporarily Open/Closed Estuaries" (TOCE), which are divided into "Intermittently Closed Estuaries" (ICEs), which have a connection to the ocean more than 50% of the time, and "Intermittently Open Estuaries" (IOEs), which are closed to the ocean more than 50% of the time (Whitfield and Bate, 2007).

Based on the aerial photographs, literature (e.g., Bottom et al., 1979; Cortright et al., 1987), and online sources (http://www.coastalatlas.net), six PNW waterbodies were classified as blind (Table 2-2). The smallest is the Winchuck River while the largest is the New River. Note that this count of blind estuaries does not include the tidally restricted coastal creeks discussed below that may also periodically close off at the mouth.

With both lagoons and blind estuaries, closure of the mouth eliminates any direct flushing with the ocean though there may be subsurface flow between the estuary and the ocean. The absence or greatly reduced exchange during closure can result in low dissolved oxygen, extended periods of low salinity at least at the surface, increased temperatures, and stratification. Fish kills have been reported during periods of closure in blind estuaries in Oregon due to the ponding of freshwater (Clifton et al., 1973 in Bottom et al., 1979). Blooms of nuisance algae can occur during periods of closure (e.g., John Gilchrist & Associates and Fall Creek Engineering, Inc., 2005). Five dogs died after swimming in Big Lagoon in 2001, which was attributed to exposure to toxic blue green algae resulting from warm weather and the heavy nutrient load caused by the lack of winter breaching (Humboldt County Dept. Health Human Ser., 2005). Bottom dissolved oxygen in the Mattole River in northern California periodically decreased below 6 mg l^{-1} after "permanent lagoon formation" and occasionally dropped to 4 mg l^{-1}, though interestingly the lowest dissolved oxygen levels occurred two weeks before the estuary closed off (Zedonis et al., 2008). Salinity in these restricted waterbodies can vary drastically depending upon the extent of breaching. When closed for extended periods, some of these waterbodies can transform into freshwater systems. The Mattole River became a freshwater lagoon after the closure of the mouth (Busby, 1991). Similarly, our measurements of salinity in the New River ranged from 0 to 1.6 psu in the summer of 2003 (H. Lee II, unpub. data).

As a class, both the lagoonal systems and blind estuaries are highly vulnerable to nutrient enrichment during periods of restricted oceanic exchange. However, both lagoons and blind estuaries experience dramatic among-year and seasonal variability in flushing and corresponding variation in water quality and salinity, confounding how to manage these systems. One strategy is to assume "worst case" scenarios of restricted or blocked ocean access. Since salmon are present in many of these systems, an appropriate beneficial use defined in Oregon for these estuaries is to support "Resident Fish and Aquatic Life, Salmonid Spawning & Rearing". The dissolved oxygen threshold for this beneficial use in Oregon for estuaries is 6.5 mg l^{-1} (http://www.sos.state.or.us/archives/rules/OARs_300/OAR_340/340_041.html). In addition, the Oregon narrative criteria states that "where a less stringent natural condition of water of the State exceeds the numeric criteria … the natural condition supersedes the numeric criteria and becomes the standard for that water body." Dissolved oxygen levels of > 6.5 mg l^{-1} may be difficult to obtain during periods of restricted flushing especially in the lagoonal systems. One management option is to artificially breach the barriers, though that may result in other impacts, such as the premature flushing of salmon smolts.

2.4.2 Tidal Coastal Creeks/Tidally Restricted Coastal Creeks

Tidal coastal creeks are creeks or streams discharging directly into the ocean and which experience input of ocean water at least during high tide. These systems are the smallest type of "estuary", some of which are essentially freshwater streams with a limited area influenced by ocean waters. However, the hydrodynamics of these systems appears to be more complex than their small size suggests. Based on field observations on a few coastal creeks and analysis of aerial photographs, most have restricted connections with the ocean during a portion of the year. We refer to these as "tidally restricted coastal creeks" which are defined by having one or more of the following characteristics at least intermittently: 1) there is a narrowing of mouth from sand movement sufficient to restrict exchange with the ocean; 2) formation of a berm that closes the mouth, separating the creek from the ocean; or 3) there is a development of a sill near the mouth sufficient to restrict exchange with the ocean.

We tentatively identified 55 waterbodies as tidal coastal creeks in the PNW, ranging from Chapman Point Creek (0.001 km^2) to Beaver Creek (0.55 km^2) (Table 2-2). Analysis of aerial photographs suggests that most of these systems have limited exchange during portions of the year, and 53 are tentatively considered tidally restricted coastal creeks. There is no clear demarcation between the larger tidally restricted coastal creeks and the tidal rivers. For example, Bear River was classified as a tidally restricted coastal creek though it alternatively could be classified as a blind drowned river mouth estuary. Several of the smaller tidal rivers in Washington, including the Waatch, Hoh, Quinault, Sooes, and Quillayute, have partially restricted mouths and may represent the transition between the tidally restricted coastal creeks and blind drowned river mouth estuaries. Part of the difficulty in cleanly separating these systems is the paucity of studies on the estuarine segments of tidal creeks. Even a relatively small field effort evaluating salinity patterns and mouth constrictions across a suite of tidal creeks would greatly promote our understanding of their dynamics.

Based on our present knowledge, flushing is presumed to be very high in estuarine portion of tidally restricted coastal creeks when they are open, and salinity near the mouth is likely to undergo extreme tidal and seasonal fluctuations. An indication of the extent of these salinity variations can be found in the larger drowned river mouth estuaries, where salinity at the mouth can vary by more than 20 psu over a tidal cycle (Figures 4-8 and 4-9), and median salinity near the mouth of highly river-dominated systems can vary by 20 psu seasonally (Figure 2-11). We suspect that salinity near the mouth of the tidal creeks can cycle from essentially marine to fresh over a tidal cycle during the periods when they are open. During such periods, this high tidal exchange reduces the accumulation of terrestrially derived nutrients in the "estuarine" segments of these waterbodies. However, these extreme salinity fluctuations also limit the ability of all but the most euryhaline species to survive. A possible reflection of this salinity stress is the paucity of estuarine emergent wetlands and aquatic beds in tidal coastal creeks <0.1 km^2 (Table 2-3).

During other periods, restriction near the mouth limits exchange in these tidal creeks. If the restriction is of sufficient duration, the "estuarine" portion of tidal creeks may develop low dissolved oxygen conditions. Additionally, an ongoing study of the Yachats River indicates that the formation of a sill near the mouth restricts exchange with the ocean to the spring high tides, essentially acting as a micro-fjord (C. Brown, unpublished data). This oceanic water advected during spring tides can form localized pockets of deeper, saline waters below a freshwater surface layer. With no or minimal exchange or turnover, the dissolved oxygen in these saline pockets declines until the next spring tide or storm. Qualitative observations on other tidal creeks suggests that the Yachats River is not unique in its structure; thus a number of tidally restricted coastal creeks may have localized areas of bottom water with low dissolved oxygen during a portion of the month.

While there are uncertainties about the dynamics and classification of these coastal creeks, we suggest that as with blind estuaries these systems could be managed on a "worst case" scenario. The most sensitive resource is likely to be the juvenile salmon that utilize many of these systems (Table 2-3). In these cases, the Oregon dissolved oxygen standard of 6.5 mg l^{-1} (http://www.sos.state.or.us/archives/rules/OARs_300/OAR_340/340_041.html) would be an appropriate criterion, except for when natural conditions prevent attainment of this standard. An ongoing study of Yachats indicates that dissolved oxygen conditions fall below 6.5 mg l^{-1} periodically in pockets of bottom water "trapped" by natural processes. The lack of typical "estuarine" resources and salmon in many of the estuaries smaller than about 0.01 km^2 suggests that the application of estuarine criteria may not be the best approach to their management. In fact, the smallest estuarine polygon in a tidal creek was only 35 m long, making application of estuarine criteria to this system dubious. A more practical strategy may be to manage the freshwater component of these systems and/or to apply human health end-points, such as water contract criteria for fecal coliforms, to the water advected onto the beach.

2.4.3 Marine Harbors/Coves

At the opposite extreme of these restricted estuaries, there is a group of coastal waterbodies that has an unobstructed connection with the ocean with relatively small freshwater inflow. These systems are separated as marine harbors/coves, and as a class should have a relatively low vulnerability to terrestrial nutrient loadings. Based on analysis of photographs and preliminary field observations, two coastal waterbodies fall into this class - Sunset Bay and Crescent City Harbor. A third system, Depoe Bay, the “world’s smallest harbor”, is tentatively classified as a marine harbor/cove. With a mouth width of about 28 meters, Depoe Bay is less open to the ocean than either Sunset Bay or Crescent City, however Depoe Bay appears to be relatively well flushed. With an open connection to the ocean and limited freshwater input, salinities in these systems will tend to remain high and undergo smaller tidal and seasonal variations than the drowned river mouth estuaries.

2.4.4 Drowned River Mouth and Bar-Built Estuaries

The remaining 31 estuaries can be split into two general classes based on geomorphology. The first are bar-built estuaries, which are formed when ocean currents and wind form coastal dunes that trap estuarine water behind them. Bar-built estuaries resemble lagoonal systems in having low freshwater inputs but differ in having greater exchange with the ocean. The two generally recognized bar-built estuaries in the PNW are Netarts Bay and Sand Lake. We classify a third estuary, the Humboldt Estuary, as bar built. Though the Humboldt has been considered a drowned river mouth estuary (e.g., Rumrill, 1998) there is no major river discharging into the estuary. More appropriately, Emmett et al. (2000) classified the Humboldt as a lagoon, and these authors pointed out that many lagoons are closed to the ocean during a portion of the year. The difficulty with classifying Humboldt as a lagoon is that it has permanent jetties maintaining an open connection with the ocean. Therefore, we suggest that with its current configuration the Humboldt more closely resembles a bar-built estuary.

The second class is the drowned river mouth estuaries, which include 28 PNW estuaries. Drowned river estuaries were created by flooding of river valleys as sea level rose during the Holocene marine transgression after the last ice ago about 10,000 years ago (Emmett et al., 2000). Drowned river mouth estuaries constitute the largest estuaries in the PNW, including the Columbia River, Willapa Estuary, and Grays Harbor. Although formed by flooding of river valleys, the geomorphology and size of many of the drowned river mouth estuaries in the PNW are influenced by the formation of ocean-built bars. The drowned river mouth estuaries with prominent ocean bars share characteristics with bar-built estuaries, and some authors have classified them as bar built. For example, Willapa and Grays (Seliskar and Gallagher, 1983) as well as the Salmon River (http://www.coastalatlas.net) have been classified bar-built estuaries. While recognizing the importance of these ocean bars to their dynamics, we classify these systems as drowned river mouth estuaries based both on their historic formation and the presence of one or more moderate to large sized rivers discharging into the estuary.

2.4.4.1 Tide- and River-Dominated Drowned River Mouth Estuaries

Drowned river mouth estuaries vary in the extent that their geomorphology and flushing are driven by tidal versus fluvial processes. For example, the Columbia River and Rogue River are considered to be strongly influenced by river flow (Cortright et al., 1987; McCabe et al., 1988) while Willapa has relatively little freshwater input for its size. While categorization of systems like the Columbia and Willapa is relatively straightforward, the difficulty comes in classifying the full suite of PNW estuaries without rigorous criteria. The Australians separate the two types of systems depending upon whether river or tide energy "dominated the evolution" of the estuary (http://dbforms.ga.gov.au/www/npm.Ozcoast.glossary?pType=Audit). In a similar vein, Elliott and McLusky (2002) define river-dominated estuaries as microtidal systems where rivers provide the majority of the sediment and tide-dominated estuaries as macrotidal systems where "there is a steady infilling of sediment provided at differing times from the sea and from the river." Such geologically based definitions help highlight the factors driving the formation of an estuary but they provide little insight into flushing, nutrient retention, or salinity patterns.

A more relevant approach to assessing estuarine vulnerability to nutrient enrichment is to separate tide- versus river-dominated systems based on the relative extent of freshwater flow through the estuary. To capture this relative input, we propose "normalized freshwater inflow" metrics. The first step in generating these metrics is to calculate the total average annual volume of rainfall over the entire watershed using the precipitation data in PRISM (http://prism.oregonstate.edu/; Daly et al., 2007), which has records from 1971 to 2000. This volume represents the total amount of freshwater impinging on the watershed, which then evaporates, percolates into the soil, or runs off into the estuary. As a first-order simplification, we assume that evaporation and percolation are similar among PNW watersheds, so that a similar fraction of the rainfall ultimately flows into each of the estuaries. Thus, this total volume of precipitation is a relative measure of the average annual freshwater flow into the estuary. Once in the estuary, this freshwater inflow mixes with ocean water transported into the estuary with the extent of mixing dependent, in part, upon estuary volume. Therefore, the next step is to normalize the total annual precipitation (m^3 of freshwater per year) by the total volume of the estuary (m^3). This metric has units of $year^{-1}$ and is referred to as the annual "volume-normalized freshwater inflow". Higher values of this metric indicate a more freshwater (river) dominated system. Among the 17 PNW estuaries where bathymetry is available, the volume-normalized freshwater inflow varies from 4 $year^{-1}$ in the Humboldt Estuary to 2501 $year^{-1}$ in the Rogue River (Table 2-4). Because bathymetry is not available for most PNW estuaries, we developed an alternative approach of normalizing the total annual precipitation by estuarine area. This annual "area-normalized freshwater inflow" metric has units of m^3 of freshwater per m^2 of estuary per year and is directly related to the volume normalized values (Figure 2-10). Since it is available for all the estuaries, we use the area-normalized metric as our approach to ranking estuaries by the extent of riverine influence.

Advantages of these watershed-scale metrics of freshwater input compared to flow data from gauged stations are that they integrate precipitation over the entire watershed rather than measuring flow in a single or limited number of tributaries, they can be calculated for estuaries that are not gauged, and they can be linked to climate change scenarios to predict altered freshwater inflow under different precipitation regimes. Advantages over simply using the ratio of the drainage area to the estuary size are that the normalized freshwater inflow metrics capture

Table 2-4. Freshwater sources into estuaries, normalized freshwater inflow, mouth width, and structure of mouth for drowned river mouth and bar-built estuaries in the PNW. Bar-built estuaries are italicized. Estuaries with area-normalized freshwater inflows less than 175 m^3 m^{-2} $year^{-1}$ are classified as tide dominated and estuaries with greater values are classified as river dominated. The river-dominated estuaries are further subdivided into "highly river-dominated systems" (dark gray) with area-normalized freshwater inflow values $\geq$400 m^3 m^{-2} $year^{-1}$ and "moderately river-dominated systems" (light gray) with values $\geq$175 m^3 m^{-2} $year^{-1}$ and <400 m^3 m^{-2} $year^{-1}$. Estuarine volume obtained from http://ian.umces.edu/neea/ with the exception of Salmon River which was from Johnson and Gonor (1982). The estimates for the normalized freshwater inflows for the Columbia River are based on the annual river discharge versus the total volume of precipitation in the watershed and are not directly comparable to the values in the other estuaries.

ESTUARY	MAJOR FRESHWATER SOURCES	NORMALIZED FRESHWATER INFLOW		MOUTH WIDTH (m)	ESTUARY MOUTH STRUCTURE
		AREA NORMALIZED (m^3 m^{-2} $year^{-1}$)	VOLUME NORMALIZED ($year^{-1}$)		
Columbia	Columbia River and multiple creeks	370	86	5038	Jetties on both side of the mouth. Note: Freshwater inflow based on river discharge and not precipitation.
Klamath	Klamath River	18,082	2435	350	Mouth constrained by bluffs to the north and south. Dunes may constrict mouth width.
Rogue	Rogue river	6,537	2501	280	Jetties on both side of the mouth. Mouth may have migrated before jetties.
Quinault	Quinault River	6,286		38	Bluff constrains mouth to the north. Dunes may constrict mouth width.
Quillayute	Quillayute River and 2 subsidiary creeks	5,220		250	Jetties on both side of the mouth. Dunes direct estuary northward of mouth.
Chetco	Chetco River	3,771		78	Jetties on both sides of the mouth.
Hoh	Hoh River	3,468		28	Mouth constrained by bluffs to the north and south. Dunes may constrict mouth width.
Queets	Queets River	2,796		70	Dunes may constrict mouth width. Forms dendritic channels behind dunes and directs estuary northward of mouth.
Smith (CA)	Smith River and 1 subsidiary creek	2,159		110	Mouth constrained by bluffs to the north. Dunes direct estuary southward of mouth.

ESTUARY	MAJOR FRESHWATER SOURCES	NORMALIZED FRESHWATER INFLOW		MOUTH WIDTH (m)	ESTUARY MOUTH STRUCTURE
		AREA NORMALIZED (m^3 m^{-2} $year^{-1}$)	VOLUME NORMALIZED ($year^{-1}$)		
Mad	Mad River	1,779		137	Dunes direct estuary southward of mouth and form a "lagoon" to the north of the current mouth. Mouth may occasionally close .
Eel	Eel River and several subsidiary creeks	997	524	150	Forms dendritic channels behind dunes. Dunes direct estuary northward of mouth.
Coquille	Coquille River	695	143	175	Jetties on both side of the mouth.
Sooes	Sooes River	539		50	Dunes direct estuary southward of mouth. Mouth constrained by land to north and south.
Nehalem	Nehalem River	499	293	180	Jetties on both side of the mouth. Dunes direct estuary northward from mouth.
Umpqua	Umpqua River and Smith River	484	219	425	Jetties on both side of the mouth.
Nestucca	Nestucca River and Little Nestucca River	420		110	Dunes direct estuary northward from mouth. Mouth constrained by land to the south.
Necanicum	Necanicum River, Neawanna Creek, and several subsidiary creeks	387		60	Dunes direct estuary northward and then southward from mouth.
Copalis	Copalis River and Cedar Creek	318		20	Dunes direct estuary southward of mouth.
Siletz	Siletz River, Schooner Creek, and 3 subsidiary creeks	284	137	100	Dunes direct estuary southward of mouth. Dunes may constrict mouth width. Mouth constrained by land to the north.
Siuslaw	Siuslaw River and North Fork Siuslaw River	227	125	225	Jetties on both side of the mouth. Dunes direct estuary southward from mouth.
Alsea	Alsea River and several creeks including Lint and Drift Creeks	211	141	140	Mouth constrained by bluffs to the south.
Salmon	Salmon River and several subsidiary creeks	177	392	40	Mouth constrained by land to the north. Dunes direct estuary southward from mouth.
Tillamook	Kilchis, Wilson, Tillamook, Trask, and Miami Rivers	116	62	360	Jetties on both side of the mouth. Dunes direct estuary southward from mouth. Dunes result in accumulation of freshwater flow from five rivers.

ESTUARY	MAJOR FRESHWATER SOURCES	NORMALIZED FRESHWATER INFLOW		MOUTH WIDTH (m)	ESTUARY MOUTH STRUCTURE
		AREA NORMALIZED (m^3 m^{-2} $year^{-1}$)	VOLUME NORMALIZED ($year^{-1}$)		
Waatch	Waatch River	101		32	Mouth constrained by bluff to the north. Dunes may constrict mouth width.
Yaquina	Yaquina River and several creeks draining into sloughs	63	42	290	Jetties on both side of the mouth.
Grays	Chehalis River and Elk, Johns, and Hoquiam Rivers and several subsidiary creeks	58	32	2750	Jetties on both side of the mouth.
Coos	Coos River and several creeks discharging into sloughs including South and Isthmus Slough, and Catching and Palouse Creek	53	14	620	Jetties on both side of the mouth. Dunes direct main stem of estuary northward from mouth.
Sand Lake	Sand Creek and Jewell Creek and 2 subsidiary creeks. Groundwater?	28		135	Dunes form mouth, with extensive sand accumulation behind dunes. Mouth may occasionally close off estuary.
Willapa	Bear, Bone, Cedar, Naselle, Nemah, Niawiakum, North, Palix, and Willapa rivers	17	6	9165	Mouth constrained by land to the north. Dunes direct estuary southward from mouth. Dunes result in accumulation of freshwater flow from rivers.
Humboldt	Freshwater Creek and Elk River and to lesser extent Jacoby Creek and Salmon Creek	12	4	842	Jetties on both side of the mouth. Estuary runs behind dunes both northward and southward. Dunes result in accumulation of freshwater flow from rivers and creeks.
Netarts	13 minor creeks	11	8	114	Dunes direct estuary southward from mouth.

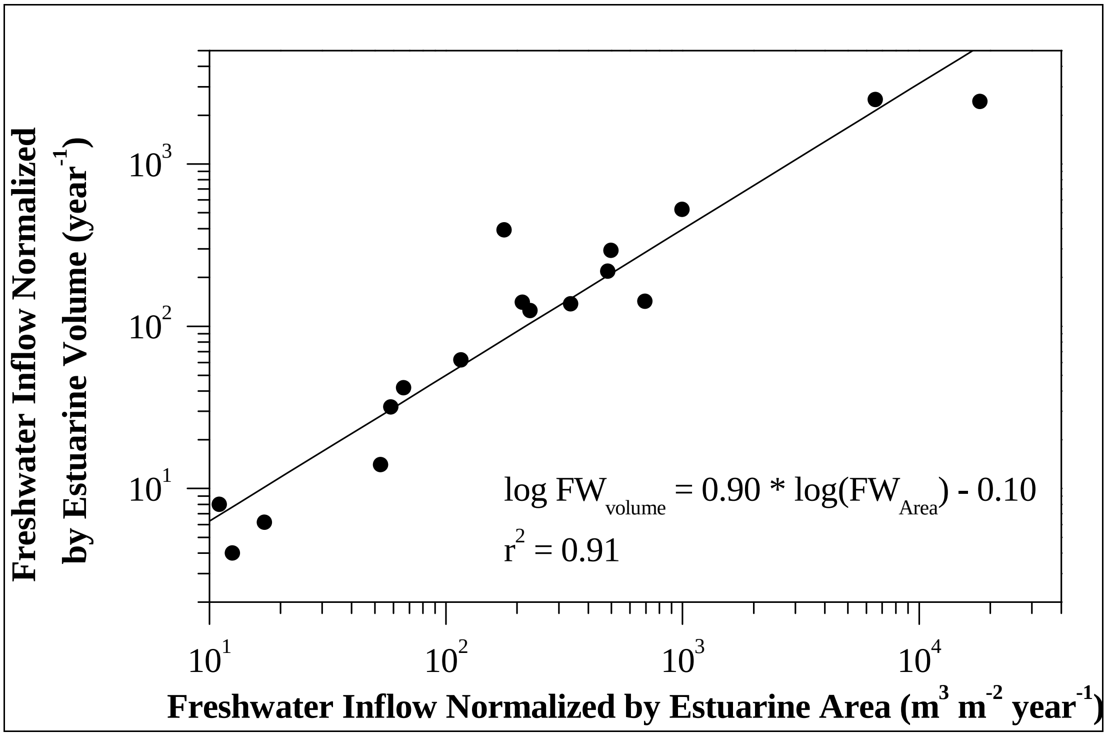

Figure 2-10. Relationship between freshwater inflow normalized by estuarine area and normalized by estuarine volume.

regional differences in rainfall, monthly values can be generated to evaluate seasonal changes in potential flushing, and, again, the metric can be linked to climate change scenarios.

The area-normalized freshwater inflow was calculated for all the drowned river mouth estuaries and bar-built estuaries (Table 2-4) except for the Columbia Estuary. Total precipitation was not available for the entire Columbia Basin, though it was possible to generate a rough estimate from the Columbia River's total annual discharge of approximately 198,000,000 acre-feet of water (http://vulcan.wr.usgs.gov/Volcanoes/Washington/ColumbiaRiver/description_columbia_river.html). This annual river discharge is equivalent to a normalized freshwater inflow of approximately 370 m^3 per m^2 per year. This estimate is a substantial underestimate compared to the values for other estuaries since it does not include any water lost to evaporation or percolation, which would be substantial over the entire Columbia Basin. While this value is not directly comparable to the other values, it indicates that the Columbia Estuary has a high river input, which is consistent with the tidal riverine segment constituting over a third of the entire estuarine area (see Table 2-2).

Area-normalized freshwater inflow values vary over a thousand-fold, from about 11 in Netarts Bay to 18,000 m^3 m^{-2} year^{-1} in the Klamath. The metric successfully separates estuaries generally considered river-dominated systems (e.g., Klamath and Rogue) from those considered tide-dominated systems (e.g., Yaquina) suggesting that it can be used to classify less well studied systems. Additionally, spatial and seasonal patterns in salinity variations in these estuaries appear to be related to normalized freshwater inflow. Figure 2-11 shows salinity versus distance from the mouth for eight estuaries where both dry and wet season salinities are available. These eight estuaries range in area-normalized freshwater inflow from 11 (Netarts) to 695 m^3 m^{-2} year^{-1}

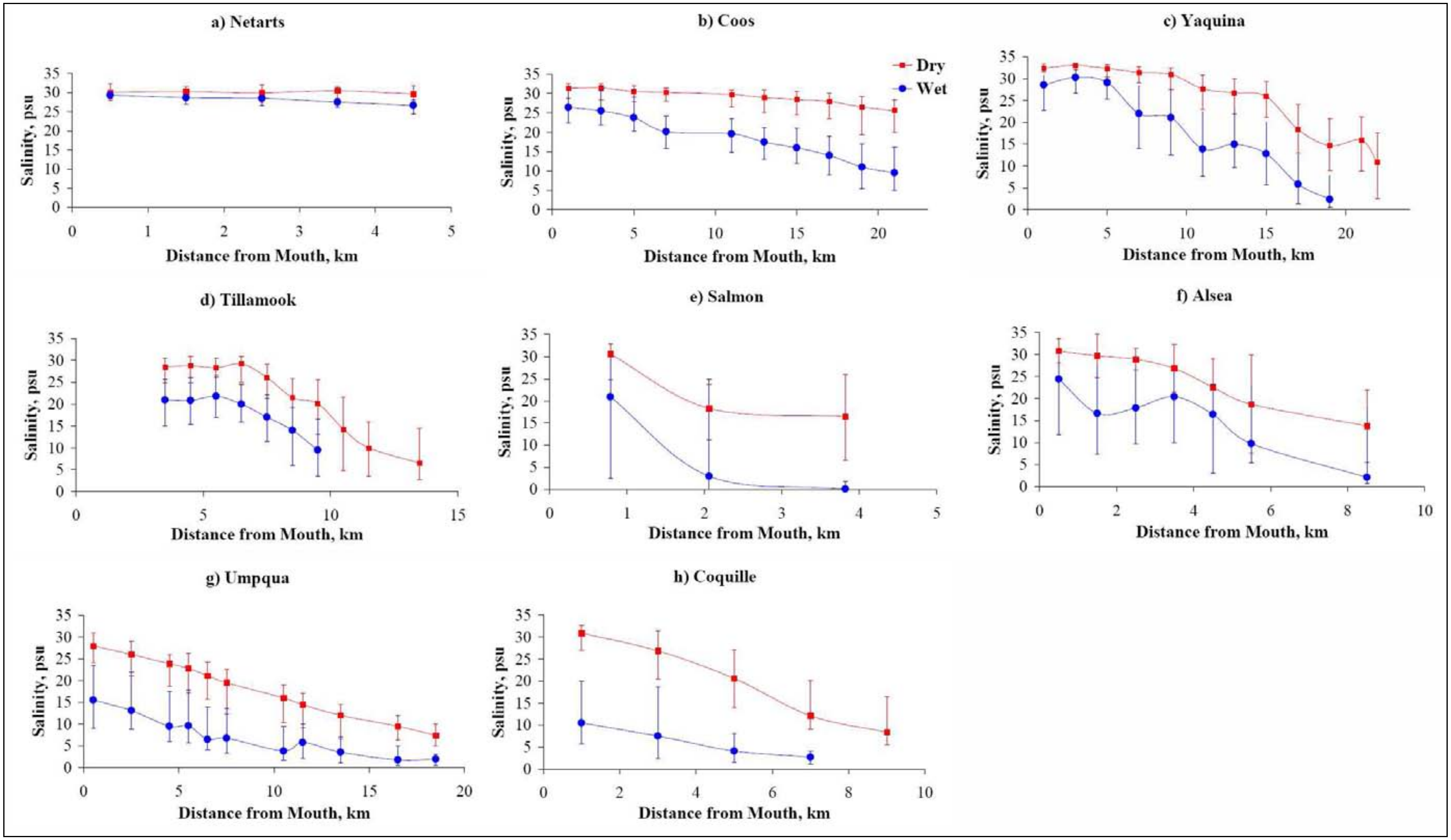

Figure 2-11. Salinity versus distance from the mouth for the wet and dry seasons for estuaries with area-normalized freshwater inflow ranging from about 11 to 695 $m^3\,m^{-2}$ $year^{-1}$. Salinity variations are provided for bar-built, tide-dominated, moderately river-dominated, and highly river-dominated systems. The symbols indicate the median values and the error bars represent the 25th and 75th percentiles. Salinity data are from the Oregon Department of Environmental Quality (http://deq12.deq.state.or.us/lasar2/) except for panel e. Data for panel e are from short-term deployments of sondes described in Chapter 4 Note that the Salmon River has the largest volume-normalized freshwater inflow, reflecting its shallowness.

(Coquille). Systems with low freshwater inflow have less salinity variation along the main axis of the estuary, salt penetrates further into these systems, and they have less seasonal variations in salinity than river-dominated systems. For the two estuaries with the highest values (Umpqua and Coquille), there is essentially a linear decline in dry season salinity with distance from the mouth. Finally, there is a positive relationship between area-normalized freshwater inflow values and variation in salinity over a tidal cycle at the mouth of the seven target estuaries (Figure 4-11). Thus, it appears that the area-normalized freshwater inflow metric captures key aspects of tide versus river domination as measured by salinity.

The next question becomes what value to use to separate tidal from riverine systems. A reasonably distinct break in values occurs between the Salmon and Tillamook estuaries, with the Tillamook value about two-thirds of the Salmon value (Table 2-4). This break is also associated with a difference in wet and dry season salinity variations (Figure 2-11e). Based on this pattern, we propose a value of $\geq$175 m^3 m^{-2} $year^{-1}$ as a preliminary threshold for defining river-dominated estuaries. There is also a moderate break in values between the Nestucca and Necanicum (Table 2-4), which is also reflected in the salinity patterns in the Salmon and Alsea versus the Umpqua and Coquille (Figure 2-11). This pattern in salinity variation suggests a subdivision of river-dominated systems into "highly river-dominated" systems with inflow values greater than approximately 400 m^3 $m^{-2}$and "moderately river-dominated" systems with inflow values between 175 m^3 m^{-2} and 400 m^3 m^{-2} $year^{-1}$.

This approach classifies 22 of the 28 drowned river mouth estuaries as river-dominated and 6 as tide-dominated (Table 2-4). Of the river-dominated estuaries, 16 are classified as highly river-dominated and 6 as moderately river-dominated. In terms of geomorphology, the river-dominated systems, and the highly river-dominated systems in particular, tend to have fewer freshwater inputs compared to the tide-dominated systems which tend to have smaller freshwater inputs distributed across several tributaries. Another pattern is that all of the small (<5 km^2) river mouth estuaries are classified as river dominated with the exception of the Waatch, while 10 of the 14 estuaries <5 km^2 are classified as highly river dominated. All three bar-built estuaries have very low normalized freshwater inflows, indicating that functionally they more resemble the tide-dominated systems.

In terms of vulnerability, our initial hypothesis is that, all other factors being equal, estuaries with higher normalized freshwater inflow will be less susceptible to terrestrially derived nutrient enrichment because of higher flushing. The higher river flow in riverine systems may actually increase the total loading from the watershed, but the high flushing should reduce the ability of phytoplankton to maintain high populations. This suggestion is consistent with the trend towards a decrease in median dry season chlorophyll *a* with increasing volume-normalized freshwater inflow (Figure 4-6). The data from this set of estuaries suggest that summer chlorophyll *a* levels in the estuaries may be determined by flushing time. Evaluation of this proposed relationship between normalized river inflow and chlorophyll *a* concentrations will require additional water quality studies across a range of estuary types as well as developing more complete water quality models to explain differences among estuaries with similar riverine inputs. Tide-dominated systems have close coupling with the coastal ocean and as a result may experience high nutrient and chlorophyll *a* levels and low dissolved oxygen conditions as a result of intrusion of ocean water into the estuary.

2.5 Classification of Estuaries by NWI Wetland Classes

The next step in classifying these drowned river mouth and bar-built estuaries was to determine similarity among estuaries based on wetland distributions. Distribution of NWI wetland types serves as an integrated response to multiple drivers, including estuarine geomorphology, watershed size and type, freshwater inflow, oceanic exchange, salinity structure, and tidal exposure. Thus, patterns of NWI wetlands classes are potentially a better indicator of environmental similarity than a classification based on any of those environmental drivers individually. Classification by NWI classes also provides a mechanism to identify estuaries with similar wetland resources, which could be used in establishing management prioritizations or assist in designing specific mitigation actions.

Utilizing the consolidated NWI codes (Table 2-1), the 31 estuaries were first clustered based on the actual areas of the wetland/habitat classes. This and the following cluster analyses are based on the NWI available in 2006 since there were only minor changes in these larger estuaries in the recent update. Since this analysis groups estuaries by habitat areas, classification by area will tend to group larger and smaller estuaries, and in terms of management is probably most useful in identifying groups of estuaries with similar extents of wetland resources. The 31 estuaries clustered into four significant classes using the default p value of 0.05 (Figure 2-12). As expected, the clusters broke out by estuarine size. One group (Cluster A) consisted of the three largest estuaries: Willapa, Grays Harbor, and the Columbia River. The next group (Cluster B) consisted of the next four largest estuaries: Humboldt, Coos, Umpqua, and Tillamook. Cluster C consisted of the 10 moderate-sized estuaries, ranging in size from 4.28 km^2 (Sand Lake) to 19.96 km^2 (Yaquina Estuary). The final group (Cluster D) consisted of the suite of smaller estuaries (0.56 km^2 to 3.1 km^2). The effect of increasing the p value was to break out the Columbia as a separate group from Grays Harbor and Willapa (at p=0.10), but further increasing the significance values did not split out any additional groups (Figures 2-13a to 2-13d).

A similar analysis was conducted using the relative proportion of the area of each of the consolidated NWI classes. Use of proportional areas removed the direct effect of estuary size and should better identify groups of estuaries that are functionally similar compared to the classifications by actual areas. Based on the relative proportions, the 31 estuaries grouped into three clusters at p=0.05 (Figure 2-14). One group (Cluster A) consisted of eight estuaries showing a wide range in size, from the Quinault (0.66 km^2) to the Columbia River (669 km^2). Though varying greatly in size, members in this cluster consist of the eight estuaries with the largest normalized freshwater inflow (Table 2-4), assuming that the Columbia ranks among the top eight. This grouping indicates that there are characteristic wetland profiles in the most highly river-dominated systems. Another group (Cluster C) consisted of the Salmon and Waatch, which showed little similarity with the other groups. These two systems separated from the other estuaries largely due to their high proportion of upper marsh habitat (estuarine, irregularly flooded, rooted class of Table 2-1).

The third group (Cluster B) consisted of 21 estuaries including both river- and tide-dominated river mouth estuaries as well as all three of the bar-built estuaries. Size varied widely in this group, from 0.56 km^2 (Sooes) to 390.9 km^2 (Willapa). The estuaries making up this cluster are so diverse that it does not seem useful in grouping systems in terms of nutrient dynamics. However, increasing the significance levels helped to separate estuaries with similar attributes

(Figures 2-15a to 2-15d). The 10% significance level separated Grays Harbor and Willapa Bay as a group (Figure 2-15a), two estuaries that share many similarities. At a significance level of 40%, 12 clusters were identified (Figure 2-15c) while the 60% significance level identified 16 clusters (Figure 2-15d). Using the criterion of combining estuaries with $\geq$ 75% similarity results in 10 "functional" groups at the 40% significance level and 12 "functional" groups at the 60% significance level (Figure 2-15d).

Choosing which of these classifications is the "best" is not a statistical question but rather depends upon the goals of the classification. Our interpretation is that the classification based on the 60% significance level along with combining estuaries with $\geq$ 75% similarity best captures ecological similarity in terms of drivers related to wetland patterns while avoiding separation of very similar systems (Figure 2-15d). Compared to the 40% significance level, the classification based on the 60% significance level separated all the tide-dominated estuaries from the highly river-dominated systems, though there is still some mixing of the tide-dominated, moderately river-dominated, and bar-built estuaries. However, the practical limitation of this classification is that it results in 12 functional groups, which may constitute too fine a division for management. An alternative might be to use the classification based on the 10% significance level which consists of four groups (Figure 2-15a).

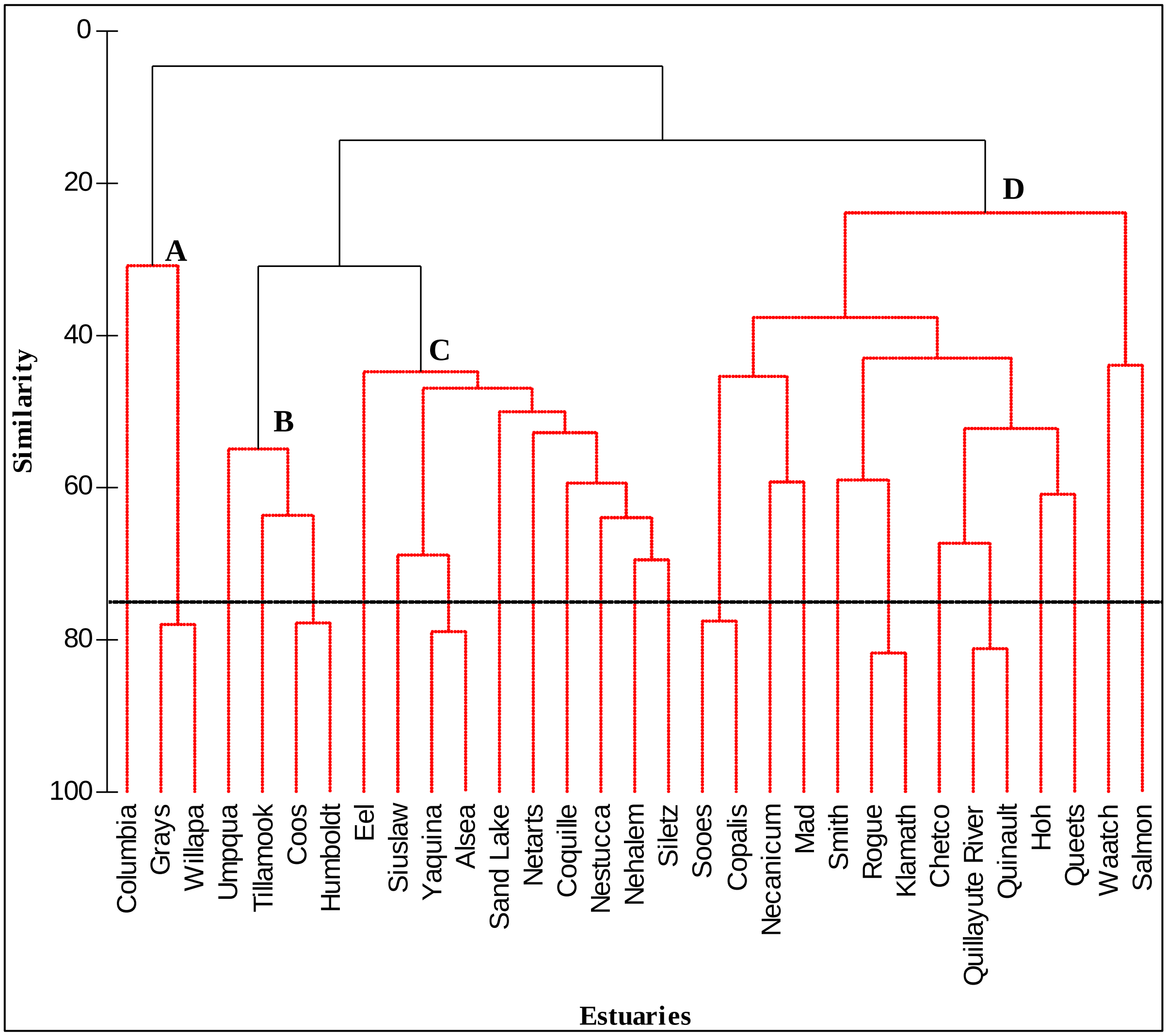

Figure 2-12. Cluster analysis of 31 PNW estuaries at the 5% significance level based on the area of the consolidated NWI habitats. Estuaries joined with red lines are not significantly different ($p>0.05$) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

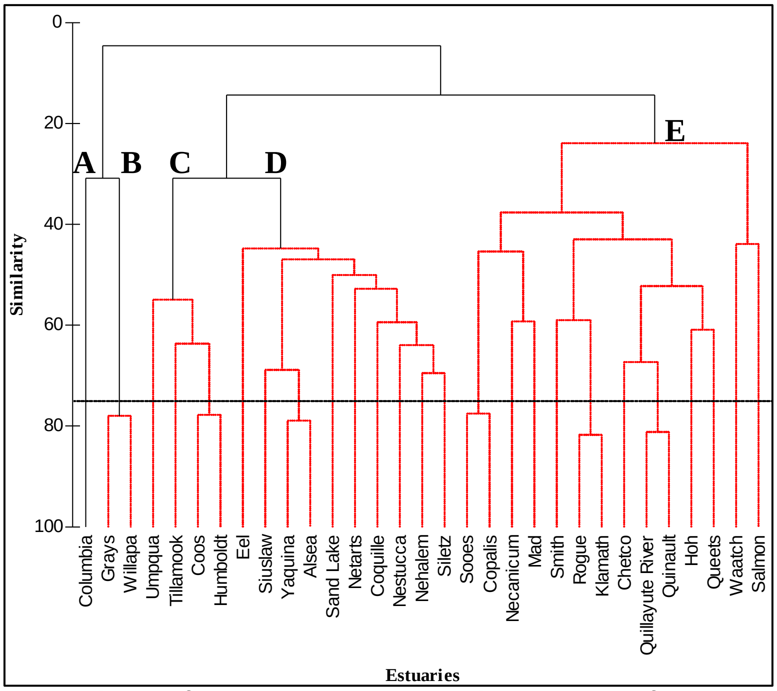

Figure 2-13a. Cluster analysis of 31 PNW estuaries at the 10% significance level based on the area of the consolidated NWI habitats. Estuaries joined with red lines are not significantly different (p>0.10) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

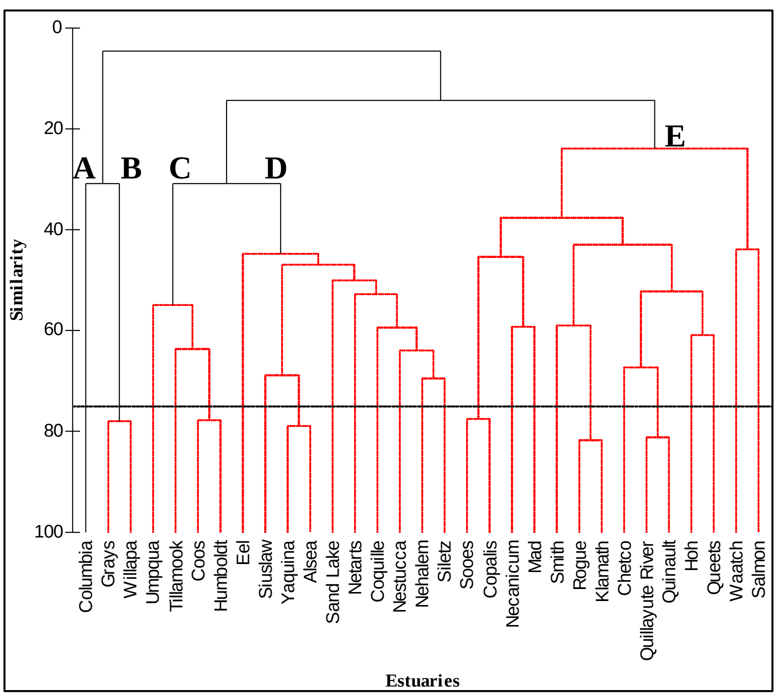

Figure 2-13b. Cluster analysis of 31 PNW estuaries at the 20% significance level based on the area of the consolidated NWI habitats. Estuaries joined with red lines are not significantly different (p>0.20) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

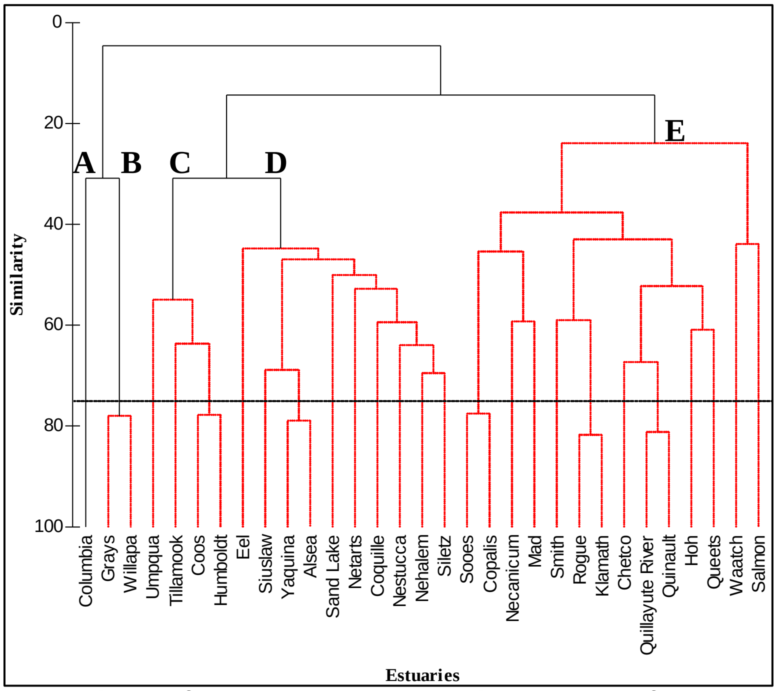

Figure 2-13c. Cluster analysis of 31 PNW estuaries at the 40% significance level based on the area of the consolidated NWI habitats. Estuaries joined with red lines are not significantly different (p>0.40) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

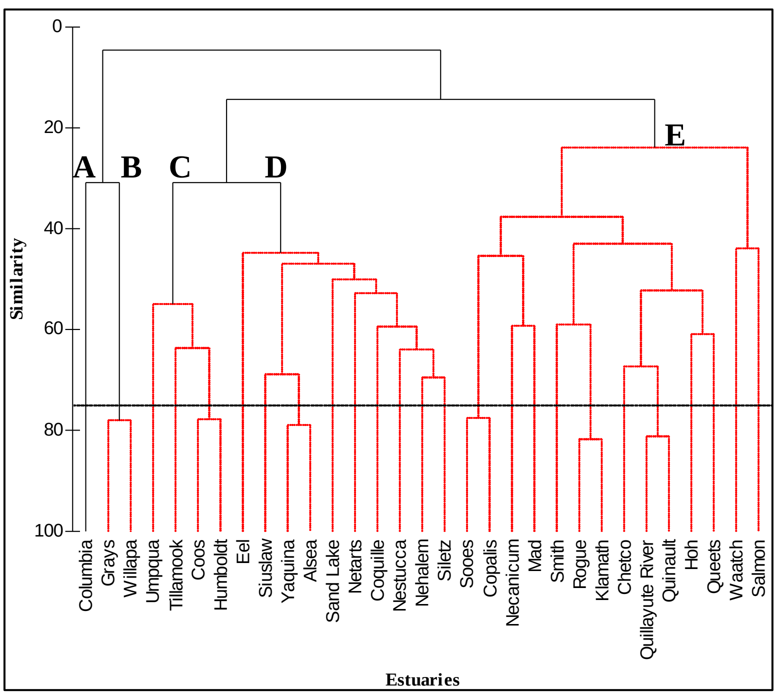

Figure 2-13d. Cluster analysis of 31 PNW estuaries at the 60% significance level based on the area of the consolidated NWI habitats. Estuaries joined with red lines are not significantly different (p>0.60) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

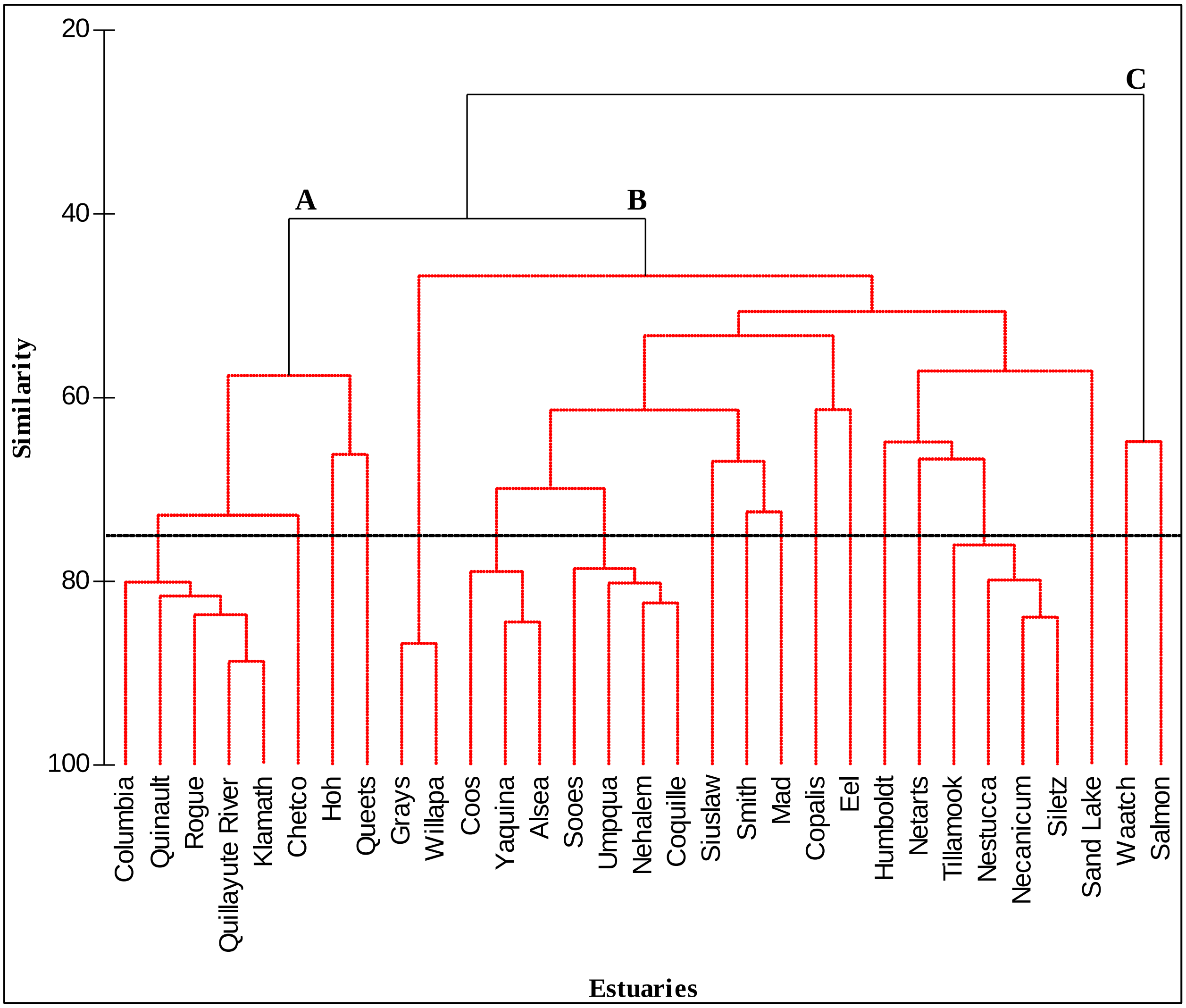

Figure 2-14. Cluster analysis of 31 PNW estuaries at the 5% significance level based on the relative proportion of the consolidated NWI habitats. Estuaries joined with red lines are not significantly different (p>0.05) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

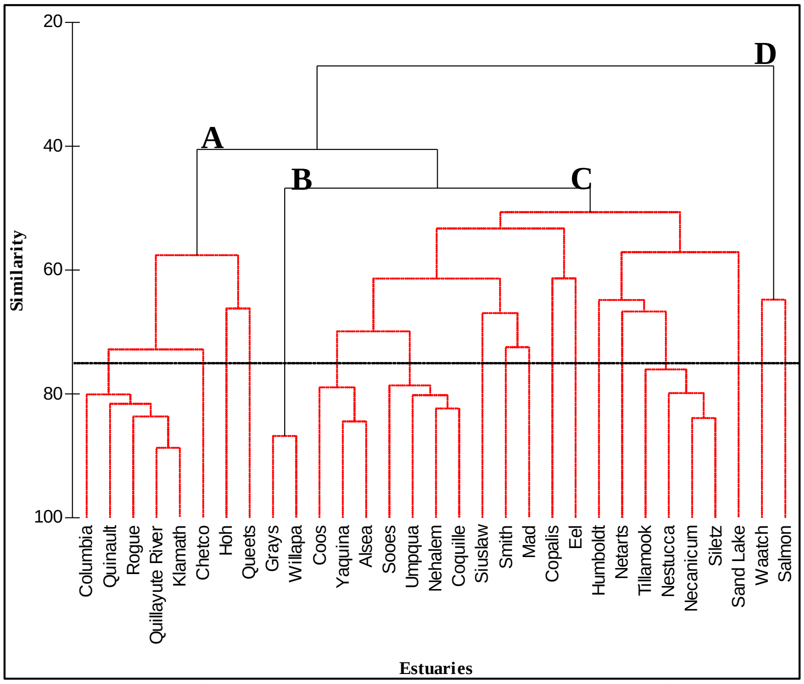

Figure 2-15a. Cluster analysis of 31 PNW estuaries at the 10% significance level based on the relative proportion of the consolidated NWI habitats. Estuaries joined with red lines are not significantly different (p>0.10) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

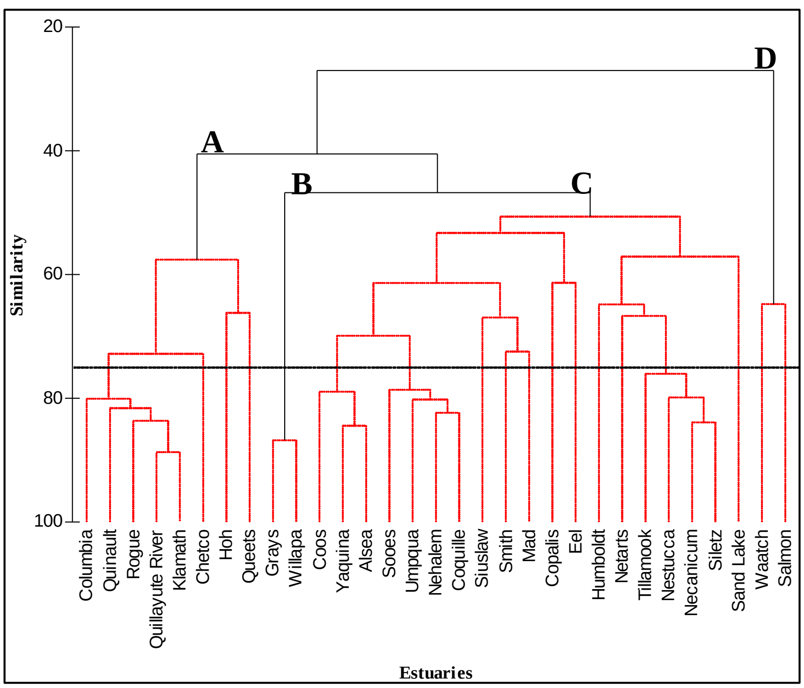

Figure 2-15b. Cluster analysis of 31 PNW estuaries at the 20% significance level based on the relative proportion of the consolidated NWI habitats. Estuaries joined with red lines are not significantly different (p>0.20) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

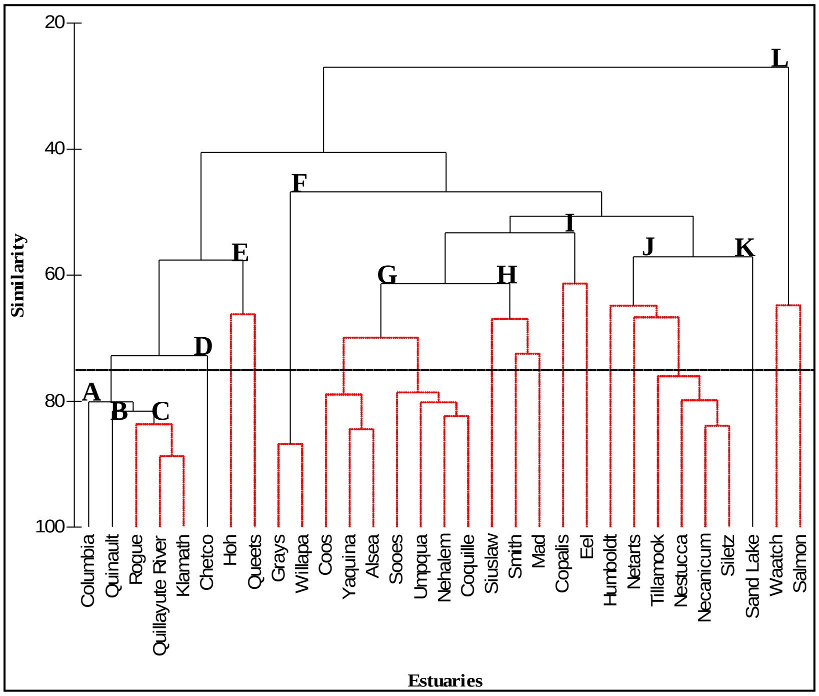

Figure 2-15c. Cluster analysis of 31 PNW estuaries at the 40% significance level based on the relative proportion of the consolidated NWI habitats. Estuaries joined with red lines are not significantly different (p>0.40) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

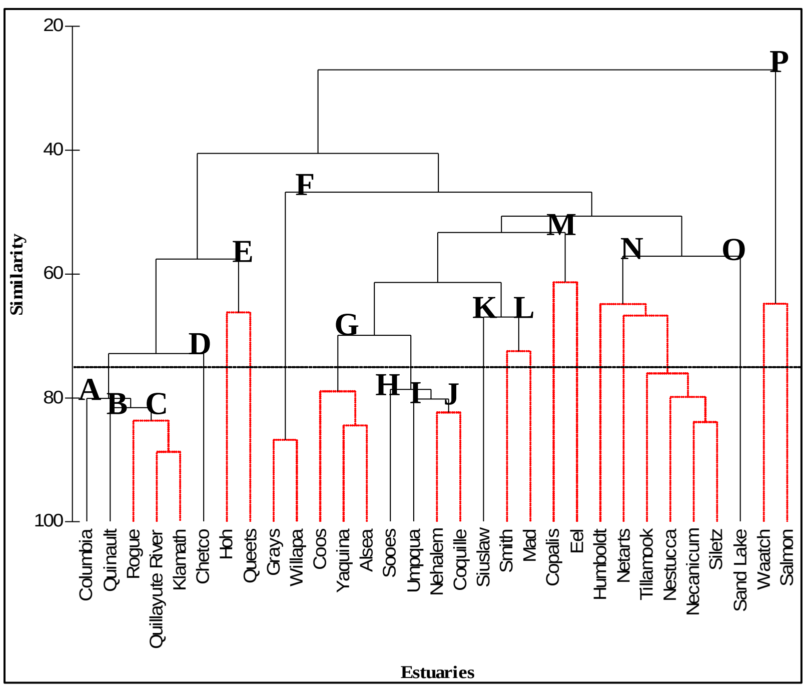

Figure 2-15d. Cluster analysis of 31 PNW estuaries at the 60% significance level based on the relative proportion of the consolidated NWI habitats. Estuaries joined with red lines are not significantly different (p>0.60) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

This classification separated out the eight highly riverine systems, Willapa and Grays Harbor, and the Salmon and Waatch. The limitation is that the fourth group consists of 19 estuaries ranging from bar-built to highly riverine dominated which appear to have different nutrient dynamics. One of the joint scientific and management challenges then becomes determining "how similar is similar enough".

2.6 Watershed Classification

Grouping estuaries by NWI classes identified suites of estuaries with similar patterns of wetland habitats and presumably similar vulnerabilities to nutrient enrichment. A different question is how the watersheds of the 31 estuaries group together. Within the same climatic regime, estuaries with similar land cover patterns in the watersheds presumably have generally similar nutrient loadings. To evaluate similarity in land cover patterns, the 31 watersheds were clustered using the land cover classes from the 2001 NOAA dataset. The NOAA coverage for the Columbia extended to approximately the Bonneville Dam, which captures the watershed adjacent to the Lower Columbia, representing about 2.5% of the entire Columbian Basin. Thus, the results for the Columbia watershed need to be interpreted in light of the fact that only a small portion of the entire land mass draining into the Columbia River was captured. The NOAA coverage for the Klamath was truncated at the northeastern segment, but as mentioned this area was filled in by using the 2001 NLCD data. A complexity in the PNW in that the presence of nitrogen-fixing alder can be an important component of nitrogen dynamics in these watersheds (see Sections 3.7 and 4.2). NOAA's "deciduous forest" and "mixed forest" classes capture alder but do not separate them out from other deciduous trees. This level of detail should be sufficient to capture general similarities among watersheds, though differences in alder coverage may result in somewhat different nitrogen loadings among otherwise similar watersheds.

2.6.1 Watershed Classes

The analysis based on the areas of the land cover classes resulted in seven clusters at a significance level of 5% (Figure 2-16). As expected, there was a tendency for larger watersheds to cluster together. The five estuaries with the largest watersheds – the highly riverine dominated Columbia, Klamath, Rogue, Umpqua, and Eel – all formed one group (Cluster A). Similarly, the three estuaries with the smallest watersheds (<100 km^2) grouped together in Cluster G while the estuaries with the next smallest watersheds (105 km^2 to 216 km^2) formed a related cluster (Cluster F). Most of the estuaries with moderate-sized watersheds grouped in Clusters B, C, and D, which had a relatively high similarity (>55%) among the three groups. The Humboldt formed a unique cluster (Cluster E) that had moderate similarity (<50%) to the other moderate-sized watersheds. Using different significance levels in the cluster analysis did not substantially change the classifications (Figure 2-17a to 2-17d), with the major changes being the separation of the Eel Estuary from the functional group consisting of the other estuaries with large watersheds (A-D in Figure 2-17d) and splitting the tide-dominated Waatch from the bar-built Sand Lake and Netarts watersheds (Figure 2-17c).

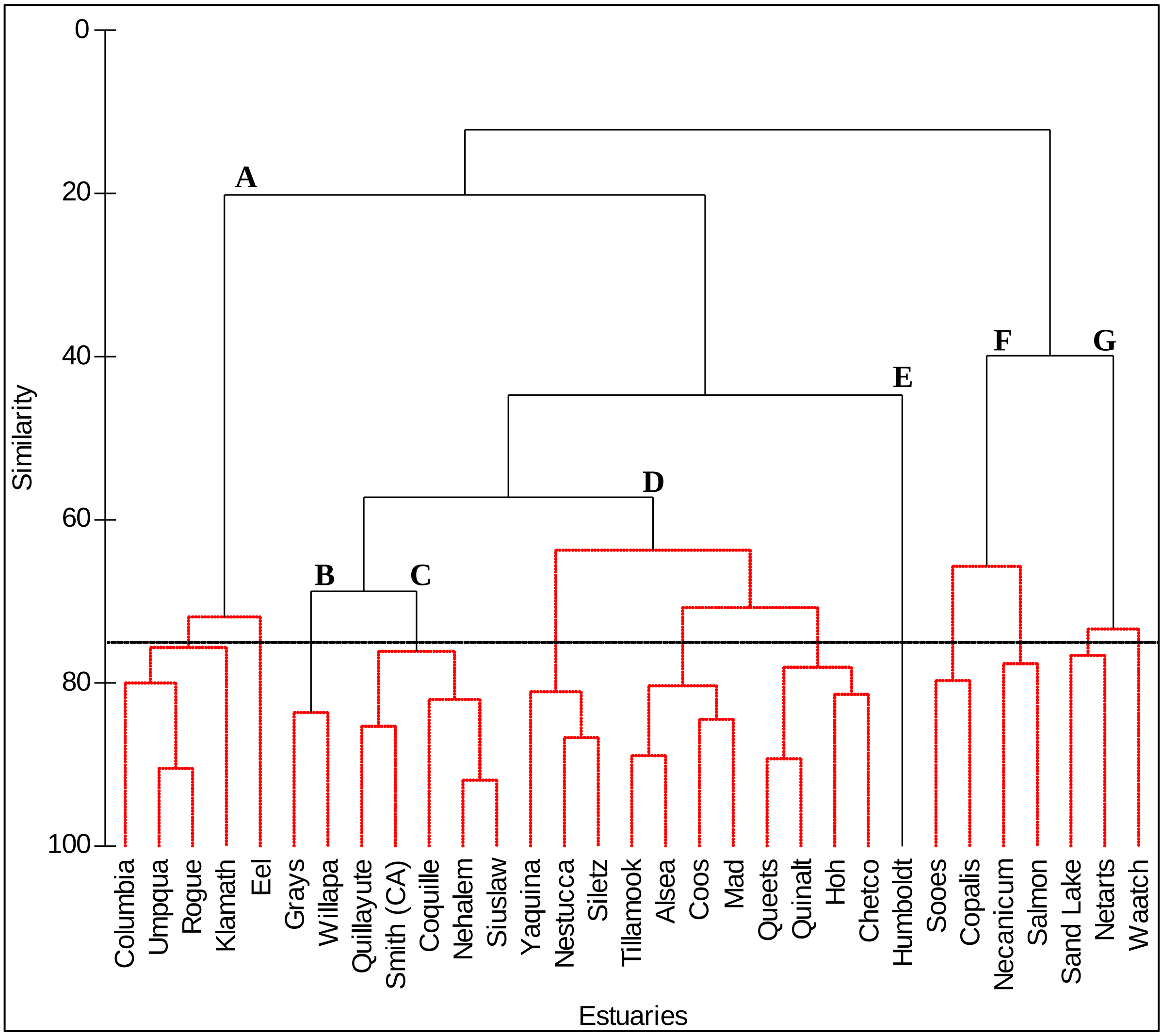

Figure 2-16. Cluster analysis of 31 PNW estuaries at the 5% significance level based on the areas of the land cover classes in the NOAA 2001 watershed data. Estuaries joined with red lines are not significantly different (p>0.05) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

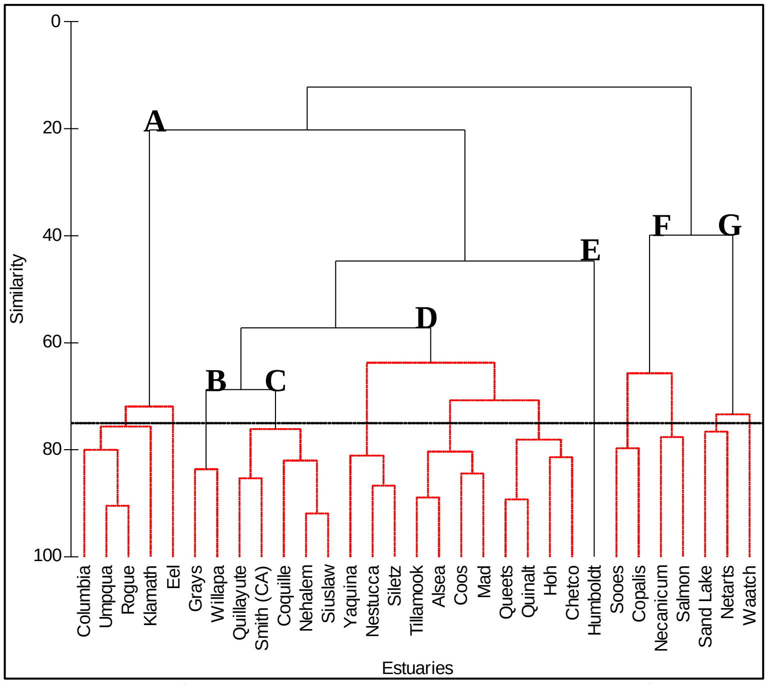

Figure 2-17a. Cluster analysis of 31 PNW estuaries at the 10% significance level based on the areas of the land cover classes in the NOAA 2001 watershed data. Estuaries joined with red lines are not significantly different (p>0.10) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

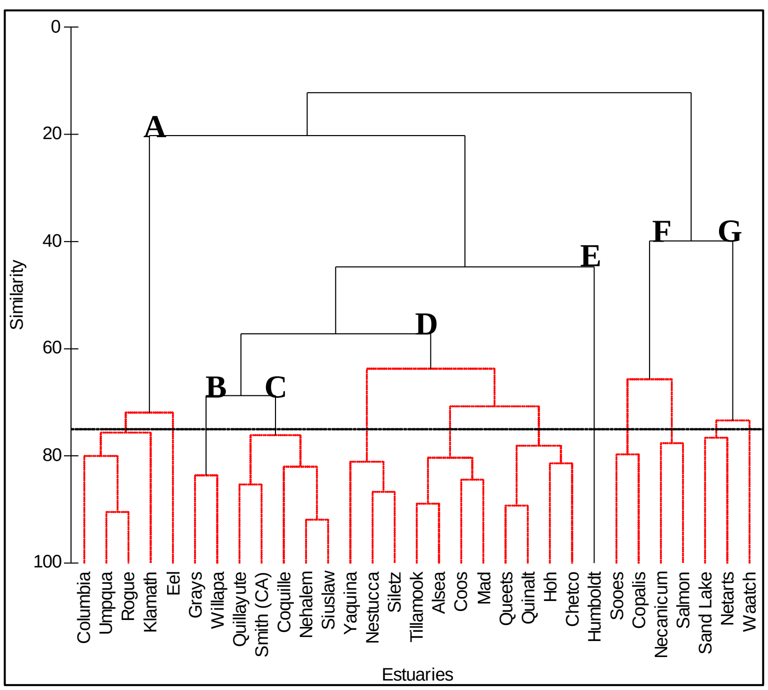

Figure 2-17b. Cluster analysis of 31 PNW estuaries at the 20% significance level based on the areas of the land cover classes in the NOAA 2001 watershed data. Estuaries joined with red lines are not significantly different (p>0.20) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

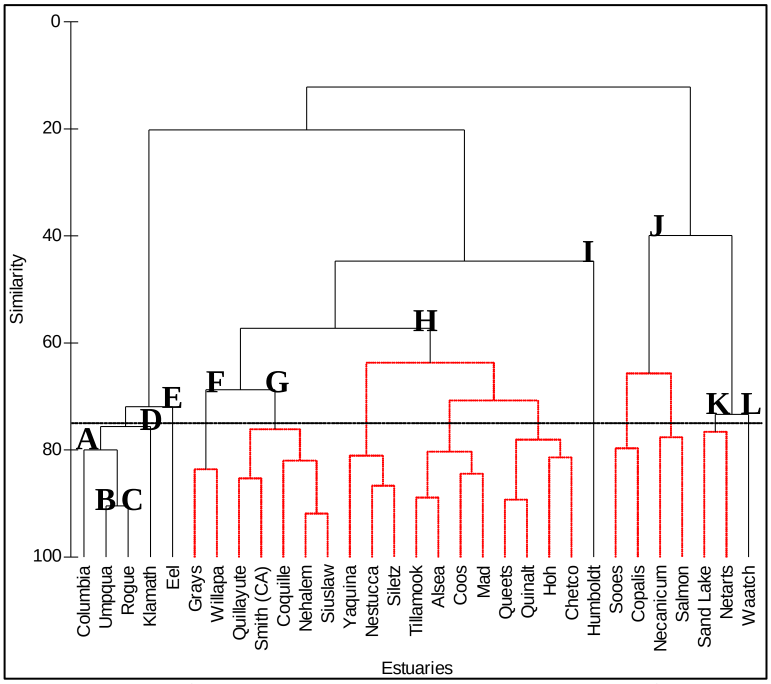

Figure 2-17c. Cluster analysis of 31 PNW estuaries at the 40% significance level based on the areas of the land cover classes in the NOAA 2001 watershed data. Estuaries joined with red lines are not significantly different (p>0.40) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

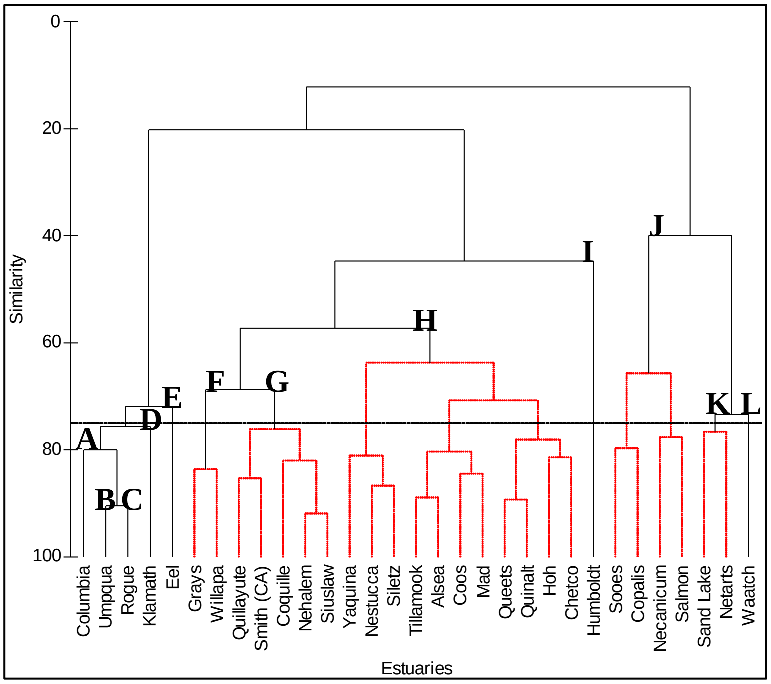

Figure 2-17d. Cluster analysis of 31 PNW estuaries at the 60% significance level based on the areas of the land cover classes in the NOAA 2001 watershed data. Estuaries joined with red lines are not significantly different (p>0.60) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

The clustering based on relative proportion of the areas of the land cover classes will tend to group estuaries associated with structurally similar watersheds. For example, clustering based on relative proportions of land use classes could group highly urbanized watersheds. In the PNW, there was a high degree of similarity (>70%) among all the watersheds (Figure 2-18). To a large extent, this similarity reflects the high percentage of the evergreen land cover class, which ranged from 36% to 80% in the watersheds. The mixed forest and scrub/shrub classes were also common across the watersheds, contributing 7% to 44% of the area. The similarity also reflects the relatively low extent of agriculture and urbanization in these PNW watersheds. Clustering at the 5% significance level resulted in nine groups (Figure 2-18) though joining clusters with >75% similarity reduced this to four functional groups. The first functional group (Clusters A-D) largely consisted of watersheds associated with river-dominated estuaries, with the notable exceptions of Grays Harbor and Willapa Estuary. Both watershed and estuary size within this group varied almost 700-fold. The second functional group (Clusters E and F) consisted of the Columbia, Eel, and Humboldt estuaries. The third functional group (Clusters G and H) was a mix of estuarine types with moderate to large watersheds. The last group consisted of Netarts Bay, a bar-built estuary, which formed a unique cluster (Cluster I). Netarts separated from the other estuaries largely due to the high percentage of unconsolidated shore, 13.8% compared to <1% in most other watersheds, but resembled other watersheds in the high proportions of evergreen and mixed forest land classes.

Increasing the significance level further divides the watersheds, with 11 groups identified at the 40% significance level (Figure 2-19c) and 13 at the 60% significance level (Figure 2-19d). These finer divisions appeared to better capture similar watersheds, such as splitting the Grays Harbor and Willapa watersheds from the Copalis. However, all of these divisions occurred at similarities >75% so they did not increase the number of functional groups from the four identified at the 5% significance level. Until future research shows the need to more finely divide watersheds based on the relative proportions of land cover classes, we suggest that the four functional groups identified at the 5% significance level (Figure 2-18) are sufficient for an initial analysis of estuarine vulnerability based on watershed structure.

2.7 Crosswalk of Wetland and Watershed Classifications

To further identify functionally similar estuaries, we conducted a matrix match ("crosswalk") of the classifications by wetlands and watersheds. Estuaries were identified that overlapped both by clustering on wetland and land cover areas (Tables 2-5 and 2-6) and by clustering on relative proportions of wetlands and land cover (Tables 2-7 and 2-8), with groups with >75% similarity joined into functional groups. The basic assumption of the crosswalks is that estuaries grouped by both wetland and watershed characteristics share similar environmental conditions. Specifically, estuaries grouped by area share similar extents of wetland resources and land cover classes. These groups are most relevant in developing management approaches that scale to area, such as calculating total watershed loadings or strategies to protect the greatest extent of a wetland class. In comparison, grouping by relative proportions more closely captures functional attributes, and is probably the better approach to developing management strategies related to nutrient vulnerability at a system level. Crosswalks were conducted based on the clustering using the 5% and the 60% significance levels. Crosswalks based on the 5% significance level are at a coarser resolution and will tend to have fewer but more variable co-clustered estuaries compared to crosswalks based on the 60% significance level.

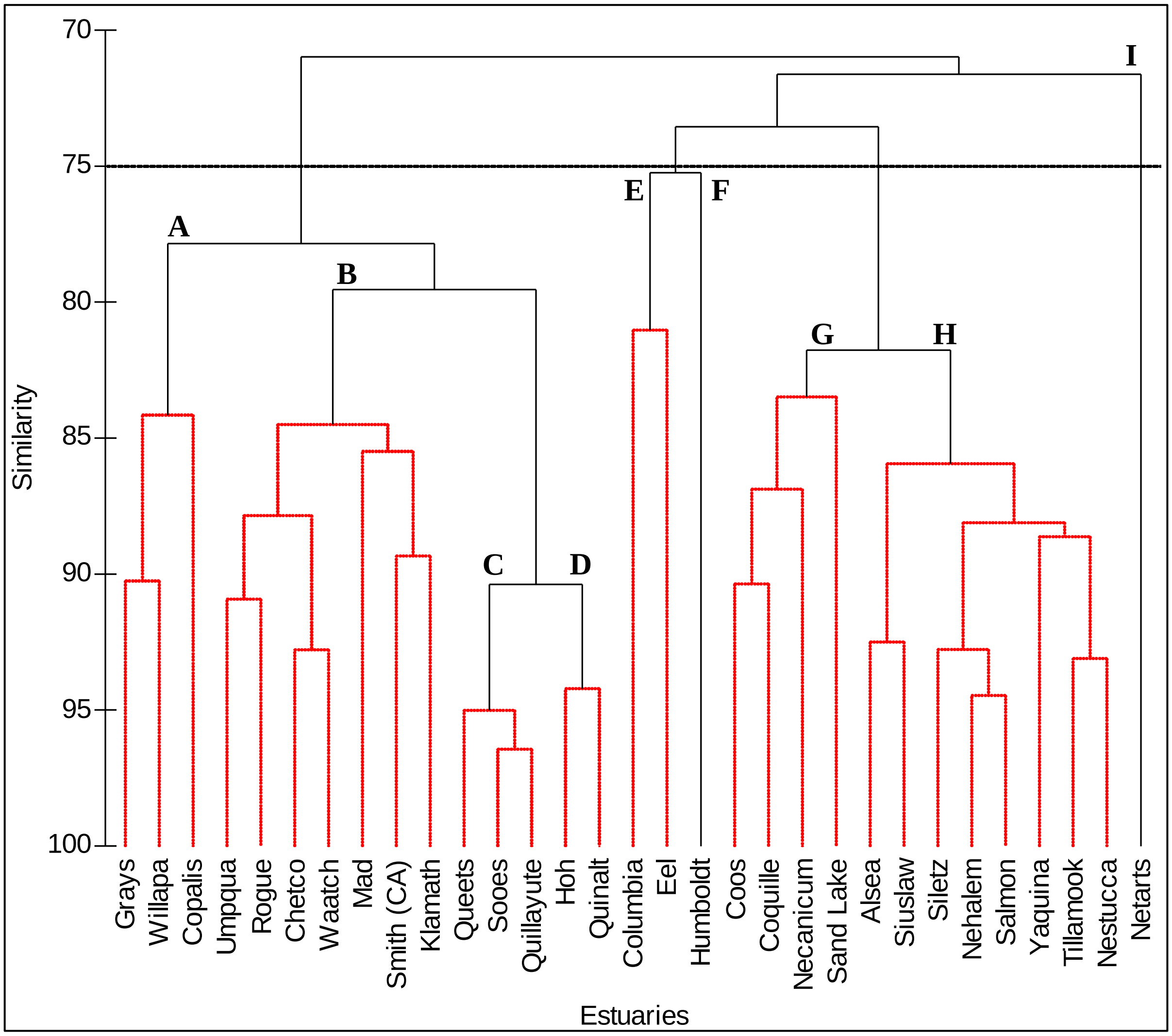

Figure 2-18. Cluster analysis of 31 PNW estuaries at the 5% significance level based on the relative proportions of the land cover classes in the NOAA 2001 watershed data. Estuaries joined with red lines are not significantly different (p>0.05) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

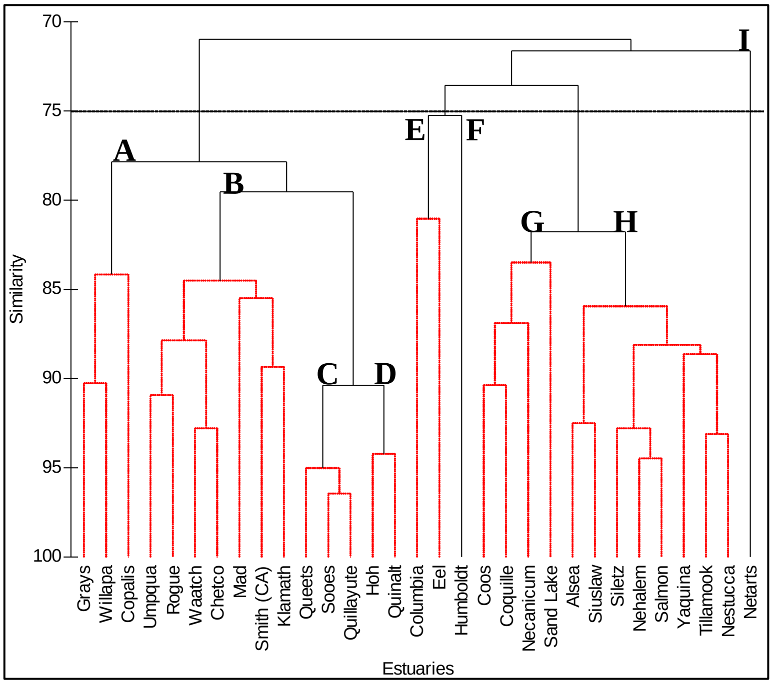

Figure 2-19a. Cluster analysis of 31 PNW estuaries at the 10% significance level based on the relative proportions of the land cover classes in the NOAA 2001 watershed data. Estuaries joined with red lines are not significantly different (p>0.10) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

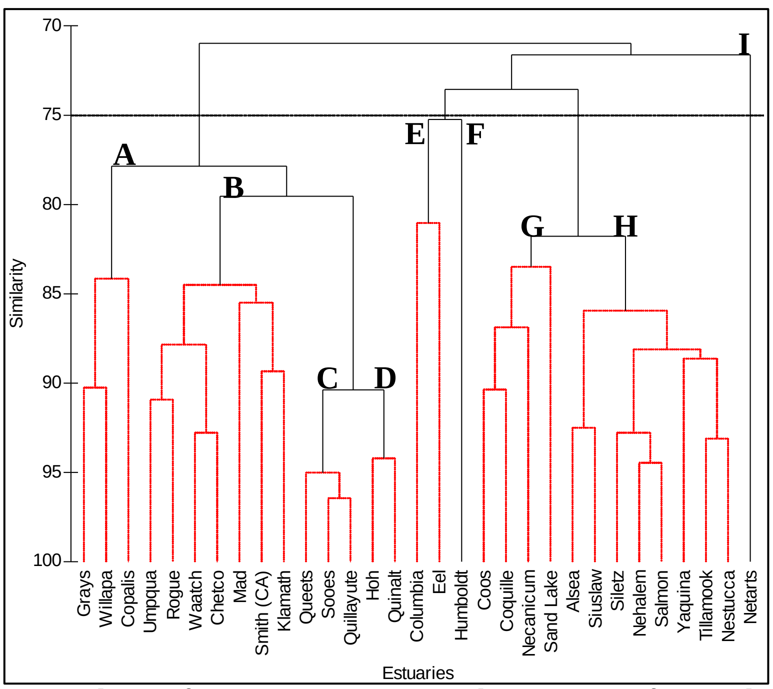

Figure 2-19b. Cluster analysis of 31 PNW estuaries at the 20% significance level based on the relative proportions of the land cover classes in the NOAA 2001 watershed data. Estuaries joined with red lines are not significantly different (p>0.20) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

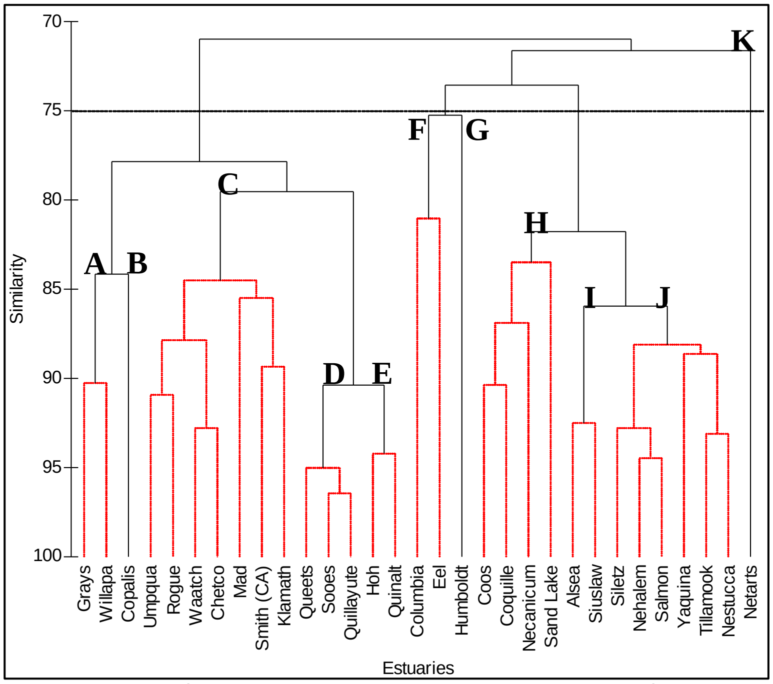

Figure 2-19c. Cluster analysis of 31 PNW estuaries at the 40% significance level based on the relative proportions of the land cover classes in the NOAA 2001 watershed data. Estuaries joined with red lines are not significantly different (p>0.40) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

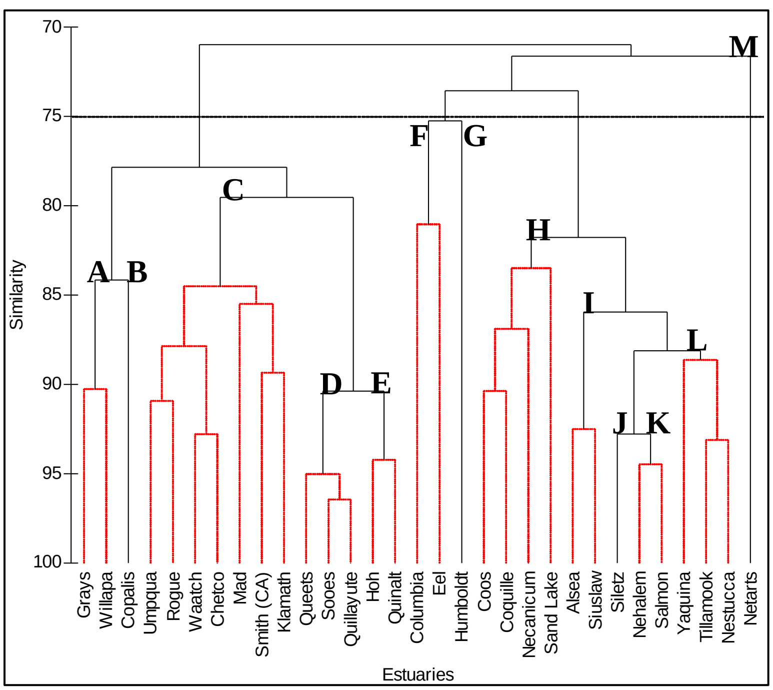

Figure 2-19d. Cluster analysis of 31 PNW estuaries at the 60% significance level based on the relative proportions of the land cover classes in the NOAA 2001 watershed data. Estuaries joined with red lines are not significantly different (p>0.60) based on SIMPROF analysis. Clusters with >75% similarity (horizontal line) are combined in the analysis.

The columns in Tables 2-5 to 2-8 are the clusters based on the watershed analysis, while the rows are the clusters based on wetland analysis. Estuaries listed within the same cell are systems that were grouped together in both the watershed and wetland analyses. For example, Coos and Tillamook were clustered together in both the wetland and land cover analyses on areas (Table 2-5). Clusters are ordered in the tables by the mean size of the watersheds or estuaries within the cluster, though proximity in the table does not necessarily connote greater similarity and Figures 2-12 to 2-19 should be consulted for the actual similarities. To avoid confusion, the clusters based on wetlands were designated as *A'*, *B'*, etc.

In the crosswalk based on areas at the 5% significance level (Table 2-5), the five highly river-dominated estuaries with large watersheds (column A) were dispersed across the four wetland clusters (rows *A'* to *D'*), indicating that there are large watersheds with large estuaries (A-*A'*) and large watersheds with small estuaries (A-*D'*). The opposite pattern was not observed, and there were no small watersheds with large estuaries (i.e., no estuaries falling into F-*A'*, G-*A'*, F-*B'*, or G-*B'*). Another pattern is that the remaining river-dominated estuaries fell into relatively few cells, and were separated from the tide-dominated estuaries with the exception of the Yaquina occurring in cell D-*C'*. Additionally, nine of the highly river-dominated estuaries fell into three cells representing moderate to small estuaries with moderate-sized watersheds (C-*C'*, C-*D*', and D-*D'*). Four of the other tide-dominated estuaries formed two pairs (Grays Harbor & Willapa and Coos & Tillamook), while the Waatch was not linked with any of the other estuaries. The pattern based on areas at the 60% significance level (Table 2-6) was the same with the exception that the Eel River and Waatch were broken into separate watershed clusters (columns).

One obvious pattern in the crosswalk by relative proportions of wetland and land cover classes at the 5% significance level (Table 2-7) is the grouping of seven of the highly river-dominated systems in a single cell (A-D – *A'*). Excluding the Columbia River, these seven estuaries had the highest normalized freshwater inflow values (Table 2-4). Thus, estuaries with the highest freshwater inflow appear to have wetland and watershed patterns that differ from other river-dominated or tide-dominated systems. With the exception of the Salmon River, the remaining moderate-sized river-dominated estuaries were grouped into two cells (A-D – *B'* and G-H – *B'*), though they were mixed with tide-dominated estuaries. At the 60% significance level (Table 2-8), the four estuaries with the highest normalized freshwater inflow values (exclusive of the Columbia) still grouped together (A-E – *A'*-*C'*) though the three estuaries with next highest freshwater inputs were separated into two cells (A-E – *E'* and A-E – *D'*) with lower average estuary size. At this higher significance level, the tide-dominated Grays Harbor and Willapa estuaries were grouped together, as were the Coos and Yaquina estuaries. Assuming that grouping by relative proportions captures functional aspects of estuaries and watersheds, we predict that the estuaries grouped at the 60% significance level will have similar nutrient dynamics. The practical difficulty in using the crosswalk based on the 60% significance level is the large number of groups, which is reduced by more than half with the crosswalk based on the 5% significance level. It is not clear whether these groups show sufficient similarity for them to be managed in a similar fashion, though research to help resolve this question is outlined in Section 2.9.

Table 2-5. Crosswalk of the cluster analyses by the area of NWI consolidated wetland classes (rows) and the area of the NOAA land cover classes (columns) at the 5% significance level. Letters refer to the cluster groups in Figures 2-12 for wetlands and 2-16 for watersheds. Mean watershed area and mean estuary area within the clusters are given in parentheses, with the area of the Columbia watershed for the Lower Columbia to the Bonneville Dam. The clusters are ordered by mean watershed or estuary size and proximity in the table does not necessarily imply high similarity. The geomorphology is indicated by color with tide-dominated estuaries in blue, moderately river-dominated estuaries in green, highly river-dominated estuaries in red, and bar-built estuaries in black.

Clusters on Area / 5% Significance		WATERSHED LAND USE CLUSTERS (Mean Watershed Area For Group)						
		A ($15161\ km^2$)	B ($3197\ km^2$)	C ($2009\ km^2$)	D ($1087\ km^2$)	E ($571\ km^2$)	F ($156\ km^2$)	G ($45\ km^2$)
NWI WETLAND CLUSTER (Mean Estuary Area for Group)	*A’* ($441\ km^2$)	**Columbia**	**Grays** **Willapa**					
	B’ ($49\ km^2$)	**Umpqua**			**Coos** **Tillamook**	**Humboldt**		
	C’ ($11.1\ km^2$)	**Eel**		**Coquille** **Nehalem** **Siuslaw**	**Alsea** **Nestucca** **Siletz** **Yaquina**			**Netarts** **Sand Lake**
	D’ ($1.5\ km^2$)	**Klamath** **Rogue**		**Quillayute** **Smith**	**Chetco** **Hoh** **Mad** **Queets** **Quinault**		**Copalis** **Necanicum** **Salmon** **Sooes**	**Waatch**

Table 2-6. Crosswalk of the cluster analyses by the area of NWI consolidated wetland classes (rows) and the area of the NOAA land cover classes (columns) at the 60% significance level. Letters refer to the cluster groups in Figures 2-13d for wetlands and 2-17d for watersheds. Mean watershed area and mean estuary area within the clusters are given in parentheses, with the area of the Columbia watershed for the Lower Columbia to the Bonneville Dam. The clusters are ordered by mean watershed or estuary size and proximity in the table does not necessarily imply high similarity. The geomorphology is indicated by color with tide-dominated estuaries in blue, moderately river-dominated estuaries in green, highly river-dominated estuaries in red, and bar-built estuaries in black.

Clusters on Area / 60% Significance		WATERSHED LAND COVER CLUSTERS (Mean Watershed Area For Group)								
		A-D ($16567\ km^2$)	E ($9536\ km^2$)	F ($3197\ km^2$)	G ($2009\ km^2$)	H ($1087\ km^2$)	I ($571\ km^2$)	J ($156\ km^2$)	K ($49\ km^2$)	L ($38\ km^2$)
NWI WETLAND CLUSTERS (Mean Estuary Area for Group)	*A'* ($669\ km^2$)	**Columbia**								
	B' ($327\ km^2$)			**Grays** **Willapa**						
	C' ($49\ km^2$)	**Umpqua**				**Coos** **Tillamook**	**Humboldt**			
	D' ($11.1\ km^2$)		**Eel**		**Coquille** **Nehalem** **Siuslaw**	**Alsea** **Nestucca** **Siletz** **Yaquina**			**Netarts** **Sand Lake**	
	E' ($1.5\ km^2$)	**Klamath** **Rogue**			**Quillayute** **Smith**	**Chetco** **Hoh** **Mad** **Queets** **Quinault**		**Copalis** **Necanicum** **Salmon** **Sooes**		**Waatch**

Table 2-7. Crosswalk of the cluster analyses by the relative proportion of NWI consolidated wetland classes (rows) and the relative proportion of land cover classes (columns) at the 5% significance level. Letters refer to the cluster groups in Figures 2-14 and 2-18. Clusters with high similarity (>75%) were combined for the analysis. Mean watershed area and mean estuary area within the clusters are given in parentheses. The clusters are ordered by mean watershed or estuary size and proximity in the table does not necessarily imply high similarity. The geomorphology is indicated by color with tide-dominated estuaries in blue, moderately river-dominated estuaries in green, highly river-dominated estuaries in red, and bar-built estuaries in black.

<table>
<tr><td colspan="2" rowspan="2">Clusters on Relative Proportion / 5% Significance</td><td colspan="4">WATERSHED LAND COVER CLUSTERS
(Mean Watershed Area For Group)</td></tr>
<tr><td>E-F
(8209 km^2)</td><td>A-D
(4482 km^2)</td><td>G-H
(1175 km^2)</td><td>I
(46 km^2)</td></tr>
<tr><td rowspan="3">NWI WETLAND CLUSTERS
(Mean Estuary Area for Group)</td><td>A'
(85 km^2)</td><td>Columbia</td><td>Chetco
Hoh
Klamath
Queets
Quillayute
Quinault
Rogue</td><td></td><td></td></tr>
<tr><td>B'
(46 km^2)</td><td>Eel
Humboldt</td><td>Copalis
Grays
Mad
Smith
Sooes
Umpqua
Willapa</td><td>Alsea
Coos
Coquille
Necanicum
Nehalem
Nestucca
Sand Lake
Siletz
Siuslaw
Tillamook
Yaquina</td><td>Netarts</td></tr>
<tr><td>C'
(2.1 km^2)</td><td></td><td>Waatch</td><td>Salmon</td><td></td></tr>
</table>

Table 2-8. Crosswalk of the cluster analyses by the relative proportion of NWI consolidated wetland classes (rows) and the relative proportion of land cover classes (columns) at the 60% significance level. Letters refer to the cluster groups in Figures 2-15d and 2-19d. Clusters with high similarity (>75%) were combined for the analysis. Mean watershed and mean estuary area within the clusters are given in parentheses. The clusters are ordered by mean watershed or estuary size and proximity in the table does not necessarily imply high similarity. The geomorphology is indicated by color with tide-dominated estuaries in blue, moderately river-dominated estuaries in green, highly river-dominated estuaries in red, and bar-built estuaries in black.

Cluster on Relative Proportion / 60% Significance		WATERSHED LAND COVER CLUSTERS (Mean Watershed Area For Group)			
		F-G ($8209\ km^2$)	A-E ($4482\ km^2$)	H-L ($1175\ km^2$)	M ($46\ km^2$)
NWI WETLAND CLUSTERS (Mean Estuary Area for Group)	*F'* ($327\ km^2$)		**Grays** **Willapa**		
	A'-C' ($135\ km^2$)	**Columbia**	**Klamath** **Quinault** **Quillayute** **Rogue**		
	G' ($29\ km^2$)			**Alsea** **Coos** **Yaquina**	
	N' ($22\ km^2$)	**Humboldt**		**Necanicum** **Nestucca** **Siletz** **Tillamook**	**Netarts**
	K' ($16\ km^2$)			**Siuslaw**	
	H'-J' ($13\ km^2$)		**Sooes** **Umpqua**	**Coquille** **Nehalem**	
	M' ($8.2\ km^2$)	**Eel**	**Copalis**		
	O' ($4.3\ km^2$)			**Sand Lake**	
	P' ($2.2\ km^2$)		**Waatch**	**Salmon**	
	L' ($1.8\ km^2$)		**Mad** **Smith**		
	E' ($1.1\ km^2$)		**Hoh** **Queets**		
	D' ($0.7\ km^2$)		**Chetco**		

2.8 Watershed Alterations and Population Patterns

The last type of landscape analysis was evaluation of the extent of watershed alteration using a suite of metrics indicative of anthropogenic "pressure" on the PNW estuaries. The detailed analysis is based on the 31 estuaries used in the multivariate analysis (Table 2-4), while the regional analysis is based on the 89 estuaries used in the original report (Lee et al., 2006) rather than the 103 estuaries from the NWI update. The 14 watersheds associated with the "new" estuaries (see Table 2-5) only constitute 0.2% of the total coastal watershed area so that their exclusion has only a minor effect on the mean values for the region.

One set of indicators of anthropogenic pressure is the relative percentages of high development, low development, and cultivated classes from the NOAA land cover data. NOAA has a class for grasslands but does not separate out natural grasses from anthropogenic grasslands such as pasture. Not to bias the estimates, we combined the 1992 NLCD classes for pasture/hay and urban/recreational grasses as the fourth land cover metric of alteration. No attempt was made to evaluate the effects of logging, which can increase non-point runoff of sediments and nutrients (e.g., Likens et al., 1970). A potential indirect effect of logging may be an increase in red alder after disturbance which may increase nitrogen fluxes from watersheds (Sections 3.7 and 4.2). While this analysis does not capture the direct and indirect effects of logging, it does identify those watersheds experiencing the greatest anthropogenic impacts from development and agriculture. Another measure of alteration is population density from the 2000 census. Densities were calculated on a watershed basis, which often cut across city or county boundaries. One threshold relating population density to nutrient fluxes is the 386 people per km^2 used in the EPA "Classification Framework for Coastal Systems" to identify watersheds with a high risk for nitrate inputs (U.S. EPA, 2004a).

The last metric of alteration is the percent impervious surfaces. High percentages of impervious surfaces within a watershed can affect water quality both by increasing the volume and rate of surface runoff, which in turn can increase erosion as well as non-point loadings of sediment, contaminants, and nutrients (U.S. EPA, 1997). Adverse impacts are first observed in surface waters in watersheds with about 10% impervious surfaces and with major impacts occurring at 25-30% (Arnold and Gibbons, 1996). In tidal creeks in South Carolina, physical/chemical alterations and changes in fecal coliforms were first detected at 10-20% impervious surfaces with impacts on the benthos, shrimp, and food webs detected at values of 20-30% (Holland et al., 2004). NOAA's "Spatial Wetland Assessment for Management and Planning" (SWAMP) model uses a threshold of impervious surfaces of <7.5% as the cutoff for "exceptional" environmental condition (Sutter, 2001).

The values of these metrics for all 89 estuaries (Table 2-9) and for the 31 estuaries used in the multivariate analysis (Table 2-10) indicate relatively low levels of alterations at the watershed scale. The percent of high development land use only exceeded 2% in Crescent City Harbor while the percent of low development only exceeded 5% in three watersheds and 1% in twelve watersheds. Pasture and urban/recreational grasslands were more abundant than the other altered land use classes in several of the watersheds. Anthropogenic grasslands exceeded 10% in the Lake Earl and Humboldt watersheds and exceeded 5% in another five watersheds including the

Table 2-9. The mean, median, and maximum values for each of the metrics of watershed alteration for the 89 coastal watersheds. The "% High Development", "% Low Development", and "% Cultivated" are the percentage of the watersheds in these land cover classes from the NOAA 2001 data. The "% Pasture & Urban Grasslands" is the combination of the pasture/hay and urban/recreational grasses classes from the 1992 NLCD data. Population Density is from the 2000 census, while the "% Impervious Surfaces" was calculated using the 30-m grids from NLCD (http://www.epa.gov/mrlc/nlcd-2001.html). The % impervious surface values and the estuary with the highest percent impervious surfaces generated from Attila (U.S. EPA, 2004c) are given in parentheses for reference.

	% HIGH DEVELOPMENT	% LOW DEVELOPMENT	% CULTIVATED	% PASTURE & URBAN GRASSLANDS	% IMPERVIOUS SURFACES (values from Attila)	POPULATION DENSITY (# km^{-2})
Mean	0.13	0.91	0.12	1.19	1.16 (2.54)	18.09
Median	0.01	0.18	0.00	0.04	0.59 (2.02)	4.24
Maximum	4.98	15.76	1.82	11.1	12.72 (16.11)	288.79
Watershed With Maximum	Crescent City	Loomis Lake	Twomile Creek South	Lake Earl	Loomis Lake (Loomis Lake)	Crescent City

Table 2-10. The mean, median, and maximum values for each of the metrics of watershed alteration for the 31 coastal watersheds used in the multivariate analysis. The “% High Development”, “% Low Development”, and “% Cultivated” are the percentage of the watersheds in these land cover classes from the NOAA 2001 data. The “% Pasture & Urban Grasslands” is the combination of the pasture/hay and urban/recreational grasses classes from the 1992 NLCD data. Population Density is from the 2000 census, while the “% Impervious Surfaces” was calculated using the 30-m grids from NLCD (http://www.epa.gov/mrlc/nlcd-2001.html). The % impervious surfaces value for the Columbia is based on the watershed up to the Bonneville Dam. The % impervious surface values and the estuary with the highest percent impervious surfaces generated from Attila (U.S. EPA, 2004c) are given in parentheses for reference.

	% HIGH DEVELOPMENT	% LOW DEVELOPMENT	% CULTIVATED	% PASTURE & URBAN GRASSLANDS	% IMPERVIOUS SURFACES (values from Attila)	POPULATION DENSITY (# km^{-2})
Mean	0.13	0.56	0.17	1.96	0.83 (2.25)	14.73
Median	0.03	0.28	0.00	1.34	0.57 (2.05)	5.96
Maximum	1.19	3.71	1.29	10.43	3.61 (4.67)	119.19
Watershed With Maximum	Humboldt	Humboldt	Columbia	Humboldt	Columbia (Humboldt)	Humboldt

Columbia. Population densities were low to moderate across all the watersheds, and no watershed exceeded the threshold of 386 people per km^2. Only six estuaries had watersheds with a population density exceeding 100 people per km^2, with a maximum density of 289 people per km^2 in the Crescent City Harbor watershed.

The percent impervious surface integrates several different types of watershed alterations and the recently available 30-m grid data from the NLCD (http://www.epa.gov/mrlc/nlcd-2001.html) allows detailed analysis of this driver. In general, the percent impervious surface is low in the coastal PNW watersheds (Tables 2-9 and 2-10). With about 12% impervious surfaces, only the watersheds associated with Crescent City Harbor and Loomis Lake Creek exceeded the lower 10% threshold value from Arnold and Gibbons (1996). As a marine harbor/cove, Crescent City Harbor is well flushed and this level of impervious surfaces is not likely to have a major effect on its water quality. We do not have information on extent of flushing in Loomis Lake Creek but Loomis Lake is classified as eutrophic based on total phosphorus concentrations (O'Neal et al., 2001). Thus, it is possible that Loomis Lake Creek may have relatively high nutrient concentrations during periods of restricted exchange with the ocean, though it is not clear what estuarine resources would be exposed to these elevated nutrient concentrations (see Table 2-3). No other watershed exceeded the NOAA SWAMP threshold of 7.5% for exceptional condition and only six other watersheds had impervious surfaces >3%. Among the 31 estuaries, the Columbia had the highest percent impervious surfaces at 3.6% but this is based on the 2001 NLCD data that extends only up to the Bonneville Dam and includes the city of Portland. Presumably this percentage would decline if more of the Columbian Basin were included.

To evaluate the similarity in alterations, the 31 watersheds were clustered using the six metrics of alteration. Because these variables are measured in different units, the values were normalized and Euclidean distance was used as the metric of similarity. Classification by watershed alterations resulted in four significant clusters (Figure 2-20). One group (Cluster A) that split off at a high degree of dissimilarity consisted of the lower Columbia River and Humboldt Estuary. These two systems have the highest percentages of high and low development land classes, anthropogenic grassland land cover classes, the highest percent impervious surfaces, and the highest population densities (Table 2-10). The Necanicum (Cluster B) broke out at a moderate degree of dissimilarity from the other estuaries. Though not as altered as the Columbia and Humboldt, the Necanicum watershed has the third highest values for the percentages of high development and low development land cover classes, population density, and percent impervious surfaces. Another group (Cluster C) that split out at a moderate degree of dissimilarity consisted of four estuaries, the Klamath, Grays, Eel, and Rogue. These four estuaries appeared to break out based on their moderately high values of both cultivated and pasture/urban grassland land cover classes. The last group (Cluster D) consists of the remaining 24 estuaries, which as a group had no discernable pattern of watershed alteration. Thus, the watersheds of the major PNW estuaries separate into four patterns of alteration, with the Columbia and the Humboldt the most altered due to higher proportions of development and agriculture.

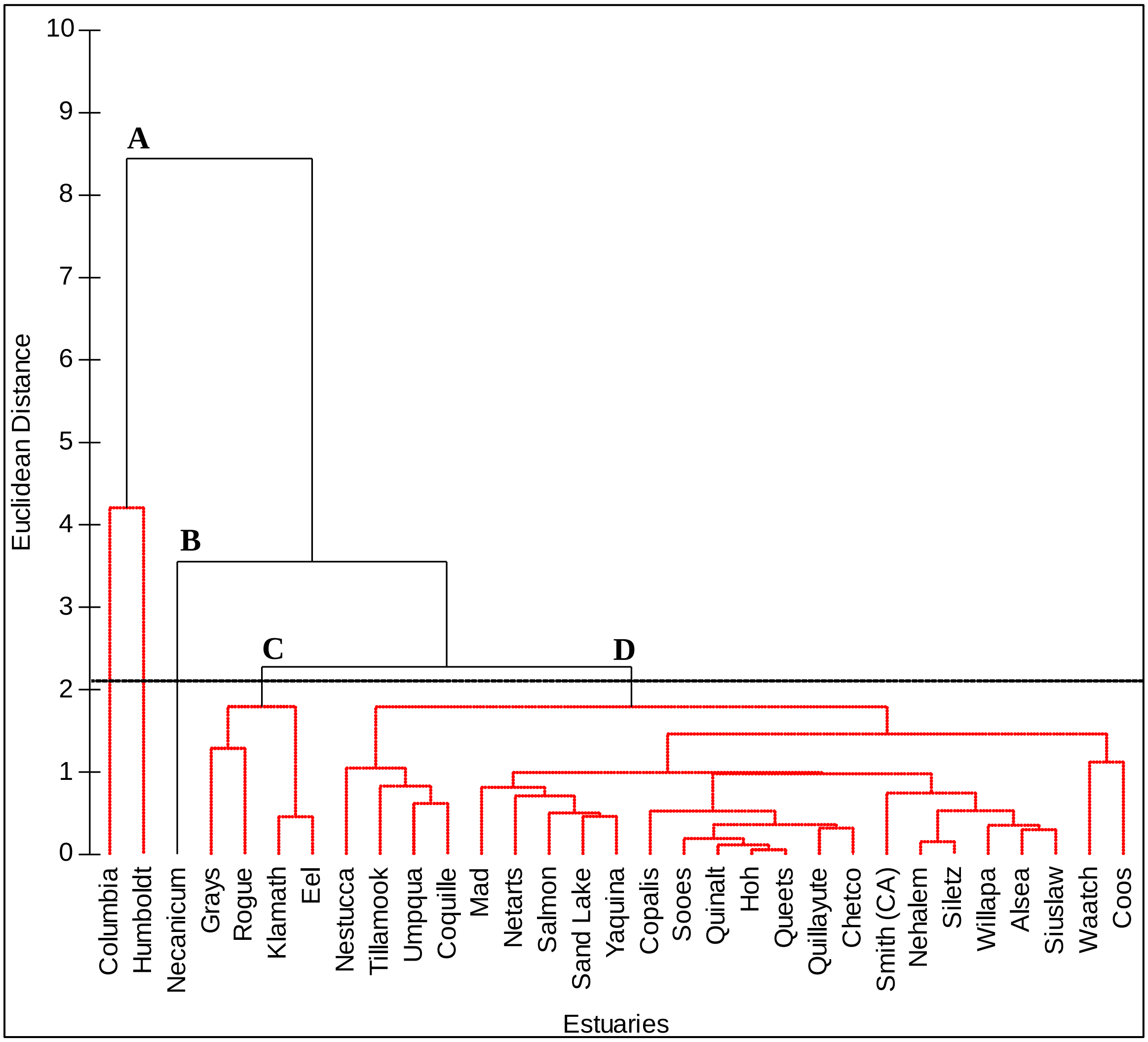

Figure 2-20. Cluster analysis of the 31 PNW estuaries based on six metrics of watershed alteration. Similarity among clusters is measured as Euclidean distance. Estuaries joined with red lines are not significantly different (p>0.05) based on SIMPROF analysis. The horizontal lines indicate 25% dissimilarity of the maximum observed dissimilarity for reference.

2.8.1 Flow-Path Analysis of Impervious Surfaces

While the percent impervious surfaces for the entire watersheds are low, the spatial distribution of watershed alterations is likely to have an important effect on the extent of nutrient, pollutant, and sediment runoff. The 30-m grid cells from the NLCD allow a flow path analysis of the percent impervious surfaces as a function of the distance from the estuary or nearest river or stream flowing into the estuary. Because of its complexity, we have only conducted a flow path analysis for the Yaquina watershed which is presented here as a potential future direction. Figure 2-21 shows the percent impervious surfaces within six concentric zones around the estuary, rivers, and streams, increasing in distance from a 0-108 meter zone to the nearest water to a 0-31,130 meter zone that represents the entire watershed. The key observation is that the percent impervious surface is higher in the portion of the watershed closer to the water than in land more distant from the water's edge. The maximum percent impervious surface of 3.78% occurs within a band 0-500 meters around the estuary, river, and streams, a value more than 4-times higher than the watershed average of 0.89%. Similar analyses should be conducted in additional watersheds, but it is likely that this general pattern of higher impervious surfaces near the water occurs in many, if not most, PNW watersheds as a result of the concentration of tourist facilities, commercial and recreational fishing activities, and housing along bay and river fronts.

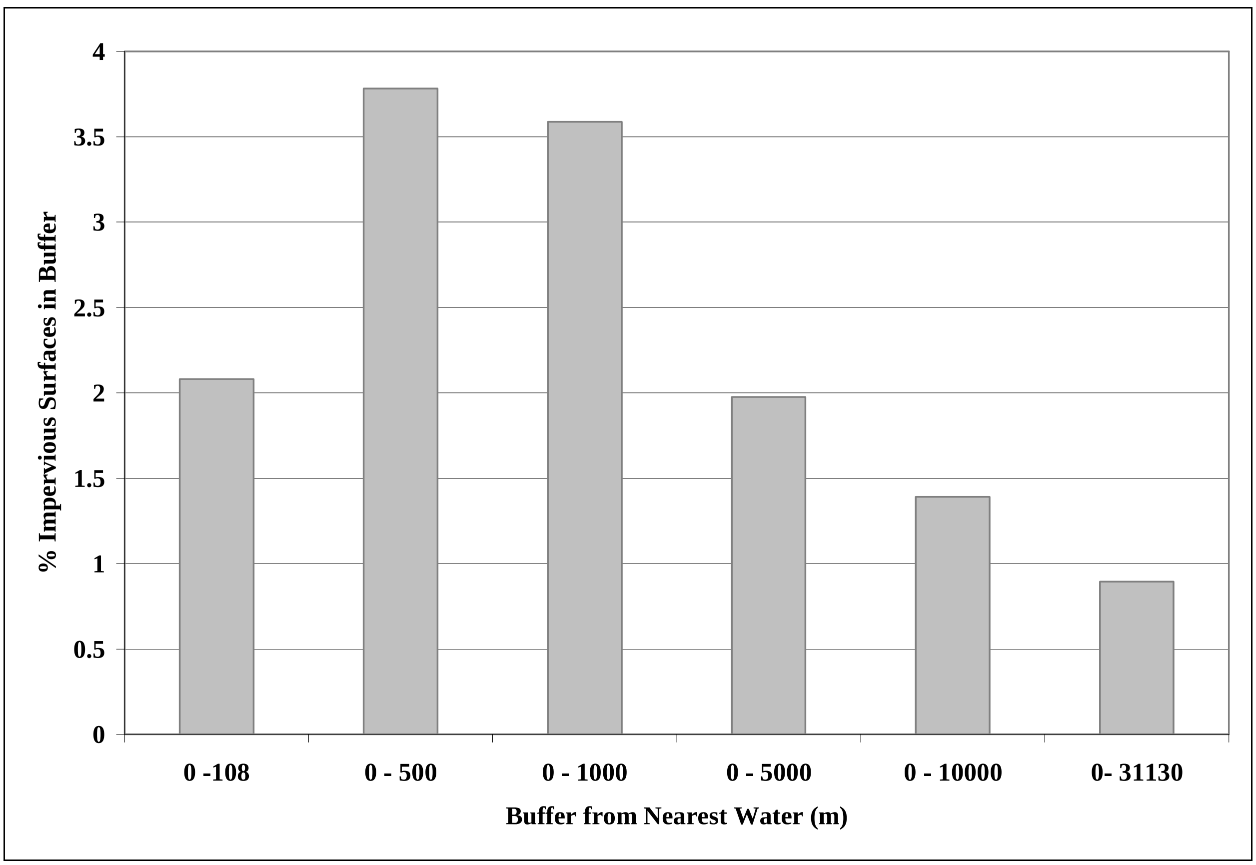

Figure 2-21. Percent impervious surfaces in the Yaquina watershed for six buffer zones of different distances from the shoreline of the Yaquina Bay or Yaquina River. The 0–31,130 m buffer represents the entire Yaquina watershed.

2.9 Synthesis of Classification Approaches and Research Needs

Five classification schemas are presented in this chapter – classifications based on geomorphology and ocean exchange (Section 2.4), wetland clustering (Section 2.5), watershed clustering (Section 2.6), crosswalks between the wetland and watershed clustering (Section 2.7), and groupings on watershed alterations (Section 2.8). Which is the "best" approach depends upon the goals of the user. Even with the specific goal of identifying estuaries with similar vulnerabilities to nutrient enrichment, it is not clear which schema is best, in part because no specific criteria have been developed as to what constitutes "similar enough" for management purposes. It is possible, however, to provide some general guidelines on how the various schemas could be used for management purposes as well as future research directions.

Of the various schemas, geomorphology provides the most general approach to grouping estuaries with similar nutrient dynamics. Assuming the ideal case of "all other factors being equal", potential differences in the extent of flushing based on ocean exchange and freshwater inflow suggest the following relative vulnerabilities (from most to least vulnerable) to equivalent increases from terrestrial runoff:

lagoon ≧ blind estuary > tide-dominated river mouth > river-dominated river mouth ≧ coves

This suggested ranking is for increased nutrient concentrations within the estuarine portion of the system and not for total loading, which may show a pattern more closely tied to the normalized freshwater inflow and presence of red alder in the watershed. Nor does this ranking account for differences in the types of resources potentially at risk across these different classes of estuaries. The tidally restricted coastal creeks are more complex than suggested by their small size and do not easily fit into this ranking because of the potential occurrence of localized pockets of low DO water resulting from periodic advection and trapping of ocean water (C. Brown, unpublished data). Nor do bar-built estuaries readily fit into this scheme. The three PNW bar-built estuaries do not share many similarities (Tables 2-7 and 2-8) and may function more like similar size tide-dominated estuaries than as a separate class of estuaries per se.

Individual clustering by wetland or land cover classes provides a more detailed breakout of estuaries than that provided by the geomorphologic classes. While this approach is useful in identifying similarities in the resources at risk (wetland clustering) or differences in potential loading (watershed clustering), we suggest that the crosswalks are more likely to identify functionally similar estuaries than clustering by either attribute alone. As discussed in Section 2.7, the crosswalks identified fine resolution separations using the 60% significance level (Tables 2-6 and 2-8) and coarser level separations using the 5% significance level (Tables 2-5 and 2-7). One of the factors in resolving which of these schemas are sufficient for management is how similar estuaries are within the groups compared to among-group differences. If, for example, variation in a key attribute within a group exceeds the differences among groups, then the classification may not be sufficiently robust for certain management purposes. Therefore, we suggest one avenue of research should be to compare within- and among-group variations in nutrient concentrations and dynamics across sets of highly similar estuaries, using the water quality survey framework described in Chapter 4. Because nutrient dynamics are influenced by drivers related to both the areas of estuaries and watersheds and to the relative proportions of wetland or watershed classes, we propose that estuaries that co-clustered based on both the areas (Tables 2-5 and 2-6) and on relative proportions (Tables 2-7 and 2-8) should show the highest similarities. This is essentially a "crosswalk of the crosswalks" to identify the most similar estuaries.

Based on the 60% significance levels, there are six pairs of estuaries that co-clustered on area and relative proportions: 1) Klamath and Rogue; 2) Grays and Willapa; 3) Coquille and Nehalem; 4) Alsea and Yaquina; 5) Hoh and Queets; and 6) Nestucca and Siletz. These pairs span a range of estuary types, including large to small estuaries as well as tide-dominated, moderately river-dominated, and highly river-dominated estuaries (Tables 2-2 and 2-4). Variation in nutrient dynamics within these pairs represents the minimum variation that can be expected from groupings based on this approach. Results from such comparisons should be assessed by environmental managers to determine whether the within-group variation relative to the among-group variation is acceptable in a regulatory context. If the levels of variation are not acceptable, then: 1) a different classification approach is required; 2) management expectations may need to be modified regarding the actual level of similarity among PNW estuaries or; 3) the desire to manage groups of "similar" estuaries in a common fashion may be unrealistic.

If the relative variation within and among these six pairs of estuaries is acceptable, the next step would be to conduct additional comparisons among sets of grouped estuaries. In particular, the three large groups in the 5% significance crosswalk based on relative proportions (A-D – *A'*, A-D – *B'*, and G-H – *B'* in Table 2-7) should be evaluated. If the relative variation falls within acceptable bounds, this classification could be used as a framework for developing management strategies for the PNW, with the caveat that additional studies would be required for the estuaries falling outside of these groups. However, failure to find ecologically relevant differences among groups at this level of significance while finding them in the six paired estuaries suggests that a finer resolution classification will be needed.

CHAPTER 3:
ENVIRONMENTAL CHARACTERISTICS
OF THE SEVEN TARGET ESTUARIES

Cheryl A. Brown and Henry Lee II

3.0 Introduction

The purpose of this chapter is to provide an overview of the environmental conditions within the Pacific Northwest (PNW) and of the seven target estuaries chosen for field surveys and aerial photography. This chapter is not meant to be a comprehensive review and the reader is referred to Emmett et al. (2000) for an overview of environmental conditions for Pacific Coast estuaries, Hickey and Banas (2003) for a review of the effects of oceanographic conditions on PNW estuaries, the Oregon Coastal Atlas (http://www.coastalatlas.net) for a description of Oregon estuaries, and the Coastal Landscape Analysis and Modelling Study (CLAMS; http://www.fsl.orst.edu/clams/) for detailed information on coastal watersheds in Oregon.

3.1 Pacific Northwest Climate

Though the PNW has a reputation for being "wet and rainy", the PNW coast essentially has a wet and dry season (see Emmett et al., 2000). From approximately October through March, the PNW experiences frequent rainfall with a peak in December through January. From approximately April through September, rainfall substantially declines often with only marginal rainfall in July through September. In addition to this seasonality, there is a geographic pattern with greater rainfall in the more northern portions of the PNW. Average annual precipitation in northern coastal Washington can reach as much as 350 cm compared to 140 cm in the Humboldt watershed in northern California. In terms of temperature, the PNW has a Mediterranean climate, with mild summers and winters. The difference between winter and summer air temperatures is small, only about 5° C near the Humboldt Estuary (Emmett et al., 2000). Freezes and snowfall at the lower elevations are relatively rare, and with the exception of the Columbia and Klamath watersheds there is relatively little snow pack in most of the coastal watersheds.

3.2 Coastal Upwelling in the Pacific Northwest

Estuaries in the PNW are adjacent to the California Current System, which exhibits strong interannual, seasonal and event-scale variability (Hickey and Banas, 2003). In this region, seasonal wind-driven upwelling advects relatively cool, nutrient rich water to the surface. The upwelling season typically commences in April and continues through September (Kosro et al., 2006). During this time period, upwelling-favorable winds from the north dominate. The upwelling conditions are interrupted by brief periods of downwelling-favorable conditions, which usually persist for several days. Previous studies have demonstrated that the oceanic inputs of nutrients and phytoplankton are important for estuaries adjacent to coastal upwelling regions, such as the west coast of the U.S. (e.g., de Angelis and Gordon, 1985; Roegner and Shanks, 2001; Roegner et al., 2002; Newton and Horner, 2003; Colbert and McManus, 2003; Brown and Ozretich, 2009). Hickey and Banas (2003) examined variations in temperature, salinity, and alongshore wind stress for three estuaries along the Oregon and Washington coast, spanning a distance of 400 km. They demonstrated that there was coherence in the fluctuations in temperature and salinity among these estuaries during the summer resulting from the large-scale patterns in along-shelf wind forcing at large scales (100s of kms). As a result, we believe that coastal ocean conditions are relatively uniform over the geographic extent of the seven target estuaries, which span 250 km (Figures 2-3 and 2-4).

3.3 Criteria for Choosing Target Estuaries and Previous Estuarine Classifications

The seven target estuaries were chosen to cover a range of estuarine and watershed conditions in the PNW. While the goal was to choose a suite of PNW estuaries representative of the range of systems, we focused this initial effort on Oregon estuaries because of logistical constraints. When this work was initiated, we had not completed our regional analysis of PNW (Chapter 2), and so relied on previous classifications and our personal experiences with these estuaries for the selection. The primary classification systems of the PNW estuaries that we utilized in estuary selection are summarized below and in Table 3-1.

3.3.1 Management Classification of Oregon Estuaries

Oregon estuaries were classified under Oregon's Statewide Planning Goal 16 (Estuarine Resources) as natural, conservation, shallow draft development, or deep draft development (Cortright et al., 1987; http://www.inforain.org/oregonestuary/oregonestuary_page5.html) with the following definitions:

> Natural: Estuaries lacking maintained jetties or channels, and which are usually little developed for residential, commercial or industrial uses. They may have altered shorelines, provided that these altered shorelines are not adjacent to an urban area. Shorelands around natural estuaries are generally used for agriculture, forestry, recreation and other rural uses.
>
> Conservation: Estuaries lacking maintained jetties or channels, but which are within or adjacent to urban areas which have altered shorelines adjacent to the estuary.
>
> Shallow Draft Development: Estuaries with maintained jetties and a main channel (not entrance channel) maintained by dredging at 22 feet or less.
>
> Deep Draft Development: Estuaries with maintained jetties and a main channel maintained by dredging to deeper than 22 feet.

The target estuaries included one natural estuary (Salmon River Estuary), two conservation estuaries (Alsea and Nestucca estuaries), two shallow draft development estuaries (Tillamook and Umpqua River estuaries), and two deep draft estuaries (Coos and Yaquina estuaries). Thus, the target estuaries represent a good cross-section of the Oregon management types.

3.3.2 Geomorphological/Hydrological Classifications

Based on the geomophological/ocean exchange classification we developed after the field surveys (Section 2.4), the seven target estuaries included three tide-dominated drowned river mouth estuaries (Coos, Tillamook, and Yaquina), two moderately river-dominated river mouth estuaries (Alsea, Salmon) and two highly river-dominated river mouth estuaries (Nestucca and Umpqua estuaries) (Table 3-1). Within this suite of estuaries, the normalized

Table 3-1. Current and historical classifications of the seven target estuaries. The current classification we developed is discussed in Section 2.4.

Estuary	Current Classification	Oregon Management Classification	Bottom et al. (1979)	Parrish et al. (2001)	Rumrill (1998)	Burt & Mcalister (1959)
Alsea	Moderately river-dominated drowned river mouth	Conservation	Drowned river valley, partially mixed	Well-flushed drowned rivers, predominantly freshwater input? or Well-flushed drowned rivers, predominantly oceanic input?	NA	Partially mixed
Coos	Tide-dominated drowned river mouth	Deep draft development	Drowned river valley, well mixed	Well-flushed drowned rivers, predominantly oceanic input	Tide-dominated drowned river mouth	Well mixed Partially mixed
Nestucca	Highly river-dominated drowned river mouth	Conservation	Drowned river valley, partially mixed	Well-flushed drowned rivers, predominantly oceanic input	NA	NA
Salmon River	Moderately river-dominated drowned river mouth	Natural	Drowned river valley, partially mixed	Well-flushed drowned rivers, predominantly freshwater input	NA	NA
Tillamook	Tide-dominated drowned river mouth	Shallow draft development	Drowned river valley, partially mixed	Well-flushed drowned rivers, predominantly oceanic input	Tide-dominated drowned river mouth	Well mixed Two-Layer (high flow)
Umpqua River	Highly river-dominated drowned river mouth	Shallow draft development	Drowned river valley, partially mixed	Well-flushed drowned rivers, predominantly freshwater input	River-dominated drowned river mouth	Two-Layer (high flow); Partially to well mixed (low flow)
Yaquina	Tide-dominated drowned river mouth	Deep draft development	Drowned river valley, partially mixed	Well-flushed drowned rivers, predominantly oceanic input	Tide-dominated drowned river mouth	Well mixed to partially mixed

freshwater inflow metric varies about 9-fold from the Coos Estuary to the Umpqua River Estuary. However, none of the estuaries with highest normalized freshwater inflow values were included (see Table 2-4). Thus, our selection of estuaries represents a reasonably wide range of drowned river mouth estuaries, but does not capture the full range expected if, for example, the Rogue River or Klamath River had been included. Additionally, no blind estuaries, coastal lagoons, marine harbor/coves, or tidally restricted coastal creeks, as defined in Chapter 2, were included.

The various geomorophological classifications of Oregon estuaries that we initially used to choose our target estuaries are summarized in Table 3-1 along with our classification schema. There is agreement among all the classifications that the seven target estuaries are drowned river mouth estuaries. In terms of riverine versus oceanic influences, there is also agreement on the three tide-dominated estuaries and two of the river-dominated systems, the Umpqua and Salmon estuaries. However, Parrish et al. (2001) classified the Nestucca Estuary as having "predominantly oceanic input" versus our classification as highly riverine dominated. It is possible that the Nestucca's division into two separate arms may affect how the normalized freshwater inflow values should be interpreted. Parrish et al. (2001) were uncertain whether to classify the Alsea Estuary as predominantly oceanic or freshwater input while we classified it as moderately river-dominated.

3.4 Estuary and Watershed Sizes

As discussed in Chapter 2, estuary size was defined as the sum of the NWI marine, estuarine, and tidal riverine polygons (http://www.fws.gov/nwi/; U.S. Fish Wild. Ser., 2002). Using this definition, estuary size among the target estuaries varied almost 20-fold, from about 3 km^2 for the Salmon River Estuary to 55 km^2 for the Coos Estuary (Tables 2-2 and 3-2). The Coos, Tillamook, Umpqua, and Yaquina are the four largest estuaries in Oregon other than the Columbia River. The only other large PNW estuaries are Grays Harbor and the Willapa Estuary in Washington and the Humboldt Estuary in northern California. The smaller target estuaries (Alsea, Salmon River, and Nestucca) were considered to capture much of the environmental range in the moderate-sized estuaries in the PNW.

As discussed in Chapter 2, PNW estuaries have extensive intertidal zones. The average percent intertidal area across all 89 PNW estuaries was 52% (Figure 2-8). The seven target estuaries bracketed that average, ranging from 32% in the Umpqua River Estuary to 87% in the Salmon River Estuary.

The sizes of the associated watersheds varied 63-fold (Table 3-2), from about 190 km^2 in the Salmon River Estuary to over 12,000 km^2 in the Umpqua River Estuary. The ratio of estuarine size to watershed size also varied by more than an order of magnitude. The Coos, Tillamook, and Yaquina all had relatively large estuaries compared to the watershed (>2.5%) while the Alsea and Salmon River had moderate-sized estuaries compared to the watershed (ca. 1%). Even though they differed 7-fold in estuary size, the Nestucca and Umpqua River had the smallest estuaries in relation to their watersheds, with estuary-to-watershed percentages of 0.61% and 0.28%, respectively.

3.5 Watershed Characteristics and Land Cover

The percent land cover values for selected classes of the watershed associated with each of the target estuaries are given in Table 3-3 based on the 2001 NOAA land cover data (see Chapter 2). Averaged across the entire watershed, none of these watersheds are highly developed. The maximum “high intensity development” land cover class was only 0.23% of the Coos watershed. The combined “high intensity development” and “low intensity development” classes exceeded 0.5% only in the Coos and Yaquina watersheds. The extent of cultivated land was also very low, with a maximum of 0.22% in the Alsea watershed. While there is little cultivated land, pasture for cattle grazing occurred within some of the coastal watersheds, in particular the Tillamook watershed. The NOAA survey does not have a land cover class specifically for pasture, but does have a class for grasslands which includes both natural grassland and pastures. To better capture the extent of pasture land, we used the 1992 National Land Cover Data (NLCD) land cover data, which has a pasture land cover class. Based on the NLCD data, the Tillamook, Umpqua River, and Nestucca watersheds each had about 3% to 4.5% pasture.

An integrative measure of land cover alteration is the percent impervious surfaces (see Chapter 2). The target estuaries showed a small range in this metric, with all the values about 2% (Table 3-3). These values are well below the 7.5% cutoff for “exceptional” environmental conditions used in the NOAA’s “Spatial Wetland Assessment for Management and Planning” (SWAMP) model (Sutter, 2001). These low percentages of altered land use and percent impervious surfaces among the target estuaries are representative of PNW watersheds in general (Tables 2-6 and 2-7).

3.5.1 Watershed Slopes

The median degree slope and median percent slope of the target estuaries are given in Table 3-3. For details on methods and datasets used to calculate watershed slope see Section B.12.4. We prefer degrees slope, which is bounded between 0 and 90 degrees, in comparison to percent slope, which can vary from 0 to infinity. Though they are the more commonly reported values, percent slopes can obtain very high values in areas with cliffs, which can skew the distribution and the mean compared to using degrees slope. The EPA Watershed Academy states that “high relief watersheds may have increments of 5-20 percent.” (http://www.epa.gov/owow/watershed/wacademy/wam/erosion.html). Using this criterion, all the target watersheds have a high relief, with median percent slopes ranging from 24.9% in the Salmon River watershed to 40.8% in the Tillamook watershed. The range in degrees slope was from about 14 degrees in the Salmon River watershed to over 22 degrees in the Tillamook watershed. Regardless of how it is measured, high relief is characteristic of PNW coastal watersheds

3.5.2 Population Characteristics

The Umpqua River and Coos watersheds had the largest human populations, about 100,000 and 39,000, respectively (Table 3-4). Although the Umpqua watershed has the largest population, the majority of the population (about 74,000) is located in the Roseburg area, which is 125 km from the coast. In contrast, the bulk of the population in the Coos watershed is located adjacent to the estuary. Normalized to watershed area, the population density in the Coos watershed was about 25 people per km^2, about 66% to 6-fold greater than the watersheds in the other target estuaries. To put the Coos watershed density in perspective, the Coos population density is one

Table 3-2. Estuary and watershed size of the seven target estuaries. Estuary area includes the marine, estuarine, and tidal riverine NWI classes. Estuarine intertidal and subtidal areas and the percent intertidal only include the estuarine NWI classes, so that the sum of the intertidal and subtidal may not equal the total estuary area. The intertidal area includes "irregularly exposed", "regularly flooded" and "irregularly flooded" habitats (approximately MLLW to MHHW).

ESTUARY	LATITUDE (deg N)	ESTUARY AREA (km^2)	ESTUARINE INTERTIDAL AREA (km^2)	ESTUARINE SUBTIDAL AREA (km^2)	% INTERTIDAL OF ESTUARINE AREA (km^2)	WATERSHED SIZE (km^2)	ESTUARY SIZE AS % OF WATERSHED
Alsea	44.4227	12.49	7.83	4.66	62.7	1222	1.02
Coos	43.4294	54.90	29.89	24.32	55.1	1575	3.48
Nestucca	45.1827	5.00	2.88	1.79	61.7	826	0.61
Salmon	45.0469	3.11	1.72	0.25	87.3	193	1.61
Tillamook	45.5130	37.48	25.61	11.23	69.6	1455	2.57
Umpqua	43.6694	33.78	8.85	18.88	31.9	12146	0.28
Yaquina	44.6205	19.96	9.05	9.77	48.1	650	3.07

Table 3-3. Watershed attributes of the seven target estuaries. Slope is given in degrees and percent slope. Land cover data for selected classes from the NOAA 2001 dataset except for the percentage for "pasture" which is from the 1994 NLCD data. The "% Impervious Surfaces" was calculated using the 30-m NLCD data (http://www.epa.gov/mrlc/nlcd-2001.html). The % impervious surface values generated from the NOAA land cover classes using Attila (U.S. EPA, 2004c) are given in parentheses for reference.

ESTUARY	LAND COVER (% WATERSHED AREA)									Median slope (degrees/ percent)
	High Intensity Development	Low Intensity Development	Cultivated	Grasslands / Pasture	Deciduous	Evergreen	Mixed Forest	Shrub	% Impervious Surfaces	
Alsea	0.02	0.13	0.22	3.31 / 1.30	3.07	57.07	25.86	6.82	0.51 (2.01)	18.1 / 32.7
Coos	0.23	0.86	0.00	7.09 / 1.39	1.45	48.64	15.89	15.43	1.16 (2.65)	18.5 / 33.5
Nestucca	0.02	0.14	0.00	5.49 / 3.01	4.63	44.60	29.22	12.15	0.55 (1.98)	14.9 / 26.6
Salmon	0.01	0.14	0.00	4.65 / 0.84	3.43	45.60	24.97	13.27	0.85 (2.15)	14.0 / 24.9
Tillamook	0.12	0.36	0.00	6.69 / 4.48	5.85	46.02	28.37	6.19	0.87 (2.30)	22.2 / 40.8
Umpqua	0.08	0.40	0.12	11.19 / 4.53	0.49	62.79	6.94	14.70	0.46 (2.20)	18.0 / 32.5
Yaquina	0.10	0.40	0.00	6.24 / 0.36	5.19	36.18	27.24	16.73	0.89 (2.38)	16.5 / 29.7

Table 3-4. Population size and density in the seven target estuaries. Population estimates based on 2000 census for the entire watershed. Population densities (# per km^2) are normalized both to the area of the watershed and to the area of the estuary.

ESTUARY	POPULATION IN WATERSHED	POPULATION DENSITY (# per km^2)		% POPULATION CHANGE FROM 1990 TO 2000
		NORMALIZED TO WATERSHED SIZE	NORMALIZED TO ESTUARY SIZE	
Alsea	4825	3.9	386	5.78
Coos	38,950	24.7	709	2.67
Nestucca	3404	4.1	681	12.28
Salmon	2881	14.9	926	20.10
Tillamook	7600	9.7	203	11.84
Umpqua	99,401	8.2	2943	6.09
Yaquina	7970	12.3	399	-4.85

to two orders of magnitude lower than in many Central and Southern California coastal watersheds, including Elkhorn Slough (339 km^{-2}), South San Francisco Bay (745 km^{-2}), San Diego (780 km^{-2}), and Anaheim (2841 km^{-2}). Population density was also normalized to the estuarine area, with the densest normalized population occurring in the Umpqua River watershed with over 2900 persons per km^2 of estuary. This high value for the Umpqua River Estuary, in part, reflects the small estuary size compared to the large size of the Umpqua River watershed (Table 3-2). However, this value is still at least an order of magnitude smaller than those the moderate and large sized estuaries in Central and Southern California.

The percent population change from 1990 to 2000 varied among the watersheds. The largest percent growth was in the Salmon River watershed which increased by 20% within a decade. At the opposite extreme, the Yaquina watershed actually showed a decrease in population of about 5%. For the Yaquina watershed it is important to recognize that other than the bay front, most of the City of Newport lies outside of the watershed. The population decrease in the Yaquina watershed over this period probably reflects a population decrease in City of Toledo, located in the upper portion of the Yaquina watershed.

3.6 Estuarine Hydrology

The seven target estuaries are relatively shallow with mean depths of 1.5-3 m (Table 3-5). The volumes of the target estuaries range from 1 x 10^6 m^3 (Salmon) to 2 x 10^8 m^3 (Coos), and are considered small estuaries compared to many in the U.S. (Hickey and Banas, 2003). These estuaries are classified as mesotidal with a mean tidal range of about 2 m (Table 3-5) and have mixed semidiurnal tides. There is a close coupling between the estuaries and the coastal ocean as a result of the large tidal prism relative to estuary volume (Table 3-5 and Hickey and Banas, 2003). Additionally, there is considerable variability in the mouth width of the estuaries, with the Salmon River Estuary having the smallest mouth (~ 40 m) and Coos and Umpqua estuaries having the largest (400 - 600 m, Table 3-5).

Reflecting the seasonal pattern in rainfall, freshwater flow is about 5-fold to almost 10-fold higher in the wet season compared to the dry season (Table 3-6). The wet season is defined as November through April while the dry season is defined as May through October. This seasonality in riverine inflow is several times greater for estuaries in the PNW compared to those along the Atlantic coast of the U.S. (Hickey and Banas, 2003). The Simmons Ratio, which is the ratio of riverflow per tidal cycle to tidal prism, is often used to classify estuaries by stratification (Simmons, 1955). When the Simmons Ratio is greater than 1, the estuary is highly stratified. When the ratio is 0.2–0.5, the estuary is considered partially mixed and when it is less than 0.1, it is considered well mixed. Using Simmons ratio (calculated using annual average inflow), Alsea and Salmon River estuaries are considered partially mixed, and the remainder of the target estuaries are considered well mixed.

Residence (or flushing) time is a measure of the retention of water within a defined boundary (Monsen et al., 2002). Residence time is often considered a mediating factor in assessing estuaries susceptibility to nutrient loading (e.g., Quinn et al., 1991; National Research Council, 2000). Estuaries with long residence times are considered more susceptible to nutrient enrichment than estuaries with short residence times. Estuarine residence times are influenced by freshwater inflow, tides, wind, mixing, stratification, and system topography. There are numerous terms and methods for calculating these time scales of transport, including flushing time, residence time, local residence time, turnover time, freshwater replacement time, and transit time (Monsen et al., 2002; Abdelrhman, 2005). Two commonly used methods for calculating residence time are the fraction of freshwater method and the modified tidal prism approach (Dyer and Taylor, 1973). The fraction of freshwater method calculates the amount of time for the freshwater inflow to replace the freshwater in the system. It can be interpreted as the average transit time for freshwater in the system and is an appropriate time scale for materials that are input to the estuary from the river (Sheldon and Alber, 2002). The modified tidal prism is more appropriate for the dry season estimates when tidal forcing dominates (riverflow small compared to tidal flow) and for estuaries such as those in the PNW where the volume is small compared to the tidal prism. The fraction of freshwater method is more appropriate during the wet season when freshwater inflows dominate. Presented in Table 3-7 are wet and dry season residence times for the seven estuaries calculated using the modified tidal prism and fraction of freshwater inflow approaches. The residence times of the seven target estuaries are short (typically less than 1 month). During the wet season, the residence time calculated using the fraction of freshwater method ranges from 2 to 10 days; while during the dry season, the residence time calculated using the modified tidal prism method varies from 1 to 48 days.

3.7 Nutrient Loading

Previously published estimates are available for nitrogen and phosphorous loading for six of the target estuaries. The nitrogen loading varies 26-fold for the seven target estuaries with the Salmon River Estuary having the lowest nitrogen loading and the Umpqua River Estuary having the highest (Table 3-8). This variability in nitrogen loading is related to differences in stream flow among the estuaries (Table 3-6). The loadings in Table 3-8 represent point sources as well as nonpoint and upstream sources. The Umpqua River estuary had the highest phosphorous loading, while the Yaquina had the lowest (Table 3-8). Point source inputs represented the largest phosphorous source in Coos and Yaquina estuaries, while in Tillamook, and Umpqua estuaries non-point inputs from forested land was the dominant source of phosphorous (Quinn et al., 1991). Nonpoint source inputs associated with forested land was the dominant nitrogen source in Coos, Tillamook, Umpqua, and Yaquina estuaries (Quinn et al., 1991). Upstream sources were the major source of nitrogen and phosphorous in the Alsea Estuary (Quinn et al., 1991). Nutrient loading estimates are often normalized by estuary area and volume to examine sensitivity to nutrient loading. When the nitrogen loading estimates are normalized to area, Yaquina and Coos have the lowest loading, while Umpqua and Alsea have the highest.

The loadings presented in Table 3-8 do not include the input of nutrients from the coastal ocean. Brown and Ozretich (2009) compared the major sources of dissolved inorganic nitrogen (DIN) to Yaquina Estuary during the wet and dry seasons (Table 3-9). There are strong seasonal differences in the nitrogen sources to the estuary. During the wet season, riverine sources dominate, while during the dry season oceanic nitrogen inputs associated with coastal upwelling dominate. In the dry season, benthic flux of DIN from the sediments into the water column is the second largest source of nutrients. Atmospheric deposition of nitrogen is a minor nitrogen source with direct deposition on the estuary only representing about 0.05% of the inorganic nitrogen input to the estuary. In addition, atmospheric deposition on the watershed is small (8%) compared to the watershed input associated with nitrogen fixing red alder in the watershed. Annual nitrogen input from wastewater treatment facility effluent is estimated to be 0.4% of the total input to the estuary. The ocean is also a large source of phosphorous to PNW estuaries; however, at this time the oceanic phosphorous loading has not been quantified.

Oregon Coast Range streams have high nitrate concentrations relative to other forested PNW watersheds (Compton et al., 2003). Wigington et al. (1998) hypothesized that forest vegetation, in particular the presence of red alder (*Alnus rubra*), is the primary control of spatial variability in stream nitrate concentrations in the Oregon Coast Range (including the Salmon River, Siletz, and Alsea watersheds). Red alder is a native species in the PNW that colonizes areas disturbed by fires, logging and landslides. Red alder have symbiotic N_2-fixing bacteria that can fix 50-200 kg N ha^{-1} y^{-1} in pure stands (Binkley et al, 1994). Compton et al. (2003) found a significant relationship between alder cover in the watershed and nitrate concentration in streams in the Salmon River watershed. Naymik et al. (2005) found a similar relationship between stream total nitrogen and broadleaf cover (which is primarily red alder in the Coast Range) in the Tillamook watershed. Brown and Ozretich (2009) estimated that > 80% of the riverine nitrogen loading to Yaquina Estuary is related to red alder cover.

Table 3-5. Hydrographic characteristics of the seven target estuaries. Sources were (1) Shirzad et al. (1988), (2) Johnson and Gonor (1982), (3) Percy et al. (1974), (4) Coastal Inlets Research Program (http://cirp.wes.army.mil/databases/inletsdb/inletsdbinfo.html), , (5) http://tidesandcurrents.noaa.gov/tides04/tab2wc1b.html, and (6) http://ian.umces.edu/neea/siteinformation.php. Mouth widths are from the Pacific Coast Ecosystem Information System (*PCEIS*; Lee and Reusser, 2006).

ESTUARY	TIDAL RANGE (m)	TIDAL PRISM (m^3)	ESTUARINE VOLUME (m^3)	RATIO OF ESTUARINE VOLUME TO TIDAL PRISM	MOUTH WIDTH (m)	MEAN DEPTH (m)	DREDGED
Alsea	1.8 (1,5)	1.15×10^7 (1)	1.9×10^7 (6)	1.6	140	2.0 (1)	No
Coos	1.7 (5)	5.27×10^7 (3)	2.1×10^8 (6)	3.9	620	1.5 (3)	Yes
Nestucca	1.8 (5)				110		No
Salmon	1.6 (2,3)	9.5×10^5 (2)	1.4×10^6 (2)	1.5	40		No
Tillamook	1.6-1.9 (5)	4.81×10^7 (1)	7.0×10^7 (6)	1.5	360	1.8 (1)	Yes (primarily prior to 1979)
Umpqua	1.6 (5)	6.23×10^7(4)	7.5×10^7 (6)	1.2	425		Yes
Yaquina	1.8-1.9 (5)	2.38×10^7 (4)	3.0×10^7 (6)	1.3	290	3.0 (1)	Yes

Table 3-6. Wet and dry season freshwater inflow for the seven target estuaries. Sources: (1) Shirzad et al. (1988); (2) Quinn et al. (1991), and (3) Percy et al. (1974).

ESTUARY	WET SEASON (m^3 s^{-1})	DRY SEASON (m^3 s^{-1})	ANNUAL AVERAGE (m^3 s^{-1})	SIMMONS RATIO = RIVERFLOW PER TIDAL CYCLE / TIDAL PRISM	ESTUARY TYPE BASED ON SIMMONS RATIO	SOURCE/NOTES
Alsea	112	15	65	0.3	Partially Mixed	(1)
Coos			82	0.1	Well Mixed	(2)
Nestucca	51	8	30 43			Nestucca River near Beaver Estimate of flow at mouth (3)
Salmon	18	4	11	0.5	Partially Mixed	Discharge near Otis
Tillamook	189	31	110	0.1	Well Mixed	(1)
Umpqua	342	71	209 210 19	0.1	Well Mixed	Umpqua discharge near Elkton Umpqua River from (3) Smith River from (3)
Yaquina	48	5	27	0.1	Well Mixed	(1)

Table 3-7. Residence time calculated using the modified tidal prism and fraction of freshwater approaches for the seven target estuaries.

ESTUARY	MODIFIED TIDAL PRISM		FRACTION OF FRESHWATER INFLOW		SOURCE
	WET SEASON (days)	DRY SEASON (days)	WET SEASON (days)	DRY SEASON (days)	
Alsea	1	4-9	NA	14 - 17	Choi (1975)
Coos	7 - 13 13-16	11 - 16 40-48	2 - 10 3 – 11	34 19 - 31	Choi (1975) Arneson (1976)
Nestucca	NA	NA	NA	NA	
Salmon	NA	1-2	NA	1-2	Askren et al. (1976)
Tillamook	2 – 3	3 – 4	3 1 – 7	9 5 - 32	Choi (1975) Colbert and McManus (2003)
Umpqua	4 - 5	5 – 10	3	5 – 8	Choi (1975)
Yaquina	6	6 -9	9	20 - 106	Choi (1975)

Table 3-8. Estimates of nutrient (nitrogen and phosphorous) loading to six of the seven target estuaries. Load estimates are not available for Nestucca Estuary, and phosphorous loads are not available for Salmon River estuary. Total load estimates for Alsea, Coos, Tillamook, Umpqua and Yaquina are from Quinn et al. (1991), while nitrogen load estimate for Salmon River Estuary estimated from flow and DIN from Compton et al. (2003).

	NITROGEN LOADING			PHOSPHOROUS		
ESTUARY	TOTAL (tons y^{-1})	NORMALIZED BY ESTUARY AREA (tons km^2 y^{-1})	NORMALIZED BY ESTUARY VOLUME (tons m^3 y^{-1})	TOTAL (tons y^{-1})	NORMALIZED BY ESTUARY AREA (tons km^2 y^{-1})	NORMALIZED BY ESTUARY VOLUME (tons m^3 y^{-1})
Alsea	3875	310.3	2.0×10^{-4}	61	4.9	3.2×10^{-6}
Coos	3054	55.6	1.5×10^{-5}	95	1.7	4.5×10^{-7}
Salmon	337	108.4	2.4×10^{-4}	NA	NA	NA
Tillamook	4315	115.1	6.2×10^{-5}	73	2.0	1.0×10^{-6}
Umpqua	8870	262.6	1.2×10^{-4}	163	4.8	2.2×10^{-6}
Yaquina	984	49.3	3.3×10^{-5}	48	2.4	1.6×10^{-6}

Table 3-9. Comparison of nitrogen sources during wet and dry seasons for the Yaquina Estuary (Brown and Ozretich, 2009).

	NITROGEN INPUT (mol DIN d^{-1})		
SOURCE	WET SEASON	DRY SEASON	ANNUAL AVERAGE
River	2.6×10^{5}	2.3×10^{4}	1.6×10^{5}
Ocean	8.8×10^{4}	$3.8 – 5.1 \times 10^{5}$	$2.3 – 3.0 \times 10^{5}$
Wastewater	1.8×10^{3}	1.5×10^{3}	1.6×10^{3}
Benthic Flux[1]	-	4.3×10^{4}	-
Atmospheric Deposition			
On Estuary	2.2×10^{2}	1.2×10^{2}	1.7×10^{2}
On Watershed	1.1×10^{4}	6.0×10^{3}	8.5×10^{3}
Source: [1]DeWitt et al. (2004)			

CHAPTER 4:
WATER QUALITY SURVEYS IN SEVEN TARGET ESTUARIES

Cheryl A. Brown and Christina L. Folger

Key Findings

- **Water quality conditions in the target estuaries during the dry season were dependent upon ocean conditions at time of sampling and fresh water inflow.**
- **The coastal ocean was the primary source of phosphate during the dry season.**
- **Estuaries received nitrogen from coastal ocean and watershed.**
- **Dry season chlorophyll *a* levels were relatively low in target estuaries, with 85% of the stations sampled having chlorophyll *a* levels < 5 µg l^{-1}.**
- **No incidence of low dissolved oxygen (< 5 mg l^{-1}) occurred at any of the stations sampled.**
- **There were some instances of poor water quality conditions (based on high dissolved inorganic nitrogen) in Tillamook Estuary.**
- **Wet season dissolved inorganic nitrogen levels appear to be related to the red alder cover within the watersheds.**

4.0 Introduction

We collected water quality data in seven target estuaries to evaluate current water quality conditions and for use in dividing each estuary into ocean- and river-dominated segments with respect to nitrogen sources. These estuaries (Alsea, Nestucca, Yaquina, Salmon River, Coos, Umpqua River and Tillamook) vary in size from about 3 km^2 to 55 km^2, and from river to ocean dominated (Table 3-1 and 3-2). Our sampling consisted of high tide and low tide water quality cruises and of short-term deployments of water quality datasondes during the months of June through September of 2004 and 2005.

Most of this chapter focuses on the dry season (May to October), because this is the time period of biological nutrient utilization. During the wet season (November to April), there is little nutrient utilization due to short residence times (associated with high freshwater inflow) and low solar irradiance. Mixing diagrams from Yaquina (unpublished data) and Tillamook estuaries (Colbert and McManus, 2003) reveal conservative transport of nutrients during the wet season. In addition, water column chlorophyll *a* and macroalgae biomasses are minimal during the wet season.

Previous research (e.g., de Angelis and Gordon, 1985; Roegner and Shanks, 2001; Roegner et al., 2002; Colbert and McManus, 2003) has demonstrated that PNW estuaries are strongly influenced by conditions occurring on the shelf, in particular wind-driven coastal upwelling during the spring and summer. The NO_3^-, PO_4^{3-}, and temperature of water entering estuaries during flood tides respond rapidly to changes in along-shore wind stress. High levels of dissolved inorganic nitrogen (primarily in the form of NO_3^-) and PO_4^{3-} entering the Yaquina Estuary lags upwelling-favorable winds by about 2 days (Brown and Ozretich, 2009). The input of phytoplankton lags upwelling favorable winds by approximately 6 days and typically occurs during downwelling conditions (Brown and Ozretich, 2009). Variations in water properties determined by ocean conditions propagate approximately 11-13 km into the Yaquina Estuary. Recently, there have been occurrences of severe hypoxia on the inner continental shelf of Oregon (Grantham et al., 2004; Chan et al., 2008). At times low oxygen water from the inner shelf is advected into the Yaquina Estuary (Brown et al., 2007).

4.1 Methods

4.1.1 Water Quality Cruises

Water quality cruises were conducted in the seven target estuaries during the summers of 2004 and 2005. During each cruise between 10 and 17 stations were sampled in each estuary, depending upon the size of the estuary, and high and low tide cruises were conducted. The time required to complete each cruise depended upon the size of the estuary, with small estuaries (Salmon River, Nestucca, and Alsea) taking 2-3 hours and larger estuaries (Coos and Tillamook) taking about 4-5 hours. Due to logistical constraints some of the riverine stations in Coos and Tillamook were sampled up to 7 hours after the beginning of the cruise. To minimize the time required to complete each cruise, two boats were used to sample the three largest estuaries (Coos, Tillamook and Umpqua River). Both the low and high tide cruises proceeded from the estuary mouth upriver to follow the propagation of the tide. For location of water quality cruise stations in each target estuary see Chapter 5. The cruises extended from the marine to the tidal fresh regions for all estuaries except Coos Estuary. For Coos Estuary, the lowest salinity sampled during the dry season was 14 psu. The dates of the cruises and the number of stations occupied in each estuary are presented in Table 4-1. We didn't sample during May or October because these are transitional months that may experience high freshwater inflow events. Additional cruises were conducted to characterize winter conditions during the wet seasons of 2006 and 2007.

Table 4-1. Dates of cruises and number of stations sampled for the seven target estuaries.

ESTUARY	DATES OF DRY SEASON CRUISES		DATES OF WET SEASON CRUISES		NUMBER OF STATIONS
	HIGH TIDE	LOW TIDE	HIGH TIDE	LOW TIDE	
Alsea	9/24/04	9/28/04	3/11/07	3/12/07	10
Salmon	7/20/04	7/16/04	3/21/07	3/27/07	10
Yaquina	6/15/04	6/21/04	2/9/06		10
Coos	8/9/05	8/8/05	2/16/07	2/15/07	17
Nestucca	8/11/04	8/20/04	1/25/07	1/27/07	15
Tillamook	7/22/05	7/23/05	3/2/07	3/1/07	15
Umpqua	6/24/05	6/25/05	2/11/07	2/12/07	13

At each station profiles of conductivity, temperature and depth (CTD; SBE 19 SEACAT Profiler, Sea-Bird Electronics, Inc., Bellevue, Washington), turbidity (Seapoint Turbidity Sensor, Seapoint Sensors, Inc., Kingston, New Hampshire), photosynthetically active radiation (PAR, LI-193 spherical sensor, LI-COR Biosciences, Lincoln, Nebraska) and *in situ* fluorescence (WETStar Chlorophyll Fluorometer, WET Labs, Philomath, Oregon) were measured. All instruments on the Sea-Bird profiler were factory calibrated by the manufacturer. The profile measurements were taken at 0.5-sec intervals from the water surface to 0.5 m above the bottom, and during post-processing the data were binned into 0.25-m intervals. If the water depth at the station was shallow (typically less than 4 m), a YSI 6600 multiparameter sonde (YSI, Yellow Springs, Ohio) was used for water quality measurements (temperature, conductivity, fluorescence, turbidity, depth, and dissolved oxygen). Dissolved oxygen measurements for the profiles were obtained using a YSI 6600 attached to the Seabird frame. Dissolved oxygen measurements (surface, mid-depth, and bottom) were collected in Tillamook, Coos and Umpqua River estuaries during both the low and high tide cruises, and during the low tide cruise in Salmon River Estuary. No dissolved oxygen measurements were collected in the Alsea, Yaquina and Nestucca estuaries. The YSI datasondes were calibrated using the methods presented in Section B.4.

Water samples were collected from mid-depth at each station using a hand-operated pump, filtered (0.45 μm filter), and frozen until analysis. The samples were analyzed for dissolved inorganic nutrients (nitrate+nitrite, ammonium, phosphate and silicate) by MSI Analytical Laboratory, University of California-Santa Barbara, CA. One-liter surface water samples were collected from each station and analyzed for chlorophyll *a*. These water samples were filtered within 2 to 4.5 hours of sample collection using 47-mm GF/F filters. The volume of water filtered varied between 250 and 500 ml. Chlorophyll *a* was extracted by sonicating the filters and soaking them overnight in 10 ml of 90% acetone. The next morning the samples were centrifuged and analyzed for chlorophyll *a* content using a fluorometer (10 AU Fluorometer, Turner Designs, Inc., Sunnyvale, CA). Four-liter water samples were collected from each cruise station and analyzed for total suspended solids (TSS). Each sample was agitated prior to filtration and filtered using ashed Whatman 47 mm GF/F filters. Filters from samples were dried overnight in an oven at 70°C and allowed to reach room temperature in a desiccator prior to weighing. More information on the water column sample analyses can be found in Section B.2.1 and B.2.2

4.1.2 Datasonde Deployment

YSI 6600 multiparameter sondes (YSI, Yellow Springs, Ohio) and Mini-CTDs (Star-Oddi, Iceland) were deployed in the seven target estuaries along the salinity gradient (see Table 4-2 for number of instruments deployed in each estuary and Chapter 5 for maps illustrating deployment locations). The YSI datasondes measured conductivity, temperature, turbidity, chlorophyll *a* (*in situ* fluorescence), dissolved oxygen, pH, and depth. Mini-CTD units measured depth, salinity and temperature. Both instruments collected data at 15 minute intervals. During 2004, we lost two instruments deployed in the lower portion of the Nestucca Estuary. In order to fill this data gap, in 2006 an additional YSI 6600 CTD was deployed at the mouth and three additional Mini-CTDs were deployed in the mesohaline region of the estuary. Calibration procedures for all instrumentation are presented in Sections B.2-B.4. In addition to the datasondes that we deployed, there were additional YSI datasondes deployed in Coos Estuary associated with the

South Slough Estuarine Research Reserve (Charleston Bridge and Valino Island) and Oregon Institute of Marine Biology (OIMB). The locations of these instruments (labeled Y5-Y7) are presented in Figure 5-10.

Table 4-2. Number of short-term datasondes deployed during the dry season in each estuary and year deployed.

ESTUARY	NUMBER OF YSI DATASONDES	NUMBER OF MINI-CTDS
Alsea	2 (2004)	3 (2004)
Coos	7 (2005)	3 (2005)
Nestucca	2 (2004) 1 (2006)	3 (2004) 3 (2006)
Tillamook	4 (2005)	2 (2005)
Salmon	2 (2004)	3 (2004)
Umpqua	4 (2005)	3 (2005)
Yaquina	5 (2004)	0

4.2 Riverine Nutrient Inputs During the Wet Season

To examine differences in riverine nutrient levels for the target estuaries, we calculated median wet season dissolved inorganic nitrogen (DIN) and PO_4^{3-} using stations with salinity < 2 psu (Table 4-3). During the wet season, the Yaquina estuary had the highest riverine DIN levels, while Umpqua had the lowest. In general, estuarine differences in wet season DIN were similar to variations in deciduous cover among the estuaries (Table 4-3). This suggests that variations in riverine DIN among the estuaries primarily result from variations in red alder cover, which is similar to the findings of Compton et al. (2003) and Wigington et al. (1998). The Nestucca estuary had the highest median wet season PO_4^{3-} (Table 4-3). The highest wet season DIN levels observed in the target estuaries were measured at Stations C4 and C12 in the Tillamook Estuary (Figure 5-12). Highest median NH_4^+ levels were measured in the Tillamook Estuary and the highest NH_4^+ values measured occurred at Stations C4 and C12 in the Tillamook Estuary.

Table 4-3. Median wet season dissolved inorganic nitrogen and phosphate calculated using stations with salinity < 2 psu. The table entries are listed from highest to lowest DIN levels.

ESTUARY	MEDIAN WET SEASON			DECIDUOUS COVER (%)
	DIN (µM)	NH_4^+ (µM)	PO_4^{3-} (µM)	
Yaquina	92.6	0.93	0.35	5.19
Tillamook	69.8	2.66	0.40	5.85
Nestucca	61.9	0.87	0.59	4.63
Alsea	35.0	0.99	0.38	3.07
Salmon	34.9	0.88	0.25	3.43
Coos	33.3	1.04	0.24	1.45
Umpqua	17.6	1.60	0.28	0.49

4.3 Coastal Ocean and River Conditions During Dry Season Sampling

The seven target estuaries span about 250 km along the Oregon Coast (Figures 2-3 and 2-4). Flood-tide water temperature can be used as an indicator of ocean conditions during our surveys and to assess the variability in upwelling along the Oregon Coast. Low flood-tide water temperatures (8 – 10°C) indicate the input of high-nutrient (NO_3^- and PO_4^{3-}) water to the estuaries associated with upwelling while warm temperatures indicate downwelling conditions and input of low-nutrient water from the ocean. Figure 4-1 shows the flood-tide water temperature near the mouth of Yaquina Estuary (Station Y1, which has been previously been demonstrated as an indicator of upwelling/downwelling; Nelson and Brown, 2008) and water temperature for the high tide cruises for the outermost estuarine stations sampled for the 2004 classification effort. Figure 4-2 shows the flood-tide water temperature near the entrance of Yaquina and Coos estuaries (which are about 140 km apart) and the water temperature from the high tide cruises from the 2005 classification effort. The close agreement between Yaquina and Coos estuaries flood-tide water temperatures and water temperatures from the classification cruises demonstrates that the ocean conditions are relatively uniform over the geographic range of estuaries that we sampled during 2004 and 2005. This agrees with the analysis by Hickey and Banas (2003) that demonstrated that water temperatures at the entrance of three estuaries along the Oregon and Washington coasts (Coos, Grays Harbor and Willapa), which spanned 400 km, were highly correlated during the upwelling season. During 2004, the Salmon River and Nestucca estuaries were sampled during low oceanic nutrient conditions, with the high tide cruise outermost stations having NO_3^-+NO_2^- levels of < 1.3 μM and PO_4^{3-} levels of < 0.5 μM. There was a delayed onset of coastal upwelling along the Oregon coast during 2005 with weak upwelling conditions occurring from late May to mid July and strong upwelling conditions not commencing until mid July (Kosro et al., 2006). During 2005, all estuaries that we sampled during high tide coincided with upwelling conditions, which is indicated by the relatively high NO_3^-+NO_2^- and PO_4^{3-} conditions at the outermost stations. The upwelling conditions were stronger during the Tillamook and Coos estuaries cruises than the Umpqua cruises (Figure 4-2). This is evident in the NO_3^-+NO_2^- and PO_4^{3-} levels during the high tide cruise at the outermost stations of the Tillamook and Coos estuaries compared with the Umpqua River Estuary. Nitrate+nitrite concentrations at the outermost stations were 19.1 and 23.6 μM for the Coos and Tillamook estuaries, respectively, compared to 12.9 μM in the Umpqua River Estuary.

We used water temperature versus NO_3^-+NO_2^- and PO_4^{3-} relationships generated using data from the inner shelf off of Newport, Oregon (Wetz et al., 2005), to determine whether the nutrient concentrations in the water entering the estuaries during flood tides were consistent with values associated with coastal upwelling (Figures 4-3 and 4-4). For all of the estuaries, except for Alsea, the high tide outermost stations fell along the relationships developed using offshore data, suggesting that the NO_3^- and PO_4^{3-} levels near the estuary mouths were consistent with coastal upwelling.

The elevated NO_3^-+NO_2^- concentrations at the mouth of the Alsea Estuary were associated with high freshwater input. There was a relatively high freshwater inflow event on September 19, 2004, which was 5 days prior to the high tide Alsea cruise. On September 19, the Alsea River gauge at Tidewater (http://waterdata.usgs.gov/or/nwis/sw) had a discharge of 1120 cubic feet s^{-1} (cfs), which is almost an order of magnitude higher than the long-term mean daily discharge for September (130 cfs). We had additional cruises of the Alsea Estuary on September 21, 2004, which had an NO_3^-+NO_2^- concentration at the most riverine station of 73.8 µM (with salinity = 0 psu). The Oregon Department of Environmental Quality (http://deq12.deq.state.or.us/lasar2/) sampled the Alsea River at a station about 20 miles upriver from the mouth of the estuary on September 22, 2004 which had a NO_3^-+NO_2^- concentration of 49.4 µM, which was the highest value measured at this station during September in the past 11 years. The median NO_3^-+NO_2^- concentration at this station for the month of September was 6.3 µM (data from 1994-2005, n = 9). The anomalous NO_3^-+NO_2^- concentrations at the Alsea Estuary were a result of this first freshwater inflow event of the fall. Interestingly, the PO_4^{3-} concentrations during the 2004 Alsea cruise were not anomalous.

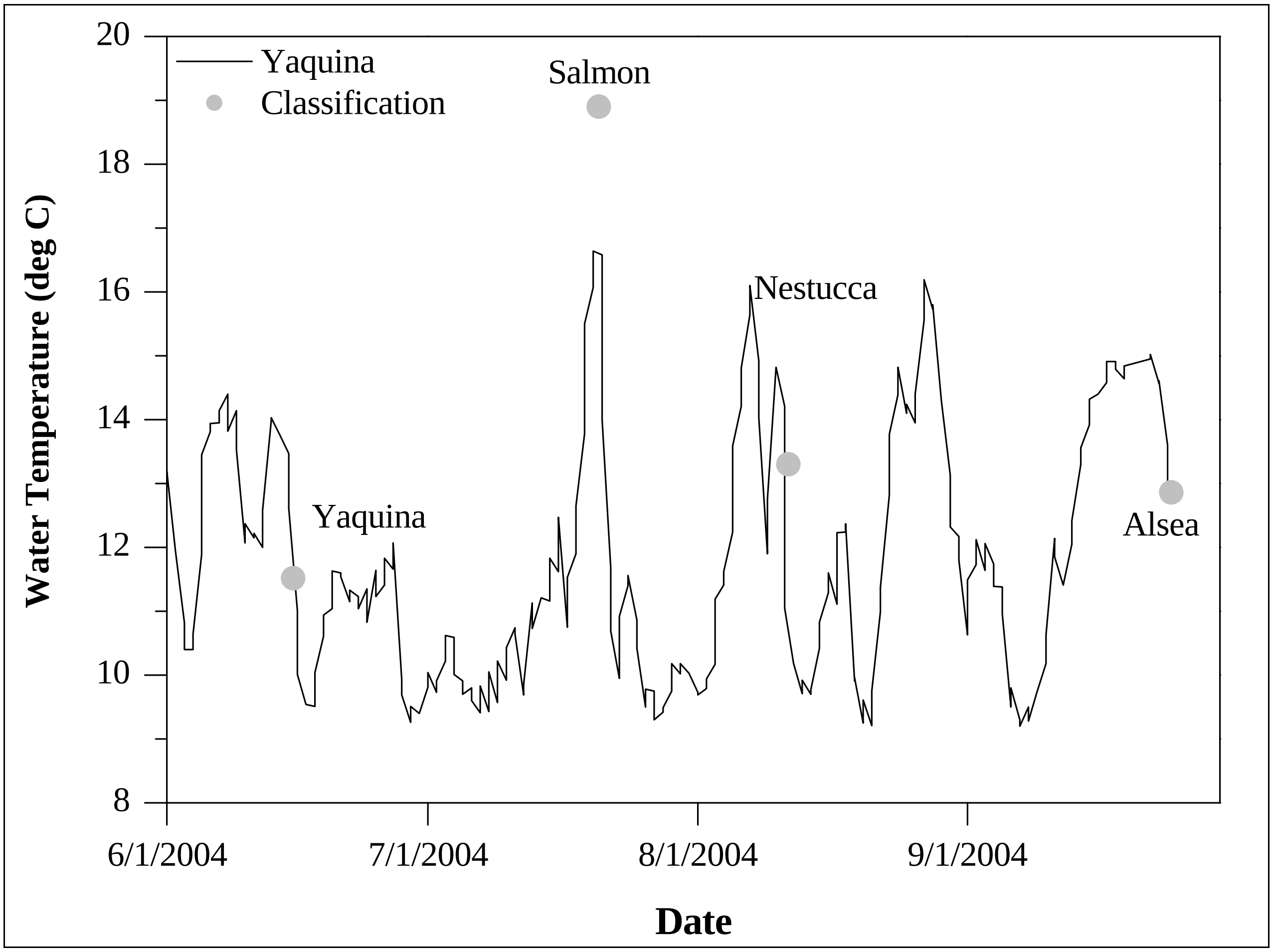

Figure 4-1. Flood-tide water temperature at Yaquina Estuary (solid line) and outermost high tide station from 2004 dry season classification cruises (filled circle). Each datapoint from the classification cruises is labeled with its estuary name.

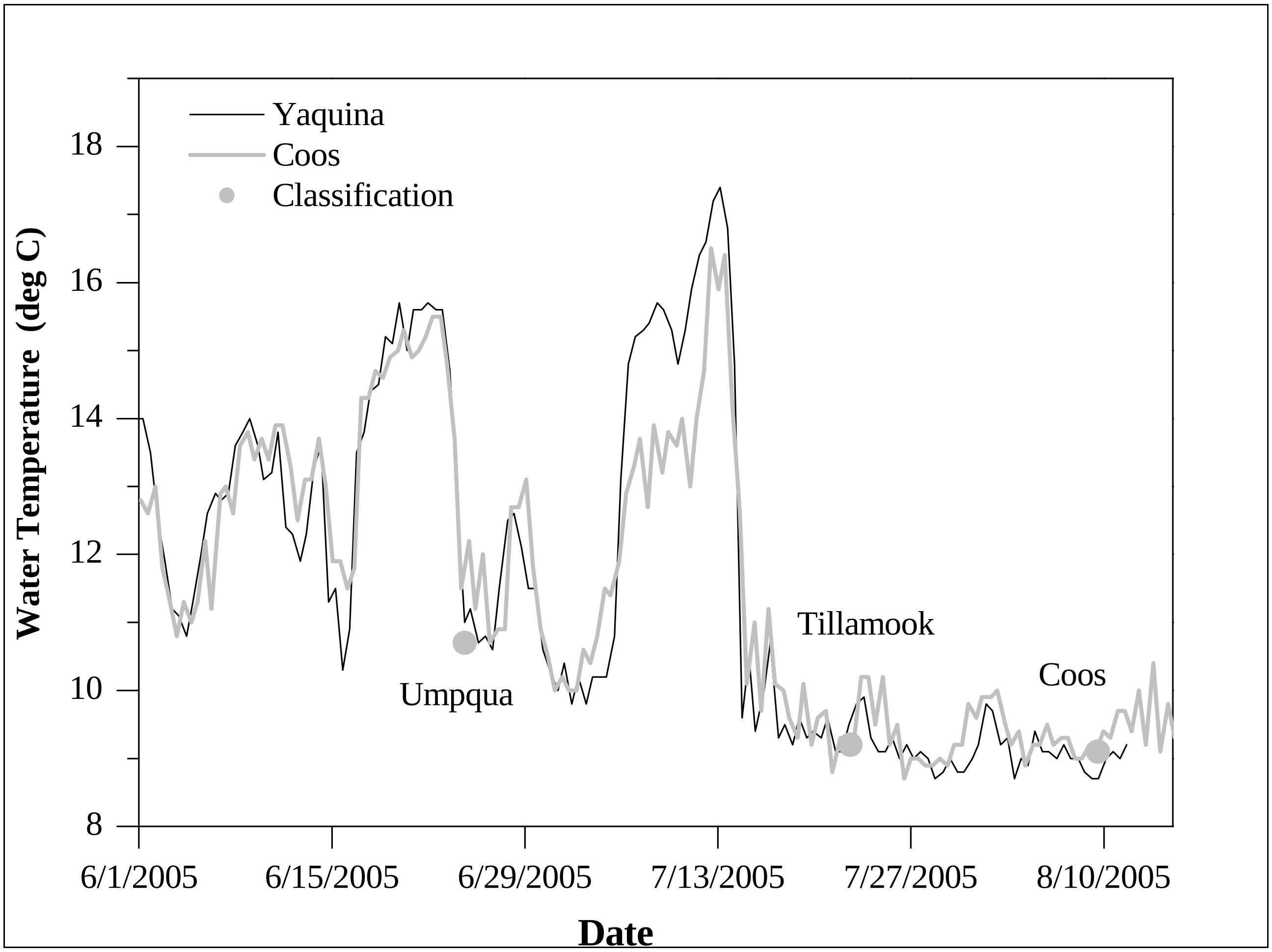

Figure 4-2. Flood-tide water temperature at Yaquina (black line) and Coos (gray line) estuaries and outermost high tide stations from 2005 dry season classification cruises (filled symbols).

4.4 Estuarine Water Quality

4.4.1 Dry Season Chlorophyll *a* and TSS Patterns

In Tables 4-4 and 4-5 we present the median dry season chlorophyll *a*, TSS, NO_3^-+NO_2^-, NH_4^+, and PO_4^{3-} for the target estuaries and rank the estuaries by these median values. Dry season chlorophyll *a* within the seven target estuaries was low with a median value of 2 μg l^{-1} (all estuaries and dry season cruises). Chlorophyll *a* was in the low to medium category for eutrophication symptoms (Bricker et al., 2003). Yaquina Estuary had the highest median chlorophyll *a* (5.6 μg l^{-1}), while the Umpqua River Estuary had the lowest (0.9 μg l^{-1}). The highest chlorophyll *a* measured was 17.3 μg l^{-1}, which occurred near the mouth of the Coos Estuary (at Station C2). The Coos Estuary had the highest median total suspended solids (13.1 mg l^{-1}), while the Umpqua River Estuary had the lowest (1.6 mg l^{-1}). Our data suggests that ocean-dominated estuaries have higher chlorophyll *a* than river-dominated estuaries. It is not possible to separate out the chlorophyll *a* associated with oceanic import from that growing *in situ* in the estuary from this dataset.

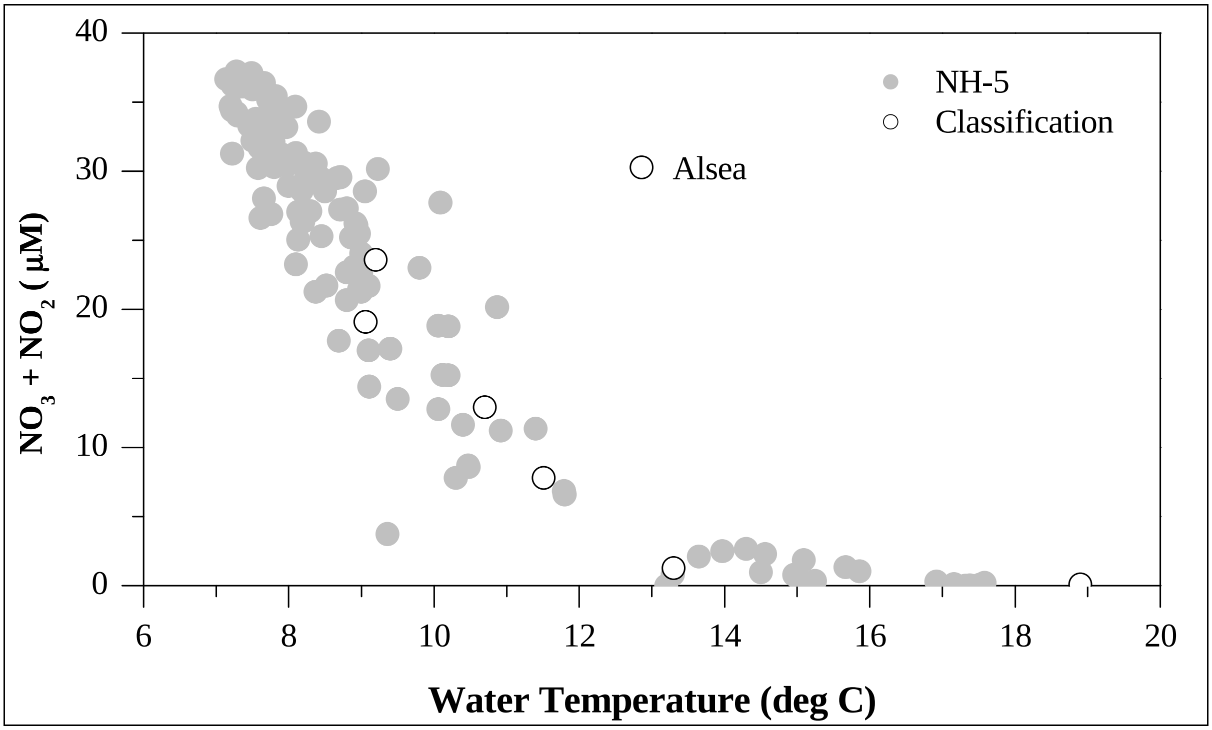

Figure 4-3. Water temperature versus $NO_3^- + NO_2^-$ relationship generated using data from a station on the shelf (NH-5, Wetz et al., 2005; gray circles) and classification high tide outermost stations (hollow circles). All data are from the dry season. Data point from Alsea is identified.

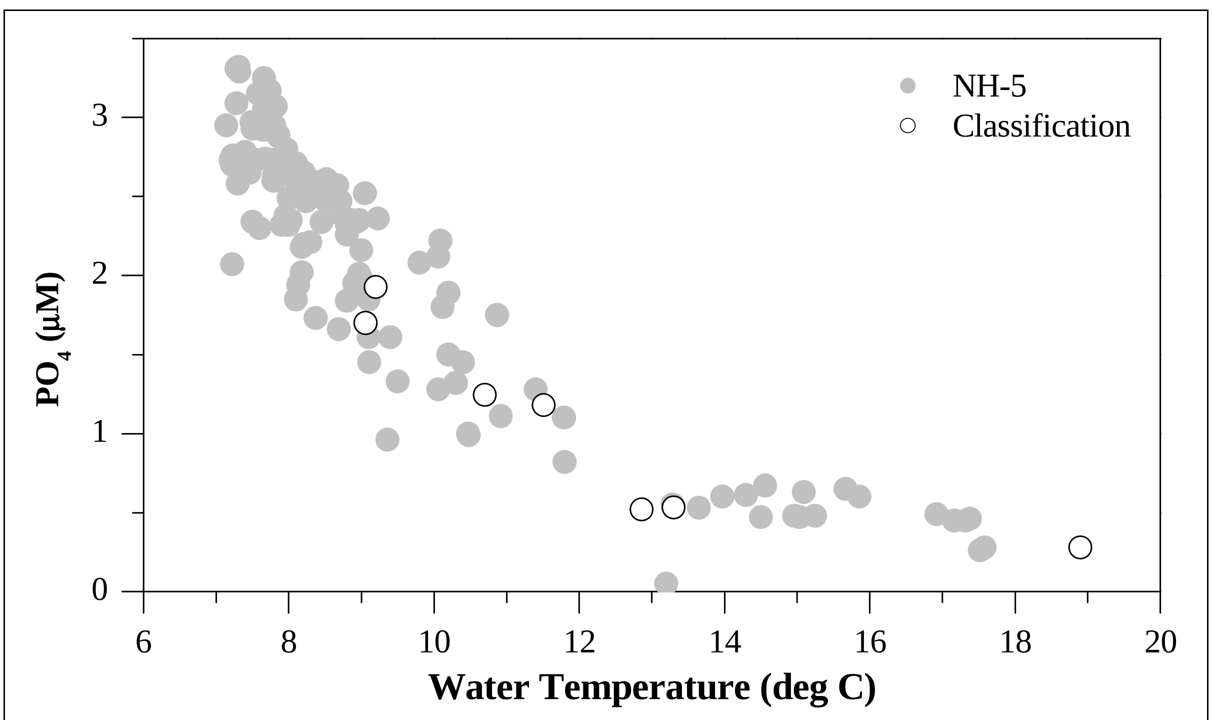

Figure 4-4. Water temperature versus PO_4^{3-} relationship generated using data from a station on the shelf (NH-5, Wetz et al., 2005; gray circles) and classification high tide outermost stations (hollow circles). All data are from the dry season.

Roegner and Shanks (2001) demonstrated that oceanic chlorophyll *a* was imported into South Slough in the Coos Estuary and that chlorophyll *a* concentrations were variable ranging from 0 to 15 μg l^{-1}. Previous research in the Yaquina Estuary (Brown and Ozretich, 2009) has demonstrated that water entering the estuary during flood tides can have peak chlorophyll *a* concentrations of 40-50 μg l^{-1}, and that the chlorophyll *a* levels in the ocean-dominated portion of the estuary are dependent upon coastal wind forcing and are quite variable. The median chlorophyll *a* imported into the Yaquina Estuary during May-September of 2002 and 2003 was 4 μg l^{-1} (n = 298).

There was no clear pattern in chlorophyll *a* (from water samples) versus salinity, except for during the high tide cruise in Coos Estuary (Figure 4-5). During the high tide cruise in the Coos Estuary there was evidence of elevated chlorophyll *a* levels at higher salinities. *In situ* fluorescence data from two of YSI datasondes in the Coos Estuary (Stations Y1 and Y3) showed the import of chlorophyll *a* from the coastal ocean to the estuary (indicated by a significant relationship between *in situ* fluorescence and salinity) and a strong tidal signal. The low chlorophyll *a* values in Umpqua were consistent with the *in situ* fluorescence data. The median values of *in situ* fluorescence at Stations Y1 and Y2 in Umpqua were 1.5 μg l^{-1}, and a tidal signal was not apparent. For comparison, the median *in situ* fluorescence values at Stations Y1 and Y3 in Coos Estuary were 8.2 and 3.7 μg l^{-1}, respectively. The median value of chlorophyll *a* observed in the Yaquina Estuary during this study was consistent with historical dry season data from this estuary (median = 4.9 μg l^{-1}, n = 1205; Brown et al., 2007). Additionally, the median value in the Alsea Estuary was similar to that from monthly sampling conducted from May to September 2004 (median of all sampling events at six stations = 1.6 μg l^{-1}, n = 30).

For PNW estuaries, we compiled all available chlorophyll *a* data, including other data collected by our laboratory, National Coastal Assessment, Oregon Department of Environmental Quality, and Washington Department of Ecology. Figure 4-6 shows the median dry season chlorophyll *a* for eight PNW estuaries (including six of the target estuaries) and the freshwater inflow normalized by estuary volume (Table 2-4). The median chlorophyll *a* patterns obtained by compiling data from numerous sources is consistent with values we obtained in our surveys. Median dry season chlorophyll *a* in the target estuaries appears to be related to freshwater inflow normalized by estuary volume. Estuaries with the low freshwater inflow normalized to volume have relatively high chlorophyll *a* with median values ranging from 3.3 - 4.9 μg l^{-1} for these tide-dominated estuaries (Yaquina, Coos, Netarts and Willapa). Estuaries with relatively high freshwater inflow normalized to volume (e.g., Alsea, Salmon and Umpqua) have low median chlorophyll *a* (1-2 μg l^{-1}). The tide-dominated estuaries may have relatively high chlorophyll *a* compared to the river-dominated system because they have less flushing due to low freshwater inflow and as a result phytoplankton growing inside the estuary reach higher levels. An alternate explanation may be that the higher chlorophyll *a* levels in tide-dominated estuaries are due to import of chlorophyll *a* from the coastal ocean. Newton and Horner (2003) demonstrated that high productivity phytoplankton blooms are imported into Willapa Estuary from the coastal ocean and have species that are of oceanic origin; however, moderate blooms also occur within the estuary which are combination of phytoplankton species of oceanic and estuarine origin. This suggests that the high chlorophyll *a* levels in the ocean-dominated estuaries maybe due to a combination of the two explanations.

Table 4-4. Median dry season chlorophyll *a* and total suspended solids (TSS) for each estuary and its ranking. In the ranking, 1 represents the estuary with the highest median value and 7 represents the estuary with the lowest.

ESTUARY	MEDIAN CHL *a* ($\mu g\ l^{-1}$)	RANKING BY CHL *a*	MEDIAN TSS ($mg\ l^{-1}$)	RANKING BY TSS
Alsea	1.0	6	7.4	3
Coos	2.4	3	13.8	1
Nestucca	2.8	2	4.9	6
Salmon	1.6	5	5.1	5
Tillamook	1.7	4	3.7	4
Umpqua	0.9	7	1.6	7
Yaquina	5.6	1	9.7	2

Table 4-5. Median dry season $NO_3^-+NO_2^-$, NH_4^+, and PO_4^{3-} for each estuary and its ranking.

ESTUARY	MEDIAN $NO_3^-+NO_2^-$ (μM)	RANKING BY $NO_3^-+NO_2^-$	MEDIAN NH_4^+ (μM)	RANKING BY NH_4^+	MEDIAN PO_4^{3-} (μM)	RANKING BY PO_4^{3-}
Alsea	30.9	2	4.6	1	0.7	5
Coos	6.1	5	3.2	2	0.9	1
Nestucca	10.4	3	2.2	3	0.7	4
Salmon	3.3	6	1.4	6	0.5	6
Tillamook	34.0	1	2.0	4	0.8	3
Umpqua	1.9	7	0.9	7	0.3	7
Yaquina	6.1	4	1.7	5	0.8	2

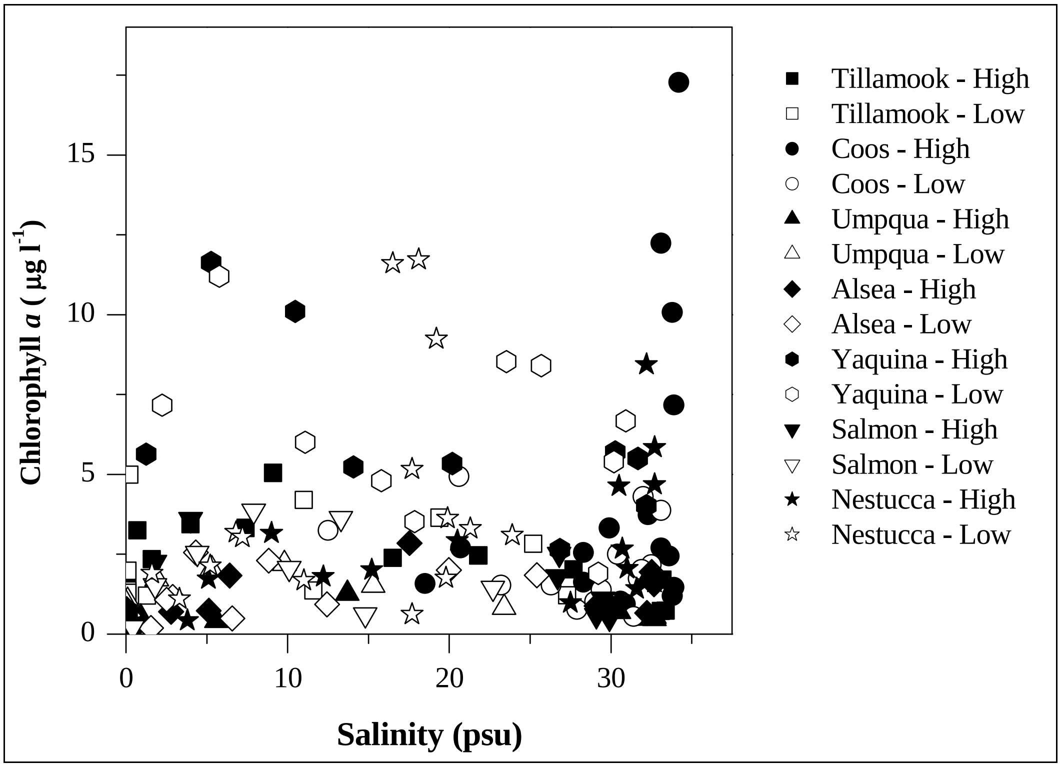

Figure 4-5. Dry season chlorophyll *a* versus salinity for all estuaries with filled symbols representing high tide cruises and hollow symbols representing low tide cruises.

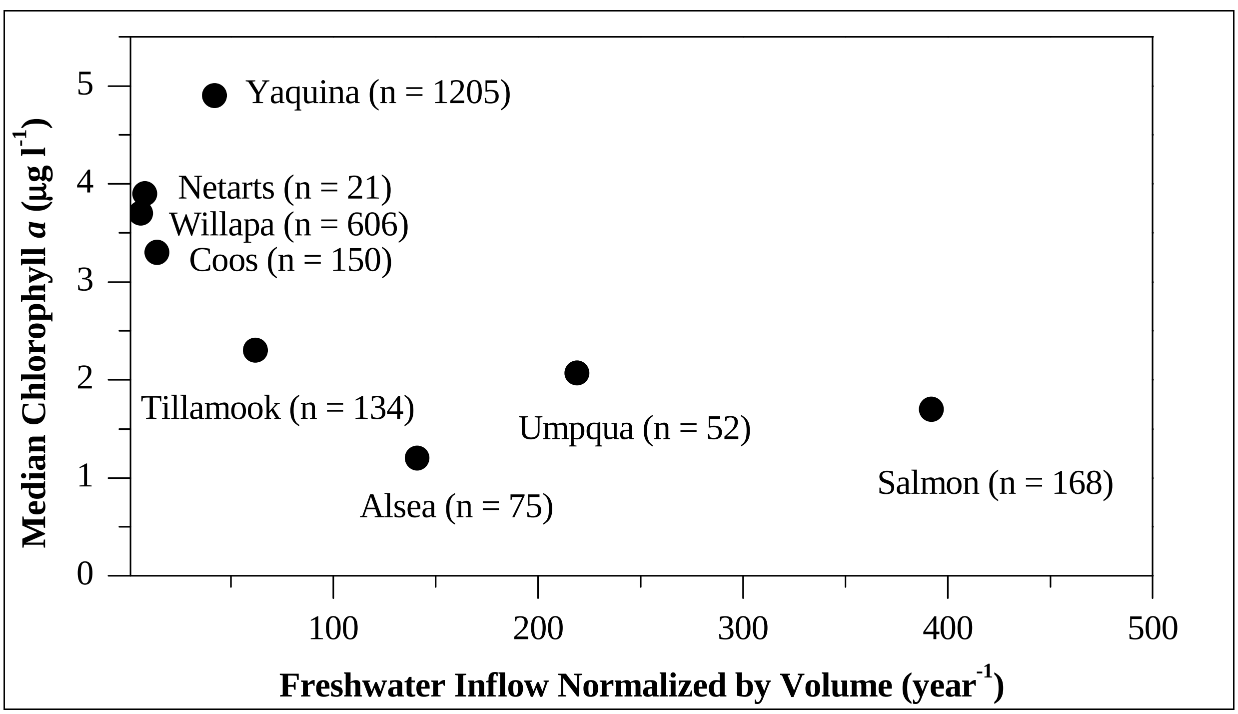

Figure 4-6. Median dry season chlorophyll *a* versus freshwater inflow normalized by estuary volume for eight PNW estuaries. The chlorophyll *a* includes data from this study, NCA, Washington Department of Ecology, and Oregon DEQ.

4.4.2 Dry Season Nutrient Patterns

The Tillamook Estuary had the highest median dry season concentration of NO_3^-+NO_2^-, while the Umpqua River Estuary had the lowest (average of all stations, low and high tide cruises; Table 4-5). The Alsea Estuary had similar median NO_3^-+ NO_2^- (rank = 2) as Tillamook, but this was due to the high freshwater inflow event prior to the cruises. Additional sampling in the Alsea Estuary suggested that the NO_3^-+NO_2^- and NH_4^+ levels were anomalously high during the dry season cruises. Monthly cruises conducted in the Alsea Estuary from May to September 2004 had a median NO_3^-+NO_2^- value of 7.5 μM and an NH_4^+ value of 1.7 μM (n = 30). The median values of the nutrients in the Yaquina Estuary were similar to medians calculated from weekly cruises (with 6 stations) conducted from May 5 - September 27, 2004 (median NO_3^-+NO_2^- = 10.9 μM, NH_4^+ = 2.7 μM, and PO_4^{3-} = 0.9 μM). In the Tillamook Estuary about 44% of the stations were located in low salinity regions of the estuary (with salinity < 5 psu), and these low salinity stations had high NO_3^-+NO_2^- levels (median = 40.3 μM). Excluding low salinity stations (< 5 psu), the median concentration of NO_3^-+NO_2^- in Tillamook was about 21 μM. We recalculated the median NO_3^-+NO_2^- only using stations with salinity ≥ 5 psu and the ranking of the estuaries by median NO_3^-+NO_2^- was Alsea, Tillamook, Nestucca, Coos, Yaquina, Umpqua River, and Salmon River (highest to lowest).

The Salmon River Estuary had the lowest NO_3^-+NO_2^- probably due to the low nutrient conditions in the coastal ocean during the sampling of this estuary. The relatively high NO_3^-+NO_2^- levels (calculated from stations with salinities > 5 psu) in the Tillamook Estuary were probably associated with strong upwelling conditions prior to the cruises (see Section 4.3) combined with relatively high watershed inputs. Colbert (2004) found median NO_3^-+NO_2^- levels of ~ 8 μM in Tillamook Estuary (using data from May – September 1998 and 1999 with salinities > 5 psu, n = 46), suggesting there is substantial interannual variability. Historical data in Umpqua Estuary from the Oregon Department of Environmental Quality confirmed the low median dry season nutrients in this estuary (median NO_3^-+NO_2^- = 1.6 μM and PO_4^{3-} = 0.3 μM , n = 15).

To examine differences in riverine nitrogen inputs among the target estuaries during the dry season, we calculated the median dissolved organic nitrogen (DIN) using only stations with salinity < 5 psu. Unfortunately, for the Coos Estuary our sampling did not extend far enough upriver to examine riverine nitrogen input. Alsea, Nestucca, Tillamook and Yaquina estuaries had similar DIN levels at the low salinity stations (< 5 psu) with median values of about 40 μM, while the Umpqua River Estuary had the lowest (2.4 μM), and the Salmon River was intermediate with median values of about 19 μM. Dry season riverine DIN input appeared to follow the trends in deciduous cover in the watersheds (Table 3-3), with Alsea, Nestucca, Tillamook and Yaquina watersheds having the highest deciduous cover ranging from 3.4 to 5.9%, and the Umpqua watershed having the lowest 0.49%.

During the dry season, there was a significant relationship between PO_4^{3-} and salinity (Figure 4-7) which demonstrates that the ocean was the dominant PO_4^{3-} source for the target estuaries. A substantial amount of the variability in the Figure 4-7 is due to temporal variations in ocean conditions. In the PO_4^{3-} versus salinity plots for individual cruises, the variability was reduced

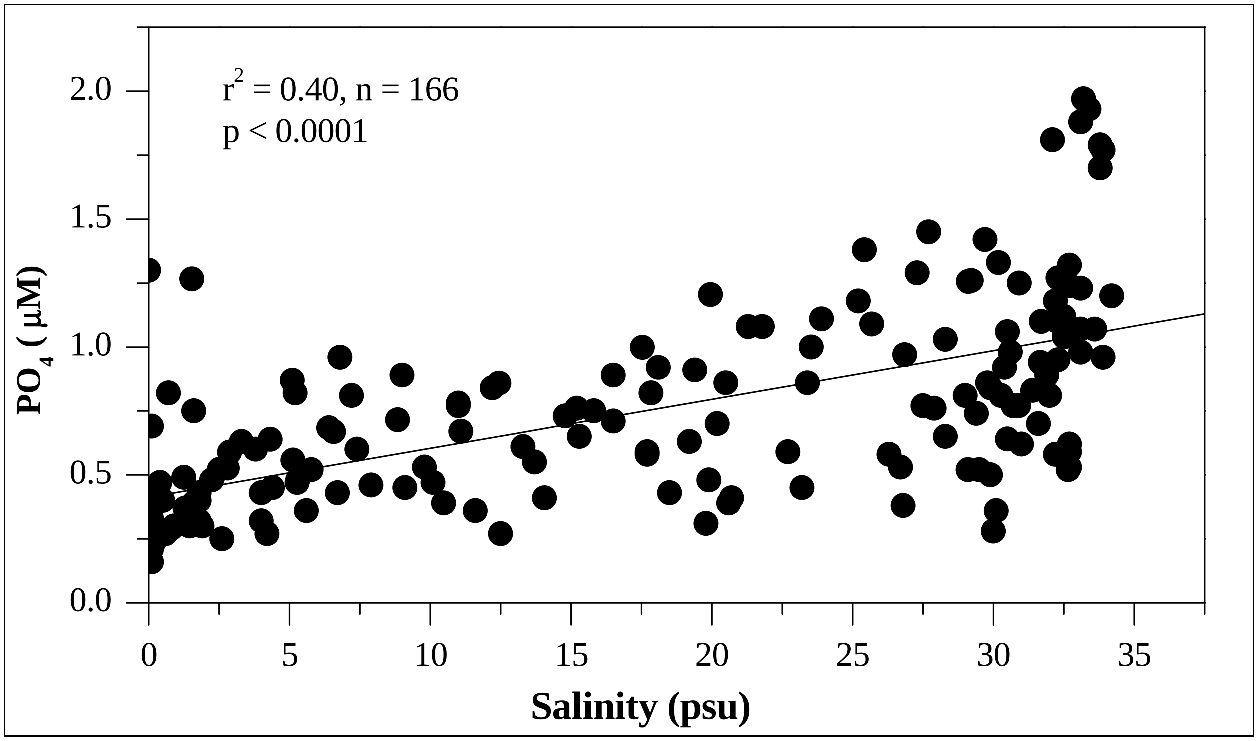

Figure 4-7. Dry season phosphate versus salinity for all classification water quality stations (all estuaries, both years, high and low tide cruises).

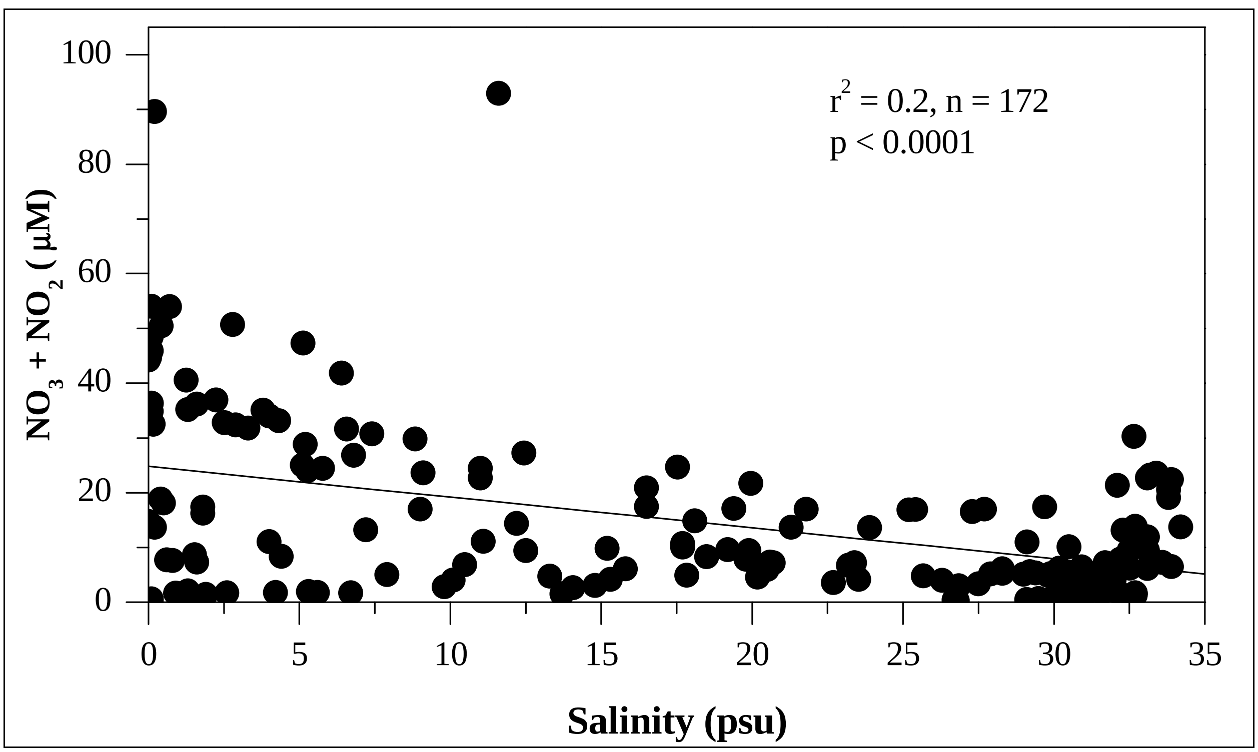

Figure 4-8. Dry season nitrate+nitrite versus salinity for all classification water quality stations (all estuaries, both years, high and low tide cruises).

and the r^2 was as high as 0.97 for some cruises. There was evidence of a PO_4^{3-} source at low salinities in all the target estuaries. Plotting NO_3^-+NO_2^- versus salinity (Figure 4-8) demonstrates that the estuaries receive NO_3^-+NO_2^- from both the river and the ocean and that the riverine concentrations are higher than those associated with the ocean. EMAP data for Oregon estuaries exhibited a significant correlation between PO_4^{3-} and salinity, but no relationship between dissolved inorganic nitrogen and salinity (Nelson and Brown, 2008). There were two anomalously high (~90 μM) NO_3^-+NO_2^- datapoints that occurred at low salinities, which correspond to two sloughs in Tillamook Estuary. These two datapoints will be discussed in Section 4.5.

4.4.3 Dry Season Salinity Fluctuations

Salinity conditions near the mouth of the seven target estuaries were compared to assess the importance of riverine versus marine dominance during the dry season. For this analysis, we examined the data from the short-term datasondes for the station nearest the mouth in each estuary. Since we lost the datasonde deployed near the mouth of the Nestucca Estuary during mid August 2004, we used data collected in late August 2006. To confirm that this was an adequate substitution, we compared the high and low tide salinity data from the cruises of the outermost station (Station C10) to the data collected in 2006. The high and low tide data from Station C10 were comparable to the high and low tide values from the datasondes deployed in 2006 (Station Y2) suggesting that this is an appropriate substitution. The datasondes were located at varying distances from the estuary mouths. The datasonde deployed at Station Y2 in the Nestucca Estuary was located at the mouth. The ones deployed in the Umpqua River and Salmon River estuaries were about 1 km from the mouth. The datasonde deployed at Station Y1 in Tillamook Estuary was about 2 km from the estuary mouth, while the ones deployed in Yaquina and Alsea estuaries were about 3 km from the mouth. In the Coos Estuary, we had a Mini-CTD deployed near the mouth; however, it malfunctioned, so our datasonde closest to the mouth was 6 km from the inlet. Additional data were provided from OIMB, which is about 1 km from the estuary mouth.

In the Yaquina Estuary, the salinity varied about 5 psu over a tidal cycle, even though this station is located about 3 km from the estuary mouth (Figure 4-9). In the Alsea Estuary, the salinity varied about 24 psu over a tidal cycle and this datasonde was deployed at a similar distance from the estuary mouth as the one in the Yaquina. This difference between the salinity variations between Yaquina and Alsea was probably due to differences in freshwater inflow. The dry season freshwater inflow to the Alsea Estuary is three times that into the Yaquina Estuary (Table 3-6). The influence of the high freshwater inflow event on September 19, 2004 was evident in the decrease in low tide salinities near the estuary mouth. Salinity data from Station C1 near the mouth of the Alsea Estuary differed by 7 psu between the high and low tide cruises, which was less than that at Station Y1 due to proximity to the estuary mouth. Even though the Salmon River datasonde was located close to the estuary mouth the salinity varied about 23 psu over a tidal cycle. These salinity variations were comparable to those in Alsea; however, the Alsea datasonde was deployed about 2 km further upstream than the one in Salmon River estuary. The salinity variations near the mouth of the Nestucca Estuary varied about 13 psu over a tidal cycle (Station Y2). Salinities near the mouth of the Umpqua River Estuary varied about 19 psu over a tidal cycle. Using the datasonde near the mouth of Tillamook (Station Y1), the salinity varied about 14.6 psu over a tidal cycle. When plotted on the same axis, the salinity variations at Tillamook were comparable to those at Nestucca (but the Nestucca datasonde was located closer to the mouth of the estuary). The salinity variations near the mouth of Coos Estuary are similar to (or slightly less than) those in the Yaquina Estuary.

Due to variability in the distance of the short-term datasondes from estuary mouths, we examined the salinity differences between the high and low tide of the station closest to the estuary mouth to rank the estuaries in terms of riverine dominance. Salinity variations at the mouth appear to be related to normalized freshwater inflows presented in Table 2-4 (Figure 4-11). Based on salinity variations and normalized freshwater inflow, the ranking of estuaries in terms of riverine dominance would be Umpqua (most riverine), Nestucca, Alsea, Salmon, Tillamook, Yaquina and Coos (most marine).

4.5 Comparison of Water Quality Data to EMAP Criteria

Dry season water quality data were compared to EPA's Environmental Monitoring Assessment Program (EMAP) West Coast criteria (Table 4-6). Data collected in the seven target estuaries were consistent with the EMAP data discussed in Section 1.3; however, since our water quality stations were not probabilistically sampled, we cannot express our results as percent of estuarine area. There was no incidence of low dissolved oxygen (< 5 mg l^{-1}) at any of the stations that we sampled. All stations (both low and high tide cruises of all estuaries) had dissolved inorganic phosphorous levels in the fair range. As discussed previously, this was primarily resulting from oceanic input of PO_4^{3-} to the estuaries (Figure 4-7). The majority (85%) of the stations sampled (all estuaries) had chlorophyll *a* levels in the good range, with the remainder in the fair range. The chlorophyll *a* was in the good range (< 5 µg l^{-1}) for all stations and both sampling dates in the Alsea, Salmon River, and Umpqua River estuaries. In the Coos, Nestucca, and Tillamook estuaries most of the stations had chlorophyll *a* in the good range, but there were a few instances of chlorophyll *a* in the fair range (Coos had 4 samples, Nestucca had 6 samples, and Tillamook had 1 sample in the fair range). In Yaquina Estuary, most of the chlorophyll *a* samples were in the fair range (14 out of 20). Interestingly, the occurrences of chlorophyll *a* in the fair category occurred near the mouth in the Coos and Nestucca estuaries suggesting import of high chlorophyll *a* water from the coastal ocean. In contrast, the relatively high chlorophyll *a* concentrations in the Yaquina Estuary extended from the mouth to the tidal fresh portion of the estuary.

Most (85%) of the stations sampled had dissolved inorganic nitrogen (DIN) levels in the good category (with 14% in the fair category and 1% in the poor category). All samples in the Coos, Salmon River and Umpqua River estuaries had DIN concentrations in the good category. All of the occurrences of DIN in the fair category occurred in the riverine portions of the estuaries with salinities less than 6.5 psu. The only estuary that had DIN concentrations in the poor range was Tillamook (Stations C4 & C12 during low tide cruise which are located in Hathaway and Dougherty sloughs with salinities of 0.2 and 11.6 psu). Unlike the anomalously high NO_3^- conditions in the Alsea Estuary during the classification cruises, there was not high river inflow to the Tillamook Estuary prior to this cruise. There was additional nutrient sampling in the vicinity of C4 and C12 by the Oregon Department of Environmental Quality during 1997 and 1998, which had similar high dry season DIN (peak values of DIN of about 100 µM compared to

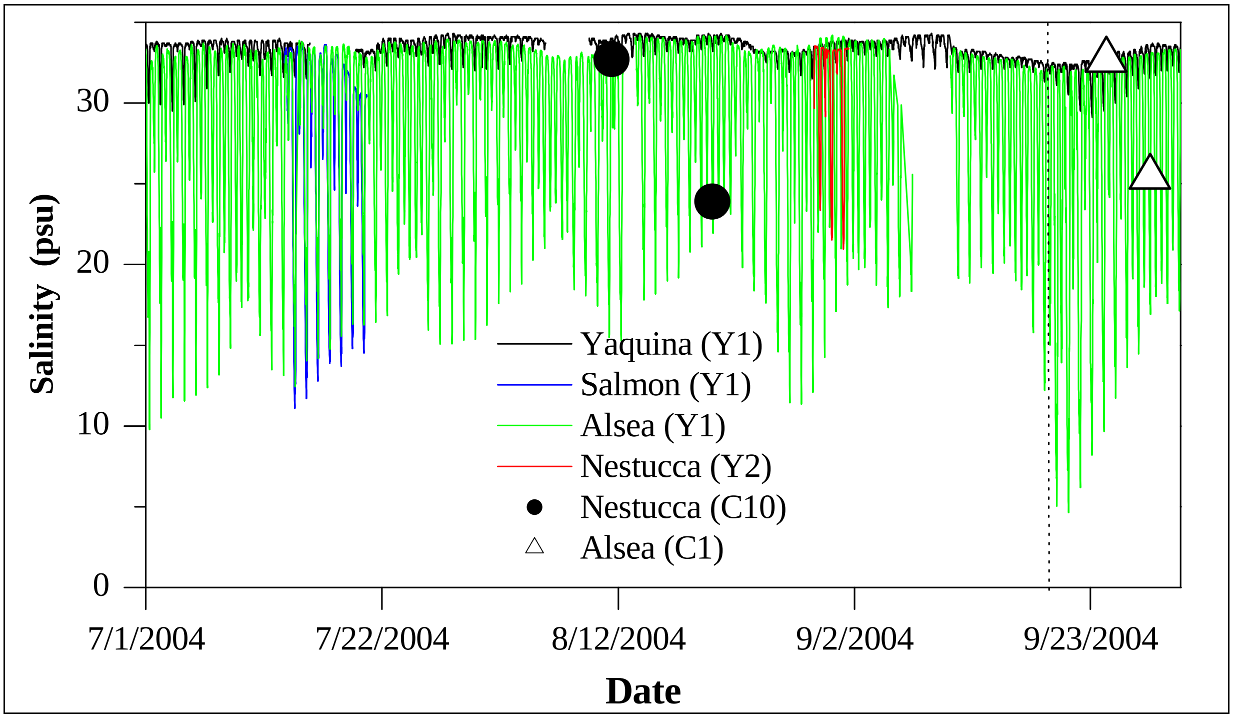

Figure 4-9. Dry season salinities near the mouths of estuaries sampled in 2004 and data collected in Nestucca (Y2) during 2006 (shifted by 2 years) to fill data gap from lost instrument. The dashed line indicates the high freshwater inflow event on September 19, 2004.

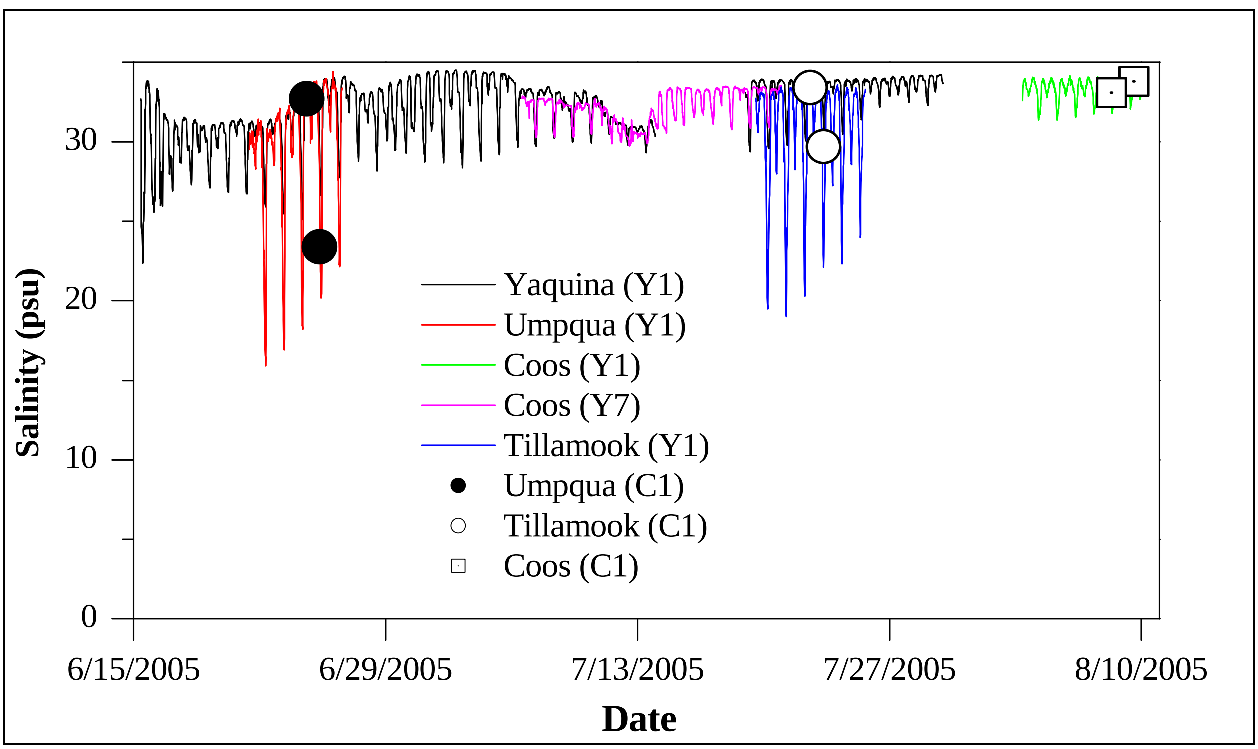

Figure 4-10. Dry season salinities near the mouths of estuaries sampled in 2005. Lines indicate short-term datasonde deployments, while symbols represent data from high and low tide cruises for station nearest the mouth of the estuary with station name present in parentheses of legend.

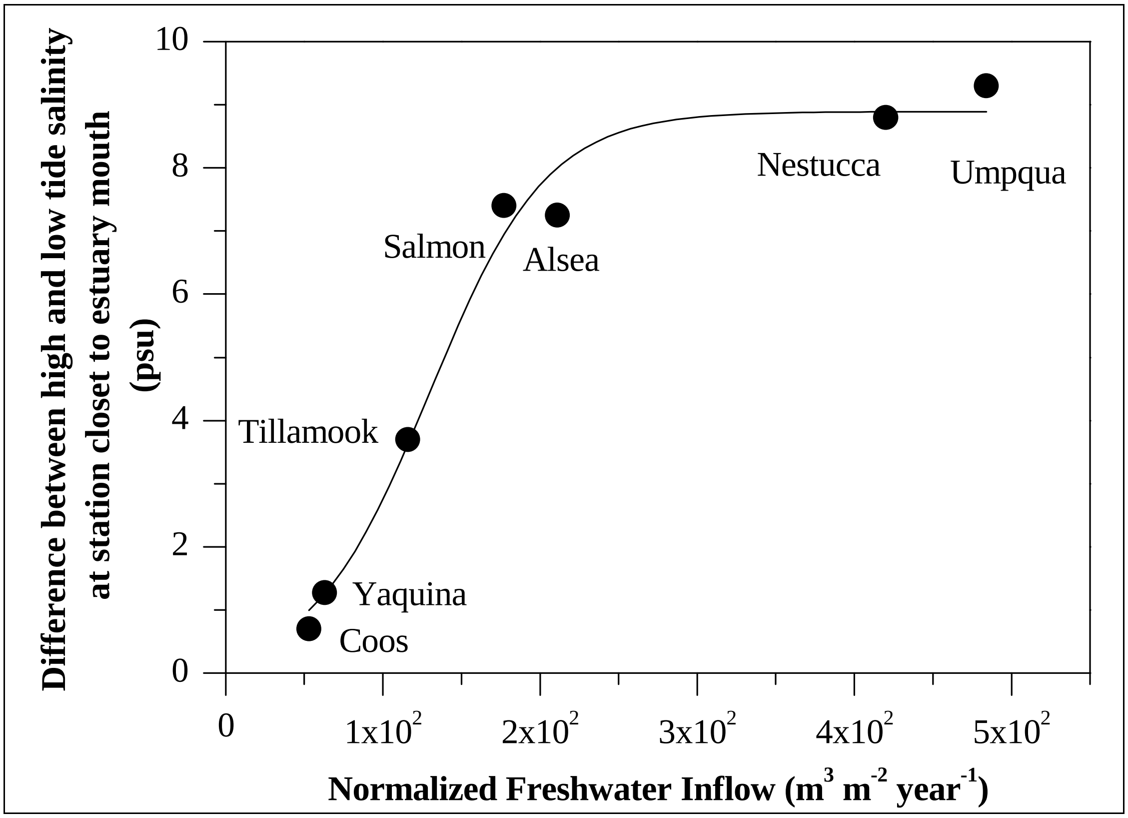

Figure 4-11. Difference between high and low tide dry season cruise salinity at station closest to estuary mouth versus area-normalized freshwater inflow.

92-96 µM observed during our low tide cruise). These two stations (C4 & C12) also had the highest wet season DIN levels observed in the target estuaries (see Section 4.2).

This suggests that Hathaway and Dougherty sloughs might be receiving high DIN inputs from upland sources. Agricultural activities occur in this watershed adjacent to the estuary, including confined animal feeding operations. The DIN levels in these sloughs were not consistent with DIN levels for other nearby Oregon streams during the month of July. The median DIN in the tidal fresh portion of the Yaquina River during July was 18 µM and the maximum concentration measured during July was 39 µM (unpublished data of C. Brown, collected 2-3 times per week during 2002 and 2003). The median DIN in the tidal fresh portion of the Alsea River during July of 2004 was 4 µM (with a maximum of ~ 6 µM, n = 5, unpublished data of C. Brown).

The degree of stratification was examined by calculating the difference between bottom and surface salinity with the surface being 0.5 m from the surface and the bottom being 0.5 m above the bottom for most stations. Strong stratification was defined as > 2 psu difference between surface and bottom readings. Due to the shallow water depth during the low tide cruises in the Salmon River and Nestucca estuaries, only one data point at mid-depth was collected; therefore, stratification could not be determined for these cruises.

Table 4-6. West Coast criteria for water quality parameters (U.S. EPA, 2004b).

PARAMETER	RANKING		
	GOOD	FAIR	POOR
Dissolved Oxygen	> 5 mg l^{-1}	2-5 mg l^{-1}	< 2 mg l^{-1}
Chlorophyll *a*	< 5 µg l^{-1}	5 -20 µg l^{-1}	> 20 µg l^{-1}
Dissolved inorganic nitrogen	< 0.5 mg l^{-1}	0.5-1.0 mg l^{-1}	> 1 mg l^{-1}
Dissolved inorganic phosphorous	< 0.01 mg l^{-1}	0.01- 0.1 mg l^{-1}	> 0.1 mg l^{-1}

In aggregate (all cruises, all estuaries, all stations) about 29% of the stations sampled exhibited strong stratification. The estuary with the highest number of stations exhibiting strong stratification was the Alsea Estuary (45% of stations). The estuaries with the lowest number of stations exhibiting strong stratification were the Tillamook and Coos estuaries (13 and 19%, respectively). The high degree of stratification in Alsea was probably related to the high freshwater inflow event discussed earlier in this chapter. For estuaries where we had stratification data for low and high tide cruises (all estuaries except for Salmon River and Nestucca), strong stratification occurred more during the high tide sampling compared to the low tide (19 occurrences during high tide versus 8 during low tide cruises). Of the stations that showed strong stratification, the majority were upriver in the low salinity region (<15 psu). This pattern was particularly evident in the Salmon River, Nestucca, and Alsea estuaries. The Yaquina and Coos estuaries exhibited strong stratification in the ocean-dominated regions.

4.6 Synthesis

During the wet season, DIN levels were highest in the Yaquina and lowest in the Umpqua. Variations in the wet season DIN appear to be related to deciduous cover. This suggests that variations in red alder are resulting in differences in riverine DIN input to the estuaries. Dry season water quality conditions in PNW estuaries were dependent upon the ocean conditions during the surveys. Flood-tide water temperature as well as nutrient versus salinity relationships generated using data from the inner Oregon shelf were useful for confirming that ocean conditions were relatively uniform over the geographic extent of the estuaries we sampled and for assessing the ocean conditions during the dry season water quality cruises. The results from the Alsea Estuary demonstrated that when assessing nutrient levels of the estuaries it is important to examine freshwater inflow immediately prior to sampling events (even during the dry season) to ensure that there was not a strong freshwater inflow event that might result in elevated DIN levels. Dry season nutrient data from the cruises demonstrated that the estuaries received PO_4^{3-} mainly from the coastal ocean, while both the ocean and rivers were DIN sources. In the seven target estuaries, there were no occurrences of low dissolved oxygen, and chlorophyll *a* levels were relatively low. The tide-dominated estuaries appear to have higher chlorophyll *a* levels than the river-dominated estuaries. The dry season PO_4^{3-} conditions were in the EMAP fair category but this is due to the input of high PO_4^{3-} water from the coastal ocean. Generally, most of the DIN concentrations observed were in the good category (85%). There were two occurrences of poor DIN conditions in two of the sloughs of Tillamook Estuary. Umpqua River Estuary had the lowest NO_3^-+NO_2^- and PO_4^{3-} of the seven target estuaries. There are several potential explanations for the low nutrients in this estuary. Of the seven target estuaries, the

Umpqua has the least amount of red alder in the watershed as evident by the lowest % Deciduous and % Mixed and the highest % Evergreen (Table 3-3). Red alder is the primary deciduous tree in the Oregon Coast Range, and NO_3^- in streams has been related to its presence (see Section 3.7). In addition, Umpqua has the highest freshwater inflow of the seven target estuaries (even during the dry season, Table 3-6) and as a result is river dominated, which may result in less oceanic nitrogen and phosphorous input to the estuary.

CHAPTER 5:
ZONATION OF OCEAN AND RIVER DOMINANCE IN SEVEN TARGET ESTUARIES

Cheryl A. Brown and James E. Kaldy III

Key Findings

- **Estuaries were divided into oceanic and riverine segments in terms of nitrogen sources during the dry season.**
- **Zonation was based upon natural abundance of nitrogen stable isotopes of green macroalgae and transport modeling.**
- **Analyses suggest that oceanic and riverine inputs are the dominant nitrogen sources for the target estuaries.**
- **Coos Estuary had the highest isotope ratios with multiple stations having isotope ratios elevated above the oceanic end member, suggesting that wastewater inputs are important in this estuary.**
- **Five estuaries (Alsea, Coos, Nestucca, Tillamook, and Yaquina) had >50% of the total estuarine area classified as ocean dominated.**

5.0 Introduction

To assess the susceptibility of estuarine resources to watershed-derived nutrient loading, we estimated the contribution of nitrogen sources to water column dissolved inorganic nitrogen (DIN). Previous research at Yaquina Estuary has demonstrated that during the dry season (May-October), the ocean is the dominant nitrogen source, while during the wet season (November – April) riverine inputs dominate (Brown and Ozretich, 2009; Table 3-9). In addition, most of the riverine nitrogen inputs are believed to be related to the presence of red alder in the watershed (see Section 3.7). Point source input associated with a wastewater treatment facility increases in importance in the up-estuary region, particularly during periods of minimal riverine inputs. The purpose of this chapter is to assess the importance of oceanic versus riverine nitrogen sources in the seven target estuaries.

We divided each target estuary into two segments, ocean dominated and river dominated, based upon the dominant dry season DIN sources. We used multiple types of data and analyses to derive the zonation of the estuaries and then translated the zonation into median salinities for use in estuaries with limited data. For the Yaquina Estuary, we had the most extensive dataset and analysis techniques for development of zonation based upon dry season nitrogen sources. We used a transport model combined with natural abundance stable isotopes ($\delta^{15}N$) of green macroalgae to identify the dominant nitrogen sources within the estuary as a function of time and location. The transport model was validated by comparing predicted isotope ratios (using the transport model to mix isotopic end members) to observed macroalgal isotope ratios at five locations. There were several reasons why attached green macroalgae were used as the bioindicator in this study. In estuaries with high flushing, macroalgae are often the dominant primary producers (Valiela et al., 1997), and their presence is often used as an indicator of eutrophication (Bricker et al., 2003). During the dry season, there are dense blooms of green macroalgae in many estuaries in the region. In addition, the attached nature of the green macroalgae is beneficial in that it integrates the nitrogen sources over time, which is advantageous when there is substantial temporal variability in nitrogen sources and loadings.

Nitrogen stable isotope ratios of green macroalgae are a useful indicator because the macroalgae take up DIN from the water column and incorporate this nitrogen into their tissue. Using the natural abundance isotope ratio of nitrogen, macroalgae can be linked to nitrogen sources with distinct isotopic signatures (McClelland and Valiela, 1998; Costanzo et al., 2001). Fixation of atmospheric nitrogen (including that by N_2-fixing root nodules associated with red alder, see Section 3.7) has $\delta^{15}N$ values slightly less than 0‰. Industrially produced fertilizer has a similar isotope ratio due to the atmospheric source of nitrogen in the Haber Process. Stable isotope signatures are particularly useful when the various sources contributing to a system are isotopically distinct (Figure 5-1). As an example, nitrogen stable isotopes may be used to distinguish fertilizer inputs from animal or human waste, but they are not useful for distinguishing human wastes (septic or wastewater treatment facility) inputs from animal wastes due to the overlap of isotopic signatures. Distinguishing nitrogen sources with overlapping isotopic signature (e.g., human waste from animal wastes) is often accomplished by comparison of nitrogen sources and land use patterns. Previous research has shown that green macroalgae integrate water column conditions that have occurred over a two week interval (Aguiar et al., 2003; Cohen and Fong, 2005). An indicator that integrates water column conditions is desirable due to the variability of coastal ocean nutrient conditions associated with variations in wind stress (see Section 4.3) and variability in water column DIN within estuaries (see Section 4.4.2).

Since it was impractical to develop a transport model for each estuary due to data limitations, we used a conservative transport two end member mixing model to estimate the contribution of riverine and oceanic DIN sources, which is possible due to their different salinity and nitrogen stable isotope signatures. Similarity between observed and predicted isotope ratios suggests that the oceanic and riverine inputs are the primary nitrogen sources to the estuary. If the observed isotope ratio is greater than that predicted based upon conservative mixing of oceanic and riverine nitrogen sources, this suggests that there may be additional nitrogen sources or denitrification may be important. If there is a wastewater treatment facility (WWTF) source in a given estuary, then we inferred the contribution of WWTF sources using Isosource (Phillips and Gregg, 2003), which calculates all feasible combinations of the three sources that can produce the observed isotope ratio at a given location and sampling time. To constrain the solutions, we selected a subset of feasible solutions, which were consistent with the salinity variations at the site. This method was validated for Yaquina Estuary through comparison to the results from the transport model. Once validated, this technique was used to estimate the contribution of DIN sources for estuaries sampled in this classification effort. In addition, salinity data were analyzed for Yaquina Estuary to determine if zonation predicted from the isotope and transport models could be approximated using salinity data. For estuaries where there were limited data available we used salinity data (collected as part of the classification effort as well as historic salinity data) to aid in the zonation of the estuaries.

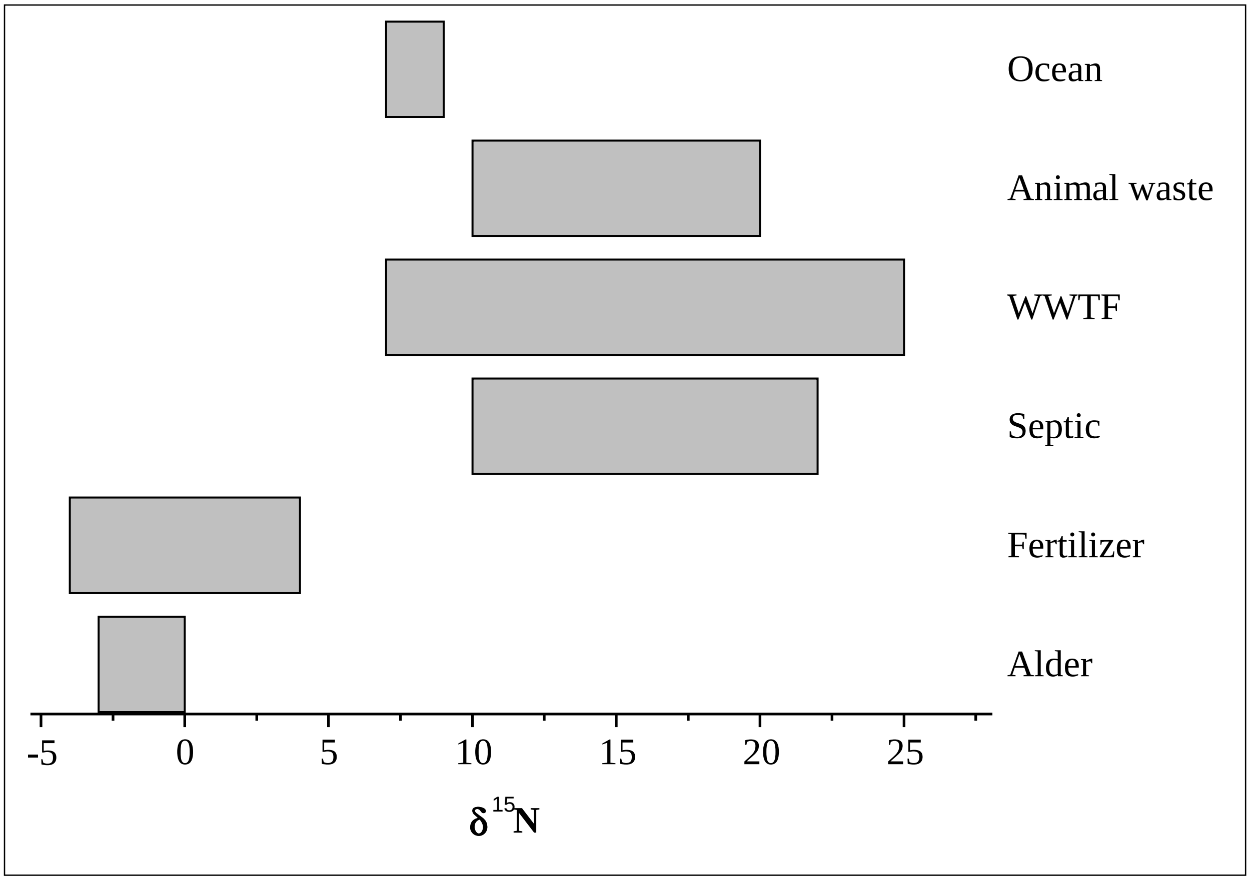

Figure 5-1. Natural abundance nitrogen isotope ratio ($\delta^{15}N$ expressed as ‰) for various nitrogen sources. Sources: ocean represents the $\delta^{15}N$ of green macroalgae from the Oregon coast (Fry et al., 2001, and Kaldy unpublished), animal waste and fertilizer is from Kendall and McDonnell (1998), septic is from Cole et al. (2004), and red alder is from Hobbie et al. (2000), Tjepkema et al. (2000), Cloern et al. (2002).

5.1 Stable Isotope Sampling

In the Yaquina and Alsea estuaries, 5 sites were sampled monthly for the isotope ratio of macroalgal tissue. Sites were located along the salinity gradient from the mouth of the estuary to the region where the system becomes mesohaline. As part of the classification effort, samples were collected for the isotope ratio of green macroalgae at two sites in the Alsea, Nestucca, Salmon River, and Yaquina estuaries during 2004. In 2005, green macroalgae were sampled at 5 sites in the Umpqua River and Coos estuaries and 4 sites in the Tillamook Estuary.

Each site was characterized by the presence of either rock or other hard substrate (e.g. pilings, tree stumps, docks) to support green macroalgae. No attempt was made to identify algae to species. The dominant macroalgae were species from the family Ulvaceae, which includes a variety of genera. At each site, five replicate samples were collected by hand from the top or sides of rocks and pilings. Only thallus material that did not contact mud sediments was collected. Individual samples were rinsed in the field with milli-Q water, transferred to labeled plastic bags and stored on ice. In the laboratory, samples were rinsed with milli-Q water to remove adhering sediments, frozen and lyophilized. Dried samples were then pulverized with a mortar and pestle. Percent nitrogen and the nitrogen isotope ratio ($\delta^{15}N$) of dried tissue were measured using a Finnigan Mat Delta Plus XP isotope ratio mass spectrometer. All isotope ratios are presented in standard δ notation.

5.2 Yaquina Estuary

5.2.1 Transport Model

A two-dimensional, laterally averaged hydrodynamic and water quality model (Cole and Wells, 2000) was used to simulate the transport of riverine, oceanic and WWTF effluent DIN sources. The transport model incorporates the temporal variability in nitrogen loading, the location of the nitrogen inputs, and the time that each nitrogen source spends in the estuary. In the model simulations, Yaquina Estuary was represented by 325 longitudinal segments spaced approximately 100 m apart with each longitudinal segment having 1-m vertical layers. The model domain extended about 37 km from the tidal fresh portion of the estuary at Elk City, Oregon to the mouth of the estuary (Figure 5-2). Model simulations were performed for 2003 and 2004 and included tidal and wind forcing, as well as freshwater inflow. Parameters simulated included water surface elevation, salinity, water temperature, and dissolved inorganic nitrogen. Each nitrogen source, riverine (N_R), oceanic (N_O), and WWTF effluent (N_W), was modeled as a separate component. The nitrogen sources were modeled as

$$\frac{dN}{dt} = transport - \mu N \tag{5.1}$$

where N is the DIN source and μ is a loss/uptake rate. The same value of μ was used for all three nitrogen sources and the value of μ was determined by fitting total modeled DIN (N_O+N_R+N_W) to observations of DIN within the estuary. The best fit to observations was found with μ = 0.1 d^{-1}. Simulations were also performed with no uptake (μ = 0) which is equivalent to conservative transport of the sources.

The results from the transport model were used to mix the three nitrogen sources using the following equation

$$\delta_M = f_R\delta_R + f_O\delta_O + f_W\delta_W \tag{5.2}$$

$$f_R + f_W + f_O = 1 \tag{5.3}$$

where f_R, f_W, and f_O are the fractions of riverine, WWTF, and oceanic DIN, respectively, and δ_R, δ_W, and δ_O are the isotope ratio of end members for riverine, WWTF effluent, and oceanic sources, respectively. The mixture of the three end members (δ_M) is assumed to be that which macroalgae utilizes. Estimates of the oceanic and riverine end members were obtained by examination of the observed isotope ratios at Stations N1 and N5. The riverine end member (δ_R = 0 to +2 ‰) was estimated from the wet season isotope ratios at Station N5, while the oceanic end member (δ_O = +8 to +9 ‰) was estimated from dry season isotope ratios at Station N1. The initial estimate for the WWTF end member (δ_W = +15 to +22 ‰) was determined from the literature (Jones et al., 2003). To arrive at the final end member isotope ratios, model

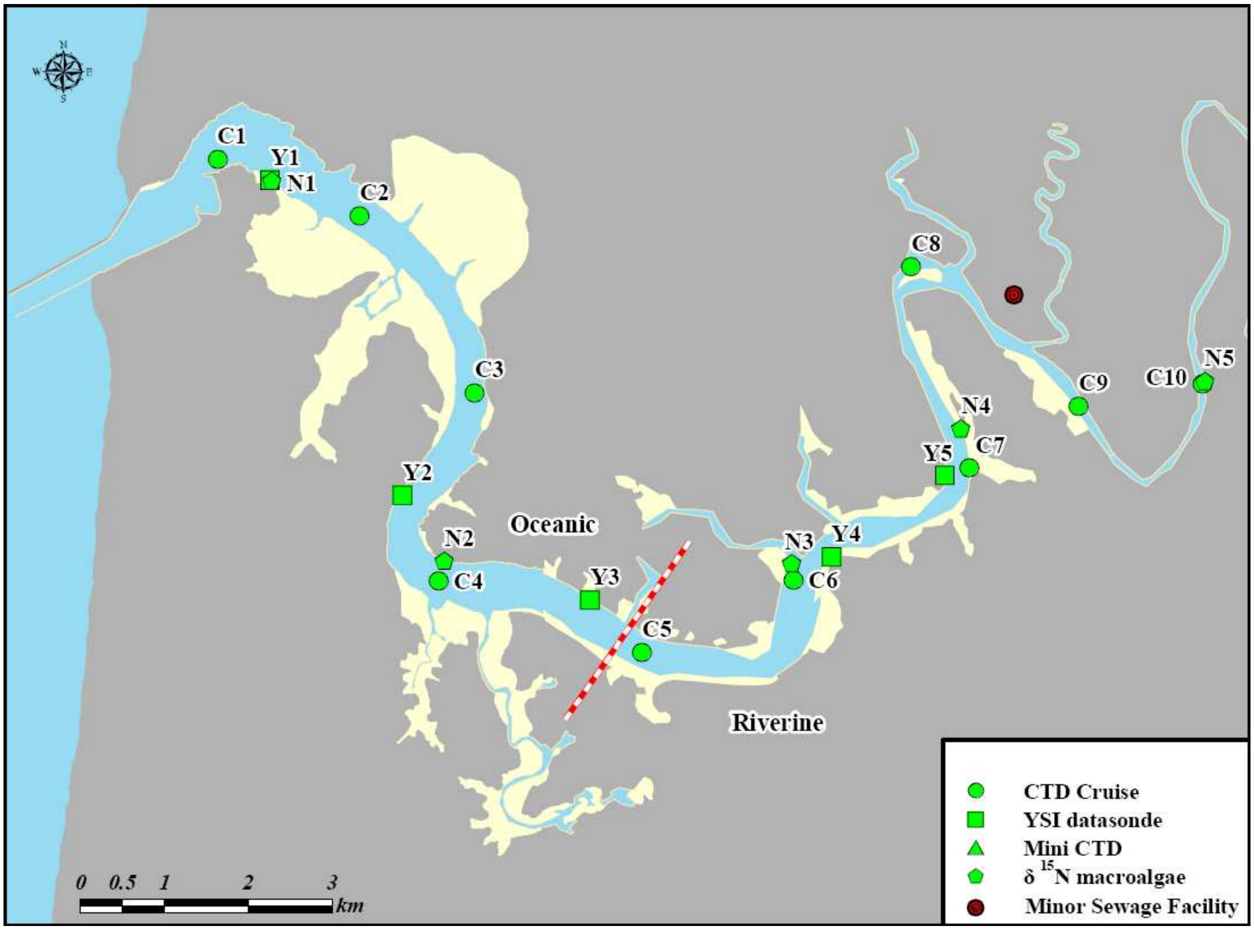

Figure 5-2. Location map for Yaquina Estuary showing the locations of datasondes, water quality stations, isotope samples, and WWTF. The red dashed line shows boundary between oceanic and riverine segments.

simulations were performed varying each end member over the range estimated from the data and literature. The final isotope ratio of end members for the three sources (δ_R=+2 ‰, δ_W=+20 ‰, and δ_O=+8.4 ‰) was determined from the best fit (minimum root mean square error, RMSE) between predicted and observed isotope ratio at the five stations N1-N5 during 2003 and 2004. The final oceanic end member selected is consistent with marine end members for the west coast of the U.S. (Fry et al., 2001), while the riverine end member is consistent with the isotope ratio expected for nitrogen associated with red alder (leaf tissue ranges between -3 and -0.5‰; Hobbie et al., 2000; Tjepkema et al., 2000; Cloern et al., 2002).

The δ^{15}N of green macroalgae collected from Yaquina Estuary (locations shown in Figure 5-2) during 2003 and 2004 exhibited strong seasonal patterns (Figure 5-3) associated with shifting nitrogen sources. During winter conditions with large freshwater inputs, there is a large degree of isotopic separation (Δ = 4‰) between samples collected near the ocean (N1) and near the riverine end-member (Station N4 or N5). This isotopic difference suggests that algae from these sites are utilizing different nitrogen sources. During the dry season, as river flow decreases and ocean upwelling becomes more pronounced, the isotope ratio of green macroalgae collected from the end-member stations becomes isotopically indistinguishable, suggesting that macroalgae are utilizing one nitrogen source. In some months during the dry season, the data suggest that a third nitrogen source, the WWTF, can become locally important (Figure 5-3). Data from 2004 also exhibit this same general pattern.

The model (Equations 5.1-5.3) reproduces much of the observed temporal and spatial variability observed in the macroalgae isotope dataset. Comparisons between simulated and observed isotope ratios at Stations N2 and N3 are presented in Figures 5-4 and 5-5. The RMSE between observed and predicted isotope ratios calculated for Stations N1 - N5 for each year ranges from 0.8 ‰ to 2.0 ‰ (with an average value of 1.3 ‰). This agreement between simulated and observed isotope ratios suggests that the model was correctly simulating the transport and mixing of the three nitrogen sources. Figure 5-6 shows the monthly average contribution of each of the three nitrogen sources (f_R, f_W, and f_O) at Stations N1-N5 and adjacent to the discharge point of the WWTF (region of maximum contribution of WWTF source).

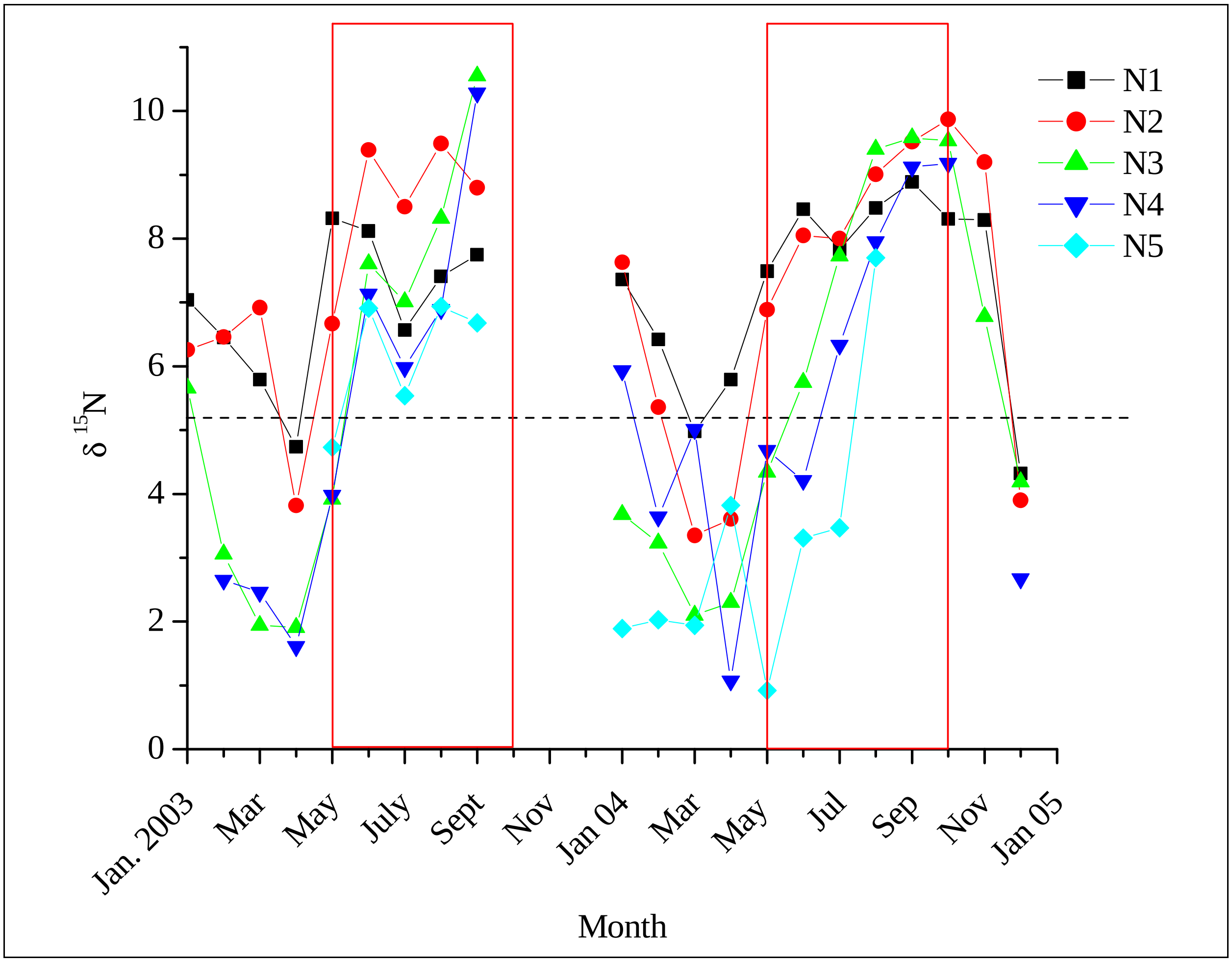

Figure 5-3. Macroalgae isotope data at five stations (N1-N5) for Yaquina Estuary with red boxes indicating dry seasons. The dashed line indicates $\delta^{15}N$ = 5.2 ‰ which is the demarcation between riverine and oceanic dominance, assuming two end member mixing of oceanic and riverine sources.

There was interannual variability in the boundary between the oceanic and riverine segments. The exact location of this boundary varies with ocean conditions (e.g., El Niño, La Niña conditions) as well as freshwater inflow. Using the modeled contributions integrated over May – September, we estimated the position of this boundary during 2003 and 2004. During 2003 the freshwater inflow was less than in 2004 and as a result the oceanic DIN reached further upstream with the boundary between the oceanic and riverine segments being located between Station N3 (which received 57% of DIN from ocean) and N4 (which received 44% of DIN from ocean). During 2004, the boundary was located slightly upstream of Y3 (Y3 received 53% of DIN from the ocean, while N3 received 32% from the ocean). In addition, during the dry season this boundary progressed up estuary as the freshwater inflow declined. This can be seen in simulation results from 2004 (Figure 5-6) which show that Station N1 was ocean dominated (fraction ≥ 0.5) during the entire dry season (May – September), while Station N2 was river dominated during May and ocean dominated from June – September. At Stations N3, N4, and N5 ocean inputs increased in importance from May – August, but didn't dominate. This was also evident in that the salinity at up estuary sites increased from May – September. The results of this analysis may appear to contradict the loading comparisons presented in Table 3-9, which shows that ocean inputs dominate during the dry season. This difference is because the analysis presented in Table 3-9 is comparing total loading to the estuary, while the analysis described in this section examined contributions to nitrogen concentrations at specific locations, which represent the nitrogen sources that would be available to primary producers. Based on comparison of salinities (modeled and observed, Table 5-1) and modeled contribution of sources, we concluded that Yaquina Estuary was ocean dominated in areas where the median salinity was ≥ 25 psu and river dominated in areas where the median salinity < 25 psu (calculated using salinity data from May – September). We used these salinity criteria to aid in the zonation of other estuaries with limited data.

Table 5-1. Summary of median observed salinities for Yaquina Estuary during May - September of 2003 and 2004. NA denotes data not available and * denotes extensive gaps in record.

STATION	MEDIAN SALINITY (psu)	
	2003	2004
Y1	31.9 (surface) 29.1 (bottom*)	32.9 (bottom)
Y2	30.7	NA
Y3	27.3	27.3
Y4	25.5	NA
Y5	18.0	19.3

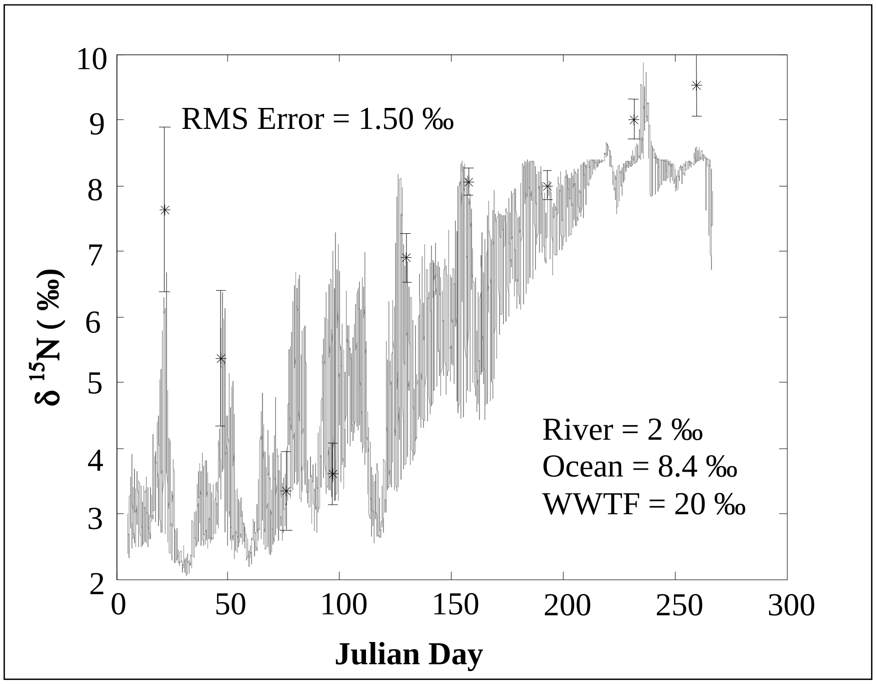

Figure 5-4. Comparison of modeled and observed isotope ratio at Station N2 in Yaquina Estuary during 2004. The grey line is model predicted isotope rate and the symbols and error bars represent mean and standard deviations of observed macroalgae isotope ratio.

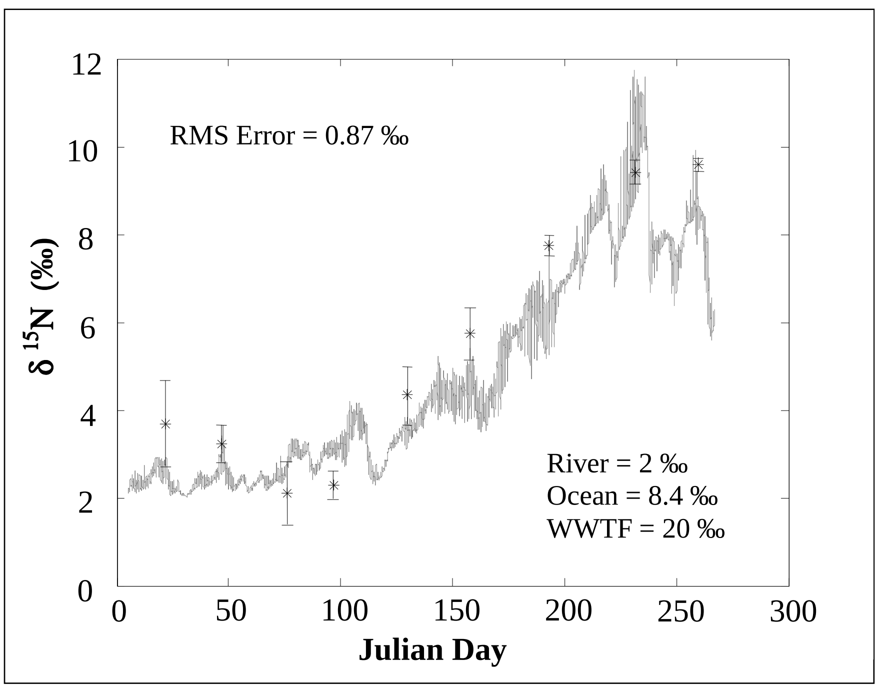

Figure 5-5. Comparison of modeled and observed isotope ratio at Station N3 in Yaquina Estuary during 2004. The grey line is model predicted isotope rate and the symbols and error bars represent mean and standard deviations of observed macroalgae isotope ratio.

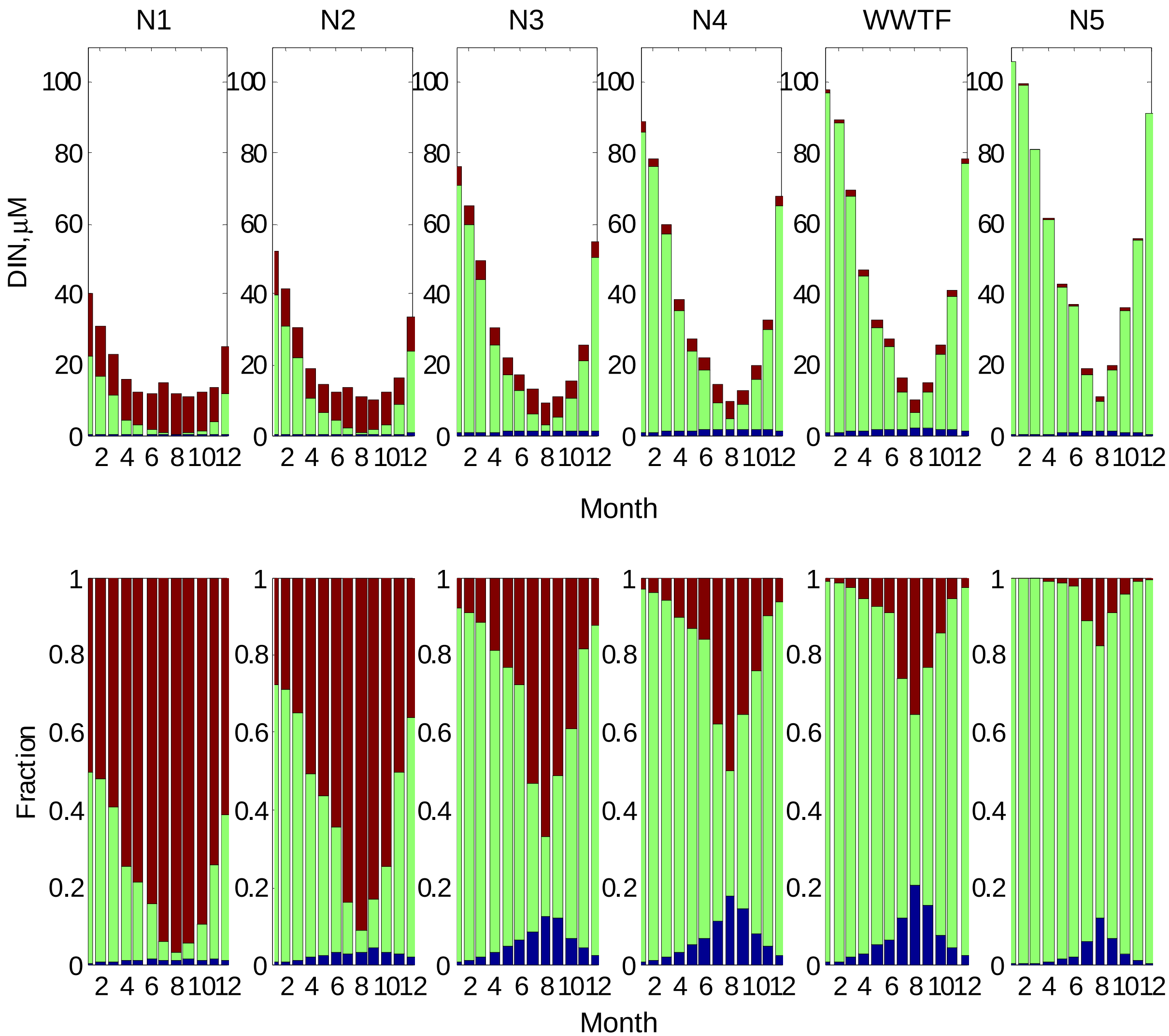

Figure 5-6. Modeled contribution of each source of dissolved inorganic nitrogen in Yaquina Estuary during 2004. The upper panels shows concentrations and the lower panels show fractions of each source with shading indicating riverine (green), oceanic (red), and WWTF effluent (blue) nitrogen sources.

5.2.2 Conservative Mixing Model

Since transport models are not available for the other target estuaries, we used a two-end member conservative mixing model to estimate the dominant nitrogen sources for different portions of each estuary. To test this analysis technique, we performed this analysis for Yaquina Estuary and compared the results to estimates obtained from the transport model (described above). Previous research has suggested that it is difficult to identify nitrogen sources due to the temporal variability of salinity and water nutrients in estuaries (e.g., Cohen and Fong, 2005). In this analysis, we incorporated temporal variability in nitrogen sources by utilizing time series of salinity data at multiple stations within the estuary to mix nitrogen sources. If time-series data were not available in the vicinity of the macroalgae sample collection, then salinity data from the high and low tide cruises from water quality stations in the vicinity were utilized. Briefly, nitrogen concentration (C_M) was modeled as the mixture of riverine and marine end-members using the following equations (Fry, 2002)

$$C_M = fC_R + (1-f)C_O \tag{5.4}$$

where C denotes concentration of DIN, the subscripts R and O indicate river and ocean end-members, respectively, and f represents the fraction of freshwater in each sample calculated from salinity (time series or high and low tide values)

$$f = \frac{S_O - S_M}{S_O - S_R} \tag{5.5}$$

where S_O and S_R are the salinity (psu) at the river and ocean end-members, respectively and S_M is the salinity measured at a specific station. The isotope ratio of mixed estuarine samples was calculated as

$$\delta_M = \frac{fC_R\delta_R + (1-f)C_O\delta_O}{C_M} \tag{5.6}$$

The same end members were used as in Section 5.2.1. Observed isotope ratios were compared to isotope ratios predicted from conservative mixing (Equations 5.4-5.6) using salinity data collected in each region of the estuary. The mean of the predicted isotope ratio (δ_M) was compared to the observations. Similarity between observed and predicted isotope ratios suggests that the oceanic and riverine inputs are the primary nitrogen sources to the estuary. If the observed isotope ratio is greater than that predicted based upon mixing of ocean and riverine nitrogen sources, this suggests that there may be additional nitrogen sources or denitrification may be important. If there is a WWTF source in a given estuary, then we inferred the contribution of WWTF sources using Isosource (Phillips and Gregg, 2003) which calculates all feasible combinations of the three sources that could produce the observed isotope ratio at a given location and sampling time. To constrain the solutions, we selected the subset of feasible solutions which were consistent with salinity variations at the site. For example, if the estimate for the ocean contribution using the conservative mixing model was 50% averaged over the two weeks prior to the macroalgae sampling, then we selected solutions with ocean contributions of 40-60%. Through this technique we can identify the segments that are dominated by oceanic nitrogen sources and identify regions where other nutrient sources (riverine and wastewater treatment effluent) dominate.

To test this technique of interpreting isotope data using temporal variations in salinity data (Equations 5.4-5.6), we compared results from this analysis to the estimate of the contribution of sources from the transport model. For both estimates of the contribution of nitrogen sources we integrated the results (transport model and conservative mixing model) over the 2 weeks prior to the macroalgae sampling date (if salinity data were available). Due to gaps in the salinity time-series or biofouling of the sensor, we were not able to compare all months at all sites. Presented in Table 5-2 is the interval of salinity data used in the conservative mixing model. To maximize the number of data points for comparison, if there were gaps in the salinity time series we used salinity data within about 4 weeks of the sampling date. Salinity data from Stations Y2, Y4 and Y5 were used to predict the isotope ratios at Stations N2, N3, and N4 (Figure 5-2). We assumed that the salinity of the river end member was zero ($S_R = 0$), and used the maximum salinity at Station Y1 during the 2 weeks prior to the isotope sampling as the ocean salinity (S_O). Riverine and oceanic end members were determined from samples collected at Y1 and Elk City (tidal fresh region of the estuary). Presented in Table 5-2 are the values of S_O, C_R and C_O used for each month.

There was good agreement between the results calculated using the transport model and those using the conservative mixing model and Isosource (Table 5-3). Using the nitrogen stable isotope ratio of macroalgae combined with short time series of salinity, we feel we are able to estimate the contribution of 3 DIN sources within ± 10%, which is acceptable for the needs of this study.

We performed sensitivity analyses to determine the effect of integration time period of the macroalgae and uptake rate using the results of the transport model (calculated for each monthly sampling at Stations N2, N3, and N4). The maximum difference in percent contribution of nitrogen sources calculated using a 2 week versus a 3 week integration period is 6% and the average absolute difference is 1%. Cohen and Fong (2005) found that *Enteromorpha intestinalis* carries its nutrient history for weeks, suggesting that a 2-3 week integration period is appropriate. The largest differences in percent contribution calculated using 2 and 3 week integration intervals occurred at Station N2, which was probably related to the greater temporal variability in predicted isotope ratio at this site (Figure 5-4). Results calculated with no uptake ($\mu = 0$) compared to those calculated with $\mu = 0.1$ d^{-1} had minimal differences during the wet season months (January – April). This is because there is little utilization of water column DIN during the wet season due to short residence time (associated with high freshwater inflow). The maximum difference in the percent contribution of the sources calculated with the two uptake rates occurred during May and June. Additionally, the riverine contribution is underestimated in the lower portion of the estuary for calculations using $\mu = 0.1$ d^{-1} compared to conservative transport. This is because the transit time for riverine DIN is maximal in the lower portion of the estuary and there is more time for loss of water column DIN. During May to September the maximal difference of riverine contribution was 14% and the average (of Stations N2, N3 and N4) underestimate of the riverine contribution was 6% (with using $\mu = 0.1$ d^{-1} compared to conservative transport). Using an uptake rate of 0.1 d^{-1} tends to overestimate the contribution of WWTF in the upper estuary (maximal difference of 7%) and had little effect in the lower estuary (0-1%). The average absolute difference during May to September in the WWTF contribution was 2%.

As a confirmation of the zonation based upon median salinities, we used the conservative mixing model (Equations 5.4-5.6) combined with mean dry season end members (C_R and C_O) to determine the salinity (S_M) where the nitrogen sources would switch from ocean to river dominance. During 2003, the mean dry season ocean end member DIN is 13.5 μM, while the riverine end member is 26 μM. Using these values the boundary between the oceanic and riverine segments occurred at a salinity of about 21 psu, which is similar to the value of 25 psu in determined in Section 5.2.1. In addition, assuming two end member (ocean and river) mixing, isotope ratios above 5.2‰ indicate oceanic DIN dominance while isotope ratios below this value indicate riverine DIN dominance. Therefore, the salinity demarking the zonation would range from 21 – 25 psu.

There are several assumptions used in this analysis that need to be valid for us to apply this technique to the seven target estuaries. This analysis assumes that macroalgae do not exhibit fractionation (preferentially take up and incorporate ^{14}N relative to ^{15}N), hence the isotope ratio of the macroalgae reflects the isotope ratio of the DIN in the water column. Cohen and Fong (2005) showed that *E. intestinalis* did not fractionate during DIN uptake and assimilation. In addition, it assumes that variability of the isotope ratio of the DIN results from a mixture of three nitrogen sources (ocean, river and WWTF) and not from fractionation associated with other biogeochemical transformations (e.g. denitrification). Another assumption is that the isotope end members used in the mixing model are applicable at a regional scale (i.e., isotope end members are the same for all seven estuaries). Stable isotope analysis of samples from across the region suggests that this is a reasonable first approximation. Also, this analysis assumes that there is minimal temporal variability in the isotope ratios of the end members. We also assume that the freshwater inflow is primarily associated with riverine inputs and WWTF inputs don't represent a significant source of freshwater. Based on riverine and WWTF flow rates, this was a good assumption even during periods of minimal riverflow (Table 3-6).

Table 5-2. Data used in the mixing model and time interval of salinity data utilized for each month. NA indicates data are not available.

SAMPLING DATE	TIME INTERVAL OF SALINITY DATA UTILIZED IN MIXING MODEL			END MEMBER VALUES		
	STATION N2	STATION N3	STATION N4	S_O (psu)	C_O (µM)	C_R (µM)
April 29	April 15 - 29	April 15 - 29	April 15 - 29	31.3	3.5 (n=33)	74.4 (n=3)
May 29	May 1-15 & 30-31	NA	May 15 - 29	31.7	4.1 (n=30)	49.5 (n=5)
June 26	June 12 - 26	June 20 - 28	June 12 - 26	32.3	9.4 (n=30)	27.8 (n=6)
July 31	July 17 - 31	NA	July 17 - 24	34.0	18.8 (n=31)	9.0 (n=4)
Aug 14	NA	July 31 - Aug 14	NA	34.0	14.5 (n=29)	11.1 (n=6)
Sept 26	Sept 1 - 16	Sept 4 - 16	Sept 23 - 30	33.6	19.1 (n=31)	15.6 (n=2)

Table 5-3. Comparison of percent contribution of DIN sources estimated using transport model and Isosource model. Value presented for the transport model represent averages, while Isosource results are ranges of feasible solutions. * denotes observed isotope ratio below river end member, therefore solution is not feasible.

MONTH	TRANSPORT MODEL			ISOSOURCE		
	OCEAN	RIVER	WWTF	OCEAN	RIVER	WWTF
Station N2						
April	24.4	74.3	1.3	8-28	71-85	0-8
May	42.3	55.8	1.9	20-40	48-62	11-19
June	80.6	16.2	3.2	63-83	5-19	11-19
July	91.9	5.0	3.1	86-100	0-9	0-6
September	93.5	3.7	2.8	87-98	0-7	2-7
Station N3						
April	8.5	89.8	1.7	0-1	99-100	0
June	47.2	43.7	9.1	32-52	35-49	12-20
August	86.8	6.2	7.0	74-94	4-18	1-9
September	75.1	14.0	10.8	76-83	0-4	17-21
Station N4						
April	4.5	93.8	1.6	*	*	*
May	6.5	90.4	3.1	0-15	79-90	5-12
June	32.1	56.6	11.3	19-39	46-60	14-22
July	48.2	31.0	20.8	65	35	0
September	61.1	22.4	16.5	70-85	0-10	15-22

5.3 Alsea Estuary

In addition to the data collected as part of this study, we also had other data sources available for the Alsea Estuary. For Alsea Estuary, we had monthly $\delta^{15}N$ of green macroalgae at five locations during 2004 extending from the ocean-dominated to the river-dominated portions of the estuary (labeled N1-N5 in Figure 5-7). As part of this study, Stations N1 and N4 were sampled for $\delta^{15}N$ of macroalgae on September 15, 2004. In addition, we had YSI datasondes deployed at three locations in this estuary from June - October 2004. Water quality cruises with six stations (S1-S6) were performed weekly from May through the end of September 2004 (Table 5-4). Historic salinity data were also available at twelve stations from the Oregon Department of Environmental Quality (with sample size at each station varying between 22 and 47).

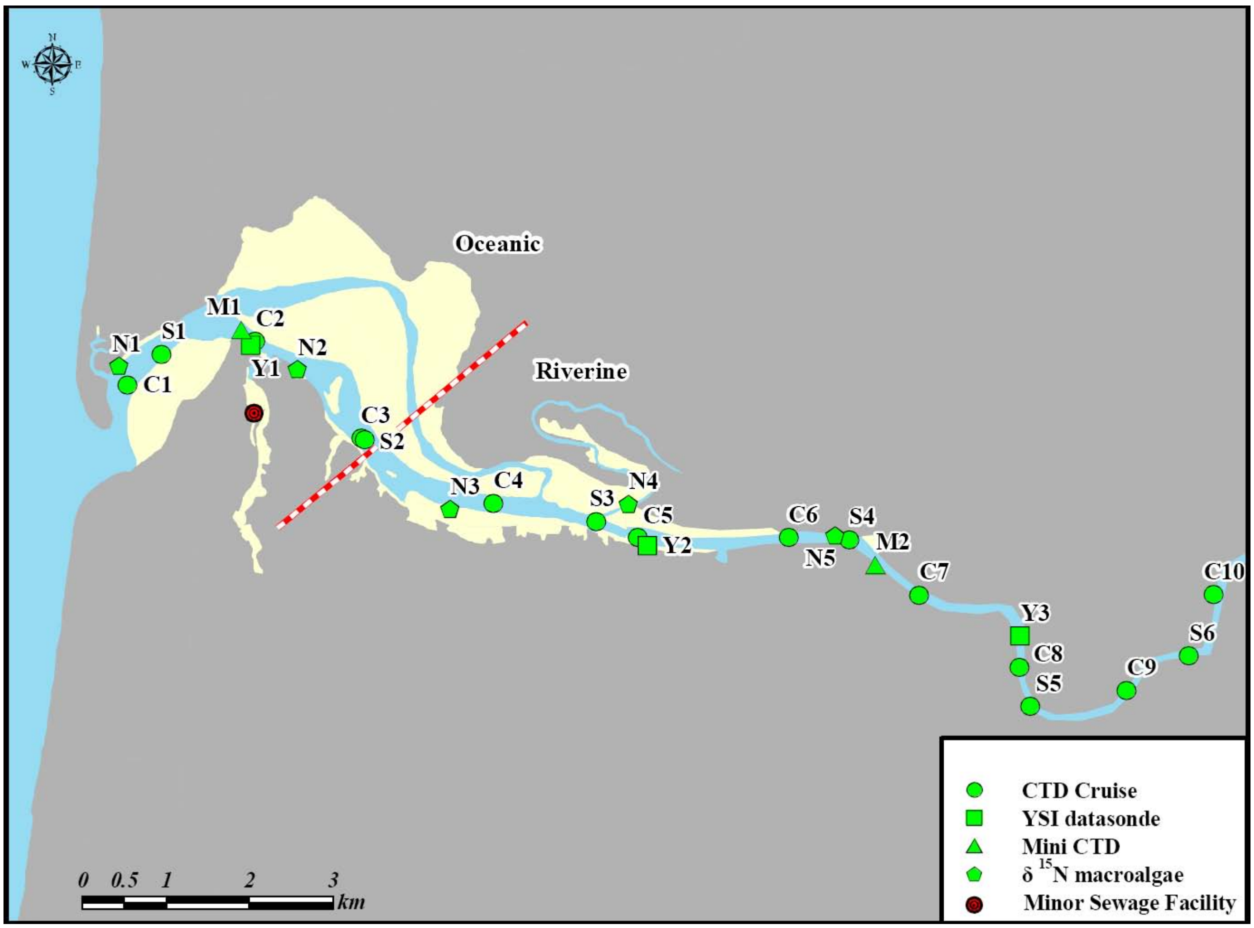

Figure 5-7. Location map for Alsea Estuary showing the locations of datasondes, water quality stations, isotope samples, and WWTF. The red dashed line shows the boundary between the oceanic and riverine segments.

From June to October, the green macroalgae $\delta^{15}N$ at Stations N1 and N2 showed similar temporal patterns (Figure 5-8) and an isotope ratio which suggested that oceanic input dominated at these two stations (average values of 7.9 and 7.5 ‰ at Stations N1 and N2, respectively). The isotope ratio at Station N3 was slightly elevated above N1 and N2, suggesting that there may be an additional nitrogen source. Station N4 had isotope ratios below 5 ‰ from January through July suggesting that riverine nitrogen sources dominated. At Station N4 during August the $\delta^{15}N$ of the macroalgae exceeded the ocean end member suggesting that there may be a third nitrogen source or denitrification may be important during the low riverflow period. At Station N5, the isotope ratio suggested that riverine nitrogen sources dominated from March through July and October, and there may have been an additional nitrogen source during August and September or denitrification had increased in importance. Riverine inflow averaged over the interval of one month prior to the isotope sampling was at its minimum during the months of August and September. During periods of minimal riverine inflow, minor DIN sources (e.g., nonpoint inputs associated with septic systems along the river) may increase in importance. In addition, denitrification may increase in importance during this period of longer residence times (although see Discussion, Section 5.9). There is a WWTF that discharges into the Alsea Estuary; however, it is located downstream of Stations N3 - N5 (Figure 5-7). Nonpoint septic inputs may have been the cause of increased macroalgal $\delta^{15}N$ since the Oregon Department of Environmental Quality (DEQ) recently initiated a program to repair failing septic systems along the Alsea River.

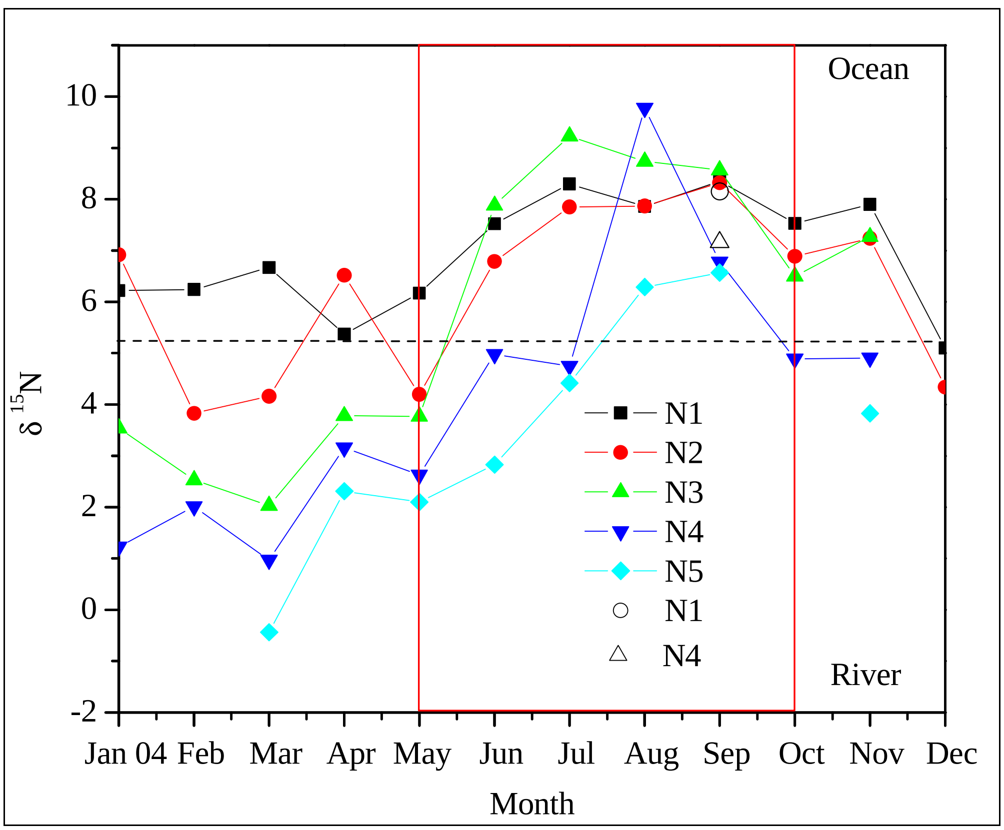

Figure 5-8. Monthly average observed isotope ratio at 5 locations in Alsea Estuary (dry season is indicated by red box). The dashed line indicates $\delta^{15}N = 5.2$ ‰ which is the demarcation between riverine and oceanic dominance, assuming two end member mixing of oceanic and riverine sources.

Alsea Estuary is located 20 km south of Yaquina Estuary. In the conservative mixing model, we used data from Yaquina Estuary for the ocean end member (C_O) DIN concentrations (sampling from Station Y1). Coastal ocean boundary conditions respond to regional phenomena such as upwelling and downwelling which encompass the entire Oregon coast. Consequently, ocean boundary conditions from a well characterized system such as Yaquina are applicable over a broad region. In Section 4.3, we demonstrated that ocean input (as indicated by flood-tide water temperature) was relatively uniform over the geographic range of the estuaries that we sampled. Riverine end member DIN (C_R) data were obtained from water samples collected at Station S6 (Figure 5-7). End member DIN concentrations were averaged over the 2 weeks prior to macroalgal sample data collection (Table 5-5). During September there were no DIN samples from Station S6 during the 2 weeks prior to our sampling so we used data from an Oregon DEQ Station (located about 20 miles from the mouth of the estuary) during the month of September from 1994-2006. We used salinity from Station Y1 (bottom) in the Yaquina Estuary averaged over the 2 weeks prior to the sampling date as the ocean salinity (S_O). Salinity data from the stations nearest to the macroalgae sampling locations were used in the analysis (Table 5-4) and in the calculations we salinity measured during the 2 weeks prior to the macroalgae sampling.

The similarity between the observed nitrogen stable isotope ratios and those predicted from the conservative mixing model at Stations N1 and N2 suggested that oceanic and riverine sources were the dominant DIN sources (Table 5-6). There was more uncertainty in the results calculated using the cruise data (Station N1) than those using the salinity time series (Station N2). At Station N2, the analysis suggested that this station receives 76-92% of DIN from the ocean. Unfortunately, there were not time series salinity data in the vicinity of Station N3 and this station is in a region of salinity changing from marine to mesohaline (compare the median salinities at Stations Y1 and Y2, Table 5-4). The limited salinity data prevented a thorough analysis of the nitrogen stable isotope ratio at this location; however, the similarity in temporal patterns in the $\delta\ ^{15}N$ during the dry season at Station N3 compared to Stations N1 and N2 (Figure 5-8), suggests that the ocean was the dominant nutrient source at this location, but there may be an additional nitrogen source as evident by $\delta\ ^{15}N$ of Station N3 > $\delta\ ^{15}N$ of Stations N1 and N2.

The macroalgal $\delta\ ^{15}N$ at Stations N4 and N5 were consistent with the two end member mixing model during the months of April through July. During August and September the observed $\delta\ ^{15}N$ at Stations N4 and N5 were elevated above that predicted from the two end member mixing model. During April and May, the mixing model suggested that Station N4 was river dominated (88-97%). During June at Station N4, there was considerable variability in the observed isotope ratio with two of the samples reflecting isotope ratios (2.8 ‰) similar to that predicted using the two end member mixing model and three having elevated isotope ratios (5.6-7.4 ‰). Assuming that the mixing model was more representative than the observed isotope ratios, Station N4 was river dominated (80%) during June. During July, the two end member mixing model suggests that Station N4 received 58% of the DIN from oceanic sources and 42% from riverine sources. There was high variability in the isotope ratio predicted using the two end member mixing model and the mean value was about 1‰ above the observed isotope ratio. Using Isosource the maximum ocean contribution that could produce the observed isotope ratio during July at Station N4 is 46%. Based on these two analyses, this station was probably located in the transition region during July. Isosource suggested that during August Station N4 received 35-55% of DIN from the ocean, 21-35% from river, and 23-31% from WWTF/septic inputs. During September, Station N4 became ocean dominated, with it receiving 50-70% of DIN from ocean, 28-42% from river, and 1-10% from WWTF. This suggested that Station N4 is in a transition region and the dominant nutrient source depends upon the month sampled (even within the dry season).

Similarity between the observed nitrogen stable isotope ratio and the two end member mixing model during the months of April through July at Station N5 suggested that the river and the ocean are the two major nitrogen sources. In addition, the observed isotope ratio suggests that this station is river dominated (70-98% of DIN) during these months (particularly during April – June). Station N5 is located between Stations Y2 and Y3, and there were considerable differences in isotope ratio predicted using these two different stations, particularly during the months of July - September. Results from Isosource suggested that during August Station N5 receives 57-71% from the river, 9-29% from ocean, and 13-21% from WWTF/septic. During September, Station N5 received 51-65% of DIN from riverine sources, 16-36% from the ocean, and 12-20% from WWTF/septic.

We used the conservative mixing model (Equations 5.4-5.6) combined with mean dry season end members (C_R and C_O) to determine the salinity (S_M) where the nitrogen sources would switch from ocean to riverine dominance. During 2004, the mean dry season ocean end member (C_O) was 11.7 μM (using data from Station Y1 in Yaquina with n = 214), while the riverine end member (C_R) was 15.4 μM (using data from Station S6 in Alsea with n =19). The riverine end member was similar to dry season average of 15.7 μM for an Oregon DEQ Station which is located at rivermile 20. Using these values the boundary between the oceanic and riverine segments would occur at a salinity of about 20 psu, which is similar to the value of 21 psu calculated for Yaquina Estuary. In addition, assuming two end members (ocean and river), nitrogen isotope ratios above 5.2 ‰ indicate that the ocean is the dominant nitrogen source while $\delta^{15}N$ ratios below this indicate river is the dominant N source.

From the stable isotope data and the conservative mixing model, it is clear that Stations N1 and N2 were ocean dominated and Station N5 was river dominated during the dry season and the boundary between ocean and riverine dominance in nitrogen sources was located between Stations N2 and N4. Using the salinity criterion developed in the above paragraph, median salinities from Stations Y2 and S3, suggested that Station N4 was river dominated during the dry season. We used historical salinity (median dry season) from Oregon DEQ at several stations located between S2 and N3 combined with the salinity criterion developed above to determine the boundary between the oceanic and riverine segments (Figure 5-7). We have a large degree of confidence in these zonation patterns because they are based on multiple lines of evidence.

Table 5-4. Summary of dry season salinities in Alsea Estuary.

STATION	MEDIAN SALINITY (psu)	5^{TH} PERCENTILE (psu)	95^{TH} PERCENTILE (psu)
Datasondes (Deployed June 17 – October 12, 2004)			
Y1	31.1	16.6	33.7
Y2	12.3	3.1	25.3
Y3	4.6	0.1	11.0
Mini-CTDs (Deployed September 14-October 6, 2004)			
M1	NA	NA	NA
M2	7.5	0.0	13.1
Water Quality Cruises (June – October 2004)			
S1	31.4	20.8	33.4
S2	25.0	4.7	33.2
S3	13.4	1.3	30.1
S4	9.0	0.0	19.9
S5	1.5	0.0	12.5
S6	0	0.0	7.3
Classification Low And High Tide Cruises			
STATION	LOW TIDE SALINITY (psu)	HIGH TIDE SALINITY (psu)	
C1	25.4	32.7	
C2	20.0	32.5	
C3	12.4	32.2	
C4	8.8	29.1	
C5	6.6	17.5	
C6	4.3	6.4	
C7	2.9	5.1	
C8	2.5	2.8	
C9	1.5	0.4	
C10	0.2	0	

Table 5-5. Ocean and river end members for Alsea calculations.

MONTH	C_o (μM)	C_R (μM)
April	5.9 (n = 4)	27.5 (n = 2)
May	7.7 (n = 11)	29.2 (n = 3)
June	13.8 (n = 25)	9.5 (n = 3)
July	10.1 (n = 23)	4.1 (n = 3)
August	6.2 (n = 6)	6.9 (n = 2)
September	14.5 (n = 20)	9.5 (n = 10)

Table 5-6. Observed and predicted isotope ratios (from two end member model) in Alsea Estuary.

MONTH	OBSERVED ISOTOPE RATIO (‰)	δ_{MIX} (‰)	STATION USED
Station N1			
April	5.4 ± 0.9	5.0 ± 1.6	S1
May	6.2 ± 0.5	4.5 ± 1.3	S1
June	7.5 ± 0.2	7.9 ± 0.2	S1
July	8.3 ± 0.4	7.9 ± 0.4	S1
August	7.9 ± 0.2	7.4 ± 0.8	S1
Station N2			
June	6.8 ± 0.5	7.4 ± 1.5	Y1
July	7.9 ± 0.2	7.9 ± 0.6	Y1
August	7.9 ± 0.1	7.5 ± 0.9	Y1
September	8.3 ± 0.2	7.7 ± 0.7	Y1
Station N3			
September	8.6 ± 0.1	6.0 ± 2.5	C4
Station N4			
April	3.2 ± 0.2	2.2 ± 0.3	S3
May	2.6 ± 0.6	2.7 ± 1.3	S3
June	5.0 ± 2.0	2.5 ± 0.2	S3
		3.3 ± 0.8	Y2
July	4.8 ± 0.6	5.9 ± 1.1	S3
		5.7 ± 1.0	Y2
August	9.8 ± 0.7	6.1 ± 1.5	S3
		4.6 ± 1.0	Y2
September	6.8 ± 1.3	5.8 ± 0.9	Y2
Station N5			
April	2.3 ± 0.2	2.2 ± 0.2	S4
May	2.1 ± 0.4	2.1 ± 0.2	S4
June	2.8 ± 2.5	2.3 ± 0.4	S4
		3.3 ± 0.8	Y2
		2.1 ± 0.1	Y3
July	4.4 ± 0.8	4.9 ± 1.2	S4
		5.9 ± 1.1	Y2
		3.9 ± 0.7	Y3
August	6.3 ± 0.2	4.5 ± 1.0	S4
		4.6 ± 1.0	Y2
		3.2 ± 0.4	Y3
September	6.6 ± 0.4	3.1 ± 0.4	M2
		5.8 ± 0.9	Y2
		3.7 ± 0.7	Y3

5.4 Salmon River Estuary

Data used to determine the zonation in Salmon River Estuary included salinity data from short-term YSI and Mini-CTD deployments in 2004 (Table 5-7), macroalgae isotope data (Table 5-8) collected as part of the classification effort at two sites (N1 & N7) and previous data collected at five locations (N2 – N6) (DeWitt and Eldridge, PCEB, unpublished data). The locations of all sampling stations are presented in Figure 5-9.

The macroalgae $\delta^{15}N$ did not show as strong an ocean signal in Salmon River Estuary as in the Yaquina and Alsea estuaries (maximum in $\delta^{15}N$ near mouth of about 6‰ at Salmon compared to 8 ‰ in Alsea and Yaquina). This is because the estuary switches from ocean- to river-dominated over a tidal cycle with the salinities at the mouth changing from 33 to 10 psu (Figure 4-8). As a result, there was not a strong oceanic signal in the nitrogen stable isotope data even at the mouth (Table 5-8). Using the salinity data from the classification data collection effort and the conservative mixing model (Equations 5.4-5.6), we examined the importance of oceanic and riverine nitrogen sources within the estuary. The concentration of DIN in the riverine end member was determined from Station C10 on the low tide cruise (C_R = 21.6 μM). This riverine end member was consistent with other data collected by U.S. EPA (Ozretich, PCEB, unpublished data), which had a mean DIN of 21.3 μM during July and August of 2001. The DIN concentration for the ocean end member was determined from flood-tide nutrient samples collected at Yaquina Estuary every flood tide during June-August 2004 (C_O = 13.4 μM). Salmon River Estuary is located 45 km north of Yaquina Estuary, and in Section 4.3 we demonstrated that ocean conditions are relatively uniform over the geographic extent of the target estuaries. We used an average value for the ocean end member since we were using isotope data from different years (2000 and 2004).

Based on the conservative mixing model, the isotope ratio predicted using the salinity data from Y1, M1 and M2 averaged between 6.5 ‰ and 7.0 ‰, which is similar to observed isotope ratios at Stations N1-N6. Using the conservative mixing model and salinity data from Y2, the predicted isotope ratio was 4.6 ‰, which is similar to the observed isotope ratio at N7. The agreement between predicted and observed isotope ratios suggested that oceanic and riverine inputs are the two main sources of nutrients to this estuary. There was better agreement between the predicted and observed isotope ratio when a short-term salinity time series was used rather than data from the high and low tide cruises. The nitrogen stable isotope ratio predicted using the average of the high and low tide cruises appeared to underpredict the macroalgal $\delta^{15}N$. The isotope mixing model suggested that Stations N1-N6 received between 70-88% of the nitrogen from the ocean, while Station N7 received about 60% of the nitrogen from the river. Based on the median salinity of 25.9 psu at Station M2 and the macroalgal $\delta^{15}N$ indicating ocean dominance at Station N6, the boundary between the oceanic and riverine segments was located slightly above Station N6. This is consistent with the boundary being located approximately at a median dry season salinity of 25 psu.

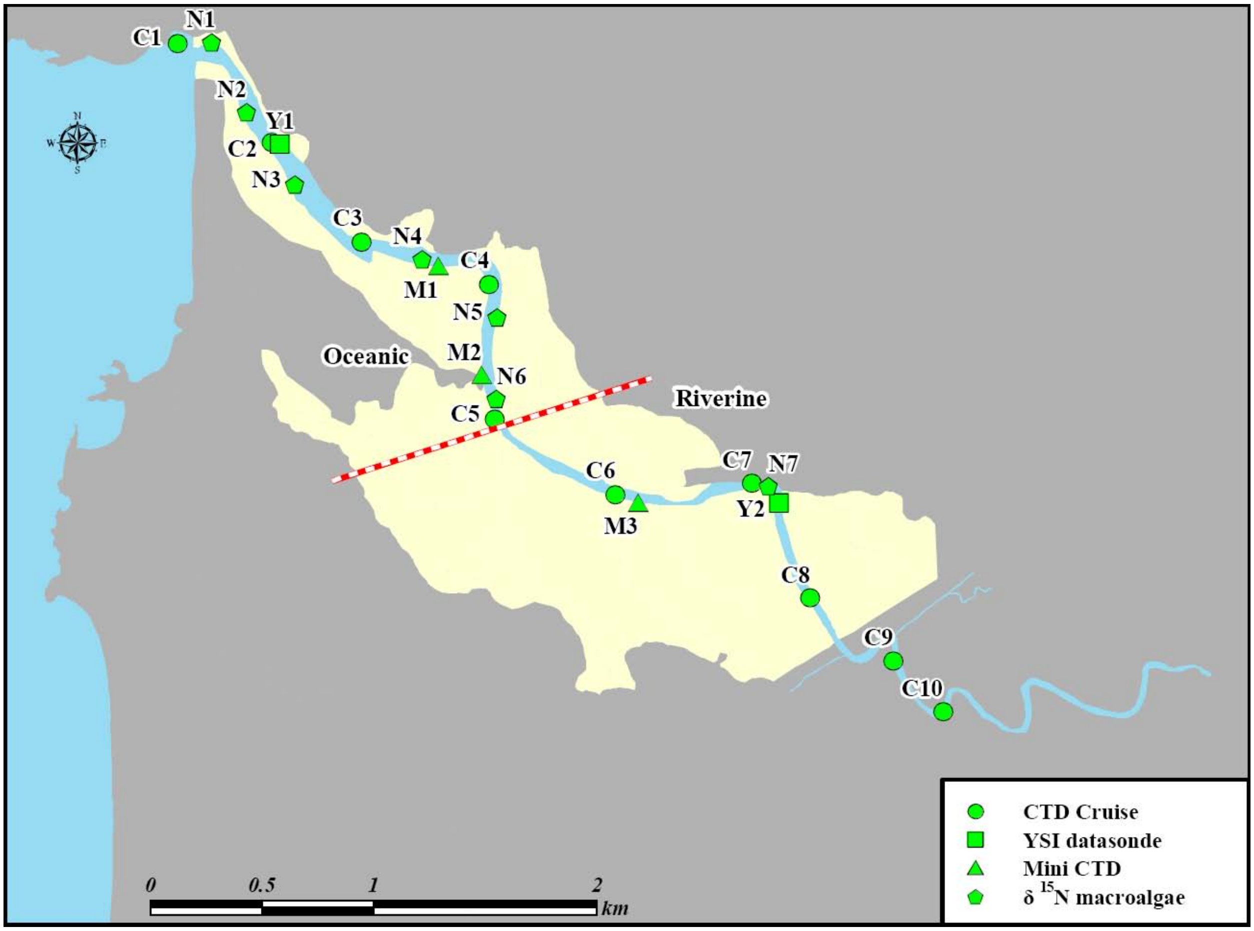

Figure 5-9. Location map for Salmon River estuary showing the locations of datasondes, water quality stations and isotope samples. The red dashed line shows the boundary between oceanic and riverine segments.

Table 5-7. Summary of dry season salinities in Salmon River estuary.

STATION	MEDIAN SALINITY (psu)	5TH PERCENTILE (psu)	95TH PERCENTILE (psu)
Datasondes (Deployed July 13-20, 2004)			
Y1	30.6	14	33.4
Y2	16.5	2.6	29.7
Mini-CTDs (Deployed July 9-29, 2004)			
M1	29.6[1]	12.1	36.1
M2	25.9	15.2	31.5
M3	16.7	4.5	33.8
[1]Evidence of biofouling after 7/18/2004. Median calculated using data from July 9-18.			

Classification Low And High Tide Cruises		
STATION	LOW TIDE SALINITY (psu)	HIGH TIDE SALINITY (psu)
C1	22.7	30.0
C2	14.8	30.1
C3	13.3	29.9
C4	10.1	29.5
C5	7.9	29.1
C6	4.4	26.7
C7		26.8
C8	1.8	4.0
C9		1.8
C10	0.4	0.5

Table 5-8. Observed and predicted isotope ratios (from two end member model) in Salmon River estuary.

STATION	SAMPLING DATE	OBSERVED ISOTOPE RATIO mean ± standard deviation (‰)	δ_{MIX} mean ± standard deviation (‰)	STATION USED FOR MIXING MODEL
N1	7/14/2004	5.9 ± 0.3	6.5 ± 1.3	C1
N2	8/23/2000	6.6 ± 0.4	5.8 ± 2.4	C2
			7.0 ± 1.5	Y1
N3	8/23/2000	5.7 ± 0.4	7.0 ± 1.5	Y1
			5.8 ± 2.4	C2
N4	8/23/2000	6.7 ± 0.3	6.8 ± 2.1	M1
			5.6 ± 2.5	C3
N5	8/23/2000	6.9 ± 0.9	5.3 ± 2.8	C4
N6	8/23/2000	6.5 ± 0.4	6.5 ± 2.4	M2
			5.1 ± 2.9	C5
N7	7/14/2004	3.6 ± 0.5	4.6 ± 1.8	Y2

5.5 Umpqua River Estuary

Data used to determine the zonation in Umpqua River Estuary included salinity (Table 5-9) and macroalgae isotope ratios ($\delta^{15}N$) collected at five sites as part this study in 2005 (Table 5-10). Unfortunately, the GPS had an error during the sampling of N2 and as a result we do not know the exact sampling location, only that the sample was collected between Stations N1 and N3. For the salinity data, we had short-term datasondes and Mini-CTD data collected in 2005 and historic salinity available from Oregon DEQ.

Using the salinity data from the classification data collection effort and a conservative mixing model (Equations 5.4-5.6), we examined the importance of oceanic and riverine nitrogen sources within the estuary. The DIN concentration of the riverine end member was determined from the Station C11 low tide cruise (C_R = 15.8 µM). The DIN concentration for the ocean end member was the mean of the high tide cruise at Stations C1 - C3 (C_O = 13.8 µM), which was similar to dry season mean values of DIN from Yaquina Estuary for 2002-2004 (average = 14 µM). Comparison of predicted and observed isotope ratios suggested that Stations N1-N3 received a mixture of ocean and riverine nitrogen (particularly using Station Y1) with the ocean contributing between 80-90% of the nitrogen at these sites. It was not possible to distinguish the effect of the WWTF located near the entrance to Umpqua Estuary (Figure 5-10) due to the variability in predicted nitrogen stable isotope ratios associated with salinity variations at Station Y1. The disagreement between observed and predicted macroalgal $\delta^{15}N$ at Station N5 suggested that there may be a third nitrogen source in this estuary. The City of Reedsport has a WWTF in the vicinity of Station N5 (distance from WWTF to Station N5 is about 2 km, Figure 5-10). Using Isosource, we calculated all of the possible combinations of the three nitrogen sources which could produce the observed isotope value of 5.4 ‰ at Station N5. Using the observed salinity at this site, we rejected the results from Isosource that had ocean contributions greater than 10% based on salinity data and the two end member mixing model. Based on this analysis, we estimated that the nitrogen sources at Station N5 were about 74-82% riverine and 15-20% associated with WWTF effluent. The observed macroalgal $\delta^{15}N$ at Station N4 was slightly elevated above the predicted isotope ratio from two end member model; however, the variability in the predicted ratio made it difficult to determine whether this elevation was associated with an additional nitrogen source or variability in the isotope ratios associated with the salinity varying from 2 to 32 psu over a tidal cycle. Based on the nitrogen isotope data, the transition from ocean to riverine dominance occurred between Stations N3 and N4. This was consistent with the data from the high tide cruise, which have high salinity (> 30 psu) water reaching Station C4 and a dramatic decrease in salinity above this station (decrease of about 17 psu between Stations C4 and C5). In addition, the high tide cruise showed the input of oceanic nutrients (NO_3^-+NO_2^- and PO_4^{3-}) to Stations C1 - C4. This was evident in that NO_3^-+NO_2^- decreases from 10 µM to 1.5 µM at between Stations C4 and C5.

Using the oceanic and riverine DIN end members (C_R = 15.8 µM and C_O = 13.8 µM) and the two end member conservative mixing model, the boundary between the oceanic and riverine segments occurred at a salinity of about 18 psu. This agrees fairly well with the salinity (including historical salinity from Oregon DEQ) and macroalgae isotopes. The Oregon DEQ database has a salinity station by Station C3 that has a median dry season salinity of 19 psu, while a salinity station by Station C4 has a median dry season salinity of 16 psu.

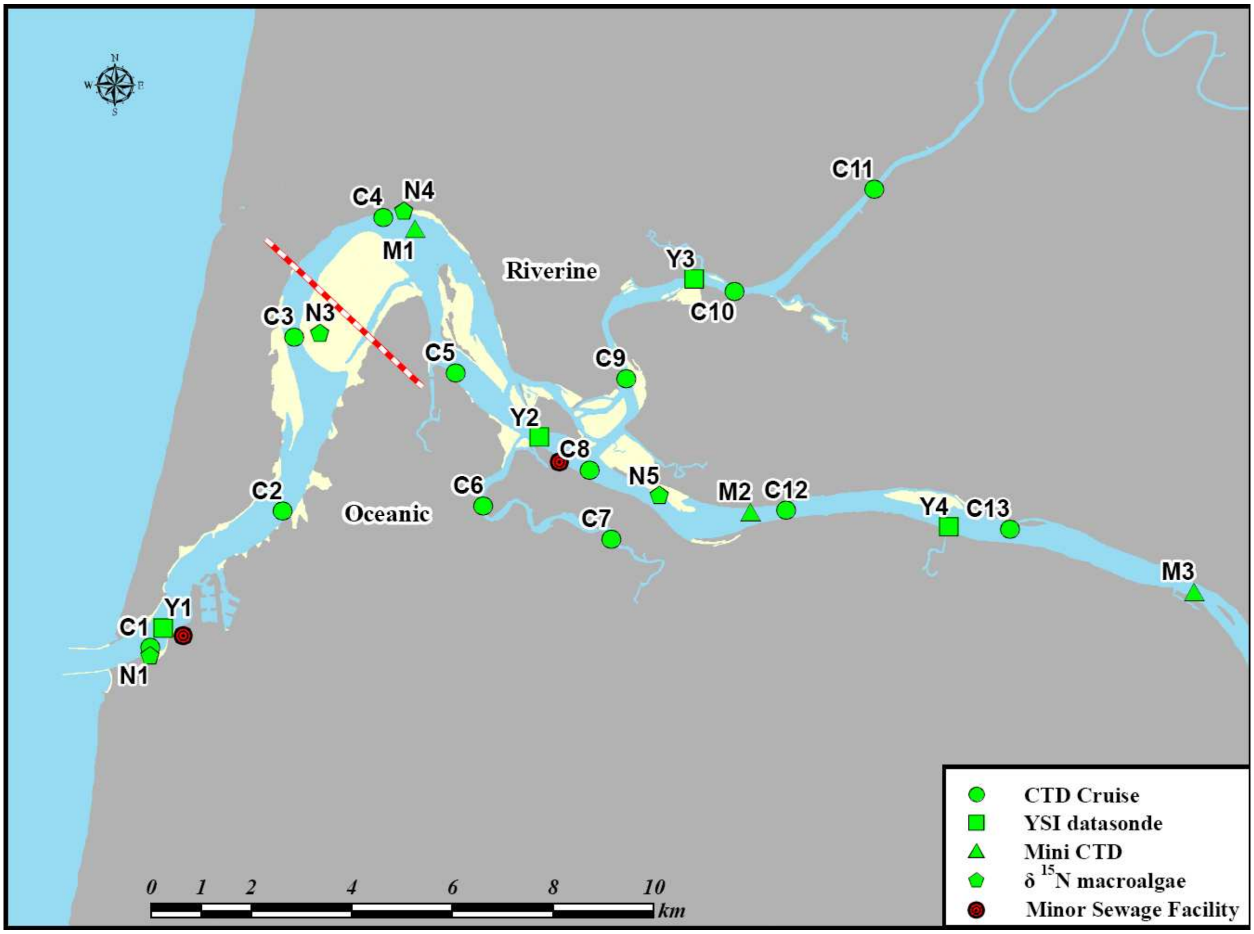

Figure 5-10. Location map of Umpqua River Estuary showing the locations of datasondes, water quality stations, isotope samples, and WWTFs. The red dashed line shows the boundary between the oceanic and riverine segments.

Table 5-9. Summary of dry season salinities in Umpqua River Estuary.

STATION	MEDIAN SALINITY (psu)	5TH PERCENTILE (psu)	95TH PERCENTILE (psu)
Datasondes (Deployed June 21 – 26, 2005)			
Y1	30.7	19.8	33.7
Y2	6.2	0.7	21.3
Y3	1.1	0.2	4.9
Y4	0.1	0.0	0.5
Mini-CTDs (Deployed June 20 – 26, 2005)			
M1	14.8	3.7	28.2
M2	0.0	0.0	4.7
M3	0.0	0.0	0.0

Classification Low And High Tide Cruises		
STATION	LOW TIDE SALINITY (psu)	HIGH TIDE SALINITY (psu)
C1	23.4	32.7
C2	15.3	32.7
C3	9.8	32.3
C4	5.3	30.5
C5	1.9	13.7
C6	1.8	4.21
C7	1.3	1.46
C8	0.1	5.6
C9	0.8	6.7
C10	0.2	2.6
C11	0.0	0.6
C12	0.0	0.9
C13	0.0	0.1

Table 5-10. Observed and predicted isotope ratios (from two end member model) in Umpqua River Estuary.

STATION	SAMPLING DATE	OBSERVED ISOTOPE RATIO mean ± standard deviation (‰)	δ_{MIX} mean ± standard deviation (‰)	STATION USED FOR MIXING MODEL
N1	6/24/2005	7.8 ± 0.5	7.6 ± 0.9	Y1
			7.3 ± 1.4	C1
N2	6/23/2005	7.3 ± 0.2	6.5 ± 2.5	C2
N3	6/22/2005	7.2 ± 0.2	5.9 ± 3.2	C3
N4	6/22/2005	5.7 ± 0.1	4.9 ± 1.6	M1
N5	6/26/2005	5.4 ± 0.2	2.2 ± 0.3	M2
			2.5 ± 0.7	C8

5.6 Coos Estuary

Coos Estuary had the highest isotope ratios of the seven estuaries that we sampled with three stations (N3-N5) having $\delta^{15}N$ values of about 10 ‰ (Table 5-11). This elevation of the isotope ratio above the marine end member may be associated with the presence of multiple WWTFs in the watershed. There are 3 major WWTFs adjacent to the estuary (Figure 5-11), each of which is permitted to discharge between 2 to 5 million gallons per day of treated effluent into the estuary (http://www.deq.state.or.us/wq/sisdata/facilitycriteria.asp). This is the only estuary that has multiple major WWTFs adjacent to the estuary.

Using the salinity data from the classification data collection effort (Table 5-12) and the conservative mixing model (Equations 5.4-5.6), we examined the importance of oceanic and riverine nitrogen sources within the estuary. Since our most riverine station (C19) had a salinity of 13 psu, we could not use our data for the river end member. We used mean dry season DIN (C_R = 13.7 μM, n = 28) from the Oregon DEQ database for a station at the Anson Roger Bridge on the Coos River (5 km upriver of Station C19). This value is similar to the concentration during the low tide cruise at Station C19 (10.3 μM). For the ocean end member, we used the mean high tide DIN (C_O = 21.3 μM) from Stations C1, C4 and C5. This is probably a good estimate of the ocean end member since there were strong upwelling conditions for about 3 weeks prior to our data collection effort (based on water temperature, see Figure 4-2). Unfortunately, the Mini-CTD near the mouth of Coos Estuary (M1) malfunctioned. Using the datasondes closest to the mouth of the estuary (Y1 and Y7) and the two end member conservative mixing model using ocean and river end members, we predict that the isotope ratio near the estuary mouth was 8.2-8.3 ‰. Unfortunately, we did not have macroalgae isotope ratio data at the estuary mouth, but Stations N1 and N2 are fairly close to the mouth and had similar isotope ratios. Assuming two end member mixing of riverine and ocean sources, the observed isotope ratios at Stations N1 and N2 suggest that the ocean provides 78 to 87% of the nitrogen at these sites. Fry et al. (2001) found that the nitrogen stable isotope ratio of green macroalgae during the dry season outside the mouth of Coos Estuary was 8.3 – 9.0 ‰ (sampled during October 1998 and July 1999) and inside the mouth of Coos Estuary the isotope ratio was 7.7 – 7.8 ‰ sampled during the same months, which is similar to our data.

The observed isotope ratios at Stations N3-N5 exceed those expected from oceanic nitrogen sources suggesting there is an additional nitrogen source. Isosource was used to calculate all possible combinations of ocean, riverine, and WWTF nitrogen sources that could produce the observed isotope ratios at Stations N3-N5. Since the short-term salinity time series provided better estimates of the isotope mixing than the average of low and high tide cruises, we used the two end member mixing model with data from Y2, Y3, and M3 to estimate the contribution of the ocean. The two end member mixing model of ocean and river sources using Station Y2 and Y3 suggested that the ocean provided about 97% of the DIN at Station N3 and N4. Using Isosource and selecting only those solutions with an ocean contribution ≥ 87%, the contribution of nitrogen sources were estimated to be 87-91% oceanic, 9-13% WWTF and < 2% riverine. The two end member mixing model of ocean and river sources using Station M3 suggested that the ocean provided about 65% of the DIN at Station N5. Using Isosource and selecting only those solutions with ocean contribution within ± 10% of that predicted from the two end member mixing model, the contribution of nitrogen sources at Station N5 were estimated to be 55-75% oceanic, 17-26% WWTF and 7-21% riverine. This may be an underestimate the contribution of the ocean and an overestimate of the contribution of the WWTF since M3 is located slightly upriver of N5. Predicted isotope ratios calculated using Stations C10 and C16 were higher than those using M3, and yield oceanic contributions of about 90%.

Based on this analysis, all of the macroalgal $\delta^{15}N$ sampling locations (N1-N5) were dominated by oceanic nitrogen sources, and the elevation of the isotope ratio above ocean end member at Stations N3 and N5 is due to the nitrogen sources being a mixture of oceanic and WWTF sources. We used a median salinity of 25 psu (as a conservative estimate) as the boundary between the oceanic and riverine segments. We used historical salinity from Oregon DEQ and South Slough Estuarine Reserve to aid in the zonation.

Table 5-11. Observed and predicted isotope ratios (from two end member model) in Coos Estuary.

STATION	SAMPLING DATE	OBSERVED mean ± standard deviation (‰)	δ_{MIX} mean ± standard deviation	STATION USED FOR MIXING MODEL
N1	8/7/2005	7.6 ± 1.8	8.4 ± 0.0	C2
			8.2 ± 0.1	Y5
			8.1 ± 0.1	Y6
N2	8/7/2005	7.0 ± 1.9	8.3 ± 0.1	Y1
			8.2 ± 0.1	Y7
			8.3 ± 0.1	C4
N3	8/5/2005	9.6 ± 0.3	8.2 ± 0.1	Y2
			8.2 ± 0.1	C7
N4	8/6/2005	10.0 ± 0.4	8.2 ± 0.1	Y3
			8.1 ± 0.2	C6
N5	8/4/2005	10.0 ± 0.1	6.1 ± 0.9	M3
			7.9 ± 0.1	C16
			7.8 ± 0.1	C10

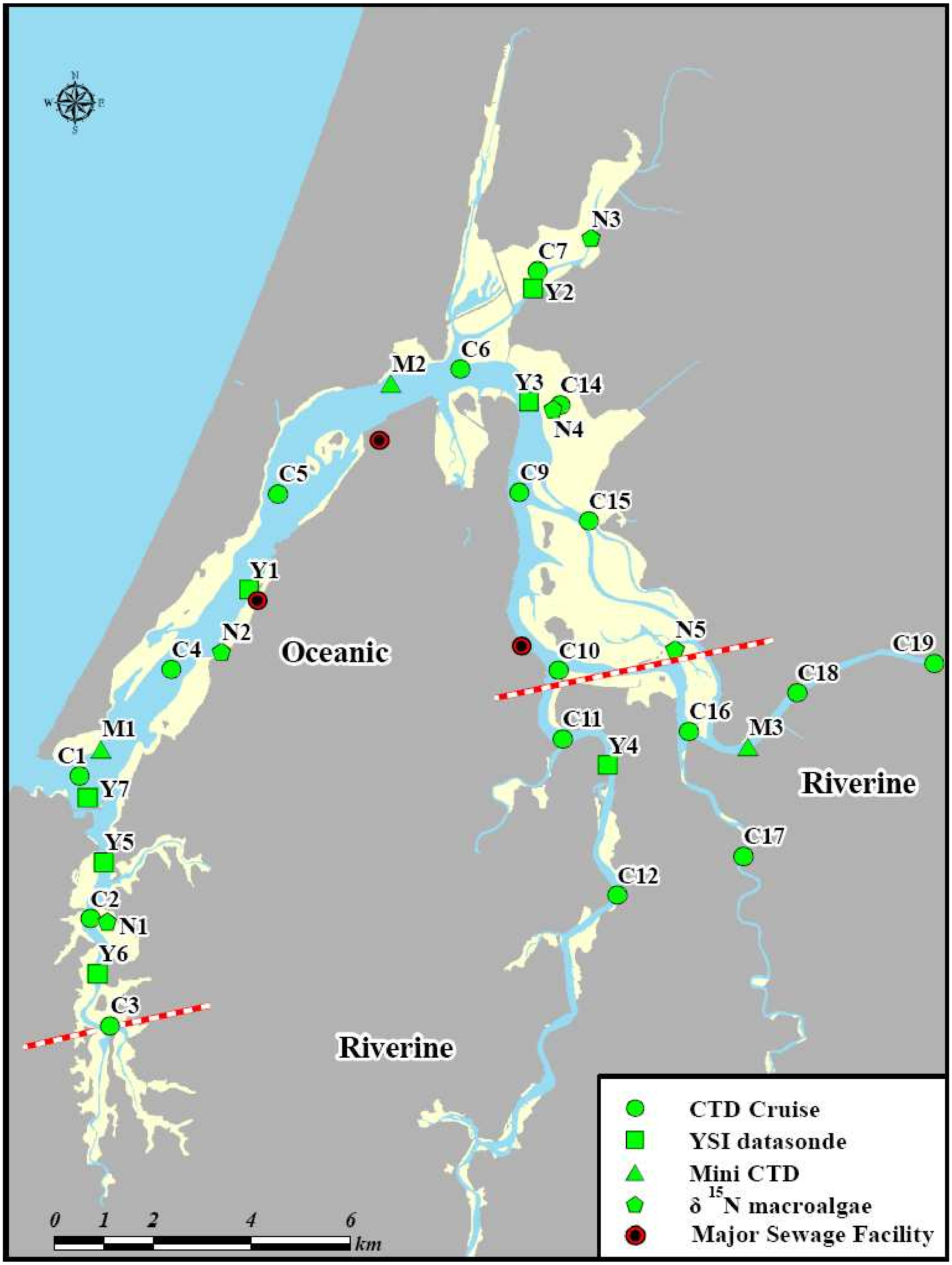

Figure 5-11. Location map of stations in Coos Estuary showing the locations of datasondes, water quality stations, isotope samples, and WWTFs. The red dashed line shows the boundary between the oceanic and riverine segments.

Table 5-12. Summary of dry season salinities in Coos Estuary.

STATION	MEDIAN (psu)	5TH PERCENTILE (psu)	95TH PERCENTILE (psu)
Datasondes (Y1-Y4 Deployed August 3 – 10, 2005; Y5 And Y6 Deployed July 26 – August 24, 2006; Y7 Deployed July 6 – 20, 2006).			
Y1	33.5	31.9	34.0
Y2	32.5	31.7	33.5
Y3	32.4	30.8	33.9
Y4	28.9	27.0	30.0
Y5	32.5	31.4	33.1
Y6	32.1	29.4	33.0
Y7	32.4	30.4	33.4
Mini-CTDs (Deployed August 3 – 10, 2006)			
M1	NA	NA	NA
M2	27.8	24.9	30.8
M3	20.6	8.6	24.5

Classification Low And High Tide Cruises

STATION	LOW TIDE SALINITY (psu)	HIGH TIDE SALINITY (psu)
C1	33.1	33.8
C2	NA	34.2
C3	NA	33.6
C4	32.5	33.9
C5	31.7	33.8
C6	30.4	33.1
C7	32.0	33.1
C9	29.8	32.3
C10	29.0	30.6
C11	27.9	29.9
C12	26.3	28.3
C14	31.9	33.9
C15	31.4	NA
C16	29.4	30.9
C17	20.6	20.7
C18	23.2	28.3
C19	12.5	18.5

5.7 Tillamook Estuary

Data used to determine the zonation in Tillamook Estuary included salinity data from short-term YSI and Mini-CTD deployments in 2005 (Table 5-13) and macroalgae isotope data (Table 5-14) collected as part of this study at four sites (N1-N4, Figure 5-12). Historic salinity data were also available at 23 stations from the Oregon DEQ (see Figure 5-13; http://deq12.deq.state.or.us/lasar2/). Using the salinity data from the classification data collection effort (Table 5-13) and the conservative mixing model (Equations 5.4-5.6), we examined the importance of oceanic and riverine nitrogen sources within the estuary. For the ocean end member in the conservative mixing model, we used the high tide DIN (C_O = 24.1 μM) from Station C1. This is probably a good estimate of the ocean end member since there were strong upwelling conditions (based on water temperature in Figure 4-2) for about 1 week prior to the macroalgae isotope sampling. Since there are five rivers discharging into Tillamook Estuary, we used different river end members depending upon the macroalgae sampling location. Stations N1 and N2 are located in the vicinity of the Miami River. For Stations N1 and N2, we used the mean DIN concentrations from Station C3 (C_R = 52.4 μM), which is in the Miami River. Station C3 had similar high and low tide DIN concentrations and a salinity of <1 psu. Colbert (2004) sampled the riverine end members of Tillamook in 1998 and 1999; their mean DIN for the Miami River during May-August was 51.4 μM, which was comparable to our river end member from Station C3. The Tillamook, Trask and Wilson rivers discharge into the southern portion of Tillamook Estuary. For Station N4, we used the mean of the low tide cruise for Stations C13-C15 (C_R = 39 μM), which had salinities ranging from 0.1-1.3 psu. Colbert (2004) sampled the Tillamook, Trask and Wilson rivers in 1998 and 1999 and found similar dry season end members (mean = 32.2 μM). Since Station N3 is located in the central portion of the estuary, we calculated the isotope ratios using the two different values for the river end member (C_R = 39 and 52.4 μM) and present results of both in Table 5-14. Interestingly, Tillamook Estuary had the highest values of riverine end members of the seven target estuaries that we sampled.

The observed isotope ratio at Station N1 was about 1‰ less than the isotope ratio predicted using the two end member mixing model of ocean and river sources (using salinity data). Based on this analysis, the contribution of oceanic and riverine nitrogen sources was 85% and 15%, respectively. Using Isosource to mix the three end members, we found that the maximum ocean contribution that could produce the observed isotope ratio was 67% (with the remainder of the nitrogen being riverine). Based on both of these analyses, Station N1 was ocean dominated.

In the vicinity of Station N2 there was high variability in the salinity which is evident in the large difference between salinities at Stations Y1 and Y2 and C2 and C3. Stations Y1 and C2 are marine, while Stations Y2 and C3 are fresh. Since we did not have salinity observations at Station N2, and considering the high variability in the nearby stations, the two end member mixing model did not produce useful results. Isosource was used to calculate all possible combinations of ocean, riverine, and WWTF nitrogen sources that could produce the observed isotope ratio. There were limited results that could produce the observed isotope ratio, so the salinity constraint was not needed. Results from Isosource suggested that the primary nitrogen source was riverine (80-94%) with oceanic and WWTF being minor components (10-20% and 3-

7%, respectively). Based on this analysis, riverine inputs were the dominant nitrogen source at Station N2.

Unfortunately, we did not have acceptable salinity time series data in the vicinity of Station N3. For the two end member mixing model, we used low and high tide cruise data from Stations C5 and C6. The isotope ratios predicted using the two end member mixing model were similar to those observed, particularly for Station C6. Based on the two end member mixing model for Station N3, the ocean contribution ranged from 65 to 82% and the riverine contribution ranged from 18 to 34%, depending upon which station and river end member was used in the analysis. Using Isosource and only selecting those results that had oceanic contributions >55%, the oceanic contribution ranged from 55 to 68%, riverine ranged from 32 to 42% and WWTF was less than 4%. Both of these analyses suggested that Station N3 was ocean dominated.

At Station N4 the observed isotope ratio was 1.7-2.6 ‰ higher than the isotope ratio predicted using the two-end member mixing model (depending upon which station is used for the salinity) suggesting that there may have been an additional nitrogen source. Based on a two end member mixing model, the oceanic contribution ranged from 9 to 23%, while the riverine contribution ranged from 76 to 91% (depending upon which station was used). There are two WWTF facilities (one major and one minor) within 4 km of Station N4 (Figure 5-12). Using Isosource and selecting only those solutions with the ocean contribution within ±10% of that predicted from the two end member mixing model, we estimated that the riverine contribution ranged from 60 to 83%, oceanic contributions were less than 30%, and WWTF contributed between 5 and 18%. Therefore, based on these analyses Station N4 was river dominated.

Due to the uncertainty in the analysis of the isotope data due to multiple riverine end members, multiple WWTF inputs, and non-ideal availability of salinity data for the mixing model relative to macroalgae sampling locations, we used recent and historic salinity data in the derivation of the zonation. Using the values of the riverine and oceanic end members from Tillamook Estuary (C_R = 39 -52 μM and C_o = 24.1 μM) and the conservative mixing model, the boundary between oceanic and riverine dominance occurred at a salinity of about 20-23 psu. The value of the ocean end member was relatively high compared to averaged dry season values for Yaquina Estuary. Using C_o = 13.4 μM (which is the June to August average for 2004 at Yaquina Estuary), the boundary would be located at about 25 psu. To be conservative in our estimates, we used a salinity criterion of 23-25 psu to divide the estuary into oceanic and riverine segments (Figure 5-13). The boundary between the oceanic and riverine segments in the northern portion of Tillamook Estuary was based upon the analysis of the isotope data at Station N3.

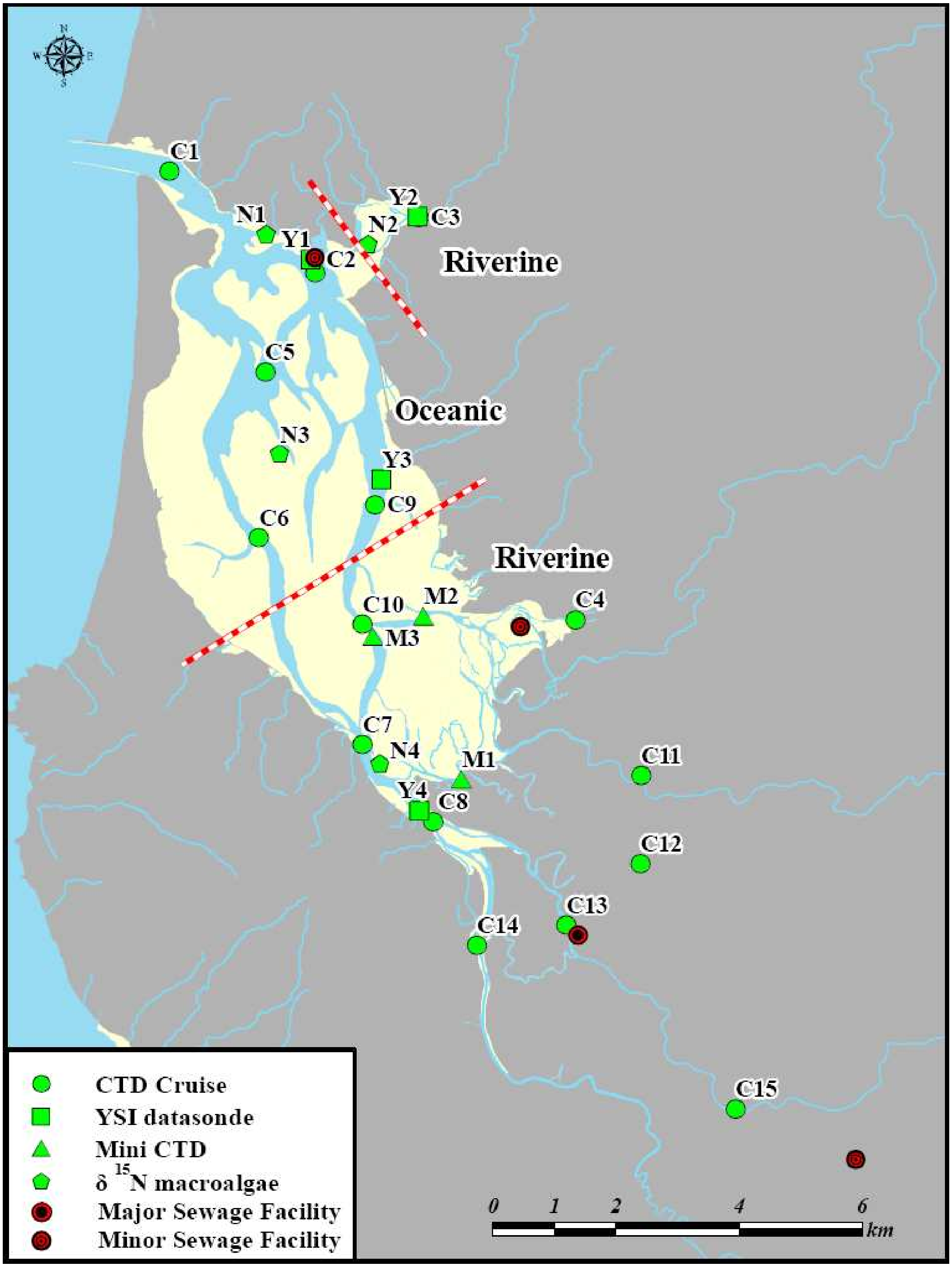

Figure 5-12. Location map for Tillamook Estuary showing the location of datasondes, water quality stations, isotope samples, WWTFs. The red dashed line shows the boundary between the oceanic and riverine segments.

Table 5-13. Summary of dry season salinities in Tillamook Estuary (NA denotes missing or bad data).

STATION	MEDIAN (psu)	5TH PERCENTILE (psu)	95TH PERCENTILE (psu)
Datasondes (Deployed July 19-25, 2005)			
Y1	33.9	25.0	34.9
Y2	2.4	0.0	27.2
Y3	29.6*	10.6*	34.1*
Y4	7.8	0.5	23.7
* Evidence of biofouling, summary statistics computed using 2 day record.			
Mini-CTDs (Deployed July 19-25, 2005)			
M1	2.4	2.2	12.7
M2	11.9	2.6	26.3
M3	17.2	2.5	25.1

Classification Low And High Tide Cruises		
STATION	LOW TIDE SALINITY (psu)	HIGH TIDE SALINITY (psu)
C1	29.7	33.4
C2	27.3	33.2
C3	0.1	0.1
C4	11.6	9.1
C5	25.2	33.1
C6	19.4	32.1
C7	NA	16.5
C8	NA	7.4
C9	11.0	27.7
C10	NA	21.8
C11	0.0	0.0
C12	0.2	0.7
C13	0.1	1.6
C14	1.3	4.0
C15	0.1	0.1

Table 5-14. Observed and predicted isotope ratios (from two end member model) in Tillamook Estuary.

STATION	SAMPLING DATE	OBSERVED mean ± standard deviation (‰)	δ_{MIX} mean ± standard deviation (‰)	STATION USED FOR MIXING MODEL
N1	7/20/2005	6.1 ± 0.9	7.5 ± 1.0	Y1
			7.7 ± 1.0	C1
			7.3 ± 1.4	C2
N2	7/21/2005	3.2 ± 0.5	7.3 ± 1.4	C2
			2.0	C3
			7.5 ± 1.0	Y1
			2.6 ± 1.4	Y2
N3	7/22/2005	6.2 ± 0.6	7.0-7.2	C5
			6.2-6.5	C6
N4	7/23/2005	5.2 ± 0.4	3.5 ± 1.3	Y4
			2.6 ± 0.5	M1

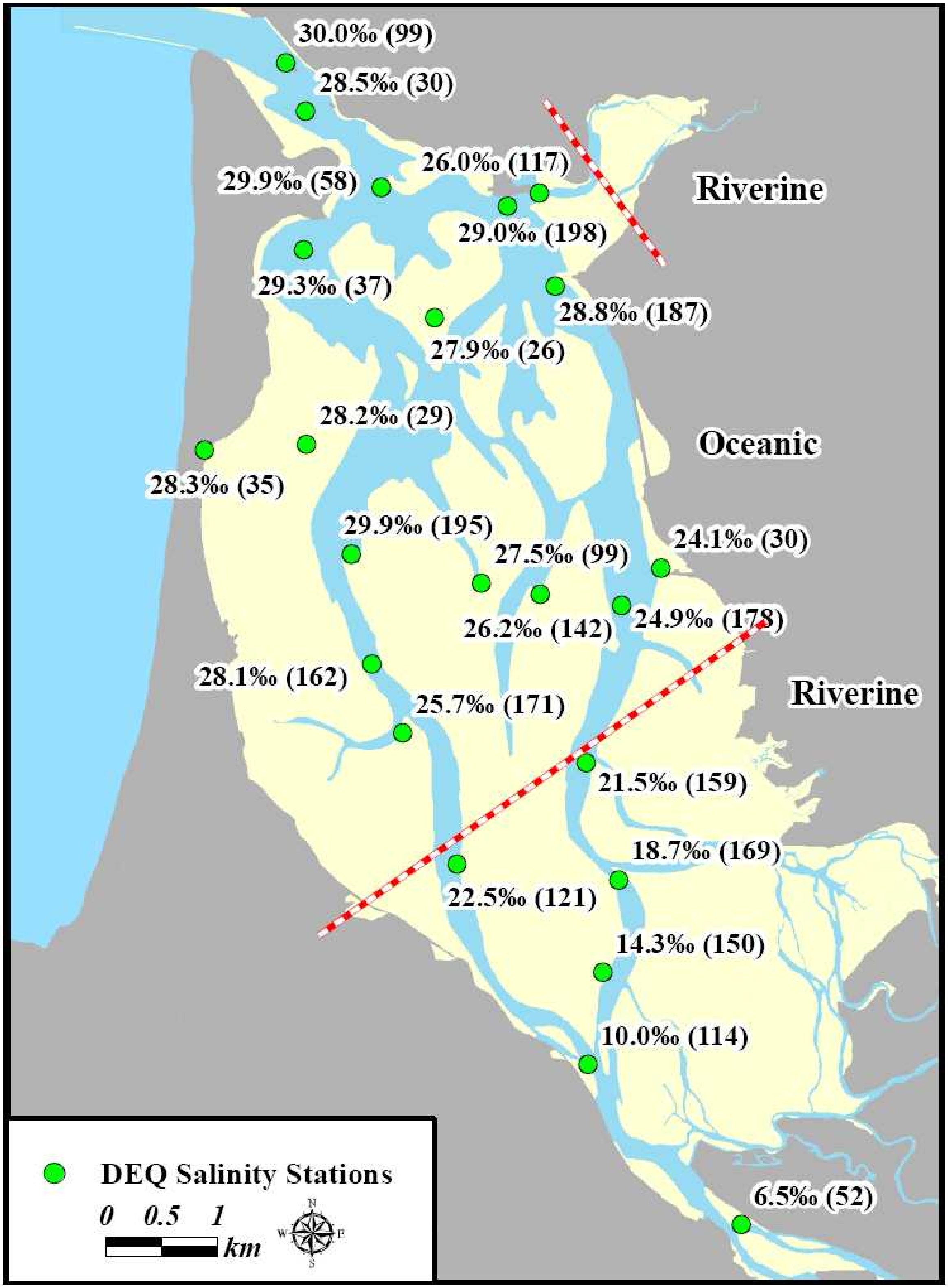

Figure 5-13. Median dry season salinities in Tillamook Estuary (with number of samples indicated in parentheses). The red dashed line shows the boundary between the oceanic and riverine segments.

5.8 Nestucca Estuary

Data used to determine the zonation in the Nestucca Estuary included salinity data from short-term YSI and Mini-CTD deployments in 2004 (Table 5-15) and macroalgae isotope data (Table 5-16) collected as part of the classification effort at two sites (N1 & N2). During our 2004 field work, we lost two instruments in the lower portion of the estuary. During September 2006, we collected additional data in this estuary for use in confirming zonation. A YSI datasonde was deployed near the mouth (Y2) and Mini-CTDs were deployed at 3 locations (M3-M5).

As discussed in Section 4.3, the Nestucca Estuary was sampled during low ocean nutrient conditions, which is reflected by DIN concentration of about 2 μM at Stations C8-C10 during the high tide cruise. Flood-tide water temperature from Yaquina during the 2 weeks prior to the isotope sampling indicated that the coastal ocean conditions were variable switching between upwelling and downwelling conditions (Figure 4-1). There were limited oceanic end member DIN data from Yaquina Estuary during the two weeks prior to the collection date (8/19/04) of the isotope sampling in Nestucca; the mean ocean end member DIN concentration from Yaquina during August 5th - 7th was 5.7 μM (n=6), and averaged over one month prior to the isotope sampling was 15.7 μM (n = 38). Due to this variability in ocean end member DIN concentrations and limited end member data available for the 2 weeks prior to sampling, we present results using C_o values of both 5.7 and 15.7 μM (Table 5-16). Since the river end member at Station C1 was fairly constant between the low and high tide cruises, we used the mean of the low and high tide cruises as the riverine end member (C_R = 38.4 μM). We did not have short-term time series salinity data in the vicinity of Station N1 during 2004 due to the loss of instruments. We used salinity data from Station C9 for the isotope mixing model and as an approximation, we also used time-series salinity data from Station Y2, which was collected in a similar time period (month) as the macroalgae samples from N1 but two years later. Using salinity data from Y2, the observed isotope ratio at N1 was similar to that predicted using the two-end member conservative transport mixing model. Based upon isotope ratio, Station N1 received about 70-90% of nitrogen from oceanic sources.

The observed isotope ratio at Station N2 was substantially higher than that predicted from the two-end member conservative mixing model (regardless of which value of C_o was used in the calculation), suggesting that there was an additional nitrogen source in the system or that denitrification was important (although see discussion Section 5.9). There is a WWTF located in the vicinity of Station N2, which discharges into the Nestucca River about 1.5 miles upstream of its confluence with Nestucca Estuary. In addition, there are two other minor WWTFs that discharge into the Nestucca River about 7 - 10 miles upstream of the mouth of the estuary. The presence of these three facilities suggests that WWTF input may be responsible for the observed deviation from the two-end member mixing model. Using Isosource constrained by the salinity data, the ocean contribution ranged between 8-13%, the riverine contribution ranged between 70-80%, and the WWTF contribution ranged between 10-20% (depending upon which end member is used).

Due to limited data availability, the southern boundary between the oceanic and riverine segments was based upon the salinity criterion (using data from Station M2). The northern boundary was based upon data from the high tide classification water quality cruises (salinity and dissolved inorganic nutrient data). Data from the high tide cruises suggested that ocean water (as indicated by high salinity and nutrient levels comparable to Station C10) reached Station C5 and there was a dramatic decrease in salinity between Stations C5 and C4 (a difference of about 19 psu). In addition, the high tide cruise NO_3^-+NO_2^- concentrations were low (< 2 μM) between Stations C10 and C5, reflecting the low ocean nutrient conditions during this cruise. There was an increase in NO_3^-+NO_2^- (of about 13 μM) between Stations C5 and C4 reflecting an increase in importance of riverine (or WWTF) nitrogen sources.

Using the conservative mixing model and the ocean and river end members for DIN (C_R = 38 μM and C_O = 15.7 μM), the boundary between ocean and river dominance occurred at a salinity of 24 psu. Based on median salinity, the estuary should be river dominated in the vicinity of Station M1 (median salinity of 12 psu). The salinity data collected during 2006 from Stations M3 and M4 confirmed that they were river dominated and that Station M5 was ocean dominated. There is some uncertainty in the zonation of this estuary (in particular the northern line) due to limited data.

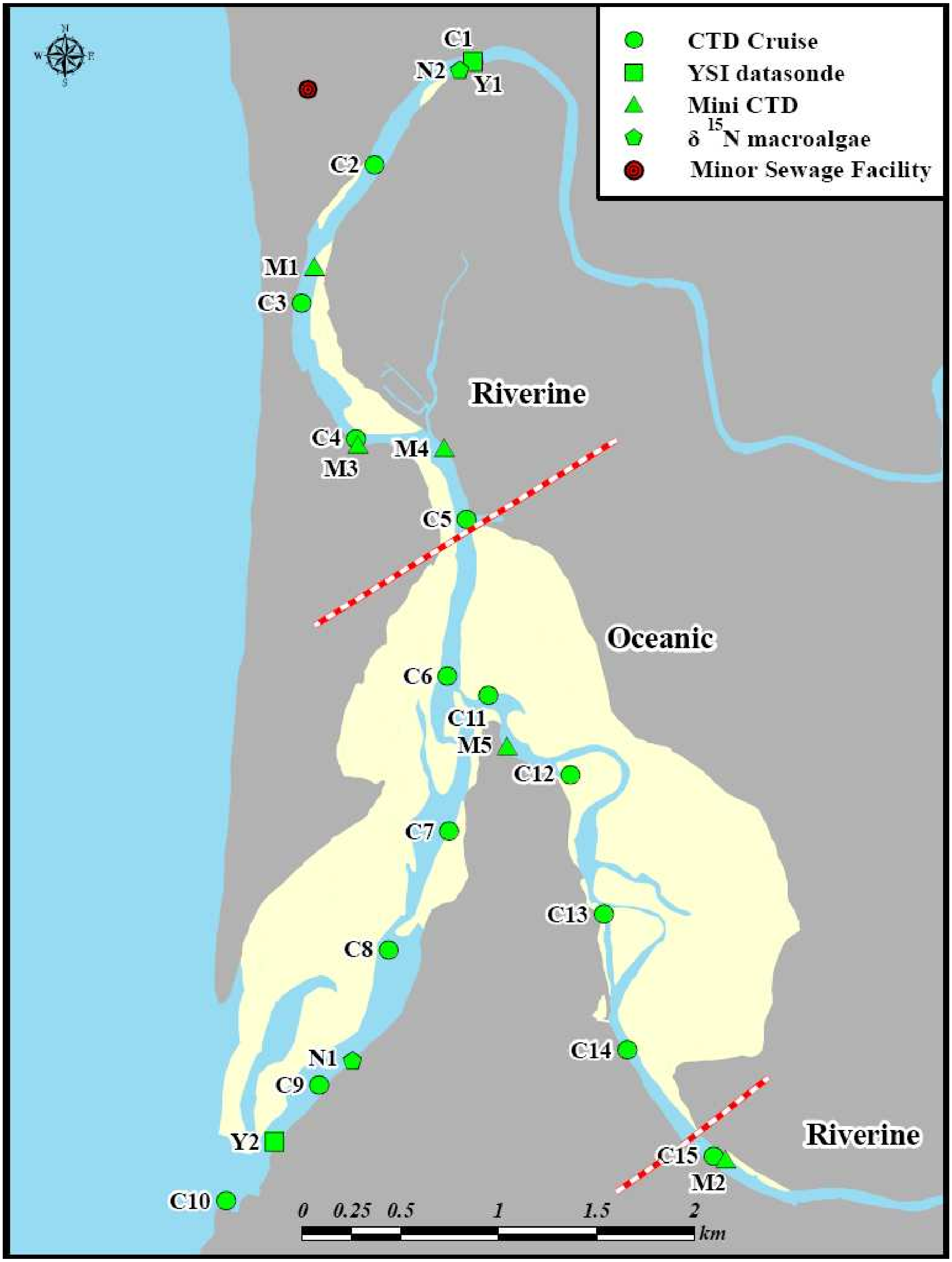

Figure 5-14. Location map of Nestucca Estuary showing the locations of datasondes, water quality stations, isotope samples, and WWTF. The red dashed line shows the boundary between the oceanic and riverine segments.

Table 5-15. Summary of dry season salinities in Nestucca Estuary.

STATION	MEDIAN (psu)	5TH PERCENTILE (psu)	95TH PERCENTILE (psu)
Datasondes (Y1 Deployed July 28 – August 19, 2004, And Y2 Deployed From August 29 – September 1, 2006)			
Y1	6.4	1.0	18.1
Y2	33.2	22.4	33.5
Mini-CTDs (M1 And M2 Deployed July 29 – August 19, 2004; M3 Deployed August 29-September 1, 2006, M4 Deployed September 1 – 6, 2006, M5 Deployed August 29 – September 6, 2006.)			
M1	11.6	0.0	32.6
M2	23.0	14.6	31.8
M3	8.1	3.1	28.3
M4	19.1	1.0	31.9
M5	28.5	17.6	32.3
Classification Low And High Tide Cruises			
STATION	LOW TIDE SALINITY (psu)	HIGH TIDE SALINITY (psu)	
C1	1.6	3.8	
C2	3.3	5.1	
C3	5.2	9.0	
C4	6.8	12.2	
C5	7.2	31.0	
C6	11.0	30.5	
C7	16.5	32.2	
C8	18.1	32.7	
C9	21.3	32.7	
C10	23.9	32.7	
C11	19.2	31.6	
C12	17.7	30.7	
C13	17.7	27.5	
C14	19.9	20.5	
C15	19.8	15.2	

Table 5-16. Observed and predicted isotope ratios (from two end member model) in Nestucca Estuary.

STATION	SAMPLING DATE	OBSERVED mean ± standard deviation (‰)	δ_{MIX} calculated using C_o = 5.7 μM (‰)	δ_{MIX} calculated using C_o = 15.7 μM (‰)	STATION USED FOR MIXING MODEL
N1	8/19/2004	6.6 ± 0.1	3.6 ± 0.4	5.1 ± 0.5	C9
			7.0 ± 1.7	7.6 ± 1.2	Y2
N2	8/19/2004	5.3 ± 1.4	2.3 ± 0.3	2.8 ± 0.7	Y1
			2.1 ± 0.1	2.2 ± 0.1	C1

5.9 Synthesis and Uncertainties in Zonation

Utilizing $\delta^{15}N$ of green macroalgae combined with short-time series of salinity, we divided the target estuaries into oceanic and riverine segments. In addition, using Isosource constrained by the salinity data we estimated the contribution of WWTF inputs. Based on our analysis, oceanic and riverine inputs were the dominant nitrogen sources within the target estuaries. The maximum contribution of WWTF estimated from stable isotope data was 30%. We feel that our analysis is robust since it uses multiple lines of independently gathered evidence which lead to similar conclusions. The analysis utilizing nitrogen stable isotopes as a tracer of oceanic and riverine nitrogen inputs appears to be consistent with zonation based upon median salinities. Depending upon the DIN concentration in the riverine end member, the zonation occurred at median salinities ranging from 18 to 25 psu. Shirzad et al. (1988) divided Oregon estuaries into three zones (seawater (≥ 25 psu), mixing (0.5-25 psu), and tidal fresh (0-0.5 psu) using average annual and depth-averaged salinity data. They included zonations for Alsea, Yaquina and Tillamook estuaries. Our oceanic zone extends further upriver (1.6, 3.2, and 8 km for Alsea, Tillamook and Yaquina, respectively) than the seawater zones presented in Shirzad et al. (1988). This is probably a result of our zonation being based upon dry season conditions, whereas the ones in Shirzad et al. (1988) are based upon average annual conditions.

Treated effluent associated with WWTF is discharged into six of the target estuaries. In Yaquina, Umpqua River and Nestucca estuaries, the WWTF effluent is discharged into the riverine segment, while in the Alsea and Coos estuaries the discharge is in the oceanic segment. Tillamook Estuary has multiple WWTFs discharging into it, one into the oceanic segment and two into the riverine segment. Effluent discharged into oceanic segments would experience strong tidal mixing, and the effluent would remain in the estuary for a shorter time period. One method that may aid in determining the susceptibility of estuaries to nutrient enrichment would be to overlay point source inputs on the ocean-river zonation.

There is some uncertainty in the estuarine zonation as a result of data gaps as well as interannual variability. This analysis was based upon one-time sampling of natural abundance stable isotope of green macroalgae. When possible, we compared our stable isotope values to other data available for the target estuaries. Coastal systems are highly dynamic, responding to environmental forcing at scales ranging from minutes (e.g. changes in tidal elevation) to days (e.g. storm tracks) to decades (e.g. the Pacific Decadal Oscillation). The zonation described here is based primarily on single sampling events or a few days of sampling. Consequently, it is imperative to recognize that these zonation schemes are intended to provide guidance at a gross scale. We are most confident in the results for simpler estuaries (e.g. one riverine source) and systems where extensive datasets were available (e.g., Alsea, Salmon and Yaquina). When possible, we compared our value of riverine and oceanic end members to values from other sources. These comparisons revealed that the river end members we utilized were consistent with historical data. For estuaries where there are multiple rivers flowing into the estuary (e.g., Coos and Tillamook) there is more uncertainty in the results given the logistical and financial constraints of sampling multiple riverine inputs.

One of the limitations of this analysis is that we cannot distinguish human (WWTF or septic) from animal waste because their nitrogen stable isotope ratios are similar. Other chemical tracer analyses can make this distinction; however, these methods were not used in this study.

Additionally, we assume that macroalgae do not fractionate (preferentially take up ^{14}N relative to ^{15}N) during nitrogen assimilation, which is supported by the recent experimental work of Cohen and Fong (2005). We are currently conducting laboratory experiments to critically evaluate this assumption for green macroalgae collected from Yaquina Estuary. In addition, our method assumes that the nitrogen stable isotope ratio is a result of a mixture of oceanic, riverine and WWTF sources, and that the end member values are the same for all of the target estuaries. We believe this is a good assumption due to the similarity in land cover within the watersheds of the seven estuaries (Table 3-3) and the similarity in ocean conditions over the geographic extent of our study (Section 4.3). Of the target estuaries, Coos and Umpqua had the least amount of alder in the watershed (See Section 4.2). Fry et al. (2001) sampled water column nitrate and green macroalgae in the Coos Estuary. The $\delta\ ^{15}N$ of nitrate at a riverine site averaged +1.7‰ and of macroalgae averaged 1.8 ‰, which is consistent with our riverine end member. Umpqua had the lowest amount of red alder in the watershed (< 0.5%), and the most riverine macroalgae sampling sites there had a minimum isotope ratio of +5.4 ‰. If the riverine end member for Umpqua was not +2 ‰, we may have over-estimated the contribution from WWTF inputs. The watersheds of the seven estuaries are primarily forested and have low population densities. If there is an additional source that we have neglected or if denitrification is important, we may have incorrectly calculated the contribution of the sources. Kendall and McDonnell (1998) recommend that the contribution of nitrogen sources from natural abundance stable isotopes be confirmed through an independent non-isotopic method. For Yaquina Estuary, we were able to confirm the isotopic contribution of sources using the transport model.

Denitrification can result in elevation of the $\delta\ ^{15}N$, which can be misinterpreted as WWTF input in our analysis. Kendall and McDonnell (1998) estimated that denitrification of fertilizer nitrate with a $\delta^{15}N$ of 1‰ can result in the residual nitrate having an isotope ratio of 15‰, which is similar to animal or human waste. The importance of denitrification in nitrogen removal in estuaries is a function of residence time (Dettmann, 2001). Denitrification represents a significant loss of nitrogen in estuaries with relatively long residence times. The residence times of the target estuaries are relatively short (less than 1 month) due to their small volume, strong tidal forcing, and high freshwater inflow (see Tables 3-6 and 3-7). We would expect that the importance of denitrification would increase as the riverflow declines. However, based on the analysis of Dettmann (2001) and residence times (Table 3-7), less than 10% of the nitrate in the target estuaries would be denitrified. Furthermore, the ability of the Yaquina transport model to reproduce the spatial and temporal patterns in the isotope data strongly suggests that the nitrogen stable isotope ratio of the macroalgae is being determined primarily by a mixture of the three nitrogen sources (oceanic, riverine, and WWTF) rather than by denitrification. Our analysis also assumes that the $\delta\ ^{15}N$ of the sources is the same across the region. Comparisons of macroalgal nitrogen stable isotope ratios between Yaquina and Alsea are similar for both the ocean and river end members (Figures 5-3 and 5-8). In addition, macroalgae $\delta\ ^{15}N$ from Coos Estuary by Fry et al. (2001) further support the validity of this assumption. This analysis also assumes that there is minimal temporal variability in the $\delta\ ^{15}N$ of the end-members. The ability of the Yaquina transport model to predict the $\delta\ ^{15}N$ reliably using fixed values for the end-members (Figures 5-4 and 5-5) indicates that this is a reasonable assumption. Finally, we assume that the WWTFs represent a minor contribution of the total freshwater input to these systems. Riverine flow rates are many times larger than WWTF inputs even during periods of low river flow, suggesting that this is a good assumption.

Since we sampled intertidal macroalgae, there is some potential that our analysis would be biased (over estimating oceanic contribution of nitrogen) due to elevation of the samples. If macroalgal samples are collected at a high elevation relative to mean lower low water, then they may only be exposed to water during higher water elevations, which would tend to occur during flood tides. Since we primarily are concerned with nitrogen sources that seagrass habitat is exposed to and most of the seagrass habitat is located in the intertidal, this is probably not a significant source of error. This error would be more significant for highly stratified estuaries, which is typically not the case for Oregon estuaries during the dry season (Tables 3-1 and 3-6). Model simulations also reveal that this error would be most important in the lower portions of the estuaries compared to the upper.

In our study area, most of the nitrogen sources to the estuary are isotopically distinct (Figure 5-1) which leads to a relatively clean 2 or 3 end-member mixing solution. As discussed in Section 3.7, watershed nitrogen inputs in the area are generally related to red alder (*Alnus rubra*) a nitrogen fixing species characterized by $\delta^{15}N$ ranging between -3 and -0.5 ‰ (Hobbie et al., 2000; Tjepkema et al., 2000; Cloern et al., 2002). Samples from the Yaquina River indicate that the $\delta^{15}N$ of the nitrate in the river water ranges from +0.4 to +2.4 ‰ (Kaldy, unpublished), which is consistent with macroalgal $\delta^{15}N$ values for our riverine sites. Literature values for the $\delta^{15}N$ of oceanic nitrate (the dominant nutrient associated with upwelling) along the PNW coast of the United States range between +6.6 and +7.7 ‰ (Kienast et al., 2002; Wankel et al., 2006). The $\delta^{15}N$ of the nitrate for water samples collected from 25 miles offshore of the Yaquina Estuary range from +6.7 to +7.6 ‰ (Kaldy, unpublished). The $\delta^{15}N$ values of green macroalgae utilizing recently upwelled water along the Oregon coast typically range between +7 and +9 ‰ (Fry et al., 2001; and Kaldy, unpublished). WWTF effluent generally has high $\delta^{15}N$ with values ranging between +7 and +25 ‰ (Heaton, 1986; Jones et al., 2003). Future work includes the measuring the isotope ratios of WWTF effluent from the local sources. Agricultural fertilizer inputs to coastal Oregon estuaries are minimal since there are no major agricultural crops cultivated along the coast. The amount of cultivated land in the watersheds of the seven target estuaries varied from 0 to 0.22% (Table 3-3). In some localized areas animal waste inputs may be substantial, notably Tillamook Estuary; however our isotope analyses cannot distinguish between animal and human waste.

The use of macroalgal $\delta^{15}N$ to identify nitrogen sources in estuaries has received attention in the recent scientific literature (Cohen and Fong, 2005). Unfortunately, the use of $\delta^{15}N$ is not always clear cut. Nitrogen dynamics are extremely complex, mediated by a variety of microbes. Different microbial biochemical transformations have specific isotope fractionation factors such that the products have different $\delta^{15}N$ values than the initial reactants (Fry et al., 2003). Consequently, caution must be used in the interpretation of $\delta^{15}N$ data, in particular neglecting the ocean end member. For example, the direct application of the $\delta^{15}N$ regression equations from Cole et al. (2004) to the data presented here would erroneously suggest that WWTF have a much greater impact on estuarine nitrogen dynamics. The background marine signal associated with upwelling is a departure from the general nutrient loading paradigm (e.g. nutrients primarily from anthropogenic watershed sources). The potential error can result from not accounting for all nitrogen sources and is likely to occur in areas where ocean upwelling is a dominant feature of seasonal nutrient inputs as it is along the Pacific Coast of the U.S. (Fry et al., 2003; Cole et al., 2004). We have used multiple lines of evidence to support our conclusions. Specifically, we have used two-end member mixing models of $\delta^{15}N$ and salinity as well as Isosource (Phillips and Gregg, 2003) and a transport model. This approach was validated for the Yaquina Estuary and we are encouraged that all three of these approaches provide very similar zonation patterns.

Based on the zonation scheme described in this chapter, five of the target estuaries had >50% of the total estuarine area classified as ocean dominated. The Salmon and Umpqua River estuaries were classified as having 29 and 30%, respectively, of total estuarine area classified as ocean dominated. Salmon River Estuary is the smallest of all the systems examined encompassing only 3.1 km^2 of total estuarine area. As discussed in Section 4.4.3, both of these estuaries switch from ocean- to river-dominated over a tidal cycle and based on this we classified them as having the strongest river influence (Figures 4-8 and 4-9). Most systems were ocean dominated, indicating that the influence of the coastal ocean and the dynamics of oceanographic phenomena cannot be ignored. The strong influence of the coastal ocean on PNW estuaries is a departure from the usual paradigm associated with nutrient loading.

Table 5-17. Summary of the total estuarine area for each target estuary and the area and percentage of total in the oceanic and riverine segments based on salinity and macroalgal isotope ratios.

AREA (KM^2)	YAQUINA	ALSEA	SALMON	UMPQUA	COOS	TILLAMOOK	NESTUCCA
Total Estuarine	19.9	12.5	3.11	33.8	54.9	37.5	5.00
Oceanic Segment	13.4	7.75	0.91	10.1	42.7	23.4	4.07
Riverine Segment	6.58	4.72	2.19	23.7	12.1	14.0	0.93
% in Oceanic Segment	67	62	29	30	78	63	82
% in Riverine Segment	33	38	71	70	22	37	18

CHAPTER 6:
AERIAL MEASURES OF ESTUARINE INTERTIDAL AND SHALLOW SUBTIDAL *ZOSTERA MARINA* COVERAGE

David R. Young, Patrick J. Clinton, Henry Lee II, David T. Specht, and T Chris Mochon Collura

Key Findings

- **Intertidal and shallow subtidal distributions of *Zostera marina* were mapped in the seven target estuaries via aerial and on-ground surveys.**
- **The extent of *Z. marina* varied among the estuaries, ranging from non-detectable via aerial surveys to about 11% of the intertidal area.**
- **The majority of the intertidal and shallow subtidal *Z. marina* is found in the oceanic segments of the estuaries.**
- **Most of the *Z. marina* habitat is found at depths ranging from -3 to 3 feet (-0.9 m to 0.9 m) above MLLW.**

6.0 Introduction

Zostera marina is a flowering marine plant that can form thick meadows or beds of perennial plants in the intertidal and subtidal sections of estuaries, providing a critical habitat and food source for numerous taxa including commercially important fish and shellfish (Heck et al., 1989; Sogard and Able, 1991; Dennison et al., 1993; Bostrom and Bonsdorff, 1997). *Z. marina* is one of the many species of seagrasses that have been severely impacted by anthropogenic activities around the world (Hemminga and Duarte, 2000; Short et al., 2001). One cost-effective approach of assessing current seagrass distributions as well as changes in distributions is through the use of aerial surveys. However, the intertidal and subtidal distribution of *Z. marina* presents challenges in mapping its distribution. Therefore, an aerial survey method of mapping the intertidal and shallow subtidal portions of the *Z. marina* habitat in PNW estuaries has been developed by Clinton et al. (2007). A method of mapping the deeper subtidal portion of the distribution not visible from the surface is still under development and testing (Dr. Ted DeWitt, pers. comm.).

The objectives of this aerial photomapping study were to 1) assess the extent and distribution of intertidal and shallow subtidal *Z. marina* in the seven target estuaries, and 2) determine the relative distribution of *Z. marina* in the oceanic and riverine segments of each of the estuaries.

6.1 Methods

6.1.1 Aerial Photography Sampling Frame and Design

Owing to the extensive eelgrass distribution in PNW estuaries, this component of the project used a remote sensing technique. The method employed aerial photographs and false-color near-infrared (color infrared, CIR) film, which provide better contrast than full-color film in distinguishing submersed aquatic vegetation (SAV) distributions (Young et al., 1999). However, because CIR film cannot resolve images more than a few cm below the water surface, the photographs must be taken during exposed intervals (i.e., daylight low tides). In addition, the weather must be cloud-free (or uniform high overcast) to obtain uninterrupted photo survey coverage. In addition, the presence of benthic green macroalgae can confound the interpretation of the SAV signals; thus, it is important to acquire the photography of target estuaries during late spring or early summer, when there is enough *Z. marina* growth and sunlight for imaging, but before the summertime bloom of benthic green macroalgae (see Section 7.2.4). Although most of the photography was conducted when the tide level was between 0 and -2 ft (-0.6 m) relative to Mean Lower Low Water (MLLW), the upper portion of the immersed eelgrass plants in the shallow subtidal zone (down to about -6 ft or -1.8 m) was floating on the surface and could be detected by the CIR film. Thus, the range of the intertidal/shallow subtidal zone sampled by this technique was from the upper margin of the zone at about +6 ft (+1.8 m) to about -6 ft (-1.8 m), the approximate lower depth limit of detection. (English units are used for most available tide level and bathymetry data in the PNW, and thus are the primary units used here). The lateral extent of a study area generally ranged from about the ocean entrance of the estuary to the upriver termination of the reported distribution of intertidal *Z. marina* in that system. In the Yaquina Estuary, the aerial photography extended only about 11 km upriver from the ocean entrance; thus, this study area did not include the entire riverine segment. However, numerous boat surveys by the authors have shown that only a small proportion of the intertidal/shallow subtidal *Z. marina* occurs upriver of the study area.

The photosurveys were conducted under contract by a commercial vendor. Maps of the required flightlines and photocenters, tables of daylight low tide windows, and specifications of the photography (e.g., large-format camera focal length, front and side overlap, maximum aircraft tilt, camera calibration, etc.) were provided by PCEB. The photoscale used in the 2004 photography was 1:10,000, and in 2005 was 1:20,000. The resultant film (or diapositive copy) was digitally scanned, and the digital photographs then were converted to digital photomaps with pixels corresponding to 0.25 m x 0.25 m on the ground using ERMapper® desktop orthorectification. These orthophotos were mosaiced for each estuary and then each pixel was classified into one of two classes, defined as: (1) eelgrass bed (>10% cover by *Z. marina*) or (2) bare substrate (≤ 10% cover by *Z. marina*).

Image classifications were accomplished with a combination of digital image processing and manual techniques. Terrestrial portions of the image were masked in ERMapper® using vector polygons, and a vegetation index algorithm was applied to mask patently unvegetated areas in the remaining imagery. A seven class unsupervised classification using ERMapper® ISOCLASS algorithm was applied to the masked imagery and the results converted to seven ArcInfo® format binary grids representing each band of the isoclassification. The grids were manually edited in ArcMap® by the photointerpreter using ArcScan® raster editing tools to remove false-positive pixels and recompiled to form the seagrass map.

An example of the distribution of *Z. marina* coverage in the Yaquina Estuary obtained from the classification of the April 2004 digital photomaps is presented in Figure 6-1. The aerial photography did not yield acceptable images of *Z. marina* beds in the Nestucca and Salmon River estuaries. For these estuaries, a vessel equipped with a differential-corrected global positioning system (DGPS) was used to position the visible edges of intertidal and shallow

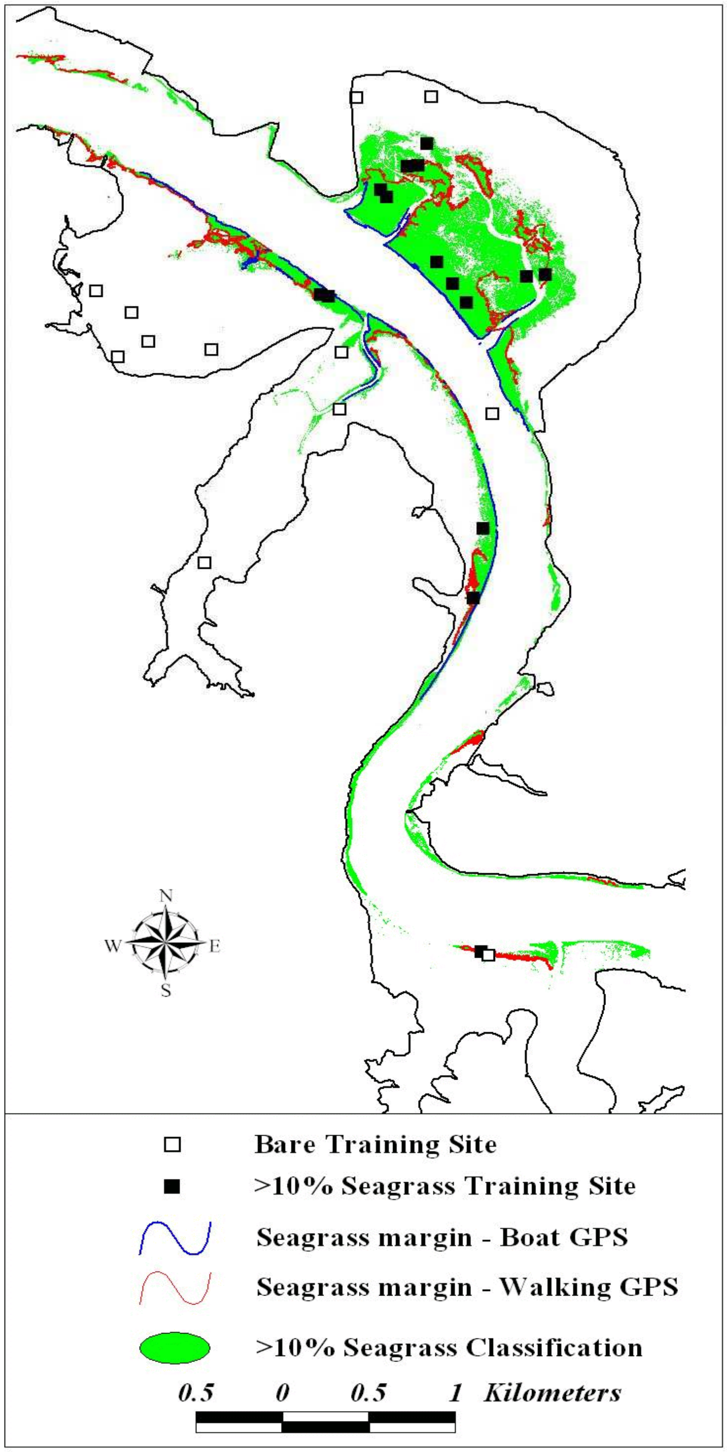

Figure 6-1. Intertidal and shallow subtidal distribution of *Z. marina* from digital image classification of aerial photos of Yaquina Estuary taken in April 2004. The eastern (upriver) edge of the survey area was selected because little intertidal *Z. marina* occurs beyond the boundary shown.

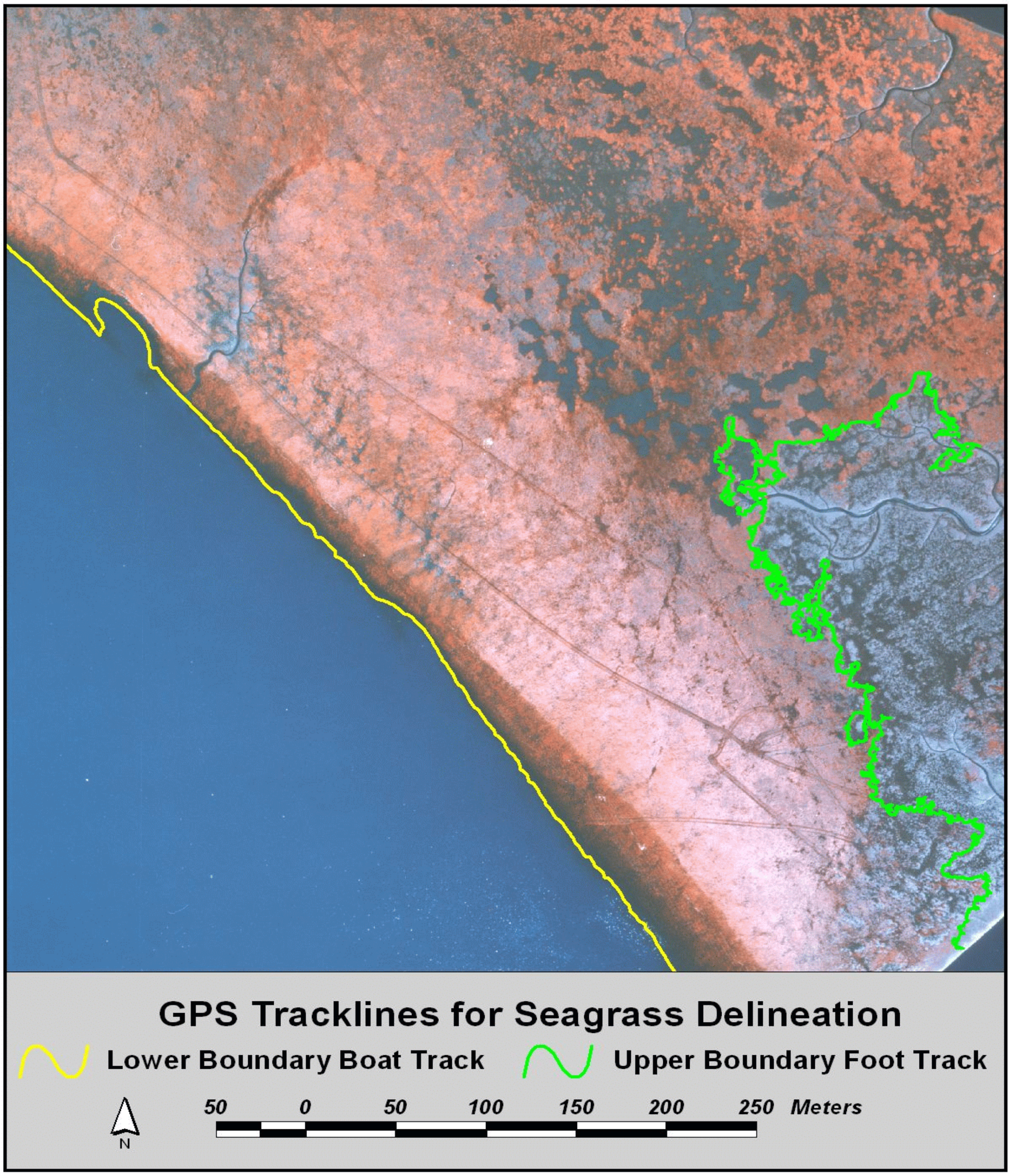

Figure 6-2. Track of lower margin of surface-visible distribution of *Z. marina* in the near-subtidal zone (using DGPS in a small boat), and of upper margin (walking with DGPS), in the Yaquina Estuary.

subtidal *Z. marina* beds within each estuary. In some cases, upper margins also were positioned by carrying a DGPS while walking this margin during exposed conditions. The root mean square error of the DGPS system used in this study was ± 0.6 m. An example of such positionings of the visible edges of an intertidal and shallow subtidal *Z. marina* meadow in Yaquina Estuary is presented in Figure 6-2.

6.1.2 Estuaries Surveyed

The sampling dates and mapping method utilized for the seven target estuaries are presented in Table 6-1.

Table 6-1. Mapping method utilized and dates of mapping for intertidal and shallow subtidal *Z. marina* distributions in the target estuaries.

ESTUARY	DATE	MAPPING METHOD
Alsea	April 9, 2004 Summer 2005	Aerial Orthophotography/ Surface Positioning
Coos	May 26, 2005	Aerial Orthophotography
Nestucca	Summer 2004	Surface Positioning
Salmon	Summer 2004	Surface Positioning
Tillamook	April 9, 2004	Aerial Orthophotography
Umpqua	July 24, 2005	Aerial Orthophotography
Yaquina	April 9, 2004 July 23, 1997 (above river mile 7)	Aerial Orthophotography

6.1.3 Ground Surveys

"Ground truth" surveys were conducted in the five estuaries surveyed by aerial photography usually in the same season that the photosurvey was conducted but sometimes a year or two later. The methodology was based upon the recommendations of Congalton and Green (1999). First, the most reliable map available of the target estuarine intertidal zone was used to separate expected *Z. marina* from bare substrate strata. One hundred stations then were positioned randomly within each stratum. Utilizing hovercraft transport and the high-resolution DGPS, 30-70 stations in each class were located during low tide, positioned, and surveyed for percent cover of *Z. marina*. A 1.25 m x 1.25 m quadrat, equipped with two orthogonal sets of five taut strings, was placed successively in the four compass quadrats around the target position. The percent cover of native *Z. marina*, green macroalgae, non-native *Z. japonica*, or bare substrate then was quantified using the point-intercept method. Following the criterion established by the NOAA Coastal Change Analysis Program (Dobson et al., 1995), stations with *Z. marina* coverage greater than 10% were classified as *Z. marina* sites. Interference by benthic green macroalgae or *Z. japonica* in the separation of ground stations into *Z. marina* or bare substrate classes was negligible in all five estuaries surveyed. Approximately 10% of the stations surveyed in each class were randomly withdrawn and used for training the geographical information system (GIS) photointerpreter conducting the image classification. The data for these stations were not used in the accuracy assessments of the resultant classifications.

6.1.4 Classification Accuracy Assessments

Following the classification of the digital photomaps, positions of the non-training stations determined on-site by DGPS were provided to the GIS photointerpreter for the five estuaries in which ground surveys were conducted. An area equivalent to 2.5 m x 2.5 m on the ground around a station's DGPS position was subsampled from the digital classification into a 10 pixel x 10 pixel array. Each pixel had been classified either as "*Z. marina*" or "bare substrate." If the number of pixels classified as *Z. marina* was greater than 10 (>10% of the total), that station was classified as *Z. marina*. Otherwise it was classified as a bare substrate station. The same criterion was applied to the ground survey data. The results from the digital classification and ground survey then were incorporated into a classical error matrix (Congalton and Green, 1999).

6.1.5 Estuary Bathymetry

In view of the importance of substrate elevation to estuarine ecology, a bathymetric model was developed for Yaquina Estuary. Several surveys of the main channel have been conducted by the U.S. Army Corps of Engineers (USACE) in recent years, but little information existed for the extensive tide flats of the estuary. Therefore, in 2002 WED/PCEB contracted with USACE to extend their surveys into the shallow sectors of the estuary. Soundings were conducted at extreme high tides near the end of the year along transect lines every 200 feet (67 m) over the general area surveyed by aerial photography. Measured water depths were related to those recorded by tide gauges within the estuary, and adjusted relative to Mean Lower Low Water. The total area covered by the several bathymetric surveys utilized in this study is illustrated in Figure 6-3. The data resulting from these surveys were used by PCEB to construct a bathymetric model of Yaquina Estuary. Survey easting, northing, and depth values were interpolated using the TOPOGRID method provided in ArcInfo Workstation. This model is discussed further in Section 6.2.3.

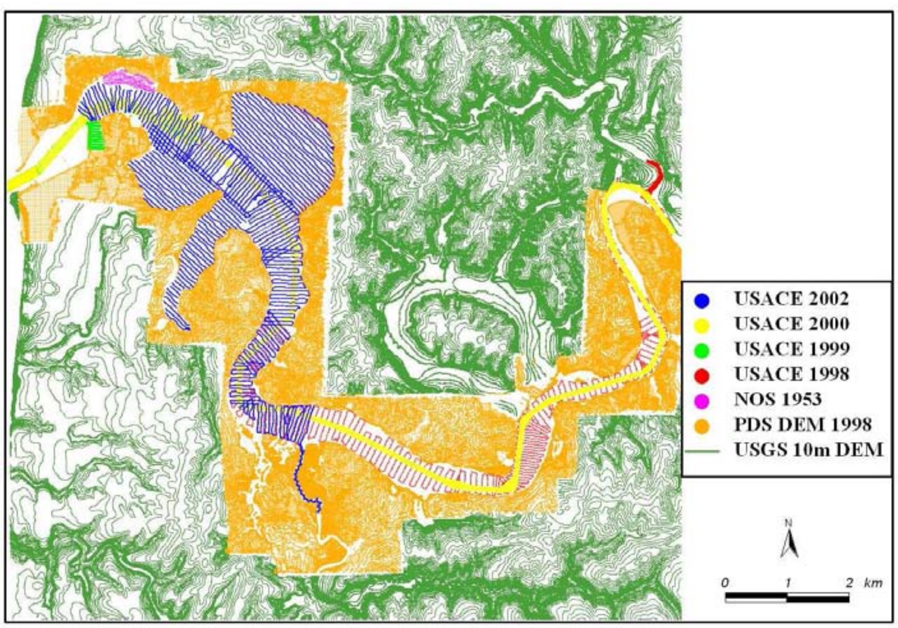

Figure 6-3. Sectors of Yaquina Estuary covered by bathymetric surveys conducted by the U.S. Army Corps of Engineers between 1998 and 2002.

6.2 Results

6.2.1 Accuracy Assessments of Photomap Image Classifications

Error matrices were prepared from the digital classification and ground survey results for the Alsea, Coos, Tillamook, Umpqua, and Yaquina estuaries. Errors of omission (failing to include a ground-truth station in the classification), commission (erroneously including a station in the classification), and overall error for the classifications of the five orthophotographs are listed in Table 6-2. These results for five of the seven target estuaries indicate that the *Z. marina* habitat and bare substrate classifications from the estuary orthophotographs have accuracies of 83% or greater, and the median overall accuracy was 89% (median overall error of 11%). For more details on the accuracy assessment of the mapping see Section B.9 in Appendix B.

Table 6-2. Summary of error matrix results for detection of *Z. marina* in five of the target estuaries based on the ground truth surveys.

ESTUARY CLASSIFIED	ERRORS OF OMISSION	ERRORS OF COMMISSION	OVERALL ERROR	NO. OF STATIONS
Alsea				
Z. marina	0%	0%	0%	44
Bare Substrate	0%	0%		60
Coos				
Z. marina	26 %	12 %	17 %	38
Bare Substrate	12 %	26 %		77
Tillamook				
Z. marina	12%	16%	14%	100
Bare Substrate	16%	12%		81
Umpqua				
Z. marina	10 %	11 %	11 %	41
Bare Substrate	11 %	10 %		91
Yaquina				
Z. marina	2 %	4 %	3 %	51
Bare Substrate	4 %	2 %		28

6.2.2 Among-Estuary Comparison of *Zostera marina* Coverage

Interpretation of the distribution and abundance of *Z. marina* from the aerial surveys and from the probabilistic field surveys discussed in Chapter 7 requires an understanding of how areas were sampled in the two approaches. The aerial photography detects *Z. marina* in both the intertidal and shallow subtidal zones. As discussed above (Section 6.1.1), it appears that the aerial photography detects *Z. marina* to a depth of approximately 6 feet (1.8 meters) below Mean Lower Low Water (MLLW). In comparison, the probabilistic field surveys used an intertidal sampling frame, approximately from Mean Lower Water (MLW) to Mean Higher Water (MHW). Thus, the probabilistic surveys would not capture the subtidal portion of the SAV population that was included in the aerial surveys. To compare directly the areal extents of *Z. marina* between the two surveys, the sampling frames used in the probabilistic surveys were overlaid on the aerial photography, generating estimates of the areas of *Z. marina* from the aerial photography within the same areas as sampled by the field surveys. Additionally, to calculate the percent of the area covered by *Z. marina* (relative cover) it is necessary to have an accurate estimate of the area actually surveyed by the aerial photography. Such areas could be calculated assuming detection to 6 feet (1.8 m) if bathymetric data were available. However, bathymetric data are currently available for only three of the seven target estuaries. Therefore, the percent cover of *Z. marina* was calculated using the portion of the aerial surveys that fell within the probabilistic sampling frames, which have known areas. These percent cover values only apply to the intertidal zone; future efforts will be directed at obtaining bathymetry for all the target estuaries, allowing percent cover estimates that also include the shallow subtidal zone.

To compare the extent of variation of *Z. marina* among estuaries (Objective 1), the absolute and relative areas of *Z. marina* in the five digitally classified estuaries, and the two surface-mapped estuaries were calculated (Table 6-3). Maps of the aerial extent of *Z. marina* for each of the target estuaries are presented in Appendix A. One comparison of interest is the percent coverage values for intertidal *Z. marina* within the probabilistic frame. Three of the estuaries – Coos, Tillamook, and Yaquina – show agreement within a factor of three (4.7 to 11.5% cover). For the Umpqua River Estuary, the percent coverage value (1.3%) is substantially lower. The Alsea, Nestucca, and Salmon River estuaries have very little intertidal *Z. marina* coverage within the probabilistic frame. At present we have no complete explanation for the near absence of intertidal *Z. marina* in the Nestucca and Salmon River estuaries. However, one possible contributing factor may be the high wave energy in these estuaries. Nestucca and Salmon River estuaries have the lowest values for median percent fines (1.9% and 7.9%, respectively; see Figure 7-10), suggesting that the intertidal zones of these estuaries are high energy environments, which may result in *Z. marina* being eroded or buried. Additionally, high tidal currents and shifting sands were commonly observed at the mouth of Nestucca during peak flood and ebb tides (see Section 8.2.5). Salinity may also be a contributing factor limiting *Z. marina* in these estuaries as discussed in Section 7.2.2.

6.2.3 Bathymetric Distribution of *Zostera marina*

A comparison of elevations predicted by the bathymetric model for Yaquina estuary with those obtained from an independent total station survey of a large embayment there (Idaho Flat) is presented in Figure 6-4. This comparison indicates that there is agreement between the measured and modeled elevations to within about 0.3 ft. Specifically, for the 167 points within the depth interval -3.0 ft to +3.0 ft, the median value for the difference between the survey and model elevations is -0.32 ft, indicating that, on average, the actual substrate elevations in this interval may be approximately 0.3 ft (0.1 m) lower than the model elevations.

Bathymetric data collected by Professor Chris Goldfinger at Oregon State University for Alsea Estuary (2002) and by the USACE for Tillamook Estuary (1995) similarly were used to obtain bathymetric models for these estuaries. The results presented in Figures 6-6 to 6-8 indicate similar bathymetric distributions for intertidal/shallow subtidal *Z. marina* in these three PNW estuaries. The aerial percentages of *Z. marina* occurring between specific depth intervals around MLLW are summarized in Table 6-4. Again, the population included in this study is *Z. marina* that is visible from the surface at low tide (MLLW), falling within the approximate depth range: -6 ft (-1.8 m) to +6 ft (+1.8 m).

Table 6-3. Estimates of *Z. marina* area in the target estuaries from the aerial/on-surface surveys. The "Total Area of *Z. marina*" is the total area of *Z. marina* detected in the aerial/on-surface surveys in the intertidal and shallow subtidal areas. The "Area of *Z. marina* within Probabilistic Frame" is the area of *Z. marina* found within the intertidal probabilistic frame, which does not include the shallow subtidal area sampled by the aerial photographs. The "% Coverage of *Z. marina* within Probabilistic Frame" is the percent coverage of *Z. marina* relative to the area of the probabilistic frame. A similar metric cannot be calculated for the entire aerial frame since the exact area sampled by the photographs is unknown. The areas of *Z. marina* in the Nestucca and Salmon River estuaries were determined by on-surface mapping rather than aerial photography.

ESTUARY	TOTAL AREA OF *Z. MARINA* IN AERIAL/ ON-SURFACE SURVEY (km^2)	AREA OF *Z. MARINA* WITHIN PROBABILISTIC FRAME (km^2)	% COVERAGE OF *Z. MARINA* WITHIN PROBABILISTIC FRAME
Alsea	0.026	0.005	0.09
Coos	2.141	1.238	4.66
Nestucca	0.004	0.0	0.00
Salmon	0.004	0.0003	0.07
Tillamook	3.27	2.38	11.46
Umpqua	0.338	0.095	1.33
Yaquina	0.809	0.635	9.91

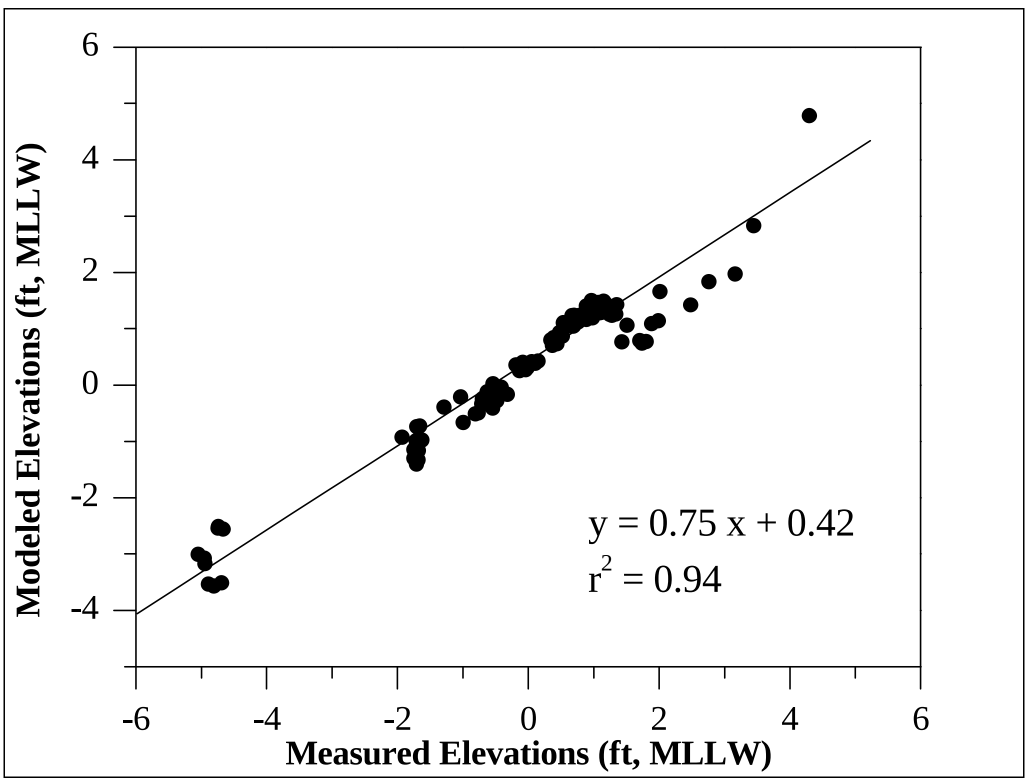

Figure 6-4. Comparison of substrate elevations from a bathymetric model and independent total station survey measurements within Yaquina estuary.

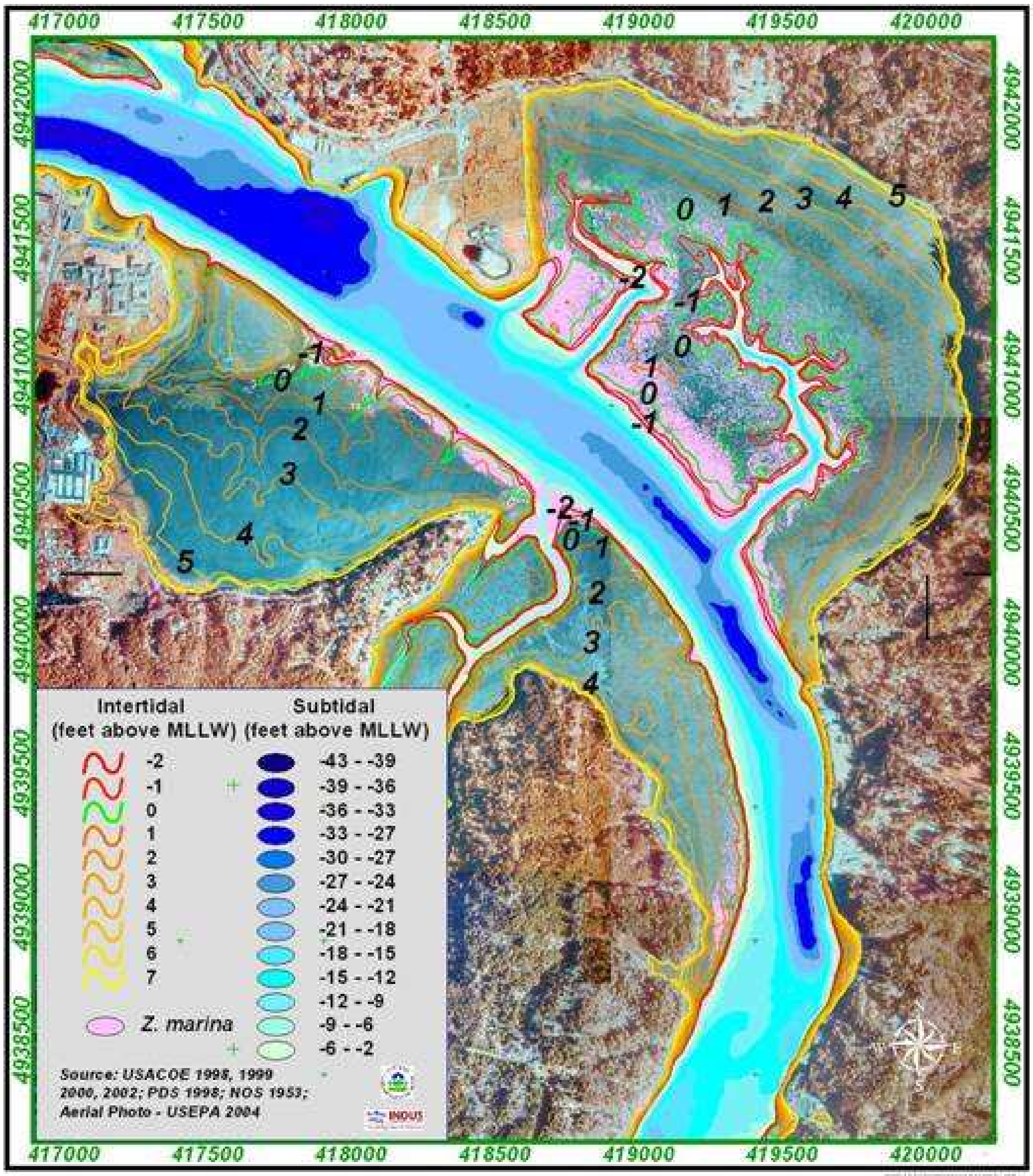

Figure 6-5. Interpolated bathymetry and *Z. marina* distribution for a section of the lower Yaquina Estuary.

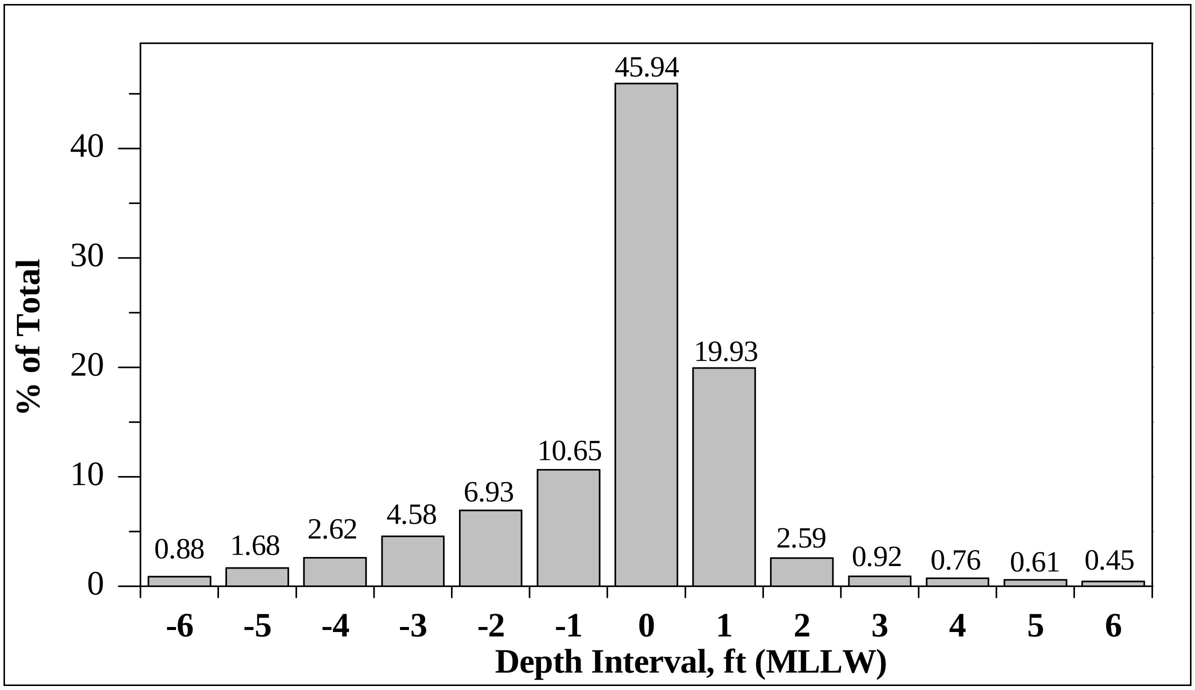

Figure 6-6. The percentage distribution of intertidal and shallow subtidal *Z. marina* habitat area classified from aerial photography that occurs in one foot intervals predicted by the bathymetric model for Yaquina Estuary (-6 ft to +6 ft).

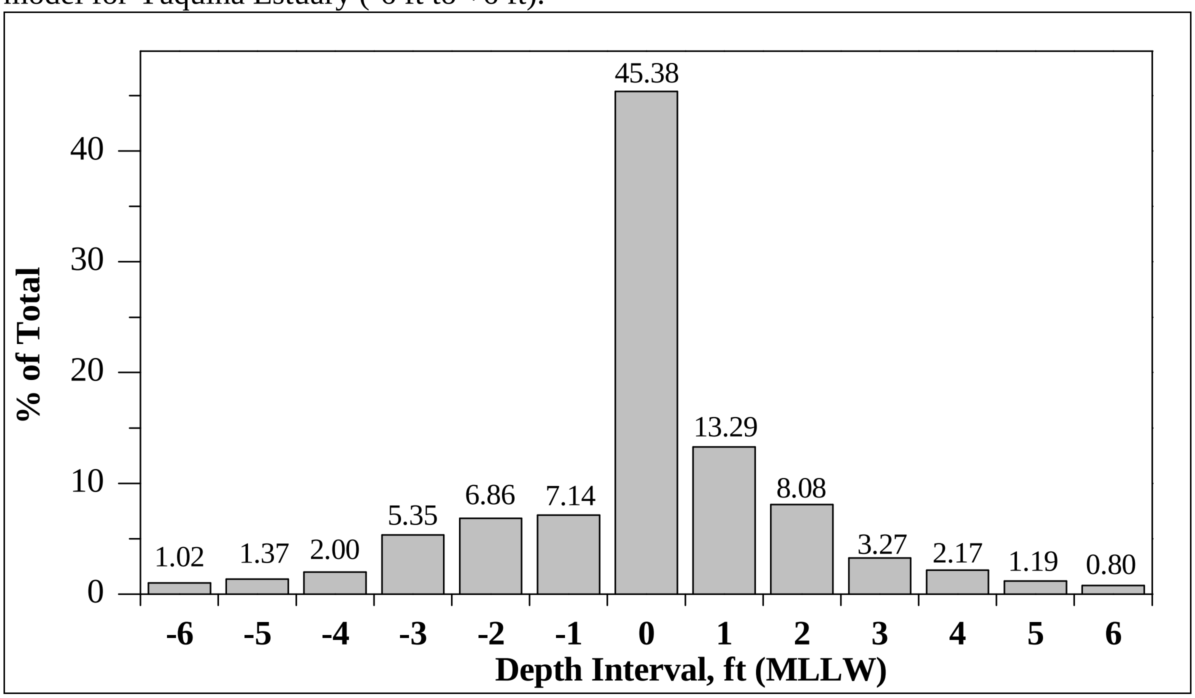

Figure 6-7. The percentage distribution of intertidal and shallow subtidal *Z. marina* habitat area classified from aerial photography that occurs in one foot intervals predicted by the bathymetric model for Alsea Estuary (-6 ft to +6 ft).

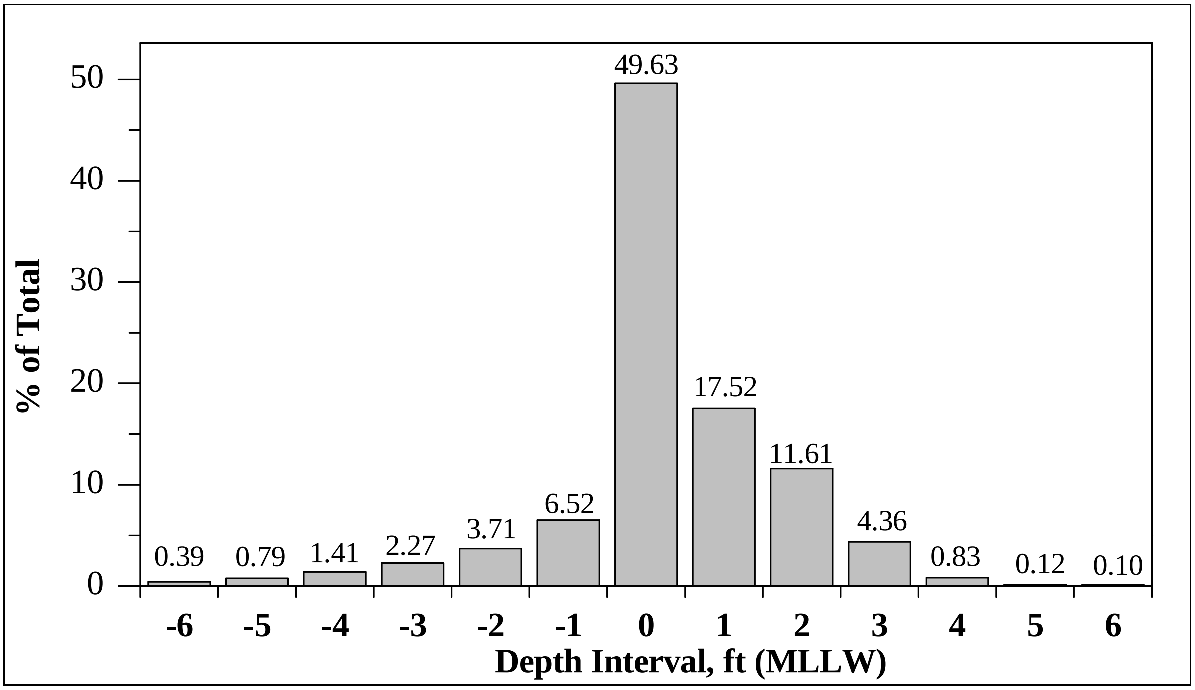

Figure 6-8. The percentage distribution of intertidal and shallow subtidal *Z. marina* habitat area classified from aerial photography that occurs in one foot intervals predicted by the bathymetric model for Tillamook Estuary (-6 ft to +6 ft).

Table 6-4. Cumulative percent area of *Z. marina* within four depth intervals relative to MLLW in three PNW estuaries.

	DEPTH INTERVAL (ft)			
ESTUARY	-1’ to +1’	-2’ to +2’	-3’ to +3’	-6’ to +6’
Yaquina	76.6	86.0	91.4	97.7
Alsea	65.9	80.3	88.8	96.8
Tillamook	73.6	88.9	95.5	99.0

As shown in Table 6-4, approximately 90 percent of the intertidal and shallow subtidal *Z. marina* classified from the orthophotography images occurred within the depth range -3.0 ft to +3.0 ft (-0.9 m to +0.9 m) around the MLLW datum. In comparison, Borde et al. (2003) reported that the upper limit of the *Z. marina* meadows observed in Coos Bay during summer 1999 was approximately +0.8 m (MLLW), in good agreement with our results. Corresponding observations for the maximum depths range from -0.2 m to -1.5 m (median: -0.85 m; Thom et al., 2003), compared to our approximate lower limit value of -0.9 m. Further, these authors reported maximum depths ranging from -0.4 m to -1.2 m (median: -0.80 m) in Willapa Bay, WA (for location see Figure 2-2), again consistent with our findings. In a subsequent study, Thom et al. (2008) reported that *Z. marina* shoot densities decreased to zero below -1.5 m in Coos Bay and Willapa Bay.

Orth and Moore (1988) concluded from their studies in lower Chesapeake Bay that both optimum and maximum depths for this species can vary considerably within a particular region, depending upon water clarity. The same observation was made for the maximum depth of *Z. marina* at numerous locations throughout the seven target estuaries surveyed in this study (Chapter 8). In general, deepest limits occurred near the mouths of the estuaries where the clearest water usually is found. The median of the maximum depth readings, relative to MLLW, for each estuary ranged from -1.82 m (for Alsea Estuary) to +0.21 m (for Coos Estuary), with an overall median of -0.76 m. (For the Tillamook, Yaquina, and Alsea estuaries alone, the overall median was -1.19 m). Again, these findings are in reasonable agreement with the lower limit value of -0.9 m we obtained for approximately 90 percent of the *Z. marina* in our bathymetric distributions for three of the seven target estuaries.

This comparison presented in Table 6-4 suggests a method of estimating the suitability of intertidal and shallow subtidal habitat for *Z. marina* as a function of bottom depth in a given estuary. The estimate, for a specified depth interval, is calculated as the area in which *Z. marina* occurs divided by the total area of the target estuary within that depth interval. These "frequency of occurrence" values obtained for one-foot (0.3 m) depth intervals around MLLW between -6 ft and +6 ft are illustrated in Figure 6-9.

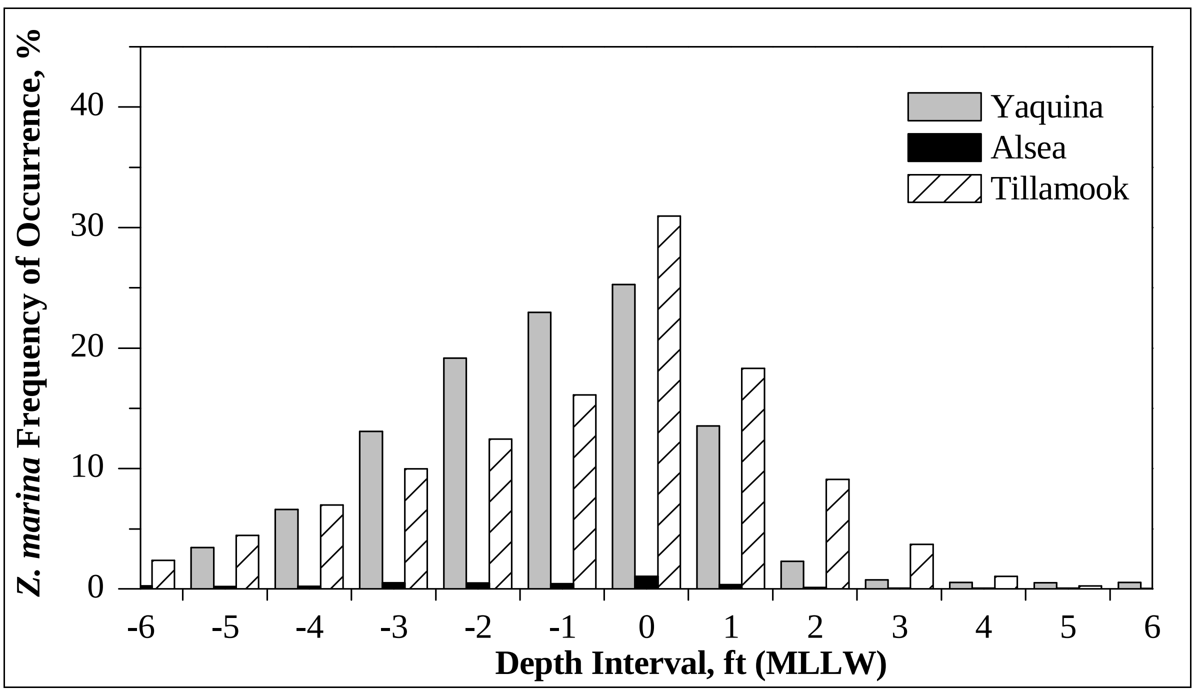

Figure 6-9. Frequency of occurrence values for *Z. marina* within one-foot depth intervals around MLLW in Yaquina, Alsea, and Tillamook estuaries (-6 ft to +6 ft).

Because the interval -3 ft (-0.9 m) to +3 ft (+0.9 m) contains approximately 90% of the *Z. marina* classified in the color-infrared aerial photomaps for each of these three estuaries, this zone is termed the primary depth interval for *Z. marina*. The average (± 1 standard error of the mean) frequency of occurrence values obtained for this interval in Yaquina, Alsea, and Tillamook estuaries are 13.9 ± 3.6%, 0.44 ± 0.12%, and 14.4 ± 3.3%, respectively. The averages for

Yaquina and Tillamook estuaries are identical (14%), but are higher than that for the Alsea Estuary (0.44%) by about a factor of 30. These relative values are consistent with those obtained from the data presented in Table 6-3. Again, further studies are required to elucidate the causes of differences in percent coverage values for *Z. marina* among PNW estuaries.

6.2.4 Within-Estuary Distribution of *Zostera marina*

As discussed in Chapter 5, the seven target estuaries were divided into riverine and oceanic segments based on their primary nutrient sources. One of the objectives of the aerial surveys as well as the probabilistic surveys (Section 7.3.1) was to assess the distribution of *Z. marina* between these two estuarine segments. To address this question, the percent of total area of intertidal *Z. marina* habitat classified from the aerial surveys (or surface mapping in the case of the Nestucca and Salmon River estuaries) was apportioned within each segment, using both the Total Aerial Frame and the Probabilistic Frame (Table 6-5).

Table 6-5. Relative distribution of the total *Z. marina* habitat area in the oceanic and riverine segments of the estuaries based on the aerial surveys. Estimates are based both on the *Z. marina* detected within the entire area covered by the aerial surveys and the portion of the *Z. marina* detected within the subset of the intertidal probabilistic frame. NA = not applicable because no *Z. marina* was detected within the probabilistic frame.

	% TOTAL *Z. MARINA* FROM AERIAL SURVEY			
	CALCULATED USING TOTAL AERIAL FRAME		CALCULATED USING PROBABILISTIC FRAME	
ESTUARY	OCEANIC	RIVERINE	OCEANIC	RIVERINE
Alsea	79.6	20.4	98.8	1.2
Coos	96.2	3.8	96.6	3.4
Nestucca	100.0	0.0	NA	NA
Salmon	87.3	12.6	99.6	0.4
Tillamook	99.8	0.2	99.7	0.3
Umpqua	52.8	47.2	88.5	11.5
Yaquina	98.5	1.5	99.7	0.3

A major implication of these results is that for all of the target estuaries, the majority of the intertidal and shallow subtidal *Z. marina* classified from the aerial (or surface mapping) surveys is found in the oceanic segment. Generally, the distributions between the oceanic and riverine segments calculated using the aerial and probabilistic frames are similar, with the exception of the Umpqua River Estuary. In this estuarine system, *Z. marina* habitat using the Total Aerial Frame is evenly divided between the oceanic (52.8%) and riverine (47.2%) segments. However, the percent-of-total value for the oceanic segment of the Umpqua using the Probabilistic Frame (88.5%) is much higher. One possible explanation for the difference between the two approaches is that there may be mis-classification of *Z. marina* in the riverine segment. In particular, two other seagrasses, *Z. japonica* and *Ruppia maritima* (widgeon grass), that occur in lower salinity areas are difficult to differentiate from *Z. marina* using remote sensing. Nonetheless, the presence of at least 53% and up to 100% of the total *Z. marina* mapped from the aerial surveys occurring within the oceanic segments demonstrates the importance of this segment to this species in the PNW.

The distribution of *Z. marina* between the oceanic and riverine segments was also evaluated using the relative percent cover of *Z. marina* as determined from the aerial photomapping (Table 6-6). As discussed in Section 7.3, relative cover is defined as the percent of the area of the oceanic segment or the riverine segment occupied by *Z. marina*. Calculation of relative cover is limited to the probabilistic frame portion of the aerial frame since the area sampled is needed to calculate the percent cover value. Higher relative cover values are assumed to indicate a better habitat for a species given that it occupies a greater percentage of the total habitat area. Based on this assumption, the oceanic segment constitutes a substantially better habitat for *Z. marina*, with relative cover 7-fold to almost 200-fold higher in the oceanic segments in the six estuaries with detectable *Z. marina* habitat. These results are consistent with the relative distributions of the total *Z. marina* population (Table 6-5) and support the view that the lower estuary is the prime habitat for *Z. marina* in PNW estuaries.

Table 6-6. Percent coverage of *Z. marina* within the oceanic and riverine segments of the estuaries based on the aerial surveys. Estimates are based on the relative cover in the probabilistic portion of the aerial survey frame. NA = not applicable because no *Z. marina* was detected within the probabilistic frame.

ESTUARY	RELATIVE % COVER OF *Z. MARINA*	
	OCEANIC	RIVERINE
Alsea	0.10	0.01
Coos	5.63	0.79
Nestucca	NA	NA
Salmon	0.08	0.002
Tillamook	15.28	0.08
Umpqua	2.86	0.26
Yaquina	12.03	0.18

6.3 Synthesis

Color infrared aerial photosurveys of the target estuaries yielded orthophotographs that were digitally classified into *Z. marina* and bare substrate classes. The classifications were checked via ground surveys using a stratified random sampling design. Classical accuracy assessments yielded individual accuracies of 83% or greater, with an overall median accuracy of 89%. This provided intertidal and shallow subtidal *Z. marina* distributions within five target estuaries, while two other estuaries were mapped via ground surveys because of the limited areal extent of *Z. marina* habitat. Comparison with bathymetric models for three estuaries showed that the one-foot depth interval bracketing MLLW contained 66% to 77% of the intertidal *Z. marina*, and that 89% to 96% of the measurable *Z. marina* occurred within the primary depth interval -3 ft to +3 ft (-0.9 m to +0.9 m). The ratio of the area occupied by *Z. marina* relative to the total estuarine area within this interval provided frequency of occurrence values representative of habitat suitability that showed Alsea Estuary contained a much lower percentage cover of *Z. marina* in the primary depth interval than in Yaquina and Tillamook estuaries. A similar result was obtained from probability-based ground surveys. Causes for such differences presently are unknown. However, the maps of *Z. marina* distribution clearly indicated that a large majority of intertidal and shallow subtidal *Z. marina* occurred in the oceanic segments of the target estuaries.

CHAPTER 7: AMONG AND WITHIN ESTUARINE DISTRIBUTIONS OF SEAGRASSES AND ECOLOGICALLY IMPORTANT BENTHIC SPECIES IN PACIFIC NORTHWEST ESTUARIES

Henry Lee II, David R. Young, Christina L. Folger, Cheryl A. Brown,
Janet O. Lamberson, Katharine M. Marko, and Faith A. Cole

Key Findings

- **The areal extents of *Z. marina*, *Z. japonica*, benthic green macroalgae, two burrowing shrimp (*N. californiensis* and *U. pugettensis*), and "bare" habitat in the intertidal zone were estimated in the seven target estuaries using probabilistic field surveys.**

- **All three tide-dominated estuaries had moderate to extensive coverage of *Z. marina*. In comparison, *Z. marina* was not found in three of the river-dominated estuaries and there was only limited coverage in the fourth. Differences in median salinity and/or salinity variability among estuary classes appear to be a major factor determining the extent of *Z. marina* though sediment movement/energy and high densities of *N. californiensis* may also be contributing factors.**

- **The nonindigenous *Z. japonica* occurred in six of the estuaries and the area occupied exceeded that of *Z. marina* in four of them.**

- **Blooms of green macroalgae in coastal PNW estuaries appear to be a natural process and not an indication of cultural eutrophication.**

- **Burrowing shrimp were a major component of the intertidal in all the estuaries, though the species had different distributions among estuaries.**

- **A multivariate analysis on the benthic habitats grouped the three tide-dominated estuaries together along with the river-dominated Alsea Estuary. The river-dominated Nestucca and Salmon formed a second group, while the river-dominated Umpqua separated out from the other estuaries.**

7.0 Introduction

Probabilistic field surveys of the intertidal habitats were conducted in the seven target estuaries with the objectives of: 1) assessing the among-estuary patterns of distribution and abundance of *Zostera marina*, *Z. japonica*, benthic macroalgae, and two burrowing shrimp, *Neotrypaea californiensis* and *Upogebia pugettensis* and 2) determining the within-estuary distribution of each of these taxa especially in relationship to their distribution between the oceanic and riverine segments within each estuary. An additional objective was to develop practical methods to quantify seagrasses, other biotic resources, and key environmental factors in regional-scale surveys.

The first objective addresses the classification of PNW estuaries based on biotic structure, providing information at finer spatial scale and with a higher level of biotic resolution than the wetland classifications conducted at a regional scale (Chapter 2). The second objective addresses one component of vulnerability, the potential exposure of the five taxa to anthropogenically derived nutrients. In particular, the objective is to test the hypothesis that the majority of the *Z. marina* occurs in the oceanic segment across a range of different types of PNW estuaries. Probabilistic survey designs are well suited to addressing this objective as they provide statistically unbiased estimates of the intertidal area occupied by each of the taxa. The distribution of *Z. marina* was also evaluated through aerial photography (Chapter 6), and each technique has its own strengths and limitations. It is beyond the scope of this document to contrast the two approaches, and in the context of assessing vulnerability, the key question is whether both approaches showed the same general patterns in *Z. marina* distributions.

In addition to the two *Zostera* species, benthic macroalgae and the two burrowing shrimp are ecologically important benthic species in PNW estuaries, affecting both the physical and chemical structure of the benthic environment and the structure of estuarine food webs (see Section 1.5 and Table 1-1). Comparison of their coverage and/or abundance among estuaries quantifies how these food web components vary by estuary type. By determining their distributions among the oceanic and riverine segments, the probabilistic surveys identify which of these food web components are most likely to be exposed to terrestrially derived nutrients, and hence be more vulnerable to nutrient enrichment.

7.1 Methods

7.1.1 Probabilistic Sampling Frame and Design

A probabilistic sample design, as utilized by the Environmental Monitoring and Assessment Program (EMAP), was used to generate statistically robust estimates of the abundance and distribution of seagrasses and other benthic resources. An overview of the probabilistic sampling approach used in previous coastal EMAP surveys in California, Oregon, and Washington can be found in Nelson et al. (2005a, b). Additional details on the probabilistic approach to describing the condition of ecological resources can be found in Diaz-Ramos et al. (1996), Stevens (1997), Stevens and Olsen (1999), and at http://www.epa.gov/nheerl/arm.

The probabilistic field surveys used the same sampling frame as the coastal EMAP survey of intertidal wetlands conducted in 2002 (Nelson et al., 2007). The estuarine intertidal sampling frame was defined as all estuarine NWI polygons that were “regularly flooded”, which was interpreted as being between Mean Lower Low Water (MLLW) and Mean Higher High Water (MHHW). NWI polygons labeled “irregularly exposed” were interpreted as being below MLLW while polygons labeled “irregularly flooded” were interpreted as being above MHHW, neither of which was included in the sampling frame. Thus, the sampling frame did not include either the shallow subtidal or emergent marshes. Areas of the sampling frame and the post-sampling division between the oceanic and riverine segments are summarized in Table 7-1.

The Alsea, Nestucca, Yaquina and Salmon River estuaries were sampled in 2004, with the Coos, Umpqua River, and Tillamook estuaries sampled in 2005 (Table 7-1). The field surveys were completed during the index period of June through mid-September, with the exception of 15 samples in the Tillamook that were taken in December 2005 because of tidal constraints. These sites were resampled in August 2006, and the updated results are used in the analysis. Nine additional samples were taken in the Alsea Estuary in September 2006 to sample a portion of the original sample frame that was inadvertently omitted. Measures of water quality and $\delta^{15}N$ stable isotopes in macroalgae were collected synoptically with the probabilistic surveys as discussed in Chapters 4 and 5.

A total of 100 random sites and a corresponding number of random replacement sites were allocated to each of the seven estuaries. In an attempt to have a sufficient number of samples in both the oceanic and riverine segments, the estuaries were *a priori* divided into sampling strata (Figures 7-1 to 7-7) with a designated number of samples allocated to each stratum (Table 7-1). The four estuaries sampled in 2004 were divided into two sampling strata based on an initial estimate of the demarcation between the oceanic and riverine segments of the estuaries. In 2005 we modified our approach, and divided the Coos, Tillamook, and Umpqua estuaries into a greater number of sampling strata based on salinity distributions. Sites that were found to be located in inappropriate habitat (e.g., marsh, rocky areas) or were inaccessible were relocated within 30 m of the original location site using a protocol developed for the EMAP wetland survey (Lamberson and Nelson, 2002). If it was still not possible to obtain a sample, the site was abandoned and the first replacement site within the same sampling stratum was chosen.

7.1.2 Data Analysis

Data from the probabilistic surveys were primarily analyzed using cumulative distribution functions (CDFs). Coastal EMAP studies have used CDFs to evaluate local and regional patterns of coastal resources (e.g., Summers et al., 1993; Strobel et al., 1995; Hyland et al., 1996; Nelson et al., 2005a, b). Details on the methods for calculating CDFs and the 95% confidence intervals can be found in Diaz-Ramos et al. (1996). Briefly, CDFs describe the percentage of the area of the sampling frame across the full range of the indicator values (e.g., percent cover, burrow hole counts) based on the cumulation of the weighted samples. The sample weight of each sample was determined by the area of the stratum divided by the number of samples taken within stratum. The CDFs were generated for each target estuary over the entire sampling frame (i.e., intertidal area of the estuary). The CDFs were also generated independently for each oceanic and riverine segment within an estuary based on the areas determined to fall into these segments after the zonation of the estuaries (Chapter 5). Since the demarcation line for the oceanic and riverine segments transected some of the original sample strata, the sample weights were recalculated based on these new strata areas and sample numbers.

7.1.3 Field Methods

Seagrass, macroalgae, and burrowing shrimp presence and abundance were estimated using two quadrat sizes (Figure 7-8). Visual estimates were made in large quadrats (2.5 m on a side or 6.25 m^2). A length of braided cord ten meters long with markers every 2.5 m was used to lay out the 2.5 m x 2.5 m sampling plot. Polyvinyl chloride (PVC) stakes were used to secure the four corners and to define the site boundaries. The presence/absence of *Z. marina*, *Z. japonica*,

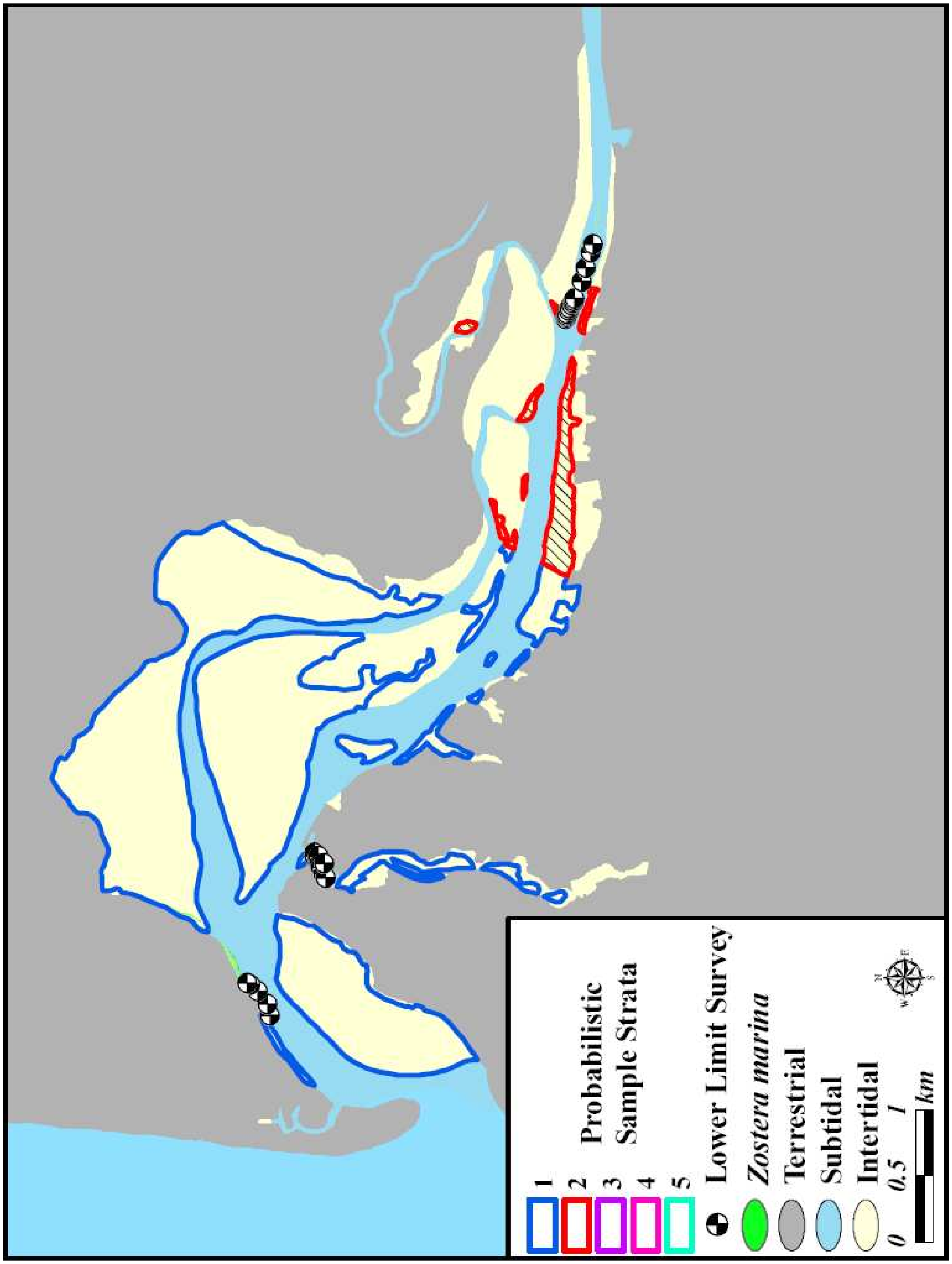

Figure 7-1. Locations of the probabilistic sampling strata and the *Z. marina* lower limit surveys (Chapter 8) in the Alsea Estuary. There were three sampling strata in the Alsea, with the third stratum sampled in 2006 indicated by the hatched area. Additional lower limit points sampled in 2006 are not indicated in the figure. The locations where *Z. marina* was detected by aerial photography are indicated in green. Also see Figure A-1 in Appendix A for a map of the *Z. marina* distribution based on the aerial surveys.

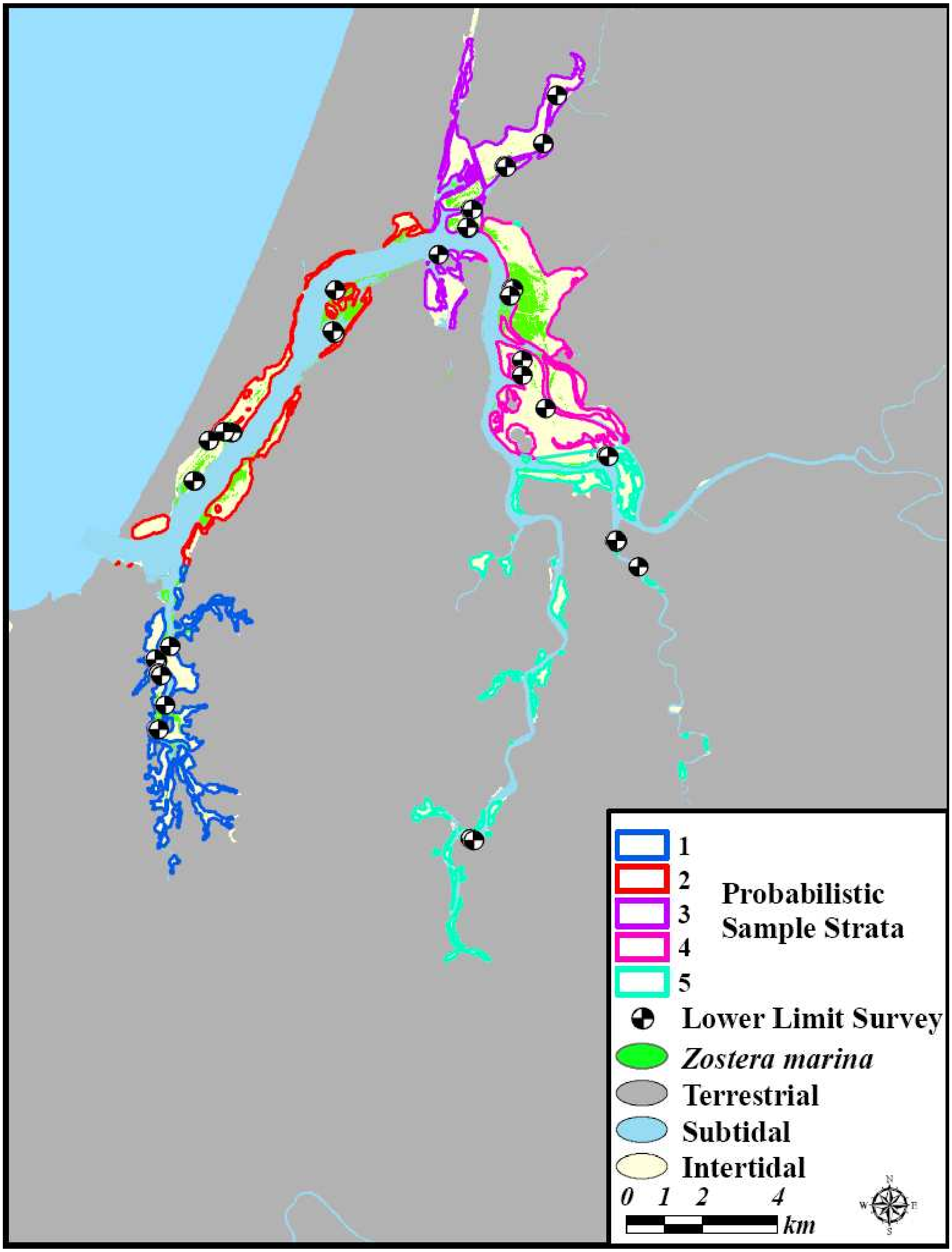

Figure 7-2. Locations of the probabilistic sampling strata and the *Z. marina* lower limit surveys (Chapter 8) in the Coos Estuary. There were five sampling strata in the Coos. The locations where *Z. marina* was detected by aerial photography are indicated in green. Also see Figure A-2 in Appendix A for a map of the *Z. marina* distribution based on the aerial surveys.

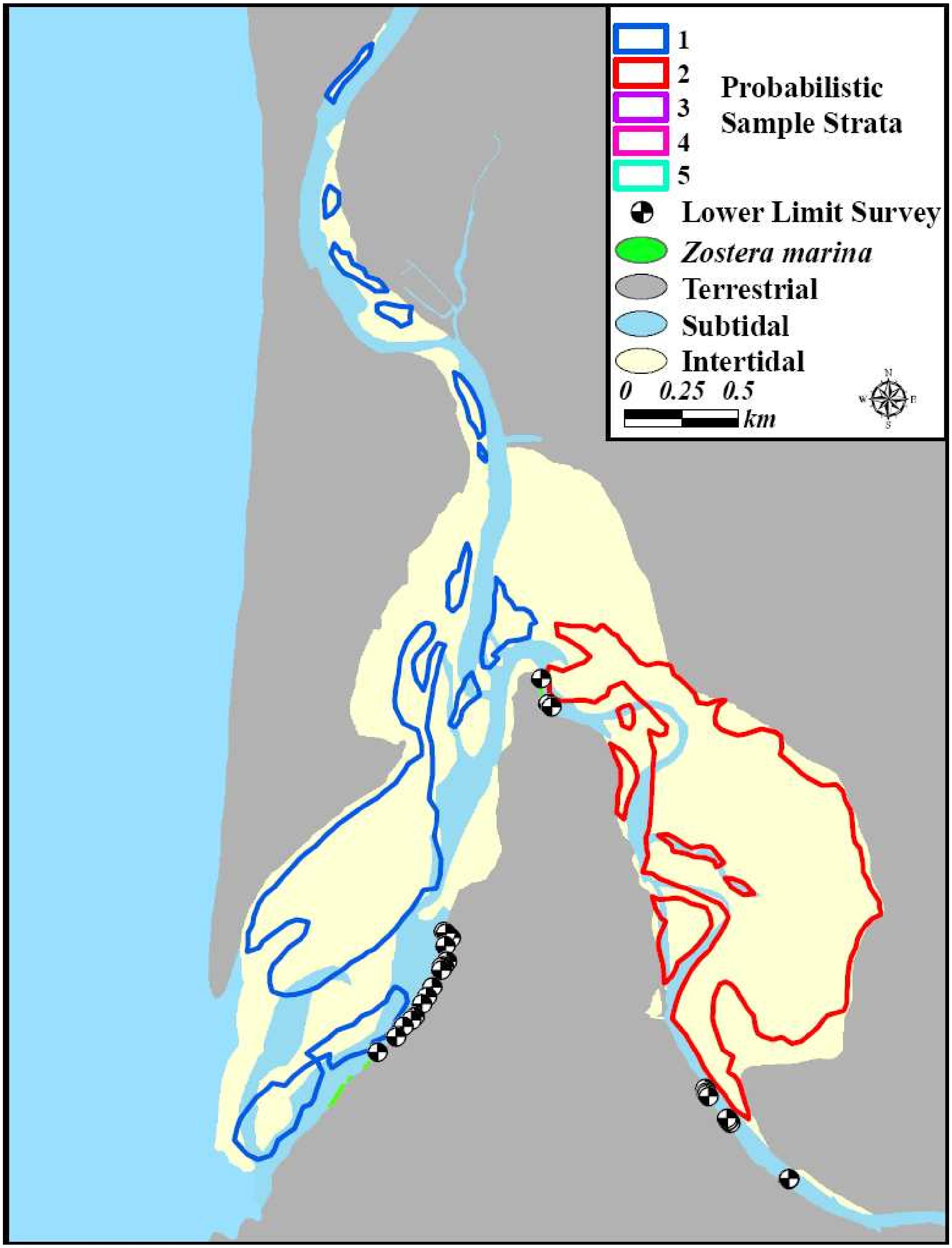

Figure 7-3. Locations of the probabilistic sampling strata and the *Z. marina* lower limit surveys (Chapter 8) in the Nestucca Estuary. There were two sampling strata in the Nestucca. The locations where *Z. marina* was detected by the on-ground survey are covered by the lower limit symbols and the reader is referred to Figure A-3 in Appendix A for a map of the *Z. marina* distribution based on the aerial surveys.

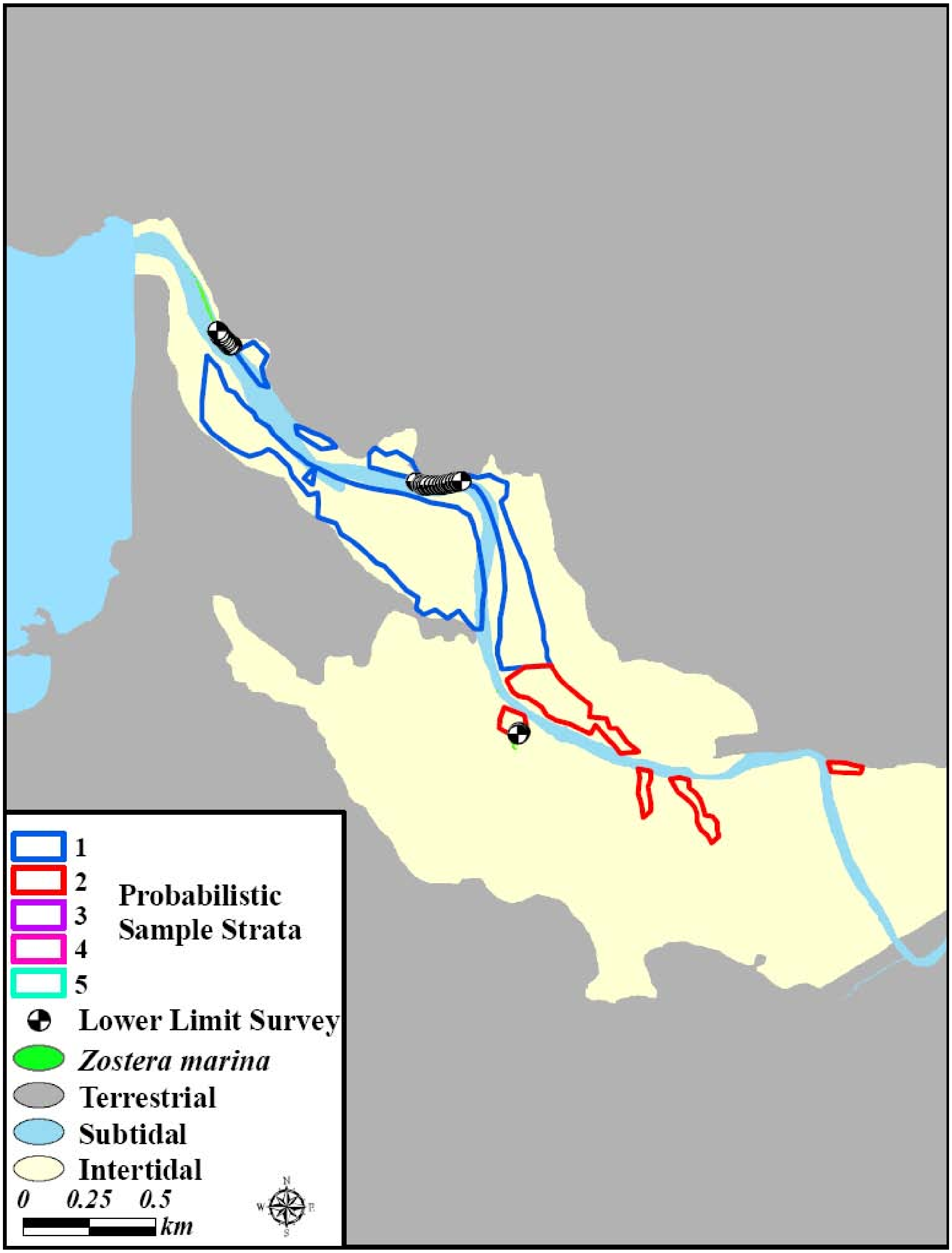

Figure 7-4. Locations of the probabilistic sampling strata and the *Z. marina* lower limit surveys (Chapter 8) in the Salmon River Estuary. There were two sampling strata in the Salmon. The locations where *Z. marina* was detected by the on-ground survey are obscured by the lower limit symbols and the reader is referred to Figure A-4 in Appendix A for a map of the *Z. marina* distribution based on the aerial surveys.

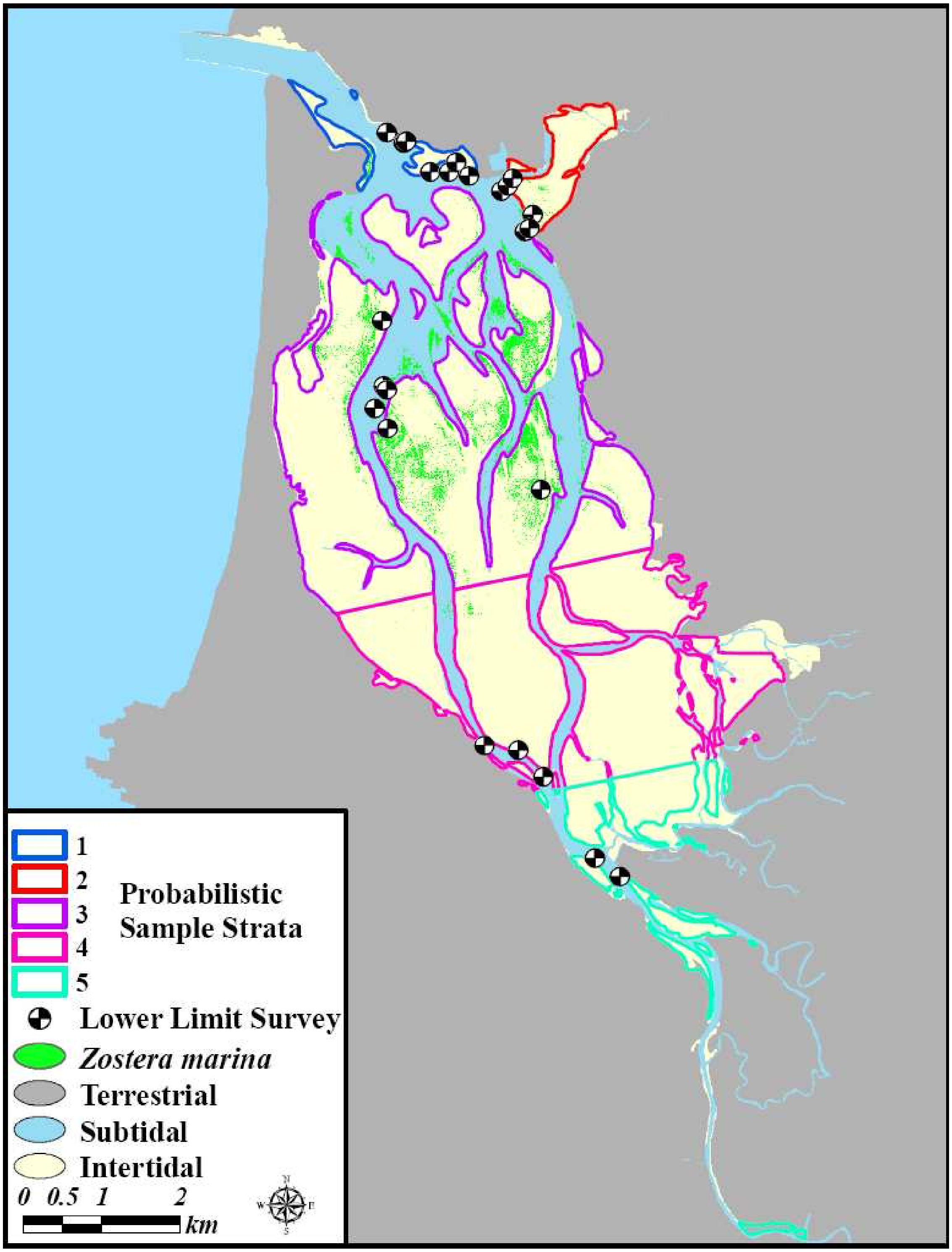

Figure 7-5. Locations of the probabilistic sampling strata and the *Z. marina* lower limit surveys (Chapter 8) in the Tillamook Estuary. There were five sampling strata in the Tillamook. The locations where *Z. marina* was detected by aerial photography are indicated in green. Also see Figure A-5 in Appendix A for a map of the *Z. marina* distribution based on the aerial surveys.

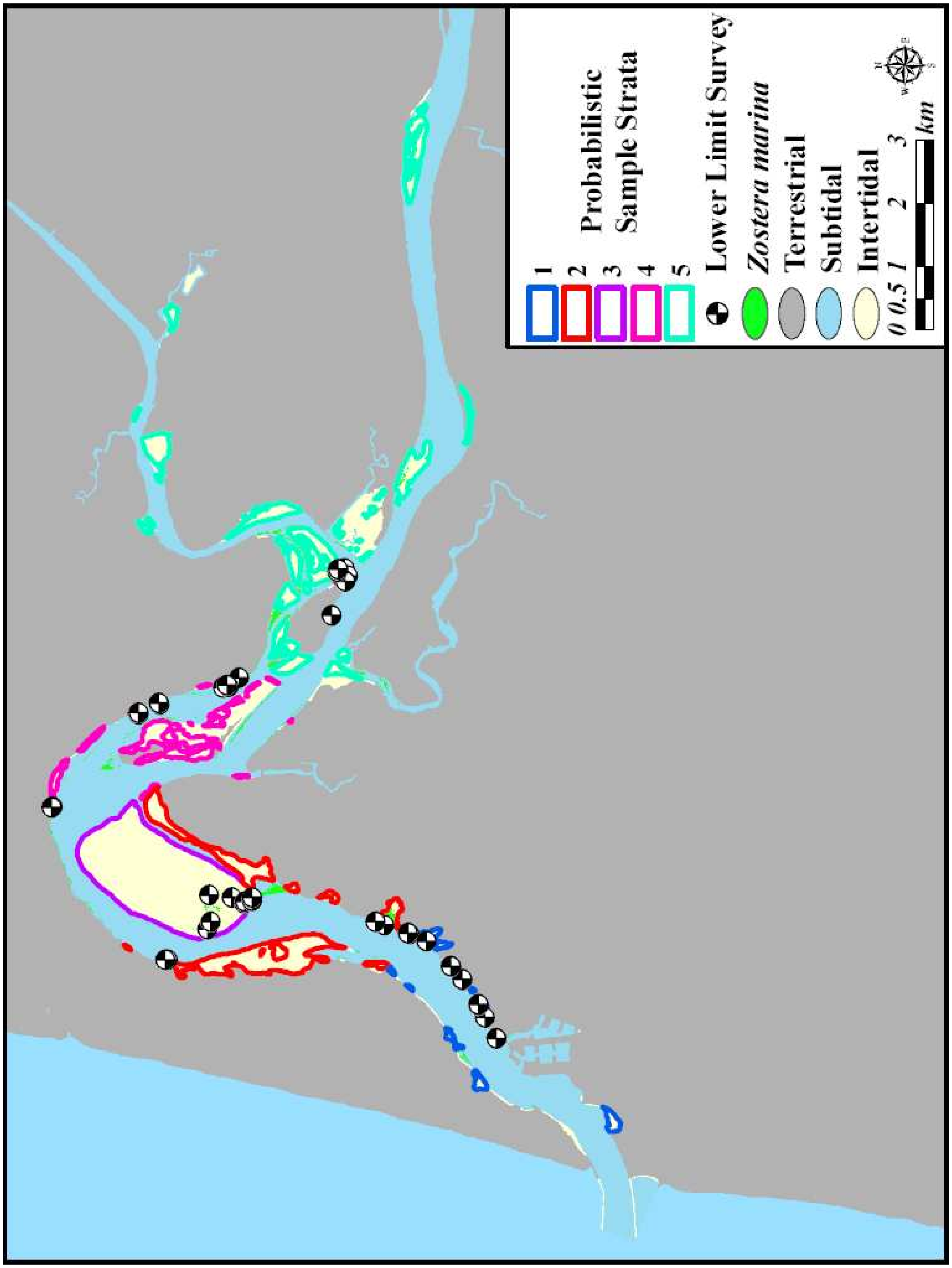

Figure 7-6. The locations of the probabilistic sampling strata and the *Z. marina* lower limit surveys (Chapter 8) in the Umpqua River Estuary. There were five sampling strata in the Umpqua. The locations where *Z. marina* was detected by aerial photography are indicated in green. Also see Figure A-6 in Appendix A for a map of the *Z. marina* distribution based on the aerial surveys.

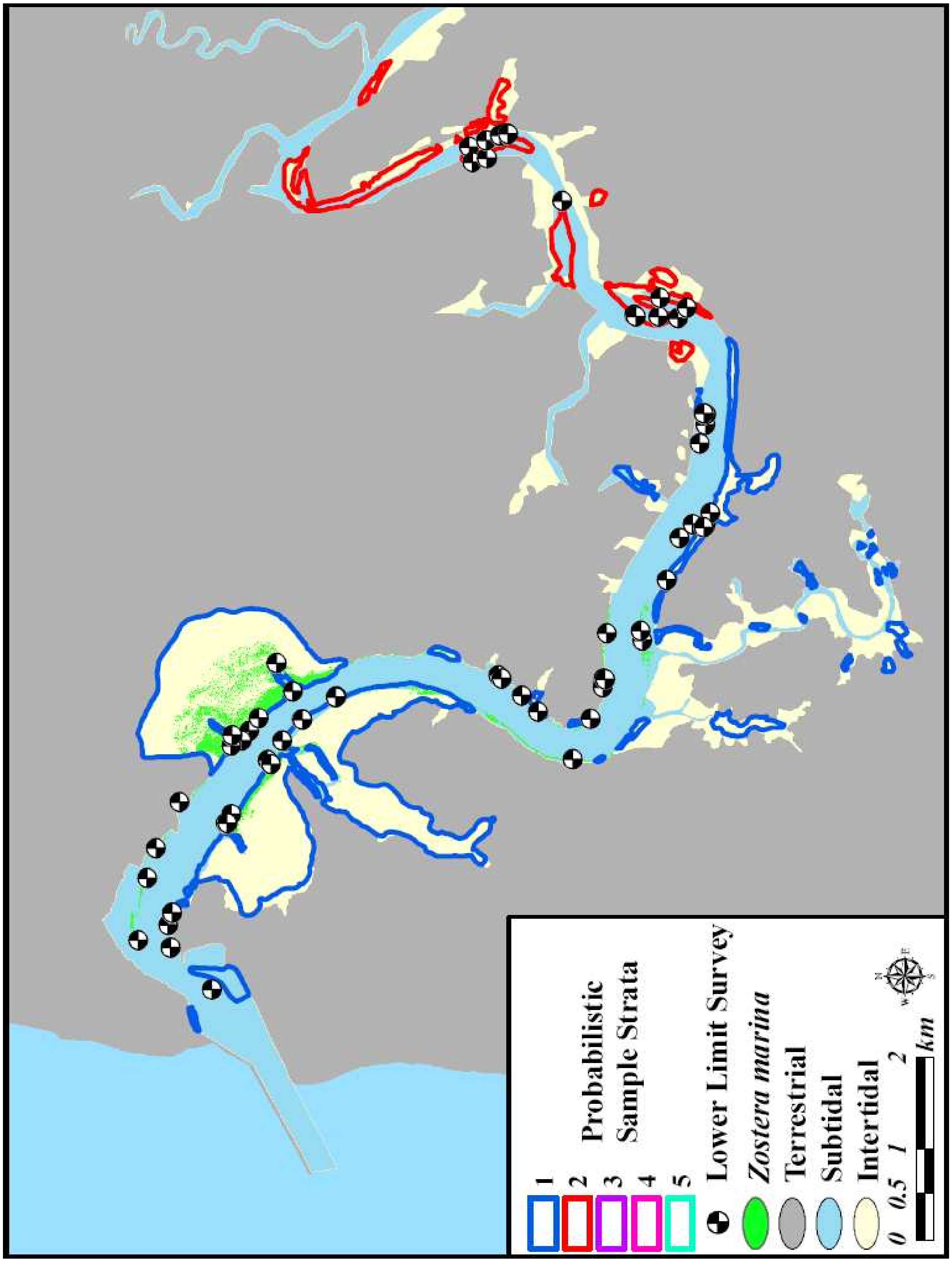

Figure 7-7. The locations of the probabilistic sampling strata and the *Z. marina* lower limit surveys (Chapter 8) in the Yaquina Estuary. There were two sampling strata in the Yaquina. The locations where *Z. marina* was detected by the aerial photograph are indicated in green. Also see Figure A-7 in Appendix A for a map of the *Z. marina* distribution based on the aerial surveys.

benthic macroalgae, burrowing shrimp, and other indicators of biological activity were visually estimated in each of these large quadrats. In 2005 and 2006, the percent cover of both seagrass species in the large quadrats was assigned to one of five cover classes (0%, 1-10%, 10-40%, 40-70%, and >70%). Two smaller 0.25 m^2 quadrats were then placed within the larger 2.5 x 2.5 m plot site, with one quadrat designated for percent plant cover and the other for burrowing shrimp. The plant quadrat was placed in the center of the sampling plot to avoid placement bias. The following measurements were taken within the plant quadrat: 1) percent plant cover of both seagrasses and macroalgae; 2) number of vegetative, reproductive and seedling shoots; 3) leaf measurements on five vegetative shoots; 4) macroalgal volume. The shrimp quadrat was placed in an area within the 2.5 m x 2.5 m plot with less macroalgae or seagrass so that thick cover would not obscure burrow holes.

We found that the design of the field sheets was critical to having the field crews accurately collect the pertinent habitat information from quadrats. To assist in future investigations, the current version of the field sheets is reproduced in Figure 7-9. Additional details on the field methods follow.

Point-intercept estimate of plant cover – Point intercepts within the 0.25 m^2 plant quadrat were used to estimate the cover of the two seagrasses and benthic macroalgae. The point-intercept estimate was based on the number of macroalgae and seagrass occurrences under 25 intercepts of two sets of equally spaced five strings on the plant quadrat. The percent cover of seagrass and macroalgae were measured independently in three dimensions (i.e., captured occurrence of plants overlaid by other plants), thus the sum could exceed 100%.

Number of seagrass shoots **–** All *Z. marina* shoots were counted within the 0.25 m^2 plant quadrat in 2004-2006. Due to the potentially high number of *Z. japonica* shoots present in a 0.25 m^2 area, the number of *Z. japonica* shoots was only approximated in the plant quadrats in 2004. In 2005 and 2006, the number of *Z. japonica* shoots was counted in a 0.01 m^2 cell within the plant quadrat. For *Z. marina*, vegetative and reproductive shoots were counted independently and a determination was made as to whether it was an annual or perennial form by the presence or absence of rhizomes.

Leaf measurements - Leaf measurements were taken on five haphazardly chosen shoots of both seagrass species within each plant quadrat when present. Leaf measurements consisted of the length of the longest blade to the nearest 1 cm, width of the same leaf at the widest point, and number of leaves per shoot. Where fewer than five vegetative shoots were present within a quadrat, the measurements were collected from all vegetative shoots present in the quadrat.

Macroalgal volume - Macroalgal volume was measured by removing all the algae from the 0.25 m^2 plant quadrat, and measuring the volume of algae using a four-liter volumetric cylinder following the method of Robbins and Boese (2002). Macroalgal volume was measured to the nearest 100 ml after macroalgae were compressed using a plunger to a point where interstitial water had been removed. A volume of 50 ml (half the detection limit) was used when macroalgae was present but not in sufficient quantity to be measured in the four-liter cylinder.

Table 7-1. Sampling design for the probabilistic field surveys and areas of the probabilistic frames. Estuarine intertidal area is the sum of the NWI “regularly flooded” estuarine polygons, which is approximately from MLLW to MHHW. The Probabilistic Frame is the actual area that was sampled. The areas of the oceanic and riverine frames are the areas within the oceanic versus riverine segments (Chapter 5) that resulted from the post-survey division of the estuaries. The samples taken in 2006 in the Alsea and Tillamook are included in the analysis.

	NWI ESTUARINE REGULARLY FLOODED AREA (km^2)	PROBABILISTIC FRAME				NUMBER OF SAMPLES			
ESTUARY		TOTAL AREA (km^2)	OCEANIC FRAME (km^2)	RIVERINE FRAME (km^2)	NUMBER OF SAMPLING STRATA	TOTAL	OCEANIC FRAME	RIVERINE FRAME	YEARS SAMPLED
Alsea	6.97	5.20	4.77	0.44	3	109	50	59	2004/2006
Coos	26.96	26.56	21.25	5.31	5	101	86	15	2005
Nestucca	2.59	2.01	1.95	0.06	2	101	99	2	2004
Salmon	1.56	0.45	0.37	0.07	2	100	52	48	2004
Tillamook	24.6	24.38	14.50	9.88	5	97	56	41	2005/2006
Umpqua	7.45	7.16	2.95	4.22	5	99	55	44	2005
Yaquina	7.87	6.40	5.26	1.15	2	100	48	52	2004

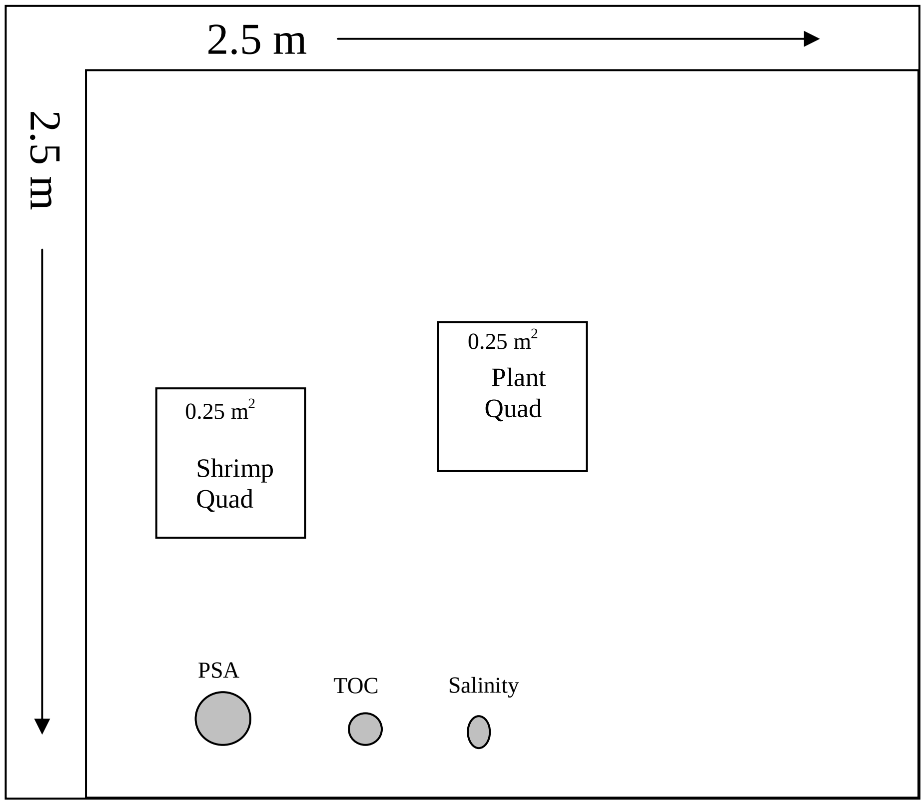

Figure 7-8. Diagram of the intertidal sampling site layout with the 2.5 x 2.5 m quadrat and the 0.25 m^2 plant and shrimp quadrats used in the probabilistic surveys. PSA = Sample for particle size analysis. TOC = sample for total organic carbon.

SITE #:	

DATE:	TIME (PDT): (Current Time)	Northing:
WEATHER:	CREW:	Easting:

2.5 X 2.5 Meter Quadrat

Percent Cover (from look up list)	*Zostera marina* Absent (0%) Very Sparse (1-10%) Sparse (10-40%) Moderate (40-70%) Dense (70-100%)		
Zostera japonica	(0%) (1-10%) (10-40%) (40-70%) (70-100%)	Green Macroalgae	(0%) (1-10%) (10-40%) (40-70%) (70-100%)
Other Rooted Plants Type	(0%) (1-10%) (10-40%) (40-70%) (70-100%)	Other Algae Type	(0%) (1-10%) (10-40%) (40-70%) (70-100%)

Debris Type	**Sediment Characteristics**
	Color: Black Tan/Brown Gray light/white Olive-green Red mottled other
	Firmness** VS S/L F/MD H/VD **Sed. Firmness Classes: VS = very soft, S/L = soft/loose F/MD = firm/ med Density, H/VD = Hard/very dense

Sediment Odor:	No Odor Hydrogen sulfide Petroleum Iodine NR
Shoreline Classes:	Marsh Rip-rap Houses Road Dunes Levee Agriculture Embankment Forest Other _________ Notes
Indicators of Biological Activity:	Unvegetated sediment Brown Diatoms Macroalgae Worms Burrowing Shrimp Clams Mounds Seagrass Dead Shellfish Amphipods Other_________________

Wave Frequency	Crests/1 m		**If Longer than 1 m:** Distance between Crests	
Sediment Temp. (C)	**Sediment Cores Circle when collected:** **TOC/PSA** **Salinity**			

NOTES:

Figure 7-9a. Front of the field sheet used to record estimates of habitat characteristics and burrowing shrimp hole counts within the 2.5 x 2.5 m quadrat.

SITE #:

Quadrat #2 Plants

Point Intercept

Zm		**Open**	
Zj		**Other Algae:**	
GM		**Other Plant:**	

Dominant Algae
Ulva
Enteromorpha
Other _____________

Volume (ml):

Seagrass Shoot Counts *Zm or Zj*

Type	# of shoots
Vegetative	
Reproductive	
Seedlings	
Perennial	

Count marked cell only

Z. marina Blade Measurements

Shoot	Length of longest Blade (cm)	Width of longest Blade (mm)	No. leaves per shoot
1			
2			
3			
4			
5			

Quadrat #3 Burrowing Shrimp

Hole Counts

Neo :	
Upo:	

% w/o algae in shrimp quad

QA Neo :	
QA Upo:	

NOTES:

Figure 7-9b. Back of the field sheet used to record plant cover, point intercepts, shoot counts, and blade widths in the 0.25 m^2 plant quadrat and burrowing shrimp hole counts in the 0.25 m^2 shrimp quadrat.

Burrowing shrimp abundance - The number of burrow holes of *N. californiensis* and *U. pugettensis* were each counted within the shrimp quadrat. Macroalgae cover was recorded within the shrimp quadrat as a possible correction factor for obscured burrow holes. At every tenth site, the number of shrimp burrow holes was counted by a second crew member as a quality assurance check.

Habitat measures - Sediment samples for grain size and total organic carbon and nitrogen (TOC/N) analysis were collected within the large quadrat (Figure 7-8). An 8-cm x 20-cm PVC core was inserted into the sediment within the 2.5 x 2.5 m quadrat to obtain a sample for particle size analysis (PSA). The top 10-cm of the sediment in the core was collected into a clean resealable plastic bag. The sediment sample for TOC/N analysis was collected in a 4-cm x 15-cm core inserted into the sediment adjacent to the PSA core and the top 5 cm were collected. A sediment sample was collected for water content by scraping the sediment surface into a 50-cc centrifuge tube. All sediment samples were stored on ice immediately after collection and frozen as soon as was practical.

Sediment particle size was determined using a Beckman Coulter LS-100Q laser diffraction particle size analyzer. The data are presented as percent fines which is the sum of percent silt and clay. The TOC/N samples were analyzed using a Carlo Erba EA 1108 elemental analyzer according to manufacturer's instructions as modified for sediment. The surface sediment sample collected in the centrifuge tube was centrifuged to extract interstitial water, and salinity of the supernatant water was measured with a bench-top salinity meter.

Sand ripples and waves were measured by laying a meter stick on the sediment surface. If sand ripples were present, the number of wave crests along the 1-meter distance was counted. If the sand ripples were larger than a meter, the distance between crests was measured using a 100-meter transect tape. Data recorded in number of crests per meter were converted to distance between crests. Sand waves were only recorded in the 2005 and 2006 field surveys. At each sampling location, photographs were taken of the quadrats, including the meter stick or transect tape to document surface structure. In addition, the type of shoreline development was visually assessed, recorded, and photographed.

Integrated Measure of Presence/Absence (P/A) of SAV, macroalgae, and burrowing shrimp and bare habitat – Several different techniques were combined to obtain an "integrated" measure of the presence/absence of *Z. marina, Z. japonica*, benthic green macroalgae, burrowing shrimp, and other plant species over a 2.5 x 2.5 m area. It was first determined if the taxon was recorded as present in either the 0.25 m^2 quadrat or the 2.5 x 2.5 m quadrat. If the taxon was not recorded in either quadrat, then the field notes and the site photographs were evaluated for indications of the presence of the taxon within the large quadrat. The taxon was recorded as present if it was observed in the site photograph. Use of the integrated P/A reduced any errors from misclassifications in the field, and is used as the best indicator of a species presence or absence. "Bare habitat" was defined as any 2.5 x 2.5 m quadrat that did not contain any *Z. marina, Z. japonica*, benthic green macroalgae, or burrowing shrimp.

7.1.4 Comparison of 0.25 m^2 versus 2.5 x 2.5 m Quadrats

As part of this study, we evaluated the efficacy of using 0.25 m^2 (small) versus 2.5 x 2.5 m (large) quadrats in estimating the distribution and abundance of seagrasses. The 0.25 m^2 quadrats are small enough to allow point-intercept estimates or counts of seagrass shoots. The 2.5 x 2.5 m quadrats have the advantages that they are same size as the minimum mapping unit in the aerial photography and are of sufficient size to "average out" small scale patchiness. Both methods are valuable, but the results need to be interpreted in light of the differences in scale. For all seven estuaries, estimates of the area occupied by *Z. marina* and *Z. japonica* from the point intercepts were lower than those from the large quadrats (Tables 7-3 and 7-4). The larger quadrats appear to generate a more comprehensive estimate of seagrass distribution because they have a higher "capture" probability for relatively sparse species. Given this higher capture probability, the large quadrats were judged the preferred approach to estimating the area occupied by the two seagrasses. While the quantitative estimates of the density class from the large quadrat are preferred over presence/absence, the percent cover classes were only collected in the large quadrats in 2005 and 2006. Accordingly, the integrated measure of presence/absence in the 2.5 x 2.5 m quadrats was used as the primary method of quantifying the intertidal area occupied by *Z. marina* and *Z. japonica*. The point-intercept and shoot-count methods were used in a supplemental fashion to estimate seagrass density.

As discussed in Chapter 6, the aerial photography has a detection limit of approximately 10% seagrass cover within a 2.5 x 2.5 m minimum mapping unit. Therefore, the most direct comparison between the probabilistic and the aerial surveys are the estimates of seagrass area from the >10% density class from the large quadrats used in Tillamook, Coos, and the Umpqua.

7.2 Across-Estuary Patterns

7.2.1 Sediment Grain Size and TOC in the Target Estuaries

Two key sediment characteristics quantified were the percent total organic carbon (TOC) and grain size, which was summarized as percent fines. The "muddiest" estuaries were the Yaquina and Coos with 50% of the area having percent fines ≥30% (Table 7-2, Figure 7-10). The sandiest estuary was the Nestucca with median percent fines of less than 2%. The Umpqua, Salmon, Alsea, and Tillamook estuaries were intermediate with median percent fines ranging from about 8% to 20%.

The median values for the percent total organic carbon (TOC) generally mirrored the results for percent fines with Yaquina and Coos having the highest median values and Nestucca the lowest (Table 7-2, Figure 7-11). The National Coastal Condition Report II (U.S. EPA, 2004b) used TOC values >5% as an indication of "poor" sediment condition and values between 2% and 5% as an indication of "fair" condition. Only 2 out of almost 700 stations across all the estuaries exceeded 5% TOC, with a maximum value of 5.3% in the Coos Estuary. All seven estuaries had TOC values >2% and <5%, though most values were at lower end of the "fair" range (<3%). The highest coverages within this range occurred in the Yaquina and Coos, where 21% and 17% of the intertidal sites were classified as "fair", respectively. Based on the minimal development in the associated watersheds (Chapter 2) and the lack of major point sources, these higher TOC values do not appear to be the result of anthropogenic loading.

Table 7-2. Sediment characteristics of the target estuaries from the probabilistic surveys. Median values based on the areal extent of the parameter in the intertidal. The 75th percentile was used for distance between wave crests because all the median values were 0. Total organic carbon (TOC) values between 2% and 5% are classified as “fair” in the National Coastal Condition Report II (U.S. EPA 2004b) while values >5% are classified as “poor”.

ESTUARY	MEDIAN % FINES	MEDIAN % TOC	% OF AREA WITH 2%< TOC <5 %	% OF AREA WITH TOC >5%	MEDIAN % N	75th PERCENTILE OF DISTANCE BETWEEN WAVE CRESTS (cm)	MEDIAN SEDIMENT SALINITY (psu)
Alsea	12.1	0.69	2.5	0	0.06	NA	28.0
Coos	30.7	0.90	17.0	<1	0.07	<1	29.7
Nestucca	1.9	0.16	7.1	0	0.02	NA	29.6
Salmon	7.9	0.34	7.3	<1	0.07	NA	32.0
Tillamook	12.4	0.41	2.6	0	0.04	12	25.1
Umpqua	19.7	0.51	9.6	0	0.05	14	19.4
Yaquina	37.9	1.30	21.4	0	0.10	NA	31.2

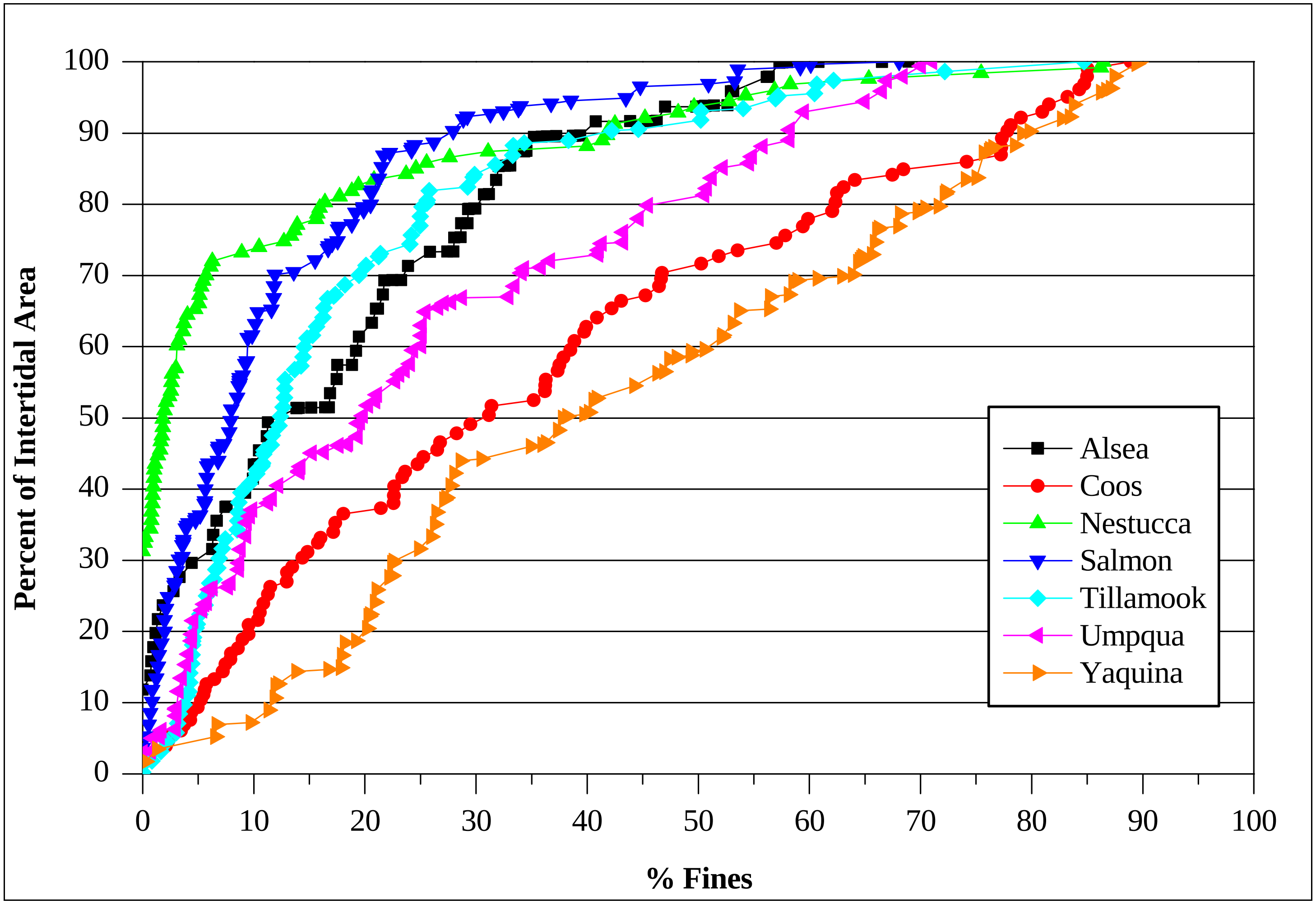

Figure 7-10. Cumulative distribution function (CDF) of the percent fines in the intertidal zone of the seven target estuaries. The nine samples taken in the Alsea in September, 2006 are not included.

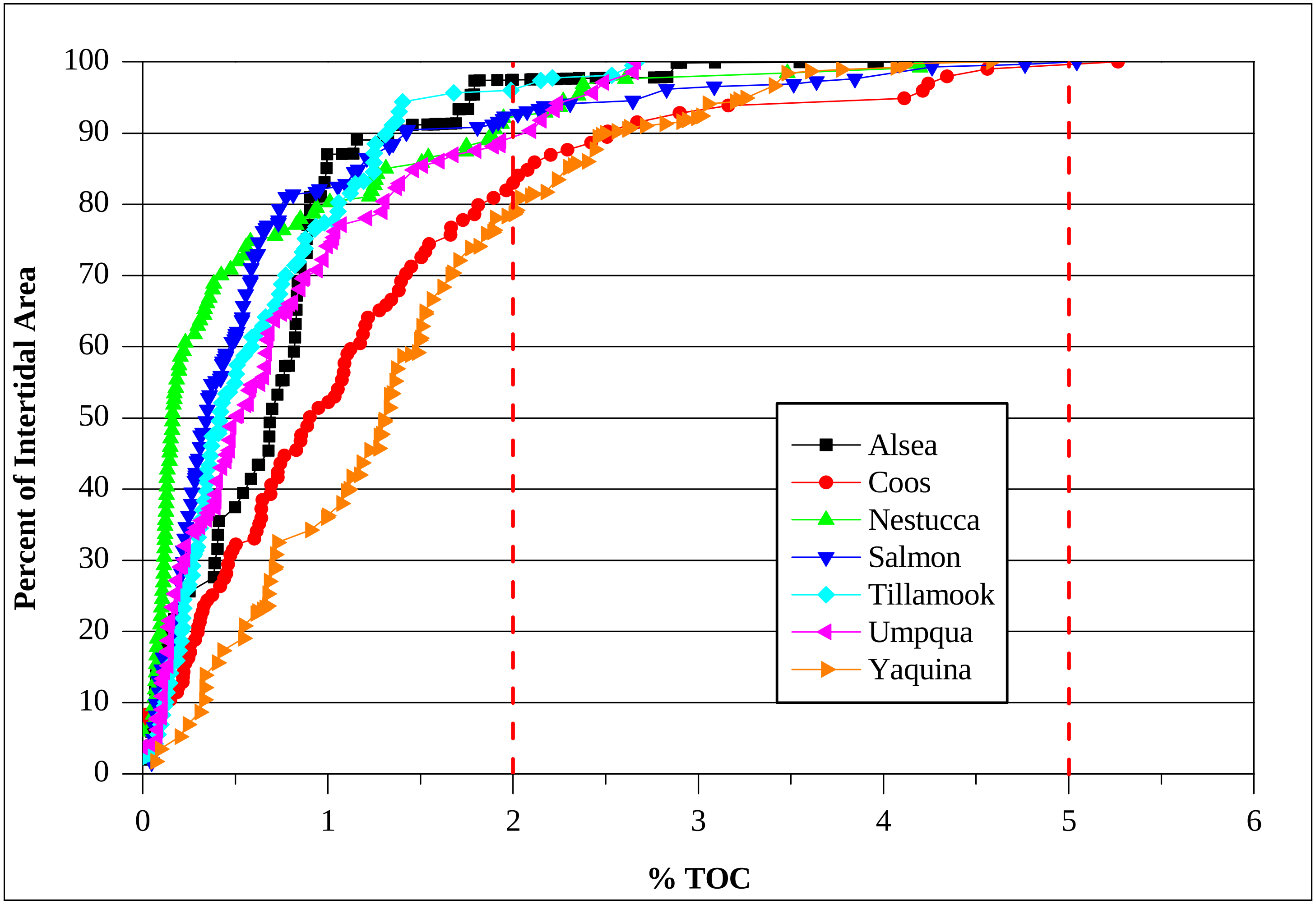

Figure 7-11. Cumulative distribution function (CDF) of the percent total organic carbon (TOC) in the intertidal zone of the seven target estuaries. The nine samples taken in the Alsea in September, 2006 are not included. The red dashed vertical lines indicates the 2% and 5% TOC thresholds used in the National Coastal Condition Report II (U.S. EPA, 2004b) with “fair” sediment conditions defined as between 2% and 5% and “poor” sediment conditions as >5%.

7.2.2 Among-Estuary Distribution of *Zostera marina*

Tillamook had the greatest areal extent of *Z. marina*, occurring over 34% of the intertidal zone (Table 7-3), while the Coos and Yaquina had coverages of about 12-17%. In contrast, the Umpqua had only about 6% coverage and no *Z. marina* was found in the probability surveys in the Alsea, Salmon, or Nestucca estuaries. Though the probability surveys did not detect *Z. marina* in these estuaries, the aerial photography detected small amounts of *Z. marina* in the Alsea and on-ground mapping found small beds in the Salmon and Nestucca (see Table 6-3). As with any sampling method, the probabilistic survey has a detection limit and even a survey of 100 sites per estuary can fail to detect seagrasses with high confidence at low densities (e.g., Bailey et al., 1998). While the probabilistic surveys failed to detect these small amounts of seagrass, the surveys were qualitatively correct in classifying these estuaries as having low coverage.

The relatively wide expanses of seagrass found in Tillamook, Coos, and Yaquina as well as the large expanses reported in Willapa Bay and Grays Harbor (e.g., Wyllie-Echeverria and Ackerman, 2003) suggest that moderate to large tide-dominated estuaries contain proportionally large areas of suitable habitat for *Z. marina.* In comparison, river-dominated estuaries have substantially less *Z. marina* habitat. Only small beds occur in the Alsea, Nestucca, and Salmon estuaries, while the Umpqua only contains one-half to one-sixth the percent cover compared to the tide-dominated estuaries (Table 7-3). Estuarine size alone does not explain this pattern, as the Umpqua and Alsea are roughly comparable in size to the Coos and Yaquina, respectively. Additionally, the Netarts Estuary, which is smaller than either the Yaquina or Alsea, has extensive seagrass coverage (Kentula and McIntire, 1986; Wyllie-Echeverria and Ackerman, 2003).

The occurrence of this pattern is supported by the low percent cover of NWI aquatic beds in the river-dominated estuaries. The average percent cover of NWI "aquatic beds" classes in the 22 river-dominated estuaries was 1.8% compared to 21.5% in the 6 tide-dominated estuaries (see Table 2-3). Limiting this comparison to estuaries >10 km^2, the percentages were 6.1% versus 25.7% in river- and tide-dominated estuaries, respectively. Though not as large a difference, the same pattern was observed in the mid-1970's survey of Oregon estuaries (Cortright et al., 1987). In this study, the average percent cover for seagrass and seagrass/algae classes was 15.7% in the three tidal estuaries compared to 7.8% in eleven river-dominated estuaries. It appears, then, that our definition of tide- and river-dominated estuaries helps to separate estuaries that support more extensive *Z. marina* populations.

We hypothesize that the most important drivers resulting in these among-estuary patterns are differences in average salinity values and, perhaps more importantly, differences in the temporal variation in salinity. In Puget Sound, *Z. marina* appears to grow best at 20-32 psu (Phillips, 1984), and higher growth rates occurred at 30 psu compared to 10 psu in experiments (Thom et al., 2001, 2003). Additionally, diversions of fresh water into estuaries have been associated with declines in seagrass (Estevez, 2000), while Montague and Ley (1993) found that increasing salinity variation was associated with declines in seagrass biomass in Florida. Coastal PNW estuaries are subject to substantial seasonal salinity variations in response to the seasonal pattern in precipitation, with winter salinities dropping below 10 psu or even 5 psu over much of the area of many estuaries (e.g., OR Dept. State Lands, 1985; Brown, unpubl. data; Figure 2-11). In particular, river-dominated estuaries appear to experience greater seasonal variations than tide-dominated systems (Figure 2-11; OR Dept. State Lands, 1985; Haertel and Osterberg, 1967). These extreme seasonal fluctuations are well illustrated in the Salmon River where differences in summer and winter median salinities approach 20 psu and where much of the estuary has a salinity <5 psu during the winter (Figure 2-11). In the Umpqua, much of the estuary has a winter salinity <10 psu (Figure 2-11). In addition to this seasonal variability, dry season salinity near the mouths of PNW estuaries can vary 10-25 psu over a tidal cycle (Figures 4-9 and 4-10). The extent of this tidal variation is related to the normalized river inflow (Figure 4-11), with the river-dominated systems demonstrating about a 2- to 10-fold greater variation in high and low tide salinities than the tide-dominated estuaries. The extensive tidal variation within the Alsea (Figure 4-10) may contribute to the paucity of seagrass in that system even though seasonal fluctuations are not as great as in the Salmon or Umpqua estuaries. At the opposite extreme, the occurrence of extensive seagrass beds in Netarts Bay (Kentula and McIntire, 1986), which is characterized by minimal seasonality in salinity (Figure 2-11), is consistent with the hypothesis that salinity variations limit seagrass abundance and distribution.

A different and potentially complementary mechanism reducing *Z. marina* in river-dominated systems is the greater current energy and sediment transport in these high flow systems, as suggested in Section 6.2.2. Yet another possible contributing factor may be the presence of the burrowing shrimp *N. californiensis*. *N. californiensis* can inhibit *Z. marina* (Dumbauld and Wyllie-Echeverria, 2003), and its extensive coverage in both the Salmon and Nestucca estuaries (Figure 7-19) may make it more difficult for *Z. marina* to become established. While these suggestions need to be evaluated further, several independent data sets indicate that river-dominated systems within the PNW have reduced *Z. marina* cover compared to tide-dominated and bar-built estuaries.

7.2.3 Among-Estuary Distribution of *Zostera japonica*

The regional distribution of the nonindigenous *Z. japonica* differed from the native seagrass in several respects. *Z. japonica* was found in six of the target estuaries in the probability surveys, including the Salmon and Nestucca (Table 7-4). The only target estuary where *Z. japonica* was not found was the Alsea, and we are unaware of any reports of this species in the Alsea. In contrast to *Z. marina*, *Z. japonica* occupied moderately large areas in both tide-dominated estuaries, such as the Coos, and the river-dominated Umpqua. Besides these six estuaries, *Z. japonica* has been reported in a wide range of PNW estuaries including the river-dominated Columbia, Siletz, Necanicum, and Coquille estuaries; the tide-dominated Willapa and Grays Harbor estuaries; and the bar-built Humboldt and Netarts (Lee and Reusser, 2006). In the probabilistic surveys, the intertidal extent of *Z. japonica* actually exceeded that of *Z. marina* in the Salmon, Nestucca, Coos, and Umpqua estuaries. Coverage of *Z. japonica* in these four estuaries ranged from 4% to 23% compared to 0% to 12% for *Z. marina* based on the integrated measure of presence/absence (Tables 7-3 and 7-4). This comparison is only for the intertidal populations, and inclusion of the shallow subtidal *Z. marina* could result in more similar percentages among the two species since *Z. japonica* is rare or absent in the shallow subtidal in these estuaries.

Table 7-3. Percent of intertidal area with *Z. marina* estimated using different methods. Estimates for presence/absence (P/A) integrate several types of data over the 2.5 x 2.5 m quadrat (see Section 7.1.3). Estimates for “Large Quadrat” are the percent of the intertidal area with >1% cover or >10% cover within the 2.5 x 2.5 m quadrats. Estimates from the point intercepts are the areas where there was at least one intercept with *Z. marina* in the 0.25 m^2 plant quadrats. NA = not available.

ESTUARY	# SITES WITH *Z. MARINA* P/A	% INTERTIDAL AREA P/A	% INTERTIDAL AREA LARGE QUADRAT COVER >1%	% INTERTIDAL AREA LARGE QUADRAT COVER >10%	% INTERTIDAL AREA POINT INTERCEPT >0
Alsea	0	0	NA	NA	0
Coos	12	11.7	10.4	2.4	1.4
Nestucca	0	0	NA	NA	0
Salmon	0	0	NA	NA	0
Tillamook	28	34.2	34.2	15.6	18.6
Umpqua	8	5.5	5.5	0.5	0.5
Yaquina	11	17.4	NA	NA	12.2

Table 7-4. Percent of intertidal area with *Z. japonica* estimated using different methods. Estimates for presence/absence (P/A) integrate several types of data over the 2.5 x 2.5 m quadrat (see Section 7.1.3). Estimates for “Large Quadrat” are the percent of the intertidal area with >1% cover or >10% cover measured within 2.5 x 2.5 m quadrats. Estimates from the point intercepts are the percent intertidal areas where there was at least one intercept with *Z. japonica* in the 0.25 m^2 plant quadrats. NA = not available.

ESTUARY	# SITES WITH *Z. JAPONICA* FROM P/A	% INTERTIDAL AREA P/A	% INTERTIDAL AREA LARGE QUADRAT COVER >1%	% INTERTIDAL AREA LARGE QUADRAT COVER >10%	% INTERTIDAL AREA POINT INTERCEPT >0
Alsea	0	0	NA	NA	0
Coos	17	19.4	19.4	10.0	12.3
Nestucca	19	23.4	NA	NA	8.6
Salmon	3	3.6	NA	NA	2.0
Tillamook	9	10.5	10.5	5.0	4.3
Umpqua	22	20.7	19.3	3.2	9.0
Yaquina	18	11.9	NA	NA	7.5

Zostera japonica has obtained these extensive within- and among-estuary distributions within the last 30 to 50 years since it was first recorded on the Pacific Coast in 1957 and in Yaquina in 1976 (Harrison and Bigley, 1982; Bayer, 1996). Further, *Z. japonica* is continuing to rapidly expand its distribution within portions of the Yaquina (Young et al., 2008). While it is apparent that *Z. japonica* is expanding, the ecological consequences of this expansion are not clear. Both the native and nonindigenous seagrass species occurred within the same coastal PNW estuaries we surveyed, but they are not presently competing within these estuaries. Of the 138 occurrences of seagrasses in the probabilistic surveys, only 2 sites (1.5%) contained both species. The low overlap of the two species is due to their different intertidal distributions. *Z. marina* was primarily found in the shallow subtidal and lower intertidal zones, while *Z. japonica* tended to colonize a band higher in the intertidal and, based on our observations, around freshwater streamlets (also see Kaldy, 2006; Ruesink, 2006). However, as *Z. japonica* continues to expand, it is possible that the two species will overlap as reported for areas in Puget Sound and British Columbia (Nomme and Harrison, 1991) and in Netarts (Dudoit, 2006), and then potentially compete for space and light (Wonham, 2003; Bando, 2006). Additionally, as *Z. japonica* expands it could eventually become sufficiently abundant to have an effect on primary production and other components of estuarine food webs (e.g., Posey, 1988; Wonham, 2003; Ruesink et al., 2006; Kaldy, 2006).

7.2.4 Composition and Seasonality of Benthic Green Macroalgae

Determining benthic green macroalgal distribution and abundance presents two additional challenges compared to the seagrasses. The first is that green macroalgae are composed of several species. In the Yaquina Estuary in 2001, benthic green macroalgae was found to be comprised on average of taxa most closely resembling *Ulva linza*: ~60%; *U. fenestrata*: ~30%; *U. flexuosa*: ~10%; *U. intestinalis*: < 5% (D. Young, unpubl. data). Given the difficulty in separating the species, all the green macroalgae are treated as a single taxon.

Another issue is the seasonality of the macroalgal blooms, which could potentially confound differences among estuaries if they are sampled at different times. To address macroalgae seasonality, we utilized data from a 1999 survey in the Yaquina that quantified seasonal changes to derive a seasonal adjustment for biomass. The 1999 survey measured percent cover and biomass of *Z. marina* and benthic green macroalgae monthly from June through December as a function of distance from estuary mouth and elevation above MLLW at six sites within the lower Yaquina Estuary (D. Young, unpubl. data). For this analysis, we took the mean of the monthly averages of both percent cover and biomass (grams dry weight [gdw] m^{-2}) for all six sites. These data illustrate the strong seasonality in macroalgae in both cover (Figure 7-12) and biomass (Figure 7-13), which increase from June through September and decline rapidly after October. At its peak in September and October, the average macroalgal dry weight exceeded 200 gdw m^{-2} and average surficial cover exceeded 50% at these non-random stations.

While aware of this seasonality, the extensive logistic effort required in the probabilistic surveys necessitated sampling from early June through mid September (Figure 7-14). The Yaquina Estuary was sampled earliest in the growing season, while the Alsea Estuary was sampled during September, the period of peak macroalgal biomass (Figure 7-14). To minimize underestimation of macroalgal occurrence during the initial phase of its growth spurt, the integrated

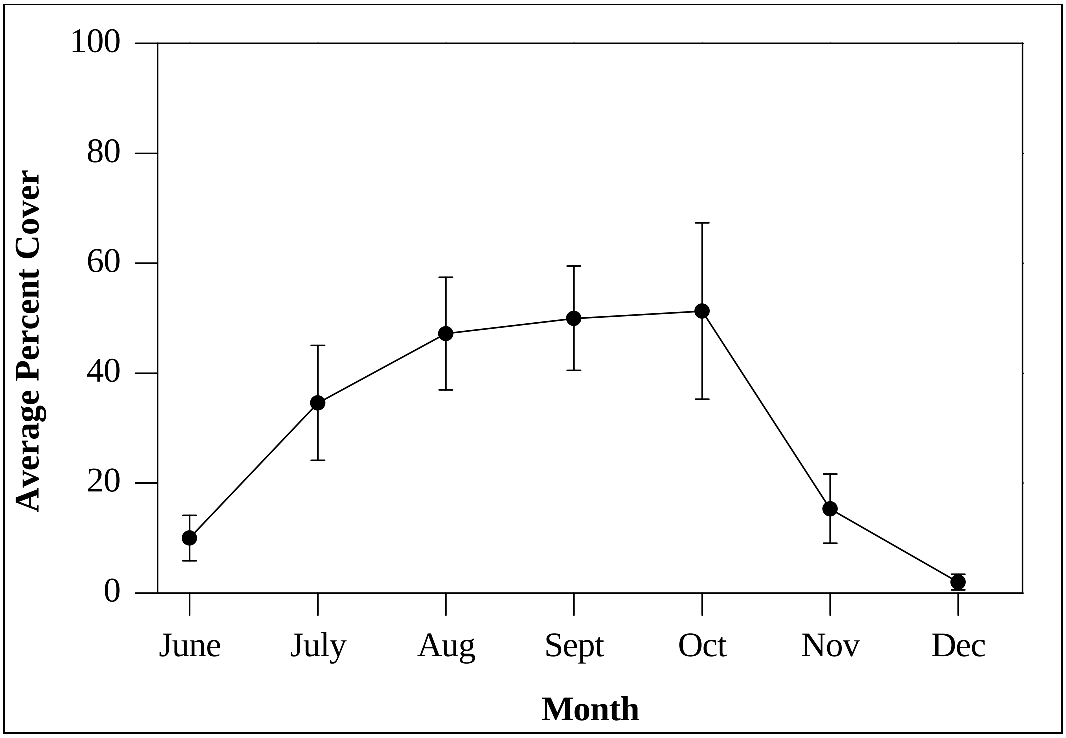

Figure 7-12. Average percent cover values (± 1 std. error) of benthic green macroalgae at six sites in the Yaquina Estuary during 1999.

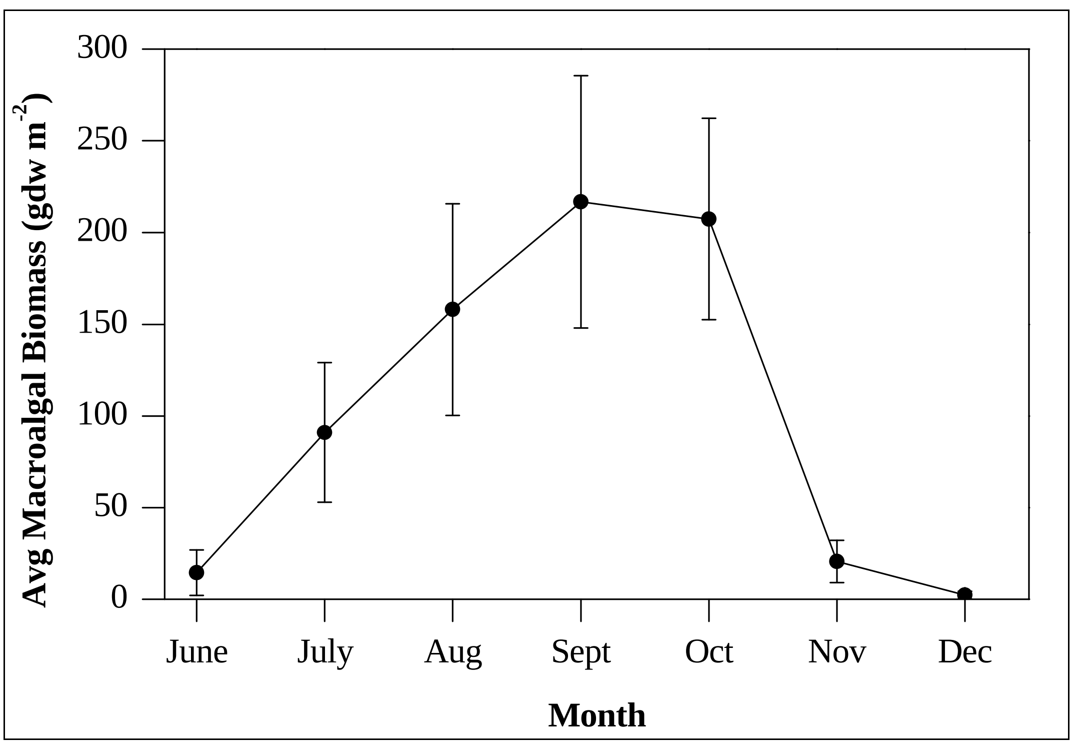

Figure 7-13. Average biomass values (± 1 std. error) of benthic green macroalgae at six sites in the Yaquina Estuary during 1999.

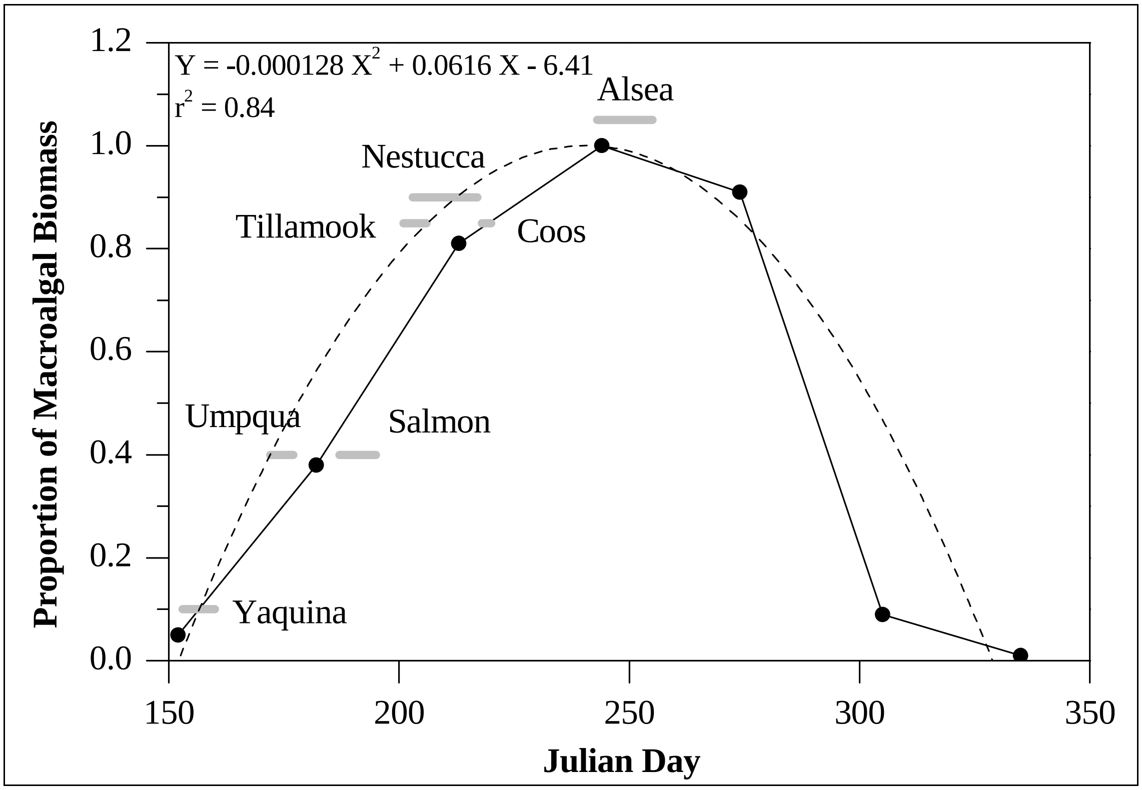

Figure 7-14. Average proportions of green benthic macroalgal biomass relative to the September peak measured in the Yaquina Estuary during 1999. Dates are given in Julian days. The solid line connects the monthly averages while the dashed line is the polynomial regression fit to the monthly data. The polynomial regression was used to derive the seasonally adjusted macroalgal biomass in all of the target estuaries based on sampling date. The horizontal gray bars indicate the sampling dates for each target estuary.

presence/absence from the 2.5 x 2.5 m quadrats was used as the primary data source for distribution since its larger sample area is more likely to detect sparse populations. To correct for seasonality in biomass, we derived a polynomial regression from the 1999 Yaquina dataset relating the relative percentage of biomass to the monthly maximum value in September (Figure 7-14). This polynomial was used to derive seasonally adjusted biomass estimates based on actual sampling dates with the assumption that similar seasonal patterns occur in all Oregon estuaries. This approach generated estimates of standing stock within the range previously observed in Yaquina in six of the estuaries. However, the model predicted biomasses greater than 12,000 gdw m^{-2} at a few Yaquina sites, several-fold higher than have been observed in our previous studies or reported in other studies in Yaquina (Davis, 1981; Kentula and DeWitt, 2003; D. Young and B. Boese, unpubl. data). It is possible that the model over predicted biomass at these stations since these predictions utilized the steepest portion of the adjustment curve (Figure 7-14). In any case, the maximum seasonally adjusted values in the Yaquina need to be evaluated cautiously

7.2.5 Among-Estuary Patterns of Benthic Macroalgae

Six of the target estuaries had areal coverages of macroalgae in the range of 41% to 64% of the intertidal zone while macroalgae occurred over 22% of the intertidal zone in the Umpqua (Table 7-5). Benthic macroalgae has been reported from a number of other coastal estuaries in the PNW as well as from Puget Sound (Phillips, 1984; Thom, 1984; Kentula and McIntire, 1986; Nelson and Lee, 2001; Thom et al., 2003), and we have observed macroalgal mats in a number of additional moderate-sized estuaries such as the Siletz. The presence of benthic macroalgae in systems ranging from Puget Sound to the Salmon Estuary indicate that macroalgae is a wide-spread feature of PNW estuaries greater than about 3 km^2 in size. Limited observations suggest that green benthic macroalgae is sparse or absent in estuaries smaller than about 0.5 km^2, while additional observations are needed for estuaries between 0.5 and 3.0 km^2 in size.

At a sufficiently high density, macroalgae can result in anoxic sediment conditions, smother benthos and seagrasses, and reduce light availability for seagrasses (e.g., Hull, 1987). In lieu of an established threshold relating percent cover to these effects, we use >70% cover as the threshold for "high" macroalgal cover. At this percent cover, macroalgae "blanket" the sediment surface though we have no direct evidence it results in detrimental effects to seagrasses. Based on the point intercepts (Figure 7-15), the estuaries divided into three groups with the Umpqua having no high macroalgal cover, the Alsea, Coos, Nestucca, Salmon, and Tillamook estuaries having moderate extents (4-7%) of high cover, and the Yaquina Estuary having an extensive area (18%) of high macroalgal cover. This high cover occurred within the Yaquina Estuary even though it was sampled the earliest during the macroalgal growth season (Figure 7-14). Results from the 2.5 x 2.5 m quadrats show a similar pattern in the three estuaries where the data are available. High macroalgal cover occurred over 6-8% of the intertidal area in the Coos and Tillamook estuaries and was not detected in the Umpqua (Figure 7-16).

Another indicator of potential macroalgal impact is standing stock, with a value of 100 gdw m^{-2} identified as a potential threshold for impacts on seagrasses in Chesapeake Bay (Bricker et al., 2003). This threshold is in the general range of observed effects from an experimental removal of the green macroalgae *Ulvaria obscura* on shoot production in subtidal *Z. marina* in Puget Sound (Nelson and Lee, 2001). In this experiment, sequential removals of about 144 gdw m^{-2}, 40 gdw m^{-2}, and 115 gdw m^{-2} over a month resulted in a slower decline in shoot production than in the controls. The percentages of the intertidal area exceeding this threshold using the unadjusted biomass (Figure 7-17) and seasonally adjusted biomass (Figure 7-18) were similar in six of the target estuaries (Table 7-6). In the Yaquina Estuary, there was about a two-fold difference between the estimates, with about 20% of the intertidal exceeding the threshold using the unadjusted biomass versus about 42% using the seasonally adjusted values. Based on either the unadjusted or adjusted values, the target estuaries break into three relatively distinct groups. The first is the Yaquina Estuary with a high percentage of the intertidal zone (20-42%) exceeding the biomass threshold, which is consistent with the pattern based on percent cover (Figure 7-15). The second group consists of the Alsea, Coos, Nestucca, and Tillamook estuaries with a moderate percentage (4-6%) of the intertidal exceeding the biomass threshold. The third group consists of the Salmon and Umpqua estuaries which had minor areas ($\leq$1%) exceeding the threshold. The Salmon differed from the Umpqua in having an extensive area of macroalgal cover though the biomass did not reach high standing stock.

Table 7-5. Percent of intertidal area with green benthic macroalgae estimated using different methods. Estimates for presence/absence (P/A) integrate several types of data over the 2.5 x 2.5 m quadrat (see Section 7.1.3). Estimates for "Large Quadrat" are the percent of the intertidal area with >1% cover or >10% cover measured within 2.5 x 2.5 m quadrats. Estimates for the point intercept are intertidal areas where there was at least one intercept with macroalgae in the 0.25 m^2 plant quadrats. NA = not available.

ESTUARY	# SITES WITH MACROALGAE FROM P/A	% INTERTIDAL AREA P/A	% INTERTIDAL AREA LARGE QUADRAT COVER >1%	% INTERTIDAL AREA LARGE QUADRAT COVER >10%	% INTERTIDAL AREA POINT INTERCEPT >0
Alsea	44	54.4	NA	NA	35.2
Coos	60	54.9	53.7	10.4	18.3
Nestucca	44	41.4	NA	NA	12.9
Salmon	71	56.2	NA	NA	43.2
Tillamook	65	63.6	54.1	14.9	17.8
Umpqua	26	22.2	22.1	3.2	2.6
Yaquina	35	58.4	NA	NA	48.2

Table 7-6. Biomass (M_B) estimates for benthic macroalgae. The percent of the intertidal area with M_B >100 gdw m^{-2} was interpolated from the benthic macroalgal cumulative distribution functions (CDFs, Figures 7-16 and 7-17). The number of sites with unadjusted and seasonally adjusted M_B >100 gdw m^{-2} indicates the number of sampling sites that increased above the threshold after the seasonal adjustment. Note that the slightly different percent areas with the unadjusted and adjusted values, even though there are the same number of samples, is due to interpolating values from different CDF curves.

ESTUARY	# SITES WITH M_B MEASUREMENTS (M_B >0 gdw m^{-2})	# UNADJUSTED SITES WITH M_B >100 gdw m^{-2}	# SEASONALLY ADJUSTED SITES WITH M_B>100 gdw m^{-2}	% INTERTIDAL AREA WITH UNADJUSTED M_B>100 gdw m^{-2}	% INTERTIDAL AREA WITH SEASONALLY ADJUSTED M_B >100 gdw m^{-2}
Alsea	24	3	3	4.0	5.7
Coos	31	6	6	4.9	5.5
Nestucca	14	5	5	3.9	4.0
Salmon	50	0	0	0	0
Tillamook	38	11	12	5.8	6.6
Umpqua	12	1	2	<0.1	0.3
Yaquina	31	12	26	18.0	42.3

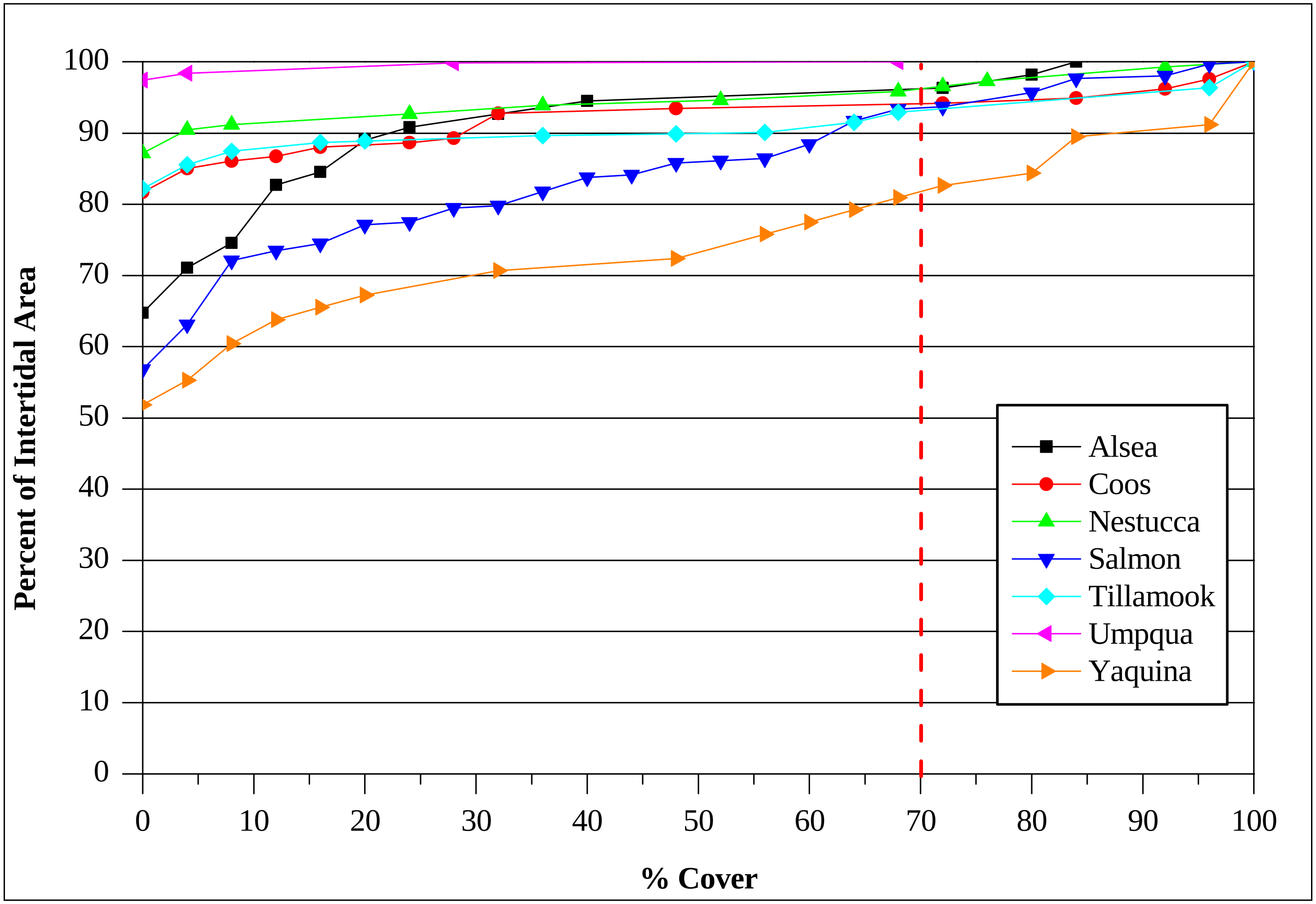

Figure 7-15. Cumulative distribution function (CDF) of the % cover of benthic macroalgae estimated from the point-intercept method using the 0.25 m^2 quadrats. The red dashed vertical line indicates 70% cover which is used as the threshold for "high" cover.

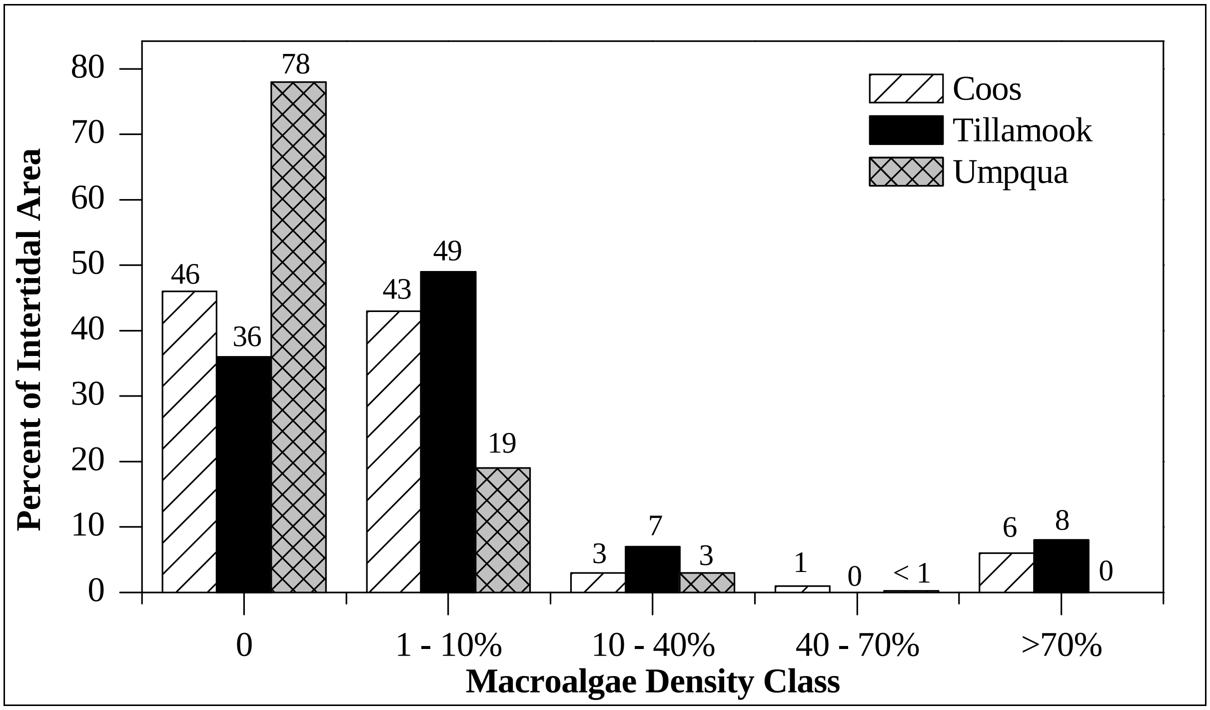

Figure 7-16. Percent of the intertidal area covered by benthic macroalgae by density class. The data are from estimates in the 2.5 x 2.5 m quadrats in the 2005 and 2006 surveys. The threshold for "high" macroalgal cover is >70%.

The overall pattern that emerges is that seasonal macroalgal blooms are a common feature across a range of different types and sizes of PNW estuaries. However, estuaries vary several fold in the extent of their intertidal areas with high percent cover (>70%) or standing stocks >100 gdw m^{-2}, with the highest values occurring in the Yaquina. One possible reason for the high macroalgal biomass in the Yaquina Estuary is its high nutrient concentrations as indicated by having the highest wet season dissolved inorganic nitrogen (DIN) concentrations as well as the second highest cover of red alder in its watershed (Table 4-3). It is possible that the macroalgae respond to this high nitrogen concentration early in the growing season when light limitation is removed, rapidly building up a high biomass. The importance of DIN is further suggested by the system with the lowest cover of macroalgae (Umpqua) also having the lowest wet season DIN (Table 4-3). Regardless of the cause(s), macroalgae may reach biomasses sufficient to adversely impact seagrasses over a moderate to substantial portion of the intertidal in the Yaquina Estuary and over lesser areas in the Alsea, Coos, Nestucca, and Tillamook estuaries, assuming the Chesapeake Bay biomass threshold is appropriate for the PNW. As discussed in Section 7.4, these blooms appear to be a natural phenomenon and not an indication of cultural eutrophication.

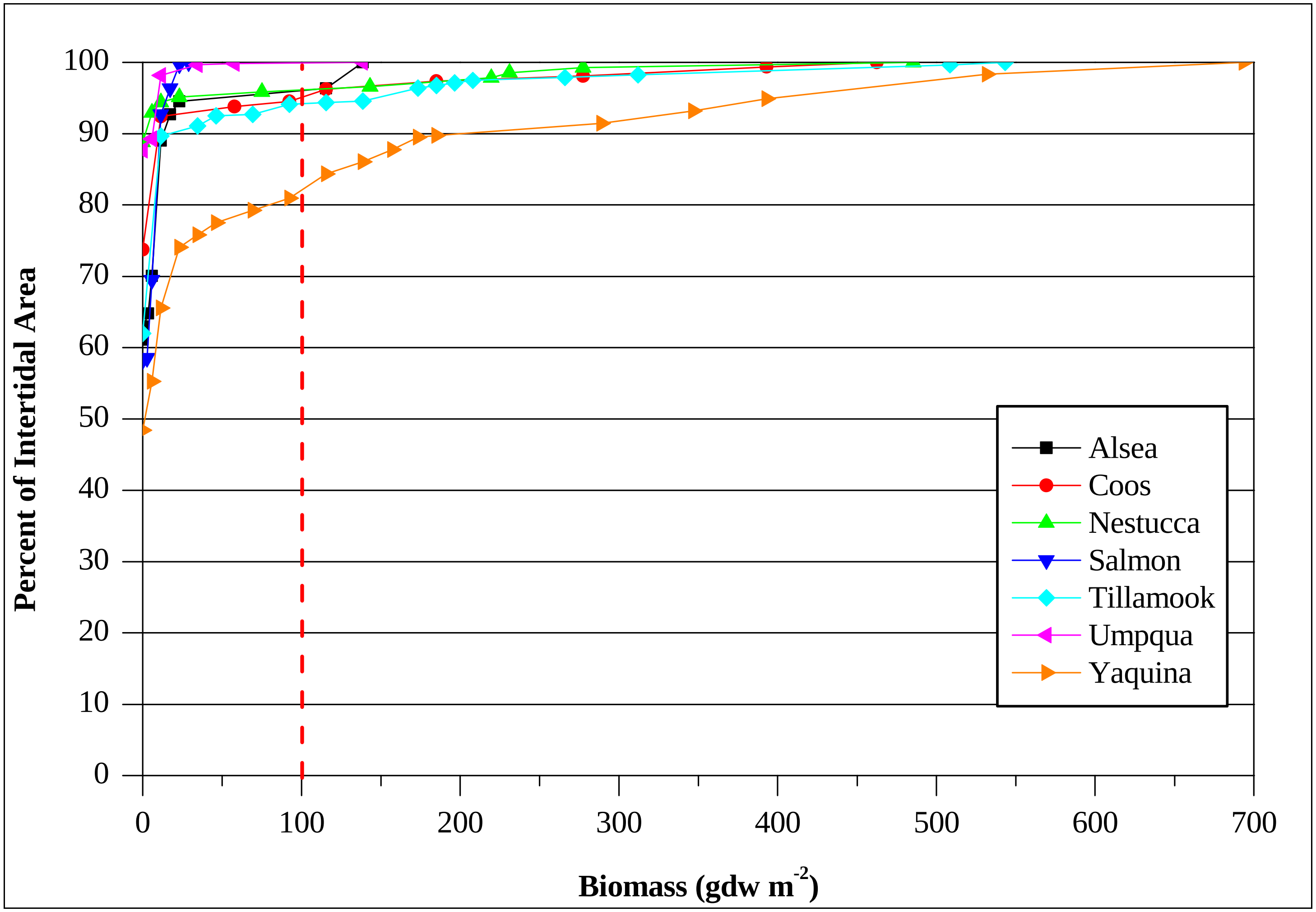

Figure 7-17. Cumulative distribution function (CDF) of the unadjusted benthic macroalgal biomass (gdw m^{-2}). The red vertical line indicates the 100 gdw m^{-2} value which has been associated with adverse impacts on seagrasses in Chesapeake Bay.

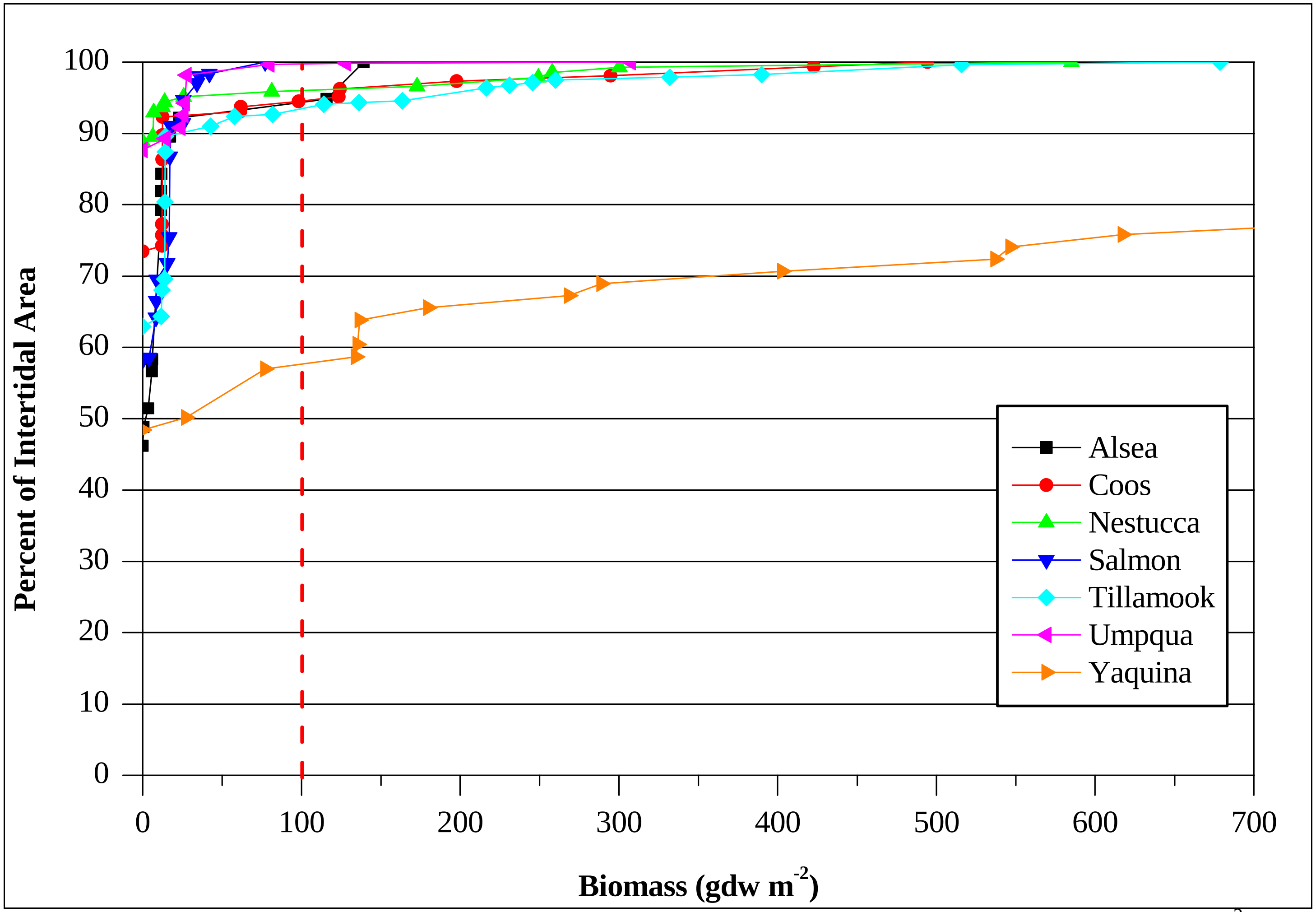

Figure 7-18. Cumulative distribution function (CDF) of the seasonally adjusted benthic macroalgal biomass (gdw m^{-2}). The red vertical line indicates the 100 gdw m^{-2} value which has been associated with adverse impacts on seagrasses in Chesapeake Bay. The Yaquina distribution extends to approximately 12,000 gdw m^{-2}.

7.2.6 Among-Estuary Patterns of Burrowing Shrimp

The spatial extent and density of burrowing shrimp were measured as burrow hole counts in the 0.25 m^2 quadrats, visual estimates using five density classes in the 2.5 x 2.5 m quadrats, and from an integrated measure of presence/absence. Because of their high density, the burrow hole counts in the 0.25 m^2 quadrats were an effective method to determine both the presence and density of both species.

Neotrypaea californiensis beds were a major feature in all the target estuaries, occurring over 21% to 74% of the intertidal area (Figure 7-19). The Salmon River had both the greatest area of the intertidal occupied by *N. californiensis* and the densest population, with burrow hole counts exceeding 100 per 0.25 m^2 at several sites. *Neotrypaea* also occupied a large extent of the intertidal in the Nestucca though populations were not as dense as in the Salmon. The areal extent and abundance of this burrowing shrimp in the other five estuaries varied from the largest in the Alsea to the lowest in the Coos Estuary. The other burrowing shrimp, *U. pugettensis,* showed a different pattern among the estuaries (Figure 7-20). Maximum coverage of *U. pugettensis* occurred in the Alsea with 53% of the intertidal area occupied and a maximum density exceeding 50 burrow holes per 0.25 m^2. The Coos, Nestucca, Tillamook, and Yaquina estuaries had moderate coverages, ranging from 17% to 33% of the intertidal area. In comparison, *U. pugettensis* occurred over less than 4% of the area in the Salmon and none were found in the Umpqua.

Our results and other studies (e.g., Feldman et al., 2000; DeWitt et al., 2004) demonstrate that burrowing shrimp are major components of the intertidal zones of estuaries larger than about 3 km^2. Of the 698 quadrats taken in the target estuaries, one or both of the burrowing shrimp occurred in 424 samples (61%). The absence of *U. pugettensis* in the Umpqua and its low coverage in both the Salmon and Nestucca (Figure 7-20) suggests that this species is less abundant in highly river-dominated estuaries, which is consistent with an earlier assessment that concluded that *N. californiensis* and *U. pugettensis* were probably absent from the Rogue Estuary (ODFW, 1979). The among-estuary pattern for *N. californiensis* does not appear to be as closely linked to extent of freshwater flushing. It has low coverage in the river-dominated Umpqua but high coverage in two other river-dominated estuaries, the Salmon and Nestucca, and its among-estuary distribution may be controlled by a number of factors. In terms of estuary size, limited observations suggest that both burrowing shrimp are sparse or absent in estuaries less than about 0.5 km^2, while additional observations are needed for estuaries between 0.5 and 3.0 km^2.

Both species directly and indirectly affect estuarine food webs within the PNW. Various fishes, birds, and even whales prey on these species (e.g., Stenzel et al., 1976; Armstrong et al., 1995; Feldman et al., 2000; Dumbauld et al., 2008). Perhaps more important are the indirect effects these bioengineering species have on nutrient fluxes and phytoplankton. The intense bioturbation and irrigation activities of these species increase the benthic flux of nitrogen (D'Andrea and DeWitt, 2003), which contributes to the total estuarine nutrient loading (Brown and Ozretich, 2009). At the same time, these species filter large quantities of the overlying water (Griffen et al., 2004), potentially decreasing phytoplankton concentrations and turbidity.

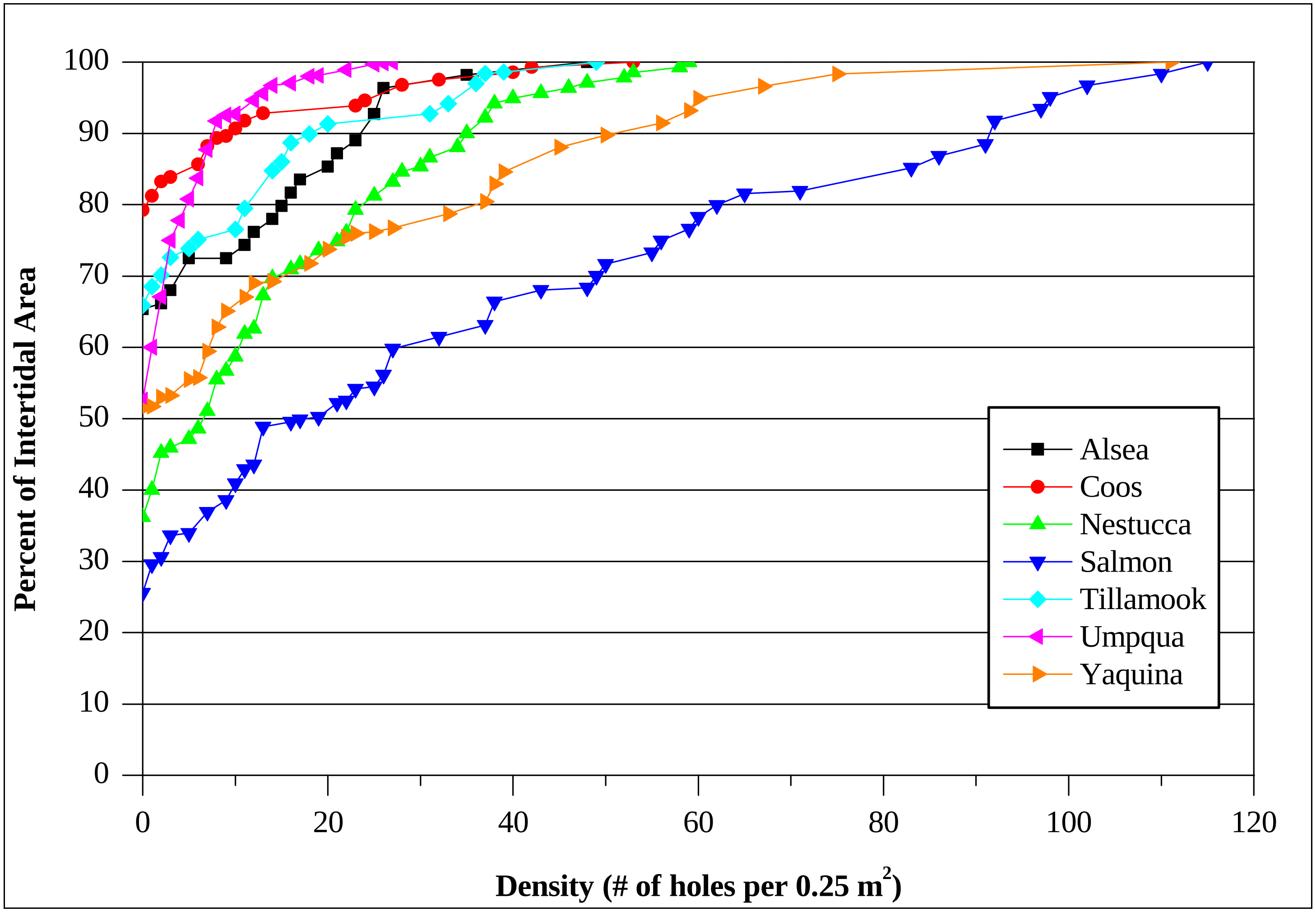

Figure 7-19. Cumulative distribution function (CDF) of the number of *N. californiensis* burrows holes per 0.25 m^2 in the seven target estuaries.

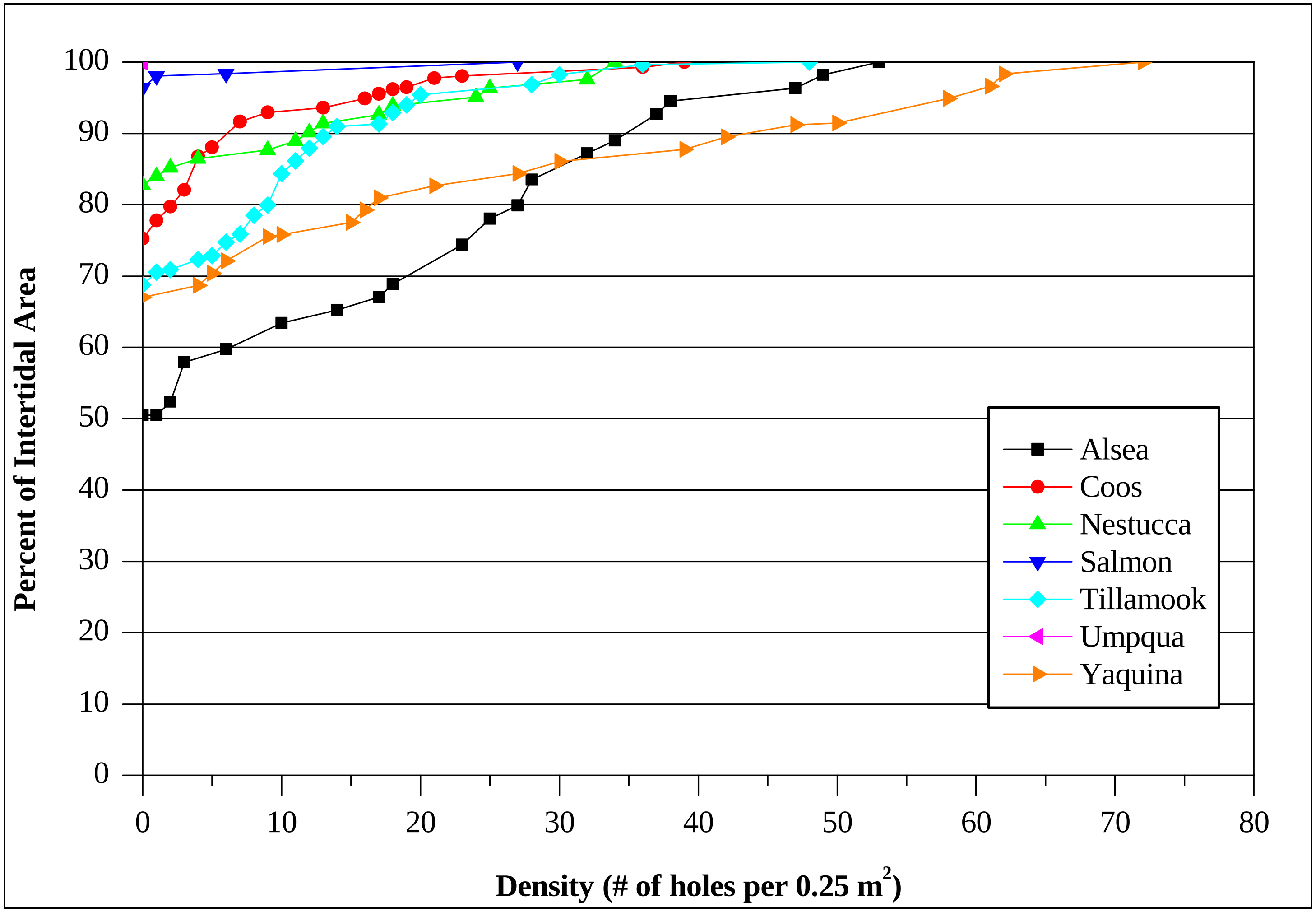

Figure 7-20. Cumulative distribution function (CDF) of the number of *U. pugettensis* burrows holes per 0.25 m^2 in the seven target estuaries.

The overall effect of these species on nutrient and phytoplankton concentrations will depend upon a number of factors, including the relative proportion of the two species and the among-estuary differences in the advection of oceanic nutrients and phytoplankton. It is important to note, however, that the role of *U. pugettensis* in the regional food webs may be diminished in response to a recent invasion by a parasitic bopyrid isopod (*Orthione griffenis*) (Markham, 2004), which may substantially reduce the density of the host shrimp within PNW estuaries (Smith et al., 2008).

7.2.7 Bare Habitat

The last benthic habitat quantified was "bare habitat", which was defined as habitat devoid of any seagrasses, macroalgae, or burrowing shrimp in the 2.5 x 2.5 m quadrats. As used here, it is meant to capture the area of the estuary that is unsuitable or at least poor habitat for these five ecologically important benthic taxa. It is not meant to imply that it is poor habitat for all species, and a number of benthic infauna characteristically inhabit "bare sediment" in the PNW (Ferraro and Cole, 2007). The coverage of bare habitat varied several fold, with only 1% of the intertidal zone of the Nestucca not having identifiable biotic structure compared to over 20% of the intertidal zone in both the Coos and Umpqua estuaries (Figure 7-21). These data indicate that at least one of the five target taxa occupied >70% to 99% of the intertidal area of these PNW estuaries.

7.2.8 Multivariate Analysis of the Target Estuaries

The differences in the areal extents of the benthic habitat types presented above allow managers to group the estuaries based on the extent of a single resource (e.g., *Z. marina*) or on a potential stressor (e.g., macroalgae). Another approach is to evaluate the overall similarity in the patterns of these benthic habitat types. Since there are a limited number of estuaries and variables (habitat types), ordination by non-metric multidimensional scaling (nMDS; Clarke and Warwick, 2001; McCune and Grace, 2002) was used instead of clustering to evaluate relationships. The nMDS analysis was conducted with Primer6 (Clarke and Gorley, 2006; http://web.pml.ac.uk/primer/primer6.htm) on the percent area occupied by the five target taxa and the bare habitat in each of the estuaries using the Bray-Curtis similarity index.

The nMDS ordination had a very low stress (<0.01) indicating that the relationships among the estuaries are well represented in two-dimensional space (Figure 7-22). The tide-dominated Coos, Tillamook, and Yaquina estuaries form a group with a very high degree of similarity (>80%), with the Alsea associated with this group at >75% similarity. The river-dominated Nestucca and Salmon estuaries form a second cluster with high similarity. These six estuaries form a single group at a moderately high (65%) similarity, with the Umpqua separate from the other estuaries. This pattern indicates that tide-dominated estuaries have very similar relative distributions of the six benthic habitats. The river-dominated Alsea is more closely aligned to this group than to the smaller river-dominated Nestucca and Salmon, suggesting that there are physical drivers related to estuary size or geomorphology that effect the biotic composition of river-dominated estuaries. The separation of the Umpqua suggests that larger highly river-dominated estuaries have different proportions of these intertidal benthic habitat types, in part because of the low areal extent of macroalgae and absence of *U. pugettensis.*

This analysis by benthic habitat type is based on a finer breakout of the intertidal habitat types than the classifications by NWI wetland classes (Chapter 2). In particular, differentiating the two burrowing shrimp habitats from unvegetated habitat and separating *Z. marina* and *Z. japonica* as discrete habitats offers a much finer resolution of the regularly flooded NWI wetland habitats. However, this analysis does not include the subtidal or emergent wetland classes that were included in the NWI classification. Even with these differences, there are a number of similarities. The Coos, Yaquina, and Alsea estuaries showed a high degree of similarity and the Umpqua was separated from the tide-dominated and moderately river-dominated estuaries in both analyses (Figures 2-14d and 7-22). The major difference is that the Salmon River showed little similarity with the other estuaries based on the NWI analysis in comparison to its relatively high similarity in the present analysis. The Salmon River separated from the other PNW estuaries in the NWI analysis largely based on its high percentage (61%) of irregularly exposed marshes, habitats not included in the present analysis.

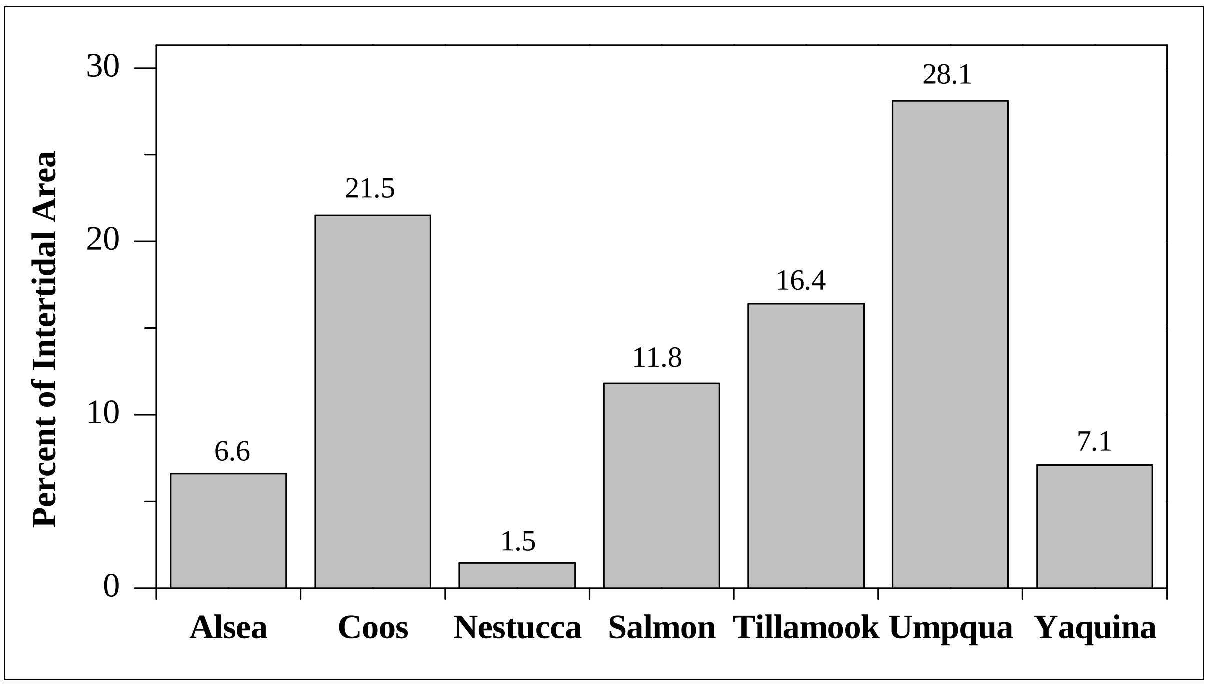

Figure 7-21. Percent of intertidal area occupied by bare habitat as defined by the absence of *Z. marina*, *Z. japonica*, benthic green macroalgae, *N. californiensis*, or *U. pugettensis* in the 2.5 x 2.5 m quadrats.

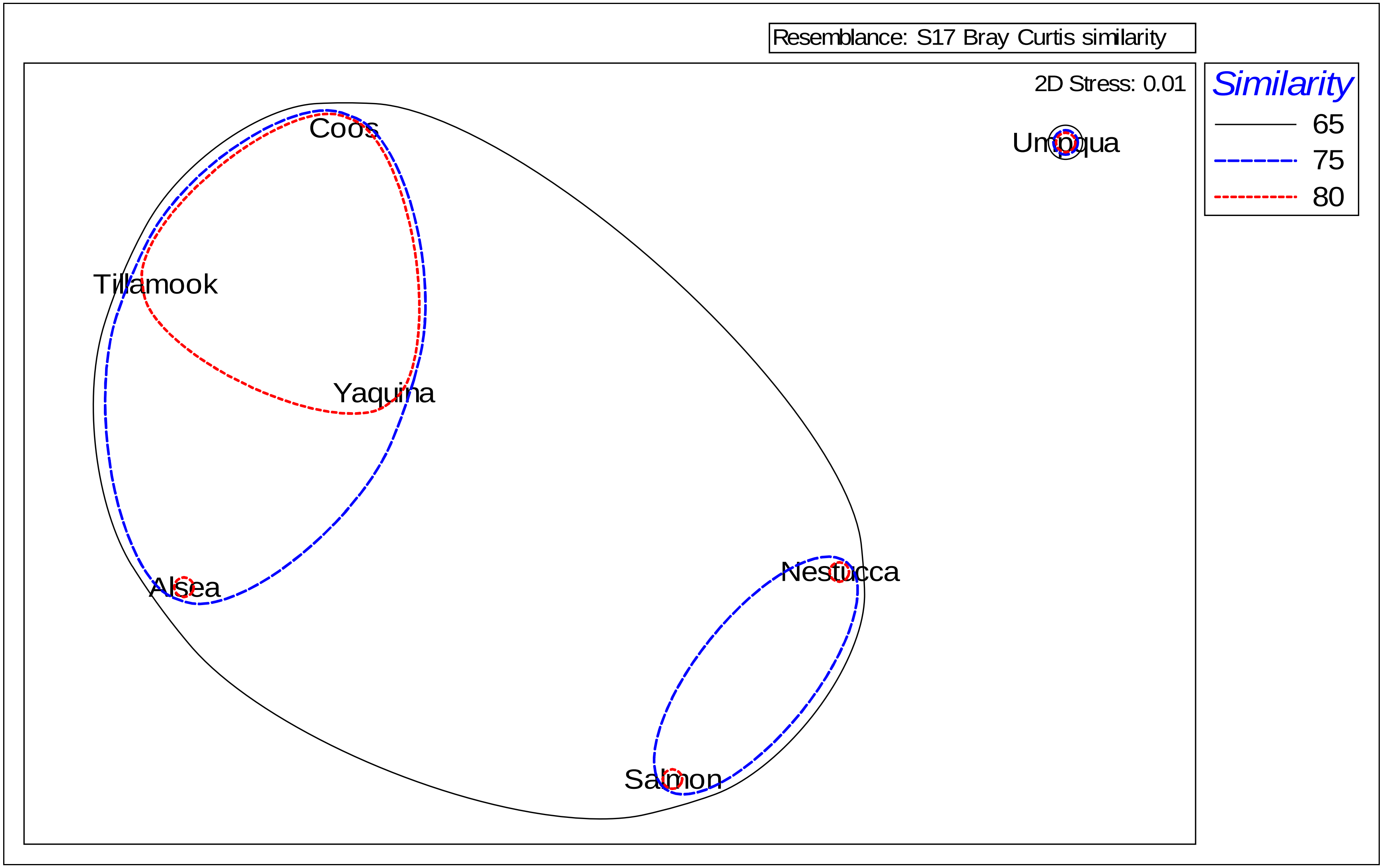

Figure 7-22. Non-metric multidimensional scaling (nMDS) of the seven target estuaries based on the percent of intertidal area occupied by *Z. marina*, *Z. japonica*, benthic macroalgae, *N. californiensis*, *U. pugettensis*, and bare habitat. Contours join estuaries at different levels of similarity as measured by the Bray-Curtis similarity index.

The comparison of the classifications based on benthic habitats versus NWI wetland classes highlights the fact that different schema can be generated depending on the suite of estuaries and variables used in the analysis. While identifying which is the “best” classification depends upon the goals of the user, a general characteristic of a successful classification is that it maximizes differences among groups while minimizing differences within groups. As discussed in Section 2.9, one approach to identifying similar estuaries for evaluation of within-group variability is to focus on estuaries that clustered together by more than one method. Of the seven target estuaries, the Coos and Yaquina estuaries show high similarity based on benthic habitats (Figure 7-22) and in the NWI and land cover crosswalk (Table 2-8). To the extent that these classifications are related to nutrient dynamics, these two estuaries should have similar nutrient and salinity patterns, which appear to be the case. Among the target estuaries, the Coos and Yaquina were ranked sequentially in the dry season concentrations of PO_4^{3-}, NO_3^-+NO_2^-, and total suspended solids (TSS) and were within two ranks for chlorophyll *a* (Tables 4-4 and 4-5). Yaquina and Coos showed a very similar extent of tidal variation at the mouth of the estuaries when plotted against the area-normalized freshwater inflow (Figure 4-11), suggesting similar short-term salinity dynamics. Additionally, Coos and Yaquina showed similar wet and dry season salinity patterns along the estuarine gradient (Figure 2-11), indicating similar seasonal dynamics. In contrast, the Umpqua was the least similar of the target estuaries in the nMDS analysis and was separated from the other target estuaries in the NWI and land use crosswalk (Table 2-8). This highly river-dominated estuary had the lowest median values for all the water quality parameters among the target estuaries during the dry season (Tables 4-4 and 4-5) and lowest wet season DIN and red alder cover in the watershed (Table 4-3), supporting the suggestion that river-dominated estuaries have different nutrient dynamics and, presumably, vulnerabilities.

7.3 Within-Estuary Distribution of Seagrasses and Other Benthic Resources

One of the objectives of the probabilistic surveys was to test the hypothesis that the majority of the *Z. marina* and other target taxa occurred within the oceanic segments (see Chapter 5) of different types of estuaries. Two nonexclusive mechanisms could generate this pattern. The first is that the total intertidal area is greater in the oceanic segments, resulting in the majority of the population occurring within the oceanic segment even if it did not constitute high quality habitat. To evaluate whether the population primarily occurred in the lower or upper estuary, we calculated the relative distributions of each taxon, which was calculated as the percent of the total area occupied by each taxon occurring in the oceanic versus riverine segments. This metric represents how the total population is split between the two estuarine segments, and sums to 100%. The second mechanism is that either the lower or upper estuary represents better habitat independent of its area. Habitat quality was estimated from each taxon’s relative cover which was independently calculated as the percent of the area of the oceanic segment occupied by the target taxon and the percent the area of the riverine segment occupied by the target taxon. For example, a taxon could occupy 10% of the area of the oceanic segment versus 25% of the riverine segment, indicating that the riverine segment provided a better relative habitat. We attempted to define sample frames *a priori* within each of the estuaries so that there was an adequate number of samples in both the oceanic and riverine segments. This initial distribution was successful in six of the seven estuaries, with $\geq$15 samples in each segment (Table 7-1).

However, only two samples were located in the riverine segment of the Nestucca, so any percentage coverages for this portion of the estuary are preliminary.

7.3.1 Within-Estuary Distributions of Seagrasses

Zostera marina was nearly exclusively found in the oceanic segments in the four estuaries where it was detected (Figure 7-23). The lowest proportion of the total area occupied by *Z. marina* in the oceanic segment was 79% in the Tillamook with 98-100% of the seagrass occurring in the oceanic segments in the Coos, Umpqua, and Yaquina. These results are qualitatively similar to those from the aerial surveys (Table 6-5). Thus, both approaches demonstrate that the majority of the native seagrass population occurs in the oceanic segments of both tide-dominated and river-dominated estuaries.

While difference in sizes of the oceanic and riverine segments contributed to this pattern (Table 7-1), the distribution of *Z. marina* is not simply a consequence of the oceanic segments being larger as indicated by its relative cover. *Z. marina* occupied very little of the intertidal zone (0-1%) in the riverine segments of the Coos, Umpqua, and Yaquina estuaries compared to its relative cover in the oceanic segments (13-17%; Table 7-7). There was a smaller difference between the two segments in the Tillamook (45% vs. 18%) but even here the relative cover in the riverine segment was less than half that in the oceanic segment. The higher relative cover occupied in the oceanic segments of all four estuaries indicates that the oceanic segments provide substantially better habitat for *Z. marina* per unit of intertidal area.

Compared to *Z. marina*, the non-native *Z. japonica* displayed both a wider within-estuary range and more variability among the estuaries. The majority of the area occupied by *Z. japonica* occurred in the riverine segments of the Tillamook and Umpqua (Figure 7-24). In contrast, the majority of the *Z. japonica* population occurred in the oceanic segments in the Coos, Salmon, and Yaquina estuaries. Averaged across all seven estuaries, 53% of the area occupied by *Z. japonica* occurred in the oceanic segments versus 47% in the riverine segments. The relative cover of *Z. japonica* was about 2- to 10-fold higher within the riverine segment than the oceanic segment in the Coos, Tillamook, and Yaquina (Table 7-7). In the Umpqua and Salmon estuaries comparable relative covers occurred in both the oceanic and riverine segments. Thus, in contrast to *Z. marina*, the upper estuary constitutes a high quality environment for this non-native species, which presumably reflects *Z. japonica*'s ability to tolerate lower salinities. The lowest sediment salinity at which *Z. japonica* was found was 0.4 psu while several sites had salinities between 5 and 10 psu. In comparison, the lowest sediment salinity for *Z. marina* was 14 psu. The greater among-estuary variation in the relative distributions of *Z. japonica* compared to *Z. marina* may be a result of the non-native species not yet reaching an "equilibrium" population within some or all of the estuaries.

Because a substantial portion of the *Z. japonica* population is in the riverine segments, this species has more exposure to terrestrially derived nutrients and hence is more vulnerable to anthropogenic nutrient sources than *Z. marina*. One possible scenario is that a moderate level of nutrient enrichment would stimulate *Z. japonica*'s growth in the riverine segment while having little effect on the lower estuary *Z. marina* population. Such a stimulation could potentially promote the establishment and spread of this invader either through enhanced vegetative growth or seed production.

Table 7-7. Relative cover of the five ecologically important benthic taxa and bare habitat in the oceanic and riverine segments in the seven target estuaries. Relative cover is the percent area of the oceanic segment or riverine segment occupied by each taxon, and ranges from 0 to 100% for each segment.

	RELATIVE COVER FOR EACH HABITAT											
	Z. marina		*Z. japonica*		Macroalgae		*Neotrypaea*		*Upogebia*		Bare Habitat	
ESTUARY	Oceanic	Riverine	Oceanic	Riverine	Oceanic	Riverine	Oceanic	Riverine	Oceanic	Riverine	Oceanic	Riverine
Alsea	0	0	0	0	52	80	36	20	54	1.3	6	8
Coos	15	0	17	30	65	15	23	11	31	0	13	50
Nestucca	0	0	24	0	40	0	64	50	18	0	2	0
Salmon	0	0	4	2	49	94	78	58	4	2	14	2
Tillamook	45	18	2	22	61	67	34	34	42	16	15	19
Umpqua	13	0	19	22	29	17	61	37	0	0	20	34
Yaquina	21	1	10	18	71	1	46	61	40	3	2	30

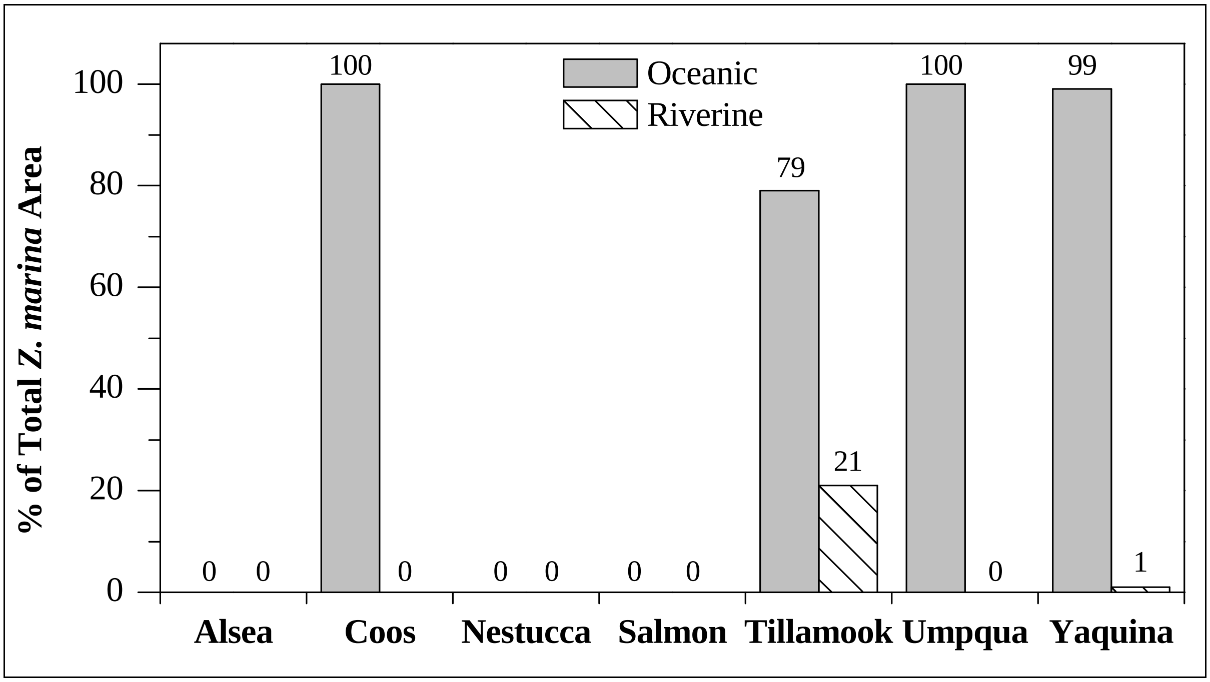

Figure 7-23. Relative distributions of the total area occupied by *Z. marina* between the oceanic and riverine segments. No *Z. marina* was found in the Alsea, Nestucca, or Salmon in the probabilistic surveys.

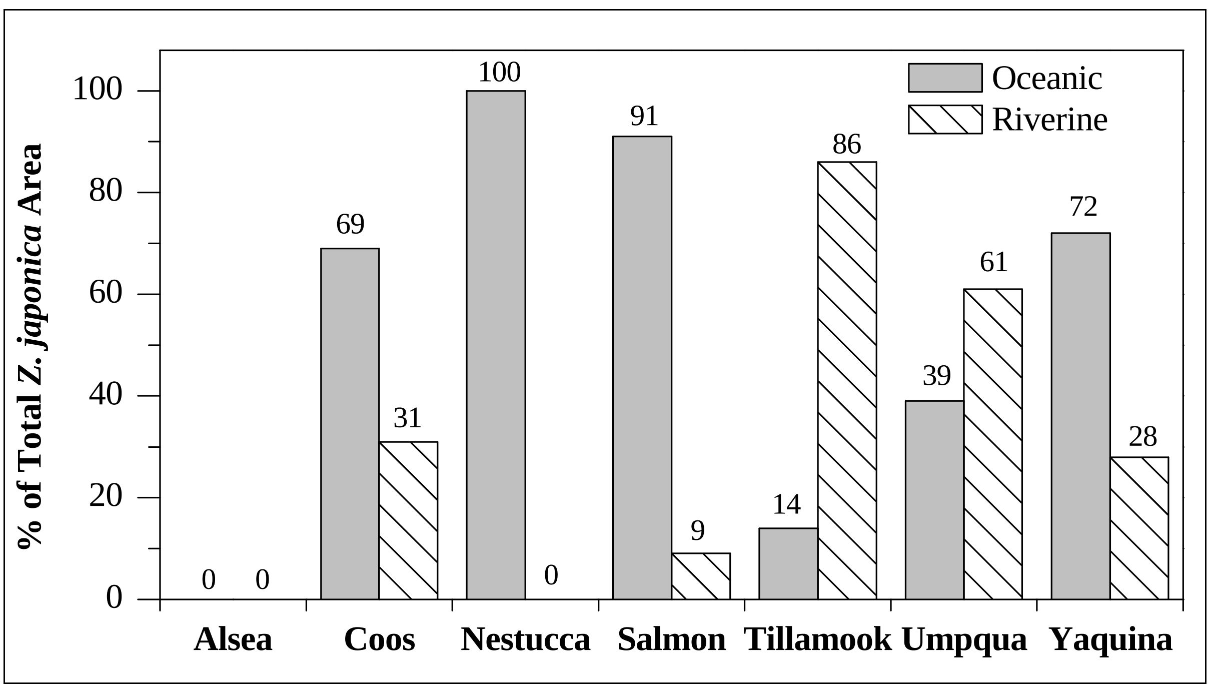

Figure 7-24. Relative distributions of the total area occupied by *Z. japonica* between the oceanic and riverine segments. No *Z. japonica* was found in the Alsea. Only two samples were taken in the Nestucca riverine segment.

At high levels of nutrient enrichment, excess nutrients could result in eutrophic conditions in the riverine segment that could potentially reduce *Z. japonica* in the upper estuary. Such reductions would be considered detrimental to the extent that *Z. japonica* provides ecosystem services. Thus, development of management strategies to reduce nutrient impacts on seagrasses will depend, in part, upon the assessment of the ecosystem services provided by *Z. japonica* versus any potential detrimental effects such as the potential for competition with *Z. marina* (e.g., Bando, 2006).

7.3.2 Within-Estuary Distributions of Benthic Macroalgae

The majority of the intertidal area occupied by benthic macroalgae occurred in the oceanic segment in all seven target estuaries (Figure 7-25). At least 88% of the intertidal area occupied by macroalgae was within the oceanic segments of the Alsea, Coos, Nestucca and Yaquina. A smaller proportion of the macroalgae occurred in the oceanic segments of the Tillamook, Umpqua, and Salmon but still $\geq$55% of the total macroalgal cover occurred in the lower estuary in these systems. Thus, the bulk of the macroalgal populations are exposed to ocean-derived nutrients during the growing season.

In terms of its relative cover, the extent of occurrence of macroalgae in the oceanic segments ranged from 29% in the Umpqua to 71% in the Yaquina (Table 7-7). In comparison to this relatively small range, the percentage of the riverine segment occupied by macroalgae varied widely across estuaries. Macroalgae occurred over approximately 94% of the riverine segment of the Salmon River versus only about 1% of the riverine segment in the Yaquina even though the Yaquina had the highest coverage within the oceanic segment. The reasons for these differences among estuaries are not clear though it is possible that differences in sampling dates may have contributed to some extent. However, eutrophication does not appear to be the cause based on the landscape analysis of these estuaries (Chapter 3). It is possible that differences in sampling dates may have contributed to some extent these differences in within-estuary spatial patterns. For whatever reason, under certain conditions the river-dominated segments provide suitable habitat for benthic macroalgae, and these upper estuary populations are more likely to be exposed to any increases in terrestrially derived nutrients.

7.3.3 Within-Estuary Distributions of Burrowing Shrimp

The populations of both burrowing shrimp species primarily occurred in the oceanic segments (Figures 7-26 and 7-27) though the two species differed in their relative concentration within the lower estuary. Seventy-nine to 100% of the estuarine area occupied by *U. pugettensis* occurred in the oceanic segments in all seven estuaries (Figure 7-27). The high percentage in the oceanic segment reflects that the riverine segments constitute a poor habitat for this species with $\leq$3% of the upper estuary being occupied in six of the seven estuaries (Table 7-7). Only the Tillamook had more than 10% of the intertidal area occupied by *U. pugettensis* in the riverine segment.

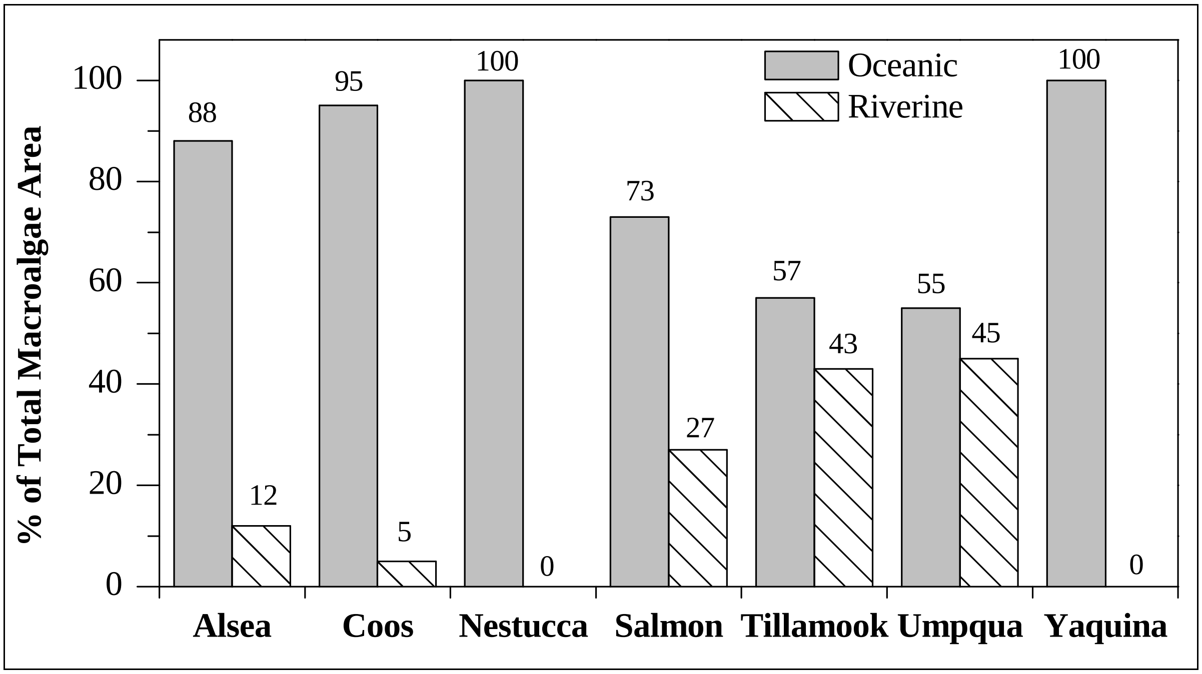

Figure 7-25. Relative distributions of the total area occupied by benthic macroalgae between the oceanic and riverine segments. Only two samples were taken in the Nestucca riverine segment.

The majority of the intertidal area occupied by *N. californiensis* also occurred in the oceanic segments (Figure 7-26), though it exhibited a wider range in its distribution among estuaries. About 78% to 98% of the total area occupied by *N. californiensis* was located in the oceanic segment in the Alsea, Coos, Nestucca, Salmon, and Yaquina estuaries. In comparison, about 55-60% of the occupied area occurred in the oceanic segments of the Tillamook and Umpqua. Another difference from *U. pugettensis* was that the riverine segments constituted relatively good habitat for *N. californiensis. N. californiensis* occupied more than 10% of the riverine segment in all seven estuaries and occupied at least 50% of the upper estuary in the Nestucca, Salmon, and Yaquina (Table 7-7). As a consequence of this greater riverine population, *N. californiensis* might be expected to have a greater vulnerability to terrestrially derived nutrient enrichments than *U. pugettensis.*

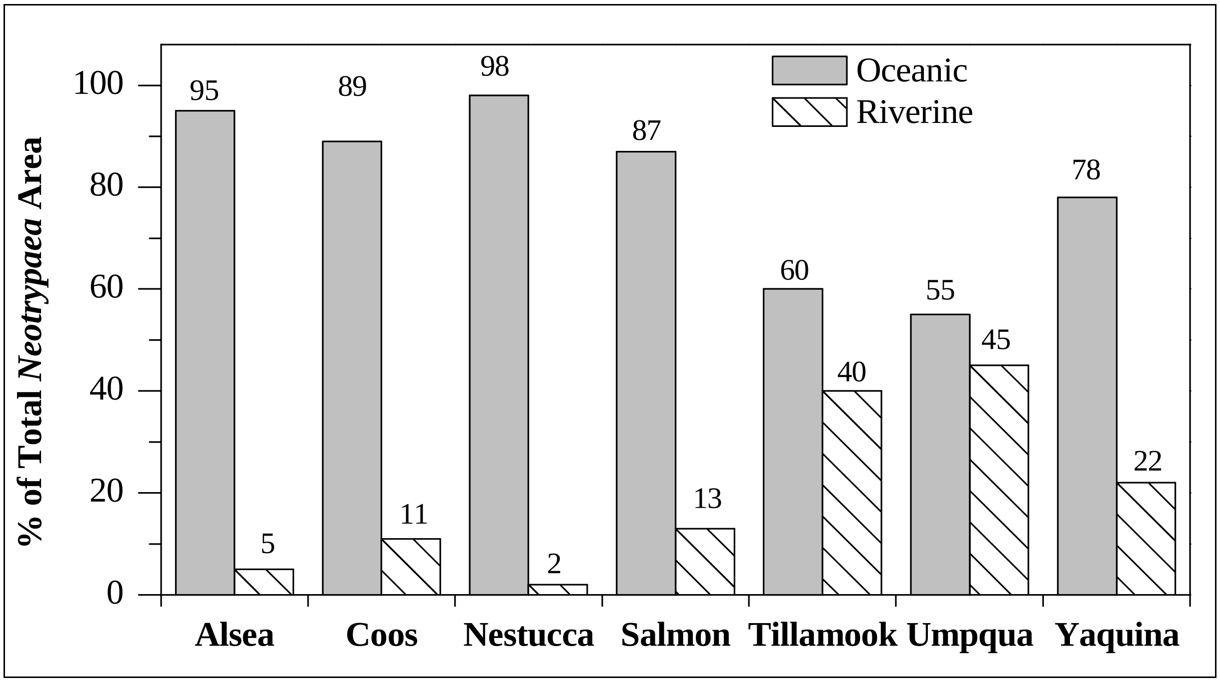

Figure 7-26. Relative distributions of the total area occupied by *N. californiensis* between the oceanic and riverine segments. Only two samples were taken in the Nestucca riverine segment.

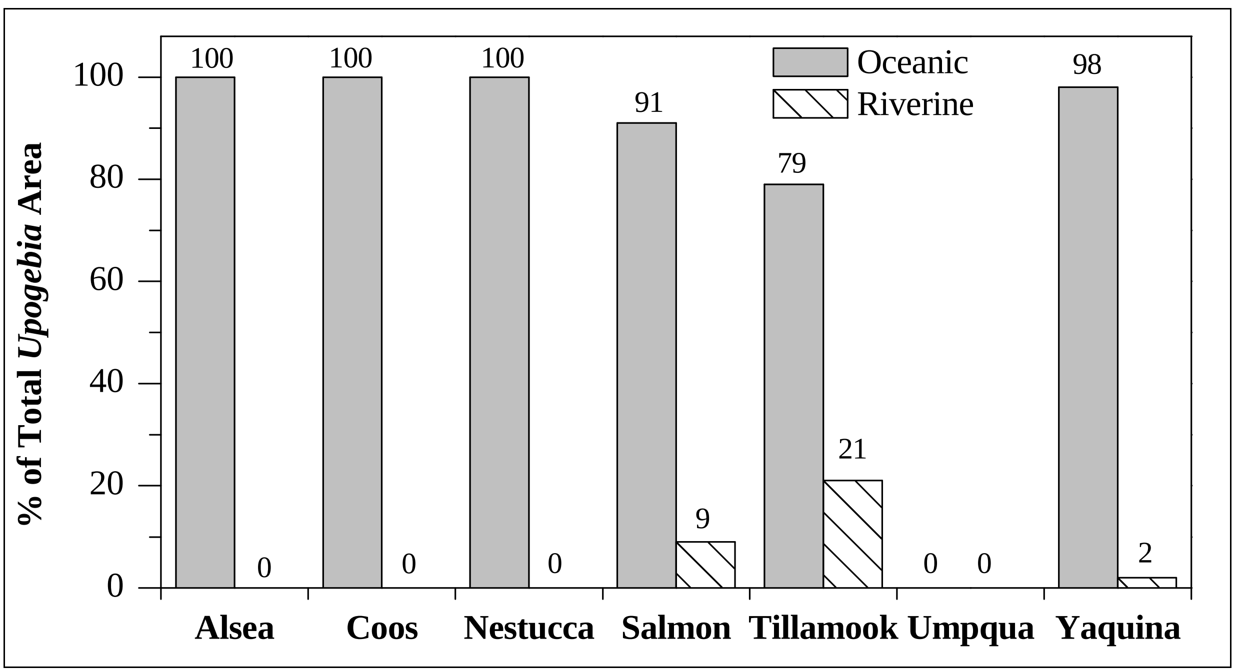

Figure 7-27. Relative distributions of the total area occupied by *U. pugettensis* between the oceanic and riverine segments. Only two samples were taken in the Nestucca riverine segment.

7.4 Benthic Macroalgae as Diagnostic Indicator of Cultural Eutrophication

High levels of macroalgae have been used in several areas of the world as a diagnostic of eutrophic conditions (e.g., Valiela et al., 1992; Bricker et al., 1999, 2003). However, applying this diagnostic to PNW coastal estuaries as an early indicator of cultural eutrophication is problematic as the seasonal macroalgal blooms appear to be a natural regional phenomenon. Summer blooms of macroalgae were observed in all seven target estuaries with macroalgae occurring over 40% of the estuarine intertidal area in six of the estuaries (Table 7-5; Figure 7-15). Macroalgal blooms have also been reported from other PNW coastal estuaries and Puget Sound (e.g., Phillips, 1984; Thom, 1984; Kentula and McIntire, 1986; Nelson and Lee, 2001; Thom et al., 2003; Bulthuis and Shull, 2006). We suggest that such a widespread occurrence of blooms is more indicative of a regional response than of localized anthropogenic enrichment.

Furthermore, it seems unlikely that all seven target estuaries are experiencing cultural eutrophication given the low level of urbanization, cultivation, and percent impervious surfaces as well as low population densities in the coastal watersheds (Section 2.8; Tables 3-3 and 3-4). The possibility that red alder may increase after logging somewhat complicates this conclusion as this nitrogen fixer contributes to the nitrogen loadings in the riverine segments (Section 3.7). However, as discussed in this chapter, and supported by the $\delta^{15}N$ ratios (Chapter 5), the macroalgae in the target estuaries were primarily exposed to oceanic- versus riverine-derived nitrogen. Thus, under present conditions, terrestrially derived nutrients could only stimulate a minority of the macroalgae in these PNW estuaries.

Based on these complementary lines of evidence, we conclude that the present level of benthic macroalgae is not an indicator of cultural eutrophication in PNW estuaries. This conclusion is specifically for coastal PNW estuaries, and certain subestuaries within Puget Sound may be experiencing anthropogenically driven increases in macroalgae (e.g., Shaffer, 2001) as are some estuaries within Southern California (e.g., Cohen and Fong, 2006). While the present level of macroalgae is not an indicator of eutrophication, increases over the existing baselines could serve as a diagnostic indicator of increasing eutrophic conditions. Increases in macroalgae in the upper estuary, where riverine-derived nutrients are the dominant source, would be the better diagnostic than changes in the lower estuary, which are more likely to reflect among-year differences in oceanic inputs. It would be important to couple measurements of abundance and extent of macroalgae with measurements of $\delta^{15}N$ stable isotope ratios to differentiate oceanic versus terrestrial nutrient sources. Measurement of macroalgal tissue nitrogen concentrations could complement the abundance and $\delta^{15}N$ measurements, especially as an integrative measure of nutrient pulses (Fong et al., 1998; Cohen and Fong, 2001). The practical limitations, however, of utilizing changes in macroalgal abundance as an indicator include its highly seasonal pattern, among-year variations, and paucity of quantitative baselines. These limitations could be mitigated as we generate additional baselines and develop a better understanding of the relative importance of the factors driving the seasonal pattern and the differences among estuaries. Until that time, however, we caution against simply using macroalgae as an eutrophication indicator in the PNW.

CHAPTER 8: LOWER DEPTH LIMIT OF *ZOSTERA MARINA* IN SEVEN TARGET ESTUARIES

Bruce L. Boese, Walter G. Nelson, Cheryl A. Brown, Robert J. Ozretich, Henry Lee II, Patrick J. Clinton, Christina L. Folger, T Chris Mochon-Collura, and Theodore H. DeWitt

Key Findings

- **Lower depth limit for colonization of *Z. marina* and light attenuation coefficients (K_d) were determined at multiple locations within the seven target estuaries.**
- **Leaf epiphyte biomass was determined seasonally at multiple locations in Yaquina Estuary.**
- ***Z. marina* generally tended to colonize deeper toward the mouth of each estuary.**
- **Lower depth limit for *Z. marina* generally followed trends in light attenuation (K_d), which tended to be greater in upriver estuarine reaches.**
- **Minimum light levels needed to maintain *Z. marina* in these estuaries was estimated to be ~13% of ambient surface irradiance.**
- **Epiphyte loads were greatest in summer and fall in the oceanic segment of the Yaquina Estuary resulting in light reductions at the leaf surface by as much as 60%.**

8.0 Introduction

Seagrass meadows are essential to the health and function of estuaries, providing critical habitat to economically and ecologically important fish, invertebrates, and birds (den Hartog, 1977; Thayer et al., 1975; Thayer and Phillips, 1977; Dennison et al., 1993; Batiuk et al., 2000). Seagrass depth distribution is dependent upon light penetration, with coastal seagrasses extending to depths receiving about 11% of the irradiance at the water's surface (Duarte, 1991). If the maximum depth that seagrasses grow in an estuary is a result of water clarity alone, then the maximum colonization depth may be a useful integrative water quality assessment measure (Dennison et al., 1993). Some have suggested its use as a monitoring tool (Sewell et al., 2001; Virnstein et al., 2002). Additionally, understanding the minimum light requirements for seagrasses is necessary for protection of existing seagrass meadows as well as restoration (Batiuk et al., 2000; Dennison et al., 1993; Fonseca et al., 1998).

In the U.S., the vast majority of seagrass research has been conducted on the Atlantic and Gulf coasts with a goal to establish water quality criteria to assure the survival and restoration of seagrass meadows (Batiuk et al., 2000). In contrast, there have been few studies on the light requirements of U.S. Pacific Coast seagrasses, predominately *Zostera marina*, with the exception of the work of Zimmerman and his colleagues in San Francisco Bay (Zimmerman et al., 1991, 1995) and Thom et al. (1998) in Puget Sound.

Light criteria have been proposed as part of the guidelines for restoring and maintaining *Z. marina* habitat in Chesapeake Bay (Batiuk et al., 2000). However, applying these values to the PNW *Z. marina* populations is problematic due to differences in tidal amplitude which tend to narrow the depth range of seagrasses (Koch and Beer, 1996) and other factors including temperature and high estuarine flushing rates. Criteria for Chesapeake Bay *Z. marina* were derived for the growing season (spring through fall) (Batiuk et al., 2000), where carbohydrates are accumulated and used to maintain plants during the winter when plants cannot sustain a positive carbon balance (Zimmerman et al., 1989). In contrast, for *Z. marina* in PNW estuaries, winter irradiance appears to be sufficient for the maintenance of a positive carbon balance and as a result plants continue to grow through the winter, albeit at a slower rate (Boese et al., 2005).

The present study measured the lower depth limit of *Z. marina* at multiple locations within seven target estuaries. These depth measurements were then compared to determine if there were spatial differences within and across estuaries. Light profiles were measured within these estuaries to determine the water clarity, expressed as a light attenuation coefficient. These measurements were then used to determine if the lower depth limits were correlated with water clarity differences within and across estuaries. Finally these depth and water clarity values were compared to literature values obtained for *Z. marina* in other estuaries.

8.1 Methods

8.1.1 Selection of Sampling Points

During the summers of 2004 and 2005, the seven target estuaries (Table 8-1) were sampled to determine the maximum depth of *Z. marina*. In each estuary, 25-45 sampling points were determined *a priori* using digital habitat maps from the Oregon Estuarine Plan Book (Cortright et al., 1987). Sampling locations were determined by randomly selecting points on a line running on the channel side of each seagrass-containing polygon using the ArcView v3.3 extensions Random Point Generator v1.2 and Mila Utilities. Randomly selected sampling points were never closer than 10 m to each other. If no seagrass was present at one of these pre-selected sampling points, while on site, a replacement sampling point was selected from the nearest seagrass bed or patch that was closest to the original *a priori* sampling point. The Yaquina Estuary was sampled in both 2004 and 2005 with the additional points randomly selected to increase spatial coverage. The Alsea Estuary was sampled in 2004 and again in 2006 to increase spatial coverage. In practice, the number of sampling points varied with the estuary and conditions found on the sampling days. The actual number of points sampled is presented in Table 8-1 with their approximate locations within each of the seven estuaries shown on Figures 7-1 through 7-7.

Table 8-1. Estuaries sampled for the lower depth limit of *Z. marina* and the datum adjustment factor used to convert lower depth limit from the mean lower low water (MLLW) datum to mean sea level (MSL) datum. *Value is mean of Oregon estuaries as specific correction was not available for this estuary.

ESTUARY	ESTUARINE AREA (km^2)	# SAMPLE POINTS	YEAR SAMPLED	MLLW TO MSL DATUM ADJUSTMENT (m)
Alsea	12.49	39	2004, 2006	+0.90
Coos	54.90	34	2005	+1.33
Nestucca	5.00	30	2004	+1.25*
Salmon	3.11	30	2004	+1.25*
Tillamook	37.48	24	2005	+1.36
Umpqua	33.78	27	2005	+1.25*
Yaquina	19.96	64	2004, 2005	+1.39

8.1.2 Sampling

The lower margin of seagrass at a sampling point was determined using a color underwater video camera (Sea-Drop 650 series, SeaViewer Cameras, Tampa, FL). The camera was secured to a fin stabilizer that was on the end of a 5-m aluminum pole which was attached to a davit mount on the research vessel (Figure 8-1). For each sampling, the boat with attached camera was initially positioned at the predetermined point and the boat was then moved approximately 50 m offshore from the visible edge of a seagrass bed. The boat would then slowly proceed shoreward on the transect line until seagrass became visible on the monitor. In 2004, when seagrass was encountered a pole was used to anchor the boat while data were collected. In 2005 and 2006 reverse thrust would be applied to the boat while the depth and GPS data were collected as quickly as possible.

In areas where the lower margin of a seagrass bed was found in water <1 m in depth, this margin was most easily found by visual inspection. As the boat approached the lower edge of a seagrass bed, an individual would exit the boat and hold it at that position while data were collected. Occasionally sites were sampled in shallow water by wading from shore to the lower bed margin.

Data recorded included the site location, water depth, and the date and time of observations. Water depth was determined using a lead line and/or hand-held depth sounder (deeper water) or using a meter stick if the depth was <0.8 m. Depth sounder accuracy was checked using a lead line during ideal conditions (i.e., no current, wind or waves). Depth values were adjusted to actual tidal heights determined from the nearest NOAA tide station. This was done by entering the estuary specific time and location adjustment factors (see: http://hmsc.oregonstate.edu/weather/tides/tideadj.html) for each measurement into the tide prediction software WXTIDE32 v4.4 and subtracting the resulting predicted tide height relative to Mean Lower Low Water (MLLW) from the depth measurement. To be consistent with seagrass lower limit values in the literature, lower limits relative to the MLLW datum were converted to lower limits relative to mean sea level (MSL) using the datum adjustment factors for each estuary if available (Table 8-1). If no correction values were available for a given estuary, the average correction value for Oregon estuaries (+1.25 m) was used. For more details on the tidal corrections see Section B.10.1.

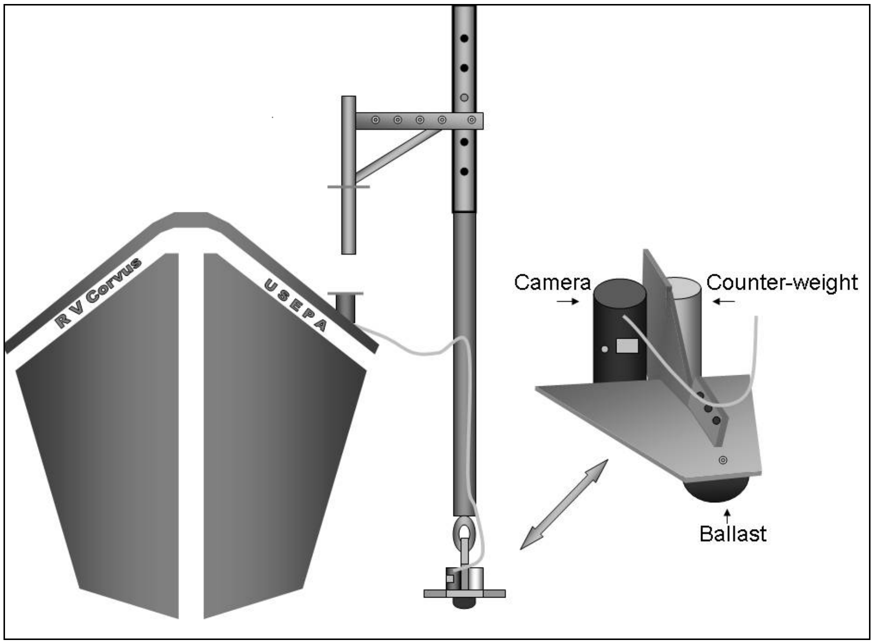

Figure 8-1. Schematic of underwater camera apparatus.

8.1.3 Light Attenuation Coefficients

Photosynthetically active radiation (PAR) is defined as irradiance in the 400- to 700-nm waveband, which encompasses the portion of the light spectrum that plant pigments use in photosynthesis. PAR profile measurements were made using LI-COR® spherical sensors. These measurements were made during single day cruises both at high and low tides at multiple locations in each estuary. At each of these locations PAR was determined at 0.25 m depth intervals from the surface to the bottom on both the down and up casts. Estimates of the overall water column light attenuation coefficient (K_d) values were determined from these profile data based on the slope of the linear regression of ln PAR vs. depth.

The K_d values were also determined in the Yaquina Estuary from long-term datasets. These values were determined in two ways. A series of nearly monthly cruises measured PAR vertical depth profiles at multiple sites from June 1998 though September 2004. These 0.25-m interval profiles were taken usually during flood tides. The K_d values for these profiles were determined by the slope of the regression of ln PAR vs. depth for the 0.50 m to 1.25 m intervals and also for the 1.00 m to 3.75 m intervals, representing near surface and near bottom light attenuations.

The second set of long-term Yaquina Estuary PAR measurements was taken at five fixed sites located at 3.9, 4.5, 11.9, 16.3, and 18.9 km from the estuary mouth[1]. Measurements at these sites were taken nearly continuously from 1999 though 2001 using two PAR sensors placed 0.75 m apart in depth. K_d values for this continuous measurement were determined from the difference between PAR measurements for the two sensors.

Once K_d was determined, the fraction of surface irradiance reaching a given depth was calculated as

$$\frac{I}{I_o} = e^{-K_d z} \tag{8.1}$$

where I is the irradiance at depth (m below MSL), I_o is the irradiance at the surface, K_d is the light attenuation coefficient (m^{-1}), and z is depth (m below MSL).

8.1.4 Epiphyte Biomass

A study of epiphytes growing on *Z. marina* leaves was conducted within the Yaquina Estuary from 2000 though 2004. Data were collected at six stations distributed between 3.5 and 17 km upriver from the estuary mouth. Leaves from collected plants were subdivided into outer (older) and inner (younger) leaves. Epiphytes were scraped from these leaf groups, and dry weights (24-36 hours at 60-70 °C) of the removed material determined for each individual plant. The effect of epiphyte cover on light (PAR) availability to *Z. marina* was estimated in the laboratory using a LI-COR® LI-190SA quantum sensor. Freshly removed epiphytes from a single leaf were washed into a Plexiglas cylinder with distilled water (60 mL). A light source was placed above this cylinder with the PAR sensor below the chamber and the amount of irradiance was determined. This value was then compared to a similarly measured irradiance value obtained using the same cylinder containing 60 mL of distilled water without epiphytes.

For analysis, dry weight data were partitioned by collection date into wet season (November-April) and dry season (May – October). Stations were also combined into two groups (oceanic and riverine segments) each with three stations based on the zonation presented in Chapter 5.

8.2 Results

8.2.1 Sampling Points/Sampling Errors

Inclement weather, tidal constraints, and channel navigation difficulties allowed for the completion of only 24 sampling stations in the Tillamook Estuary. Seven of these points were missing corresponding GPS data and therefore the sampling times for correcting depth were estimated from field notes. Two points in the Coos Estuary also had no corresponding GPS values. These latter locations were estimated from field notes using their relative positions to other sampling points where GPS values were obtained.

Electronic depth sounders were tested in the laboratory against a lead line to determine accuracy. Depth sounder readings were consistently within 0.1 and 0.2 meters of the lead line reading. The

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[1] The distance from the mouth of an estuary is here defined as the distance from the apparent ocean shore line, not from the end of the projecting jetties.

sonar scattering effect of benthic macroalgae and seagrass was a possible source of error in depth data. To help minimize this effect depth soundings were taken from within 50 cm of visible *Z. marina* shoots and not from directly over them. This precaution could not be taken when shoots were not clearly visible to the naked eye; roughly 40% of transect depth soundings were too deep (i.e. ≥ about 1.5 meters, depending on turbidity) to consider this precaution reliable. In many cases seagrass occurred on or just above benthic slopes estimated to exceed 45 degrees. When this occurred, given the difficulty in stopping the boat it was unlikely that depth was measured exactly above the edge of the seagrass bed. In these worse case conditions, we estimate that the accuracy of the depth measurement was ± 60 cm.

8.2.2 Depth Distribution

Figures 8-2 through 8-8 show the lower depth limits of *Z. marina* as a function of distance from the mouth within the seven target estuaries. The lower depth limit of *Z. marina* varied considerably within a given estuary, but tended to be greater toward the mouth of each estuary, with the exception of the Nestucca Estuary (Table 8-2).

The lower depth limit was also significantly different among estuaries (Table 8-3). The overall mean lower depth limits for *Z. marina* in the Coos and Nestucca estuaries were not statistically different from each other, but were different from the other five estuaries. The lower limit of *Z. marina* at any particular site might depend upon several factors other than depth such as sediment type, current velocity, salinity and bathymetry profiles, and the increased among-estuary variance resulting from these factors makes it more difficult to detect the effects of light availability. We attempted to reduce this variance added by other factors by reanalyzing only the deepest 1/3 of the depth limit data. Although this procedure changes mean depth, standard error, and the rank order of the estuaries, the re-analysis did not alter the results of the pairwise comparisons (Table 8-3).

Table 8-2. Linear regression coefficients of maximum *Z. marina* depth (meters below MSL) versus distance (km) from the mouth (apparent shore line) of each of the seven target estuaries. Significant constants and slopes are indicated by bold text and probability value. Note that Nestucca's significant regression is opposite of the general trend in that the lowest depths observed are farthest from the estuary mouth.

ESTUARY	CONSTANT	SLOPE	r^2
Alsea	**4.33 (p<0.001)**	**-0.297 (p<0.001)**	**0.50**
Coos	**1.36 (p<0.001)**	-0.0056 (p=0.77)	0.003
Nestucca	**1.11 (p<0.001)**	**0.134 (p<0.001)**	**0.58**
Salmon	**3.46 (p<0.001)**	**-0.889 (p<0.001)**	**0.57**
Tillamook	**3.21 (p<0.001)**	**-0.137 (p=0.049)**	**0.16**
Umpqua	**2.62 (p=0.001)**	-0.0251 (p=0.505)	0.0018
Yaquina	**3.70 (p<0.001)**	**-0.118 (p<0.001)**	**0.32**

Table 8-3. Overall mean depth (meters below MSL), mean of deepest 1/3 of depth values, and maximum depth that *Z. marina* was observed in each of the seven target estuaries. Mean values are mean ± standard error (n). Bold indicates that two of the seven estuaries were statistically different from the other five estuaries but not significantly different from each other (Home-Sidak method of pairwise comparisons, p ≤ 0.05).

ESTUARY	OVERALL MEAN DEPTH (n)*	DEEPEST 1/3 OF DEPTH VALUES (n)*	MAX. DEPTH (m)
Alsea	2.95 ± 0.19 (39)	4.26 ± 0.26 (13)	6.55
Coos	**1.28 ± 0.12 (34)**	**2.09 ± 0.15 (11)**	3.03
Nestucca	**1.49 ± 0.07 (30)**	**1.85 ± 0.02 (10)**	1.94
Salmon	2.25 ± 0.16 (30)	3.37 ± 0.06 (10)	3.70
Tillamook	2.46 ± 0.23 (24)	3.72 ± 0.39 (8)	5.17
Umpqua	2.45 ± 0.16 (27)	3.36 ± 0.28 (9)	4.71
Yaquina	2.44 ± 0.13 (64)	3.62 ± 0.11 (24)	4.53

*statistical difference in mean values (ANOVA, p<0.001).

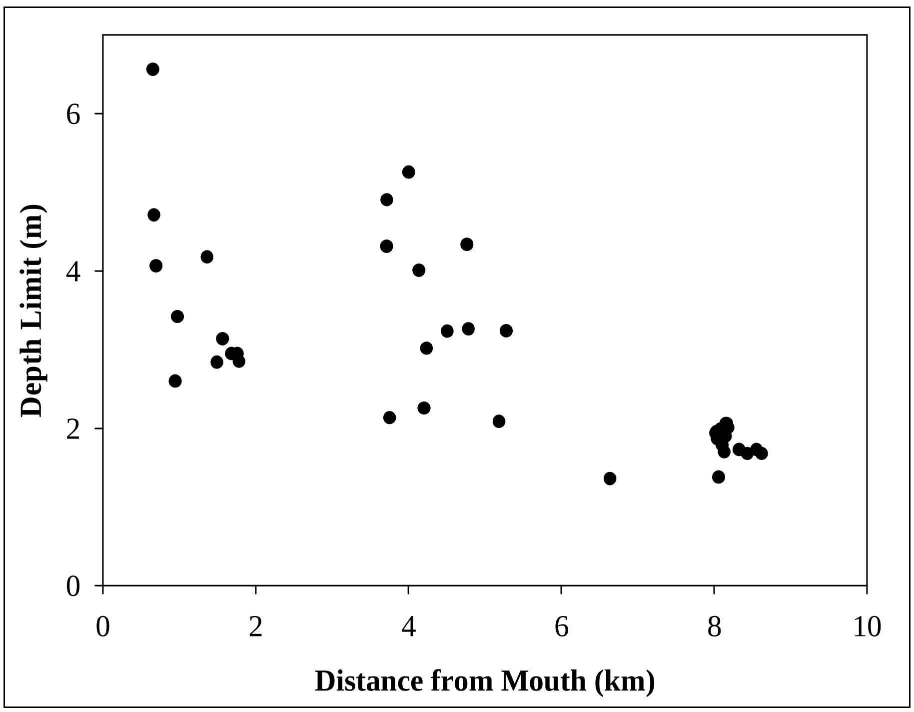

Figure 8-2. Lower limit of *Z. marina* (m below MSL) versus distance (km) from the mouth of the Alsea Estuary.

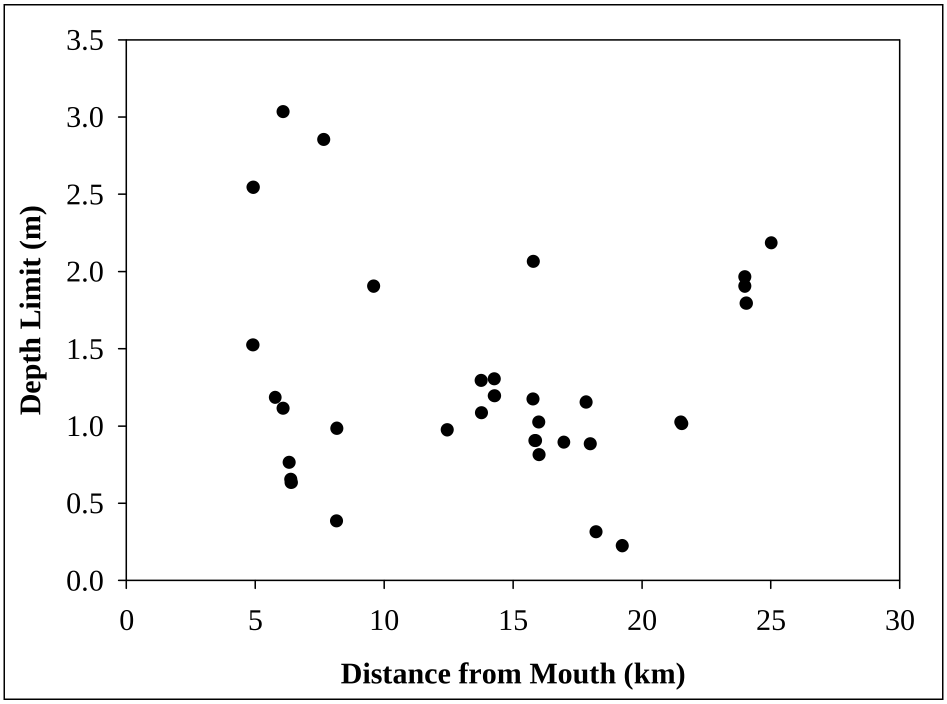

Figure 8-3. Lower limit of *Z. marina* (m below MSL) versus distance (km) from the mouth of the Coos Estuary.

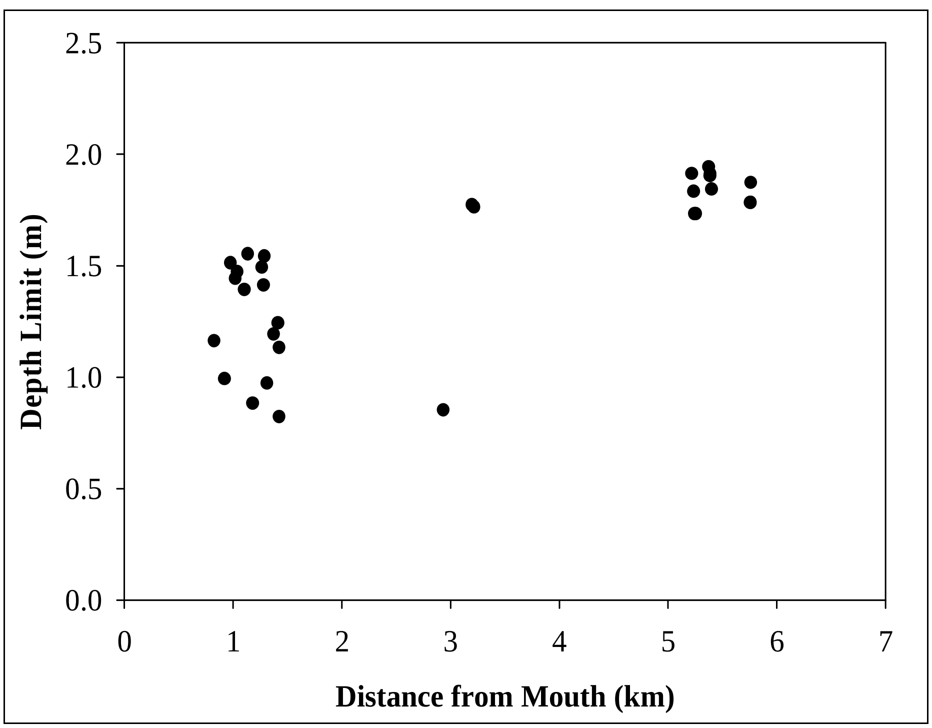

Figure 8-4. Lower limit of *Z. marina* (m below MSL) versus distance (km) from the mouth of the Nestucca Estuary.

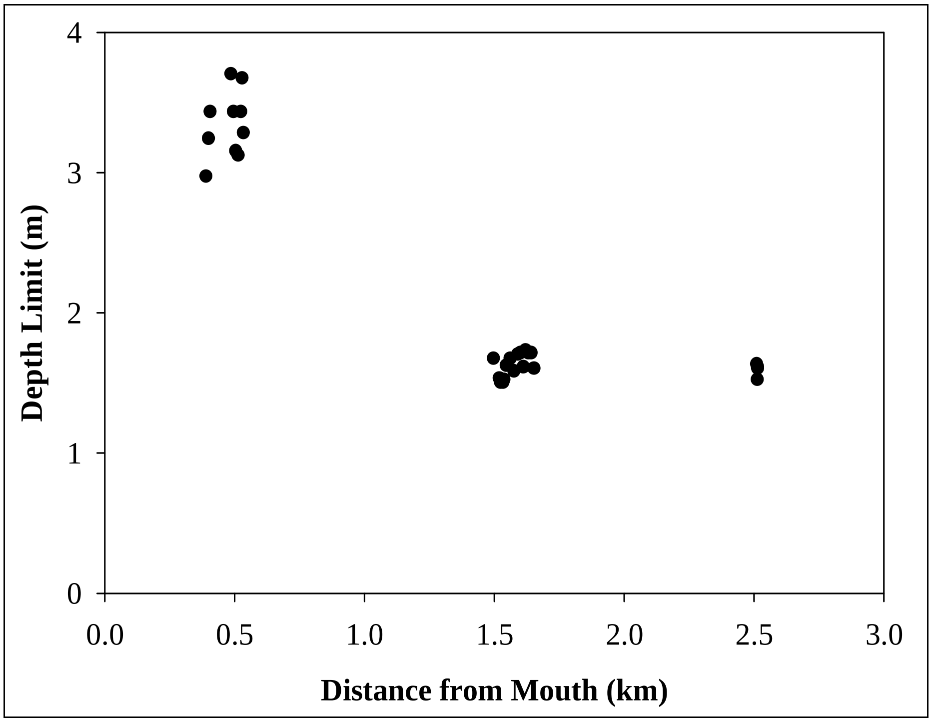

Figure 8-5. Lower limit of *Z. marina* (m below MSL) versus distance (km) from the mouth of the Salmon River Estuary.

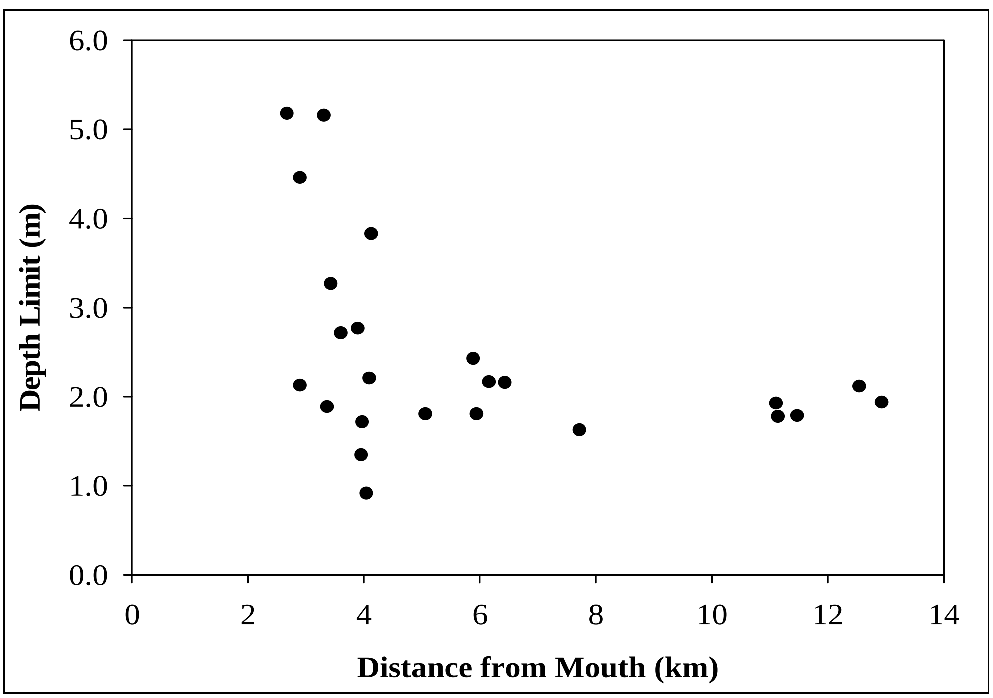

Figure 8-6. Lower limit of *Z. marina* (m below MSL) versus distance (km) from the mouth of the Tillamook Estuary.

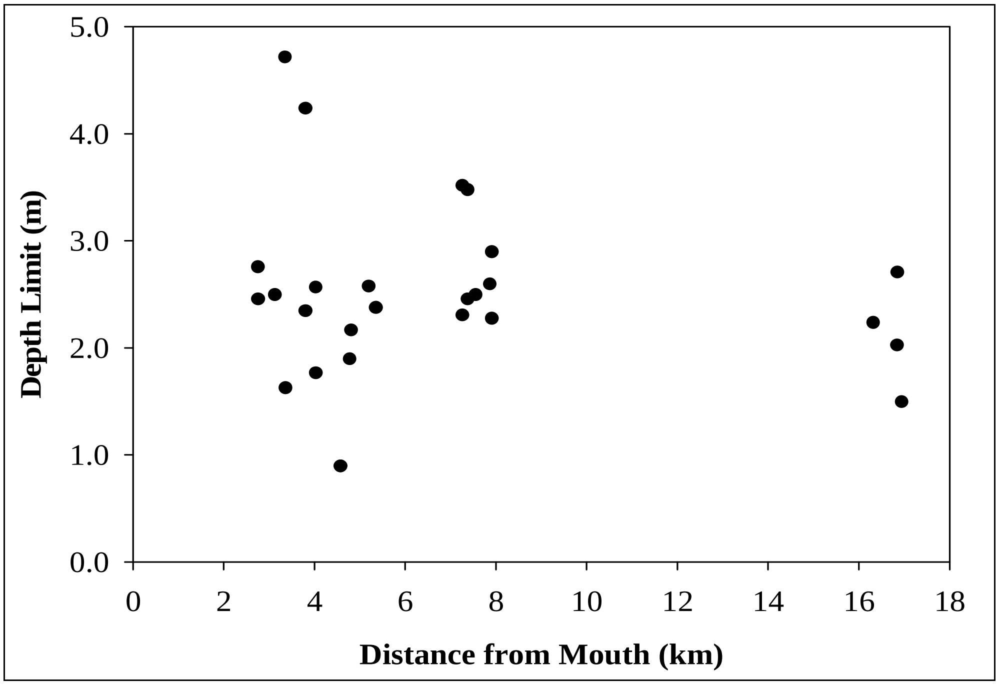

Figure 8-7. Lower limit of *Z. marina* (m below MSL) versus distance (km) from the mouth of the Umpqua River Estuary.

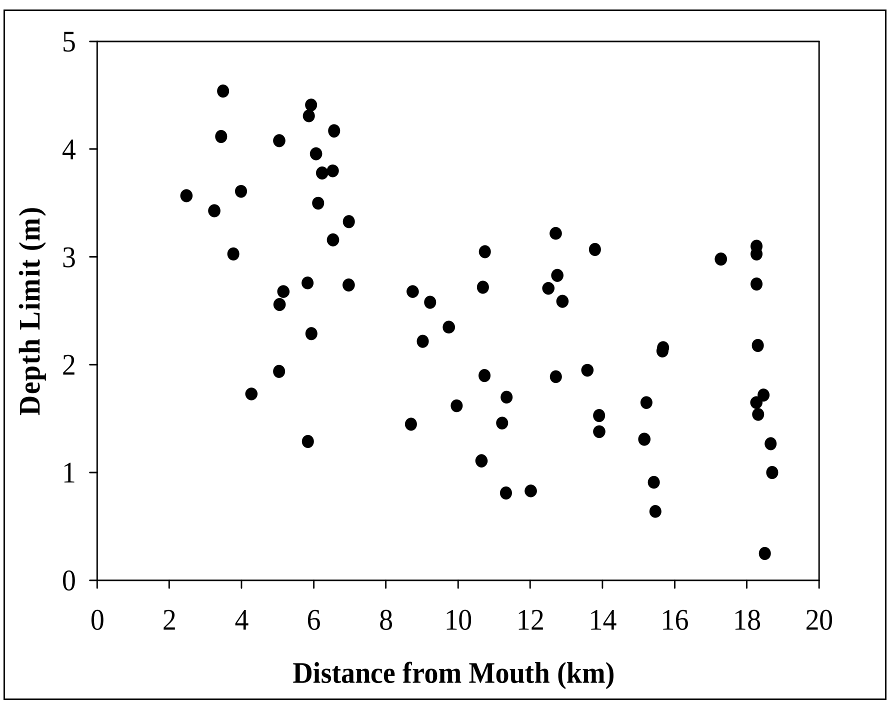

Figure 8-8. Lower limit of *Z. marina* (m below MSL) versus distance (km) from the mouth of the Yaquina Estuary.

8.2.3 K_d vs. Distance from Estuarine Mouth

The long-term irradiance datasets from the Yaquina Estuary showed strong linear relationships between K_d and distance from the mouth (Figures 8-9 and 8-10). These results indicate that water clarity generally decreases from the ocean-dominated mouth into river-dominated reaches of the estuary. These annualized results are similar to data obtained on high and low tide cruises conducted in the Yaquina Estuary on June 15 and June 21, 2004 (Figure 8-11).

Unfortunately, the same trend in K_d observed during the one day cruises within the Yaquina Estuary were generally not seen or were not statistically significant in the other estuaries examined in the same manner (Figures 8-12 through 8-16)). In the Alsea Estuary, there was a large tidal difference in K_d values (Figure 8-12). While the K_d values tended to increase in the upriver portions of the Nestucca and Salmon River estuaries (Figures 8-13 and 8-14), these trends were not significant and low tide K_d values were not determined in the Nestucca due the shallowness of this estuary during the low tide cruise. In the Tillamook Estuary, low K_d values on both the high and low tide cruises were observed at the station farthest up the estuary resulting in a non-significant regression (Figure 8-15). Tillamook Estuary K_d values were further complicated by highly variable weather conditions which resulted in an extremely variable ambient light field during many of the PAR sensor casts. Few K_d values were determined for the Coos Estuary due to instrument malfunction. Only in the Umpqua River Estuary was a significant relationship found between distance from the estuary mouth and K_d (Figure 8-16).

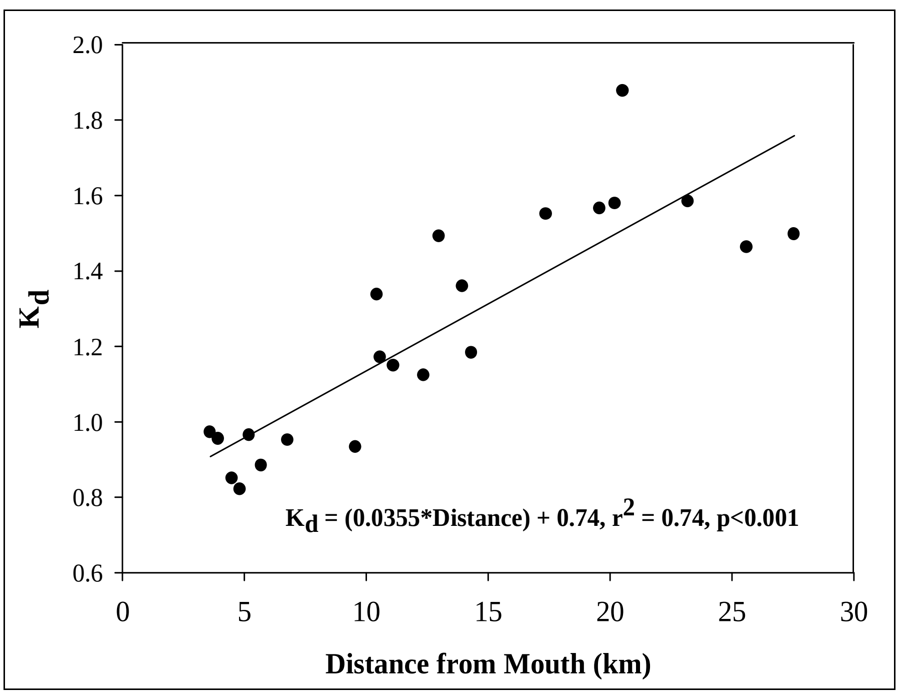

Figure 8-9. Light attenuation coefficient (K_d, m^{-1}) versus distance from the mouth (km) of the Yaquina Estuary generated using long-term cruise dataset. K_d values are means determined from PAR depth profile measurements taken approximately monthly from June 1998 though August 2004.

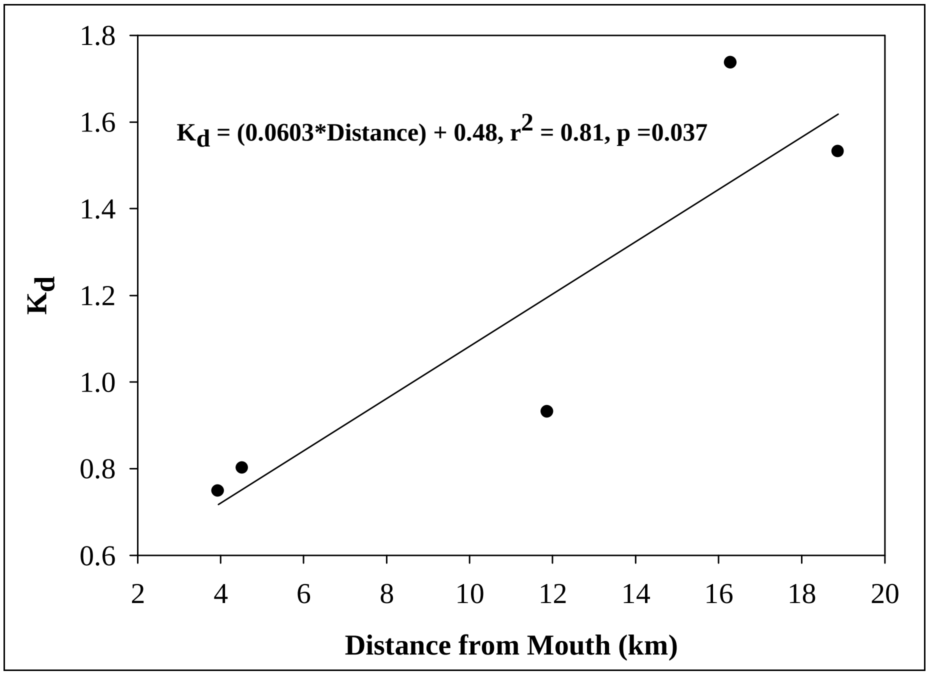

Figure 8-10. Light attenuation coefficient (K_d, m^{-1}) versus distance from the mouth (km) of the Yaquina Estuary generated using continuous dataset K_d values are means determined from PAR measurements that were taken at five sites which were monitored continuously from 1999 though 2001.

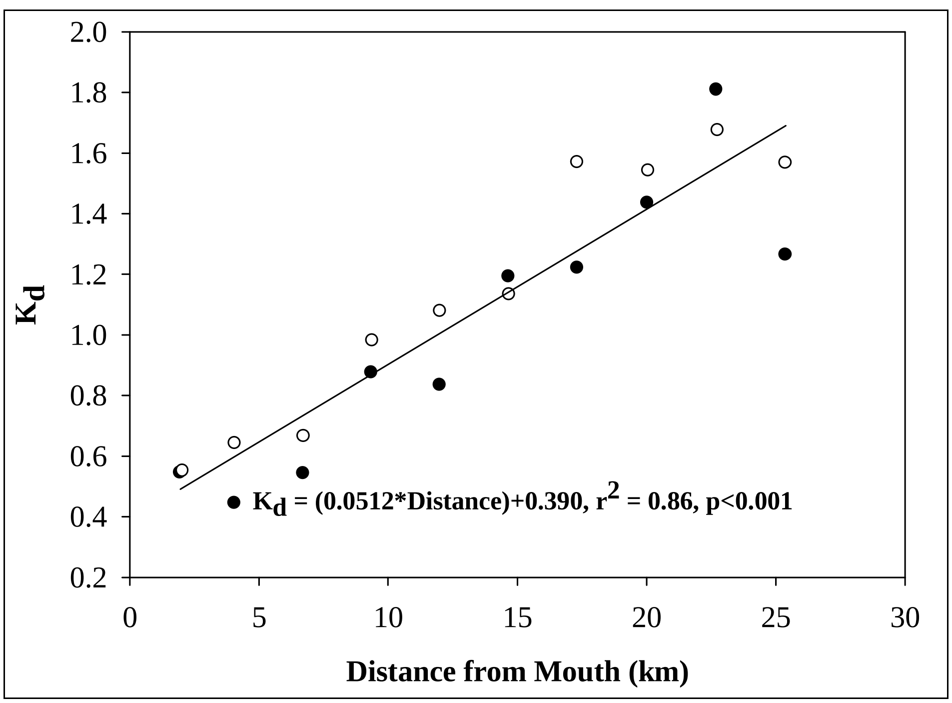

Figure 8-11. Light attenuation coefficient (K_d, m^{-1}) versus distance from the mouth (km) of the Yaquina Estuary for classification dataset. K_d values determined from light vs. depth profiles measured in the Yaquina Estuary during a single high (●) and low (o) tide in June 2004.

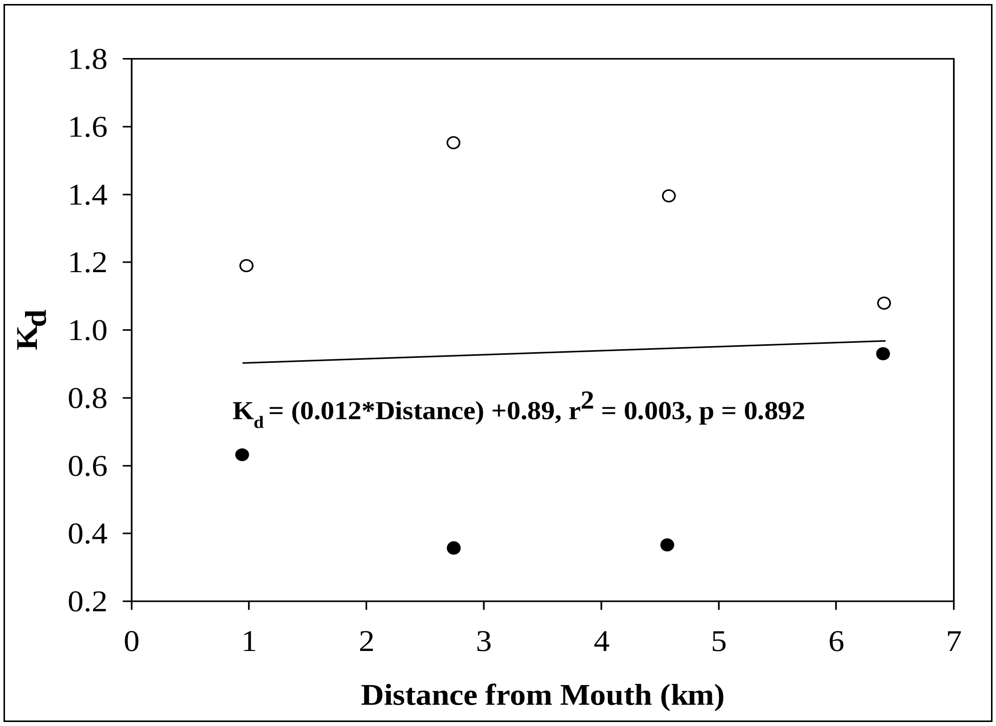

Figure 8-12. Light attenuation coefficient (K_d, m^{-1}) versus distance (km) from the mouth of the Alsea Estuary. Values were determined during high (●) and low (o) tides.

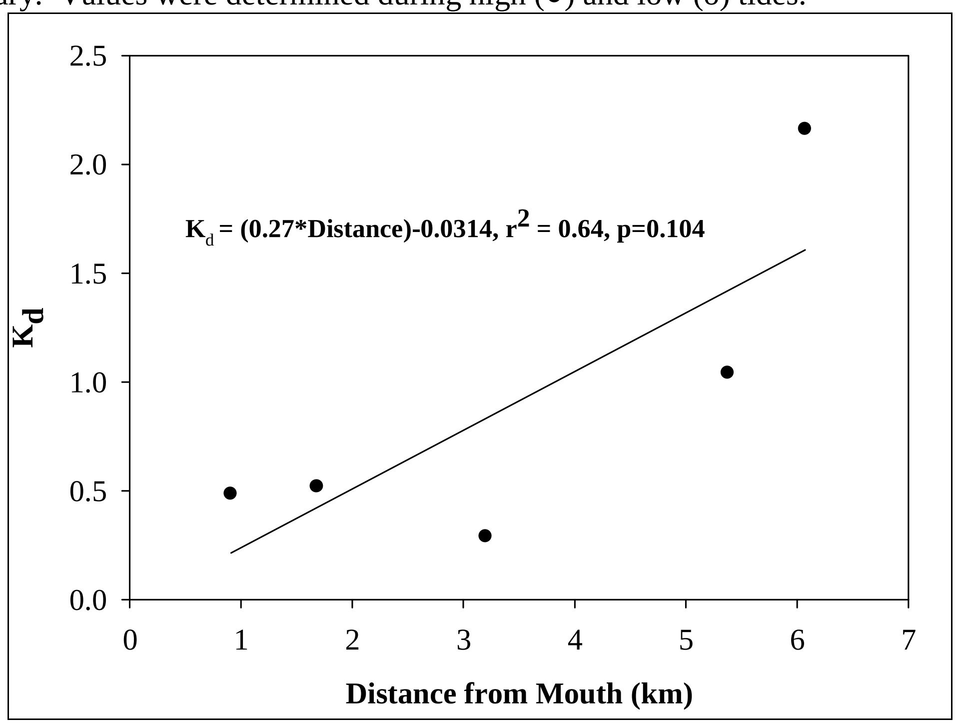

Figure 8-13. Light attenuation coefficient (K_d, m^{-1}) versus distance (km) from the mouth of the Nestucca Estuary. Values were determined during high tide.

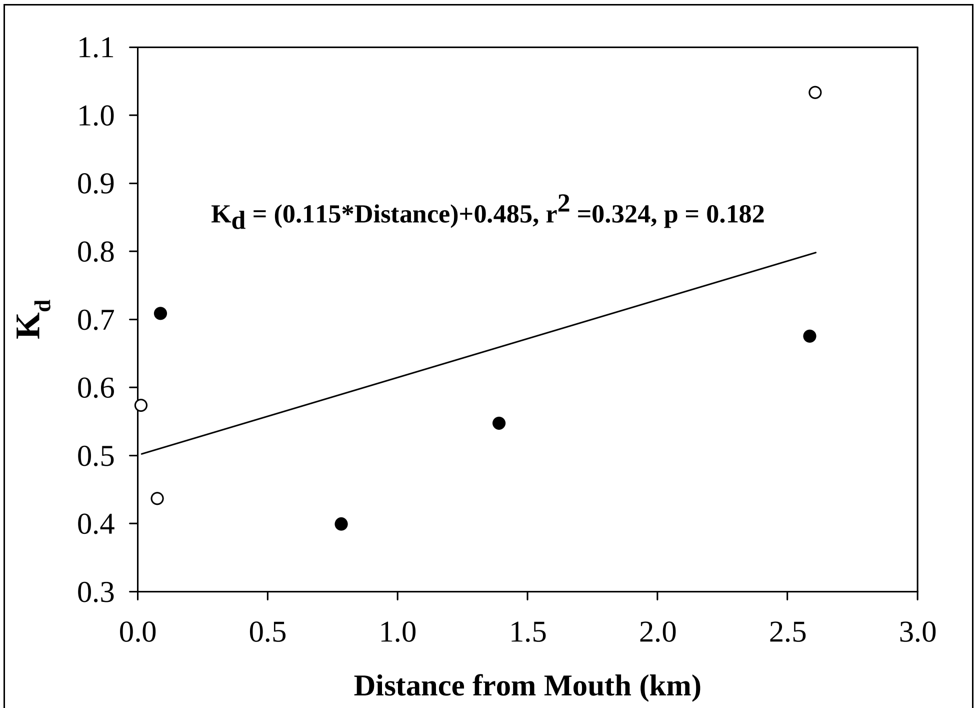

Figure 8-14. Light attenuation coefficient (K_d, m^{-1}) versus distance (km) from the mouth of the Salmon River Estuary. Values were determined during both high (●) and low (o) tides.

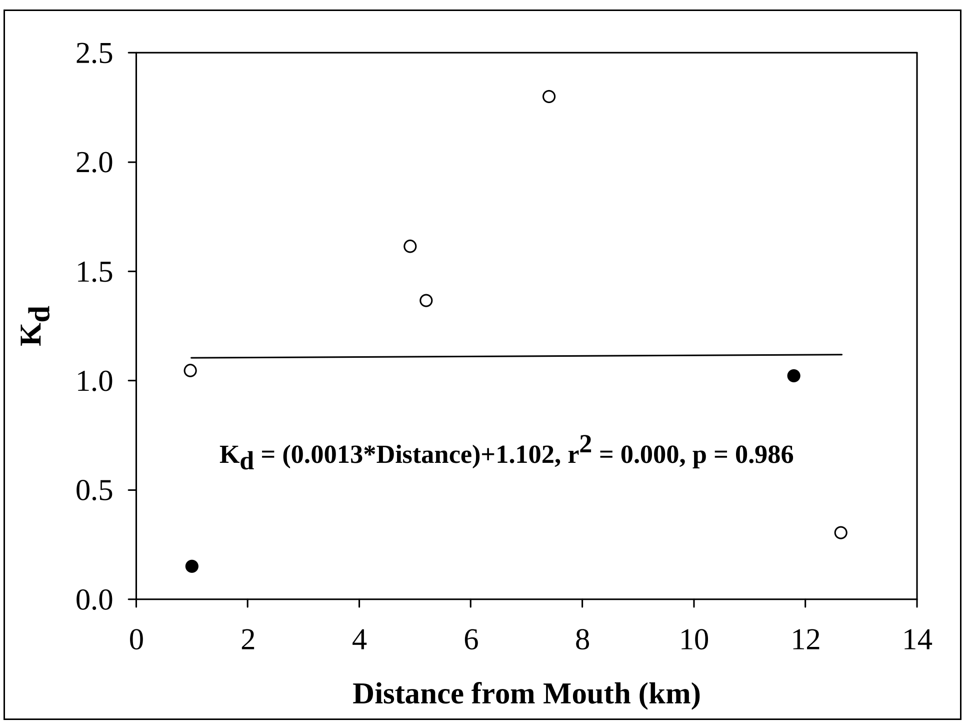

Figure 8-15. Light attenuation coefficient (K_d, m^{-1}) versus distance (km) from the mouth of the Tillamook Estuary. Values were determined during high (●) and low (o) tides.

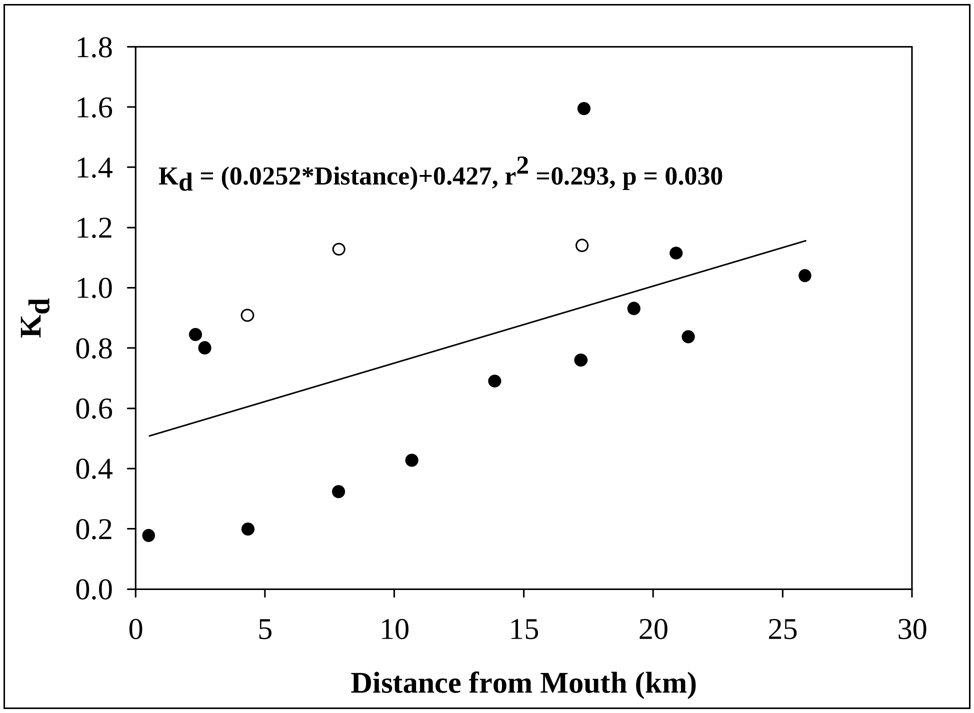

Figure 8-16. Light attenuation coefficient (K_d, m^{-1}) versus distance (km) from the mouth of the Umpqua River Estuary. Values were determined during high (●) and low (o) tides.

8.2.4 Maximum *Zostera marina* Depth, K_d and Light Relationship

As there was a significant relationship between K_d and distance from the mouth in the Yaquina and Umpqua River estuaries, it was possible to attempt to derive a relationship between the lower limit for *Z. marina* and the estimated K_d values (Figure 8-9 and 8-16). Although there was a significant relationship between K_d and lower depth limit of *Z. marina* for the Yaquina Estuary (Figure 8-17), the relationship was not significant within the Umpqua Estuary ($r^2 = 0.034$, $p=0.34$). Using Equation 8.1, the amount of surface irradiance reaching the observed lower depth limits for *Z. marina* within the Yaquina Estuary was estimated. These values ranged from 2 to 85% with a mean ± standard error of 13.4 ± 0.2 % (n = 64). Similar values for the other estuaries sampled in this study were not determined due to the lack of reliable K_d estimates.

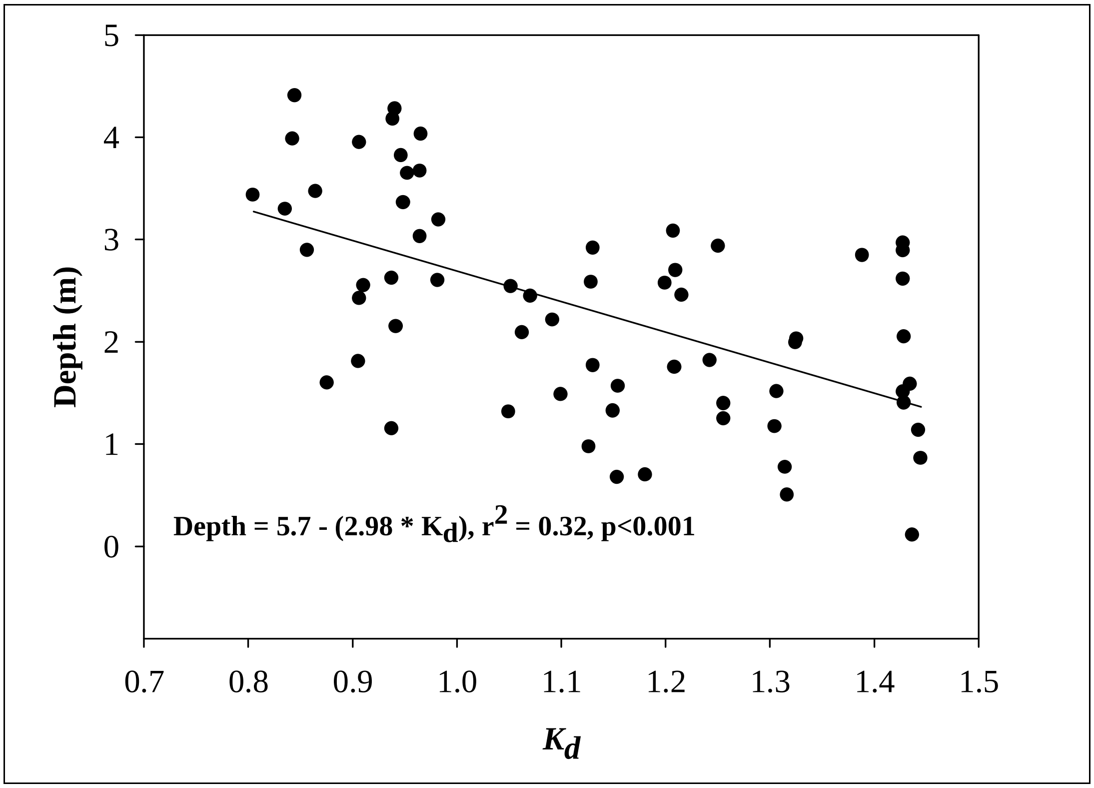

Figure 8-17. Relationship between *Z. marina* lower depth limit (m below MSL) and light attenuation coefficient (K_d, m^{-1}) in the Yaquina Estuary.

8.2.5 Epiphyte Patterns and Impact on Light

In the Yaquina Estuary, there was a general annual pattern in 2000 though 2003 in which epiphyte biomass increased in the spring to maximal amounts in the summer and fall. This statistically significant parabolic relationship was most clearly seen on the older, external seagrass blades (Figure 8-18). In the Yaquina Estuary, epiphyte biomass per unit surface area of seagrass leaves was higher in the oceanic segment than in the riverine segment in both wet and dry seasons (Figure 8-19), although only the dry season differences were statistically significant. Epiphyte biomass per unit leaf surface area was higher in the dry season than the wet season within both segments. There is a significant positive linear relationship between percent light reduction and log+1 transformed epiphyte biomass (Figure 8-20).

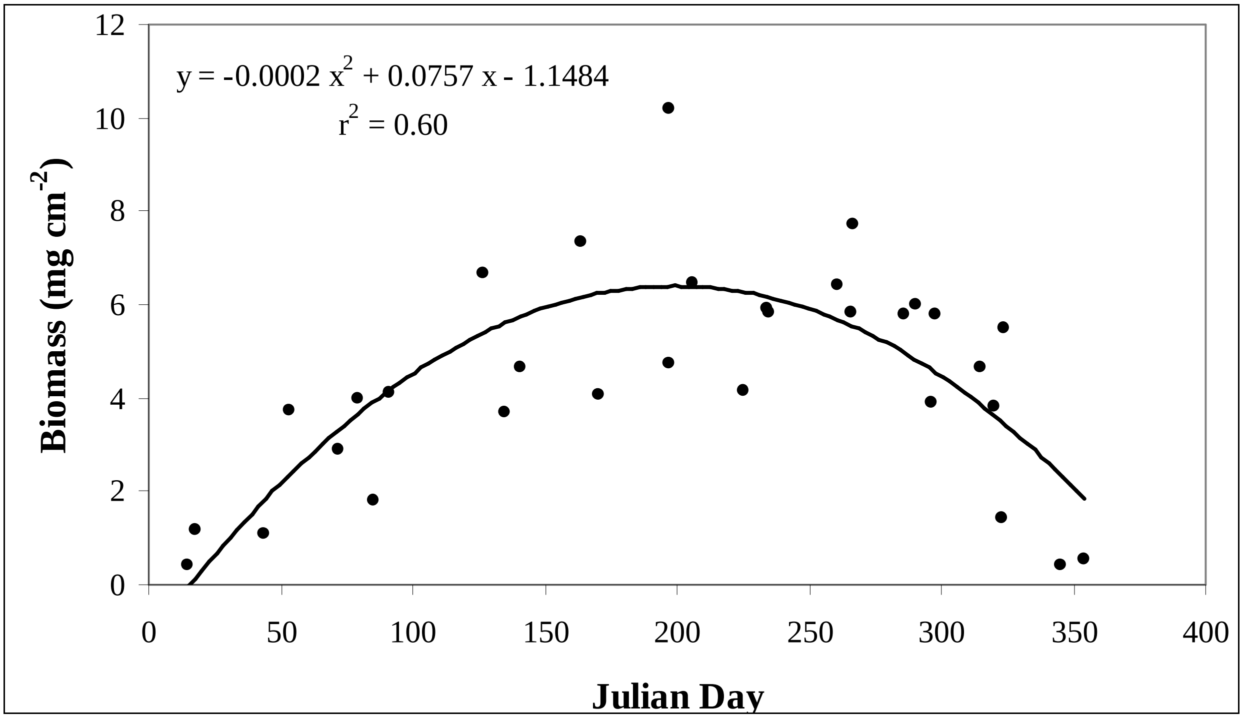

Figure 8-18. Temporal relationship of epiphytic biomass per unit leaf area on *Z. marina* external leaves in the Yaquina Estuary. Values are for 2000 though 2003.

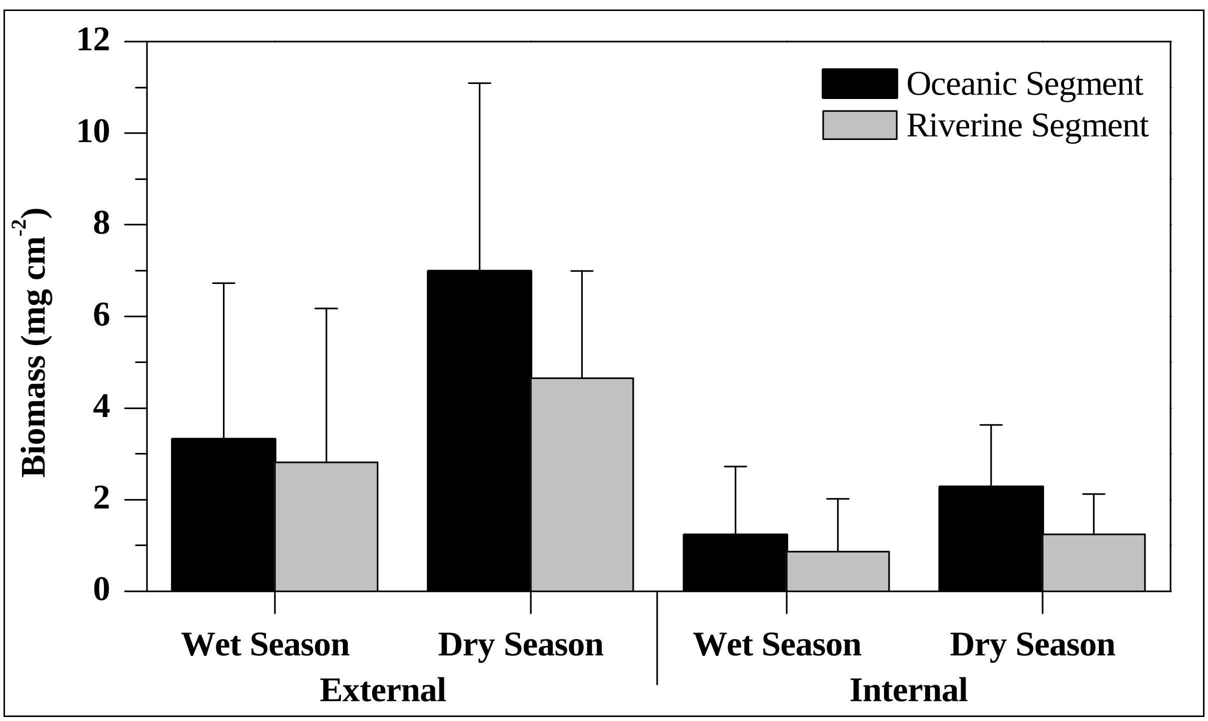

Figure 8-19. Epiphyte biomass per unit leaf area on older (external) and younger (internal) *Z. marina* leaves by season (wet or dry) in the oceanic and riverine segments of the Yaquina Estuary. Wet season is from November through April. Dry season is from May through October.

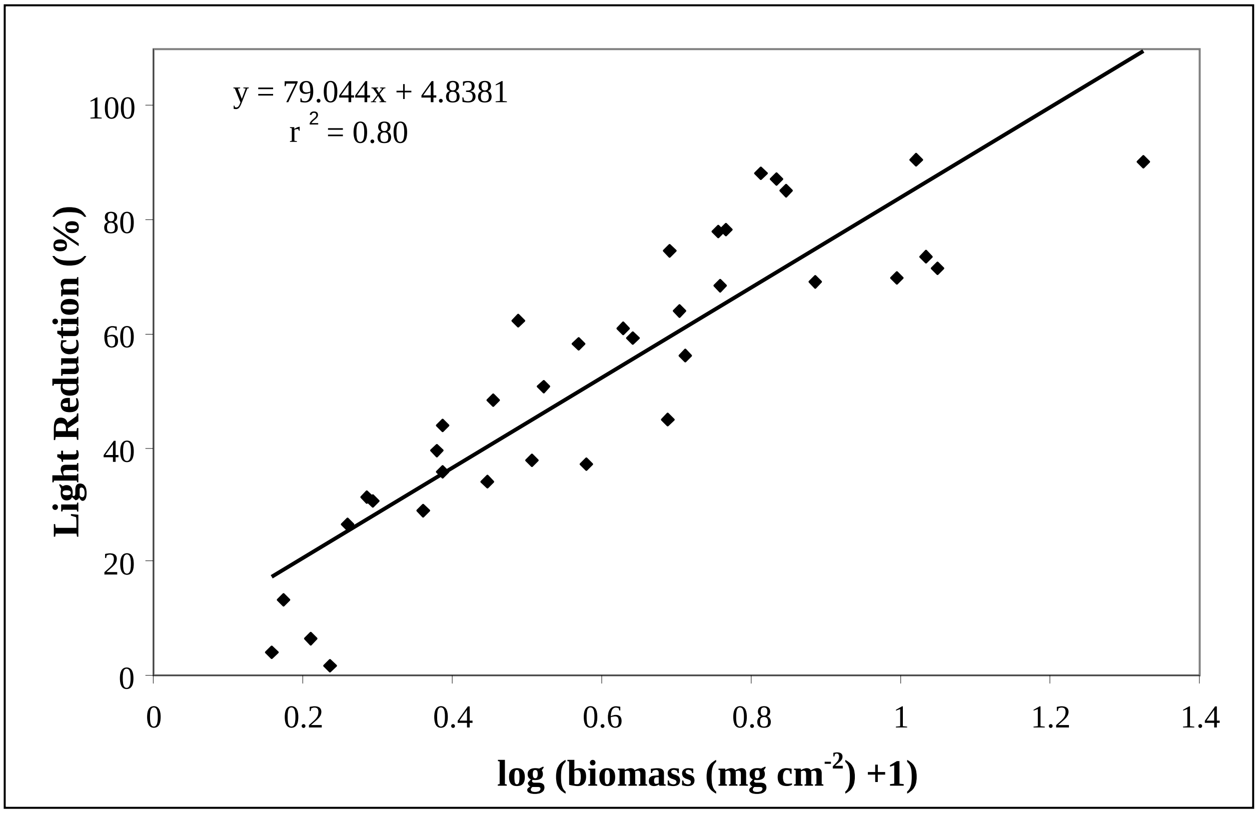

Figure 8-20. Linear regression relationship between the percent of light reduction to log(x+1) transformed epiphyte biomass per unit *Z. marina* leaf surface area.

8.3 Discussion

8.3.1 Sampling Method

The video camera sampling technique was difficult to conduct in typical field conditions. High winds and strong currents made maintaining position or tracking a straight line difficult. Stopping the vessel in time to mark the location of a *Z. marina* patch was frequently challenging, as the vessel would drift off location during the time required to set anchor or stabilize position before recording GPS position. Water clarity was also a factor which introduced variability in estimating the lower limit of *Z. marina* because visibility was often as low as 0.2 to 1 m.

Measuring the lower depth margin of *Z. marina* using manual sampling methods appeared to be more precise than the video collection technique. While this method could only be employed during low tides and in areas where the *Z. marina* lower margin was shallow (< 1.5 m), when the researcher exited the vessel quickly, the position and depth of the edge of the seagrass bed could be determined with a very high degree of accuracy and precision.

8.3.2 Depth Distributions

In general, the maximum colonization depth (Z_c) limit of *Z. marina* is ultimately determined by water clarity (Duarte, 1991). However, we observed a high variability (Figure 8-2) in these maximum depth values within the seven target estuaries which cannot be explained by water clarity relationships alone. Factors, such as current velocity, sediment characteristics, and other non-optical water properties, play a role in determining where and how deep seagrasses grow (Koch, 2001; Virnstein et al., 2002), possibly restricting the plants to shallower waters. Although some variability in the depth distribution data were reduced by only analyzing data for the deepest one-third of all the patches found within in each estuary (Table 8-3), this method introduces a degree of subjectivity. However, given this caveat, there was a general trend (i.e., in six of the seven estuaries) for the maximum depth of *Z. marina* colonization to be deepest nearest the mouth (Figures 8-2 through 8-8) and a statistically significant linear model could be fit to four of these estuaries (Table 8-2). The exception to this trend was within the Nestucca Estuary where the opposite trend appeared to occur. This latter result may be due to the overall shallowness of this estuary when compared to the others, which may have contributed to its general lack of extensive *Z. marina* beds. Relative to the other estuaries in the present study, *Z. marina* was not found near the mouth of the Nestucca Estuary where presumably mean water clarity may be greater. This lack of *Z. marina* was most likely due to high tidal current velocities and shifting sands, which we commonly observed at the mouth of this estuary during peak flood and ebb tides. In addition, the loss of multiple instruments near the mouth of the Nestucca Estuary supports this assessment that this is a high-energy environment. Upriver *Z. marina* was limited to fringing patches next to deeper water channels, where it is also possible that high seasonal current velocities may preclude deeper colonization depths.

Comparing these maximum colonization depths to other PNW estuaries is difficult. Although several studies have measured these values (Thom et al., 2001; Thom et al., 2003; Selleck et al., 2005), the data are presented in graphical form making it difficult to compare mean and maximum colonization depths directly. Data from Thom et al. (2003) indicate that the deepest *Z. marina* observed in the Coos and Willapa estuaries was approximately 1.8 m and 2.8 m below MSL, respectively. Although these depths are shallower than those observed in the present study, there is no indication that Thom et al. (2003) made a concerted effort to find the maximum colonization depth. Their study was instead focused on determining the depth of maximum shoot density and relating this to salinity and irradiance.

Nonetheless, based on the present study and those of Thom and his colleagues (Thom et al., 2001; Thom et al., 2003), maximum colonization depths for *Z. marina* within coastal PNW estuaries are shallower than those observed in Puget Sound (Thom et al., 1998; Dowtry et al., 2005; Selleck et al., 2005), where *Z. marina* is commonly found 5 m below MSL, with some areas having shoots present to three times that depth (Selleck et al., 2005).

8.3.3 Water Clarity Relationships

A priori we assumed that overall water clarity would decrease in the more riverine portions of estuaries and this trend in water clarity would be apparent in light attenuation measurements taken over an interval of only a few days. This assumption appeared to be true for the Yaquina Estuary, as long-term water clarity measures averaged over a number of years were similar to those measured during two days in 2004. However, this trend was not consistently observed in the other estuaries examined. It is likely that our short-term measurements of irradiance at depth are not adequate to define the spatial variability within PNW estuaries. For example, in the Alsea Estuary, K_d values at a given location were almost 4-fold greater during low tide than high tide. This variability is probably a natural occurrence in many PNW estuaries. At the four locations within the Yaquina Estuary where K_d values were determined nearly continuously from 1999 to 2001, the coefficient of variation ranged from 33 to 66%, indicating considerable tidal and seasonal variability.

Only within the Yaquina Estuary were there sufficient long-term measurements of K_d to derive a relationship with *Z. marina* colonization depth. This linear relationship (Figure 8-17) was consistent with the idea that some portion of the reduction in colonization depth with increasing distance from the mouth of the Yaquina Estuary is related to a reduction in water clarity. This upriver trend is consistent with that reported by Thom et al., (2003) who noted an upriver reduction in *Z. marina* abundance in the Coos and Willapa estuaries, which they attributed to increased turbidity and reduced salinity.

Differences in *Z. marina* colonization depth between PNW outer coast estuaries and Puget Sound bays are also likely due to water clarity, as K_d values for Puget Sound are generally lower (Thom et al., 2001, 2003). The trend for greater colonization depth with increased water clarity is evident for all species of seagrasses, where the maximum colonization depth corresponded to approximately 11% of surface irradiance (Duarte, 1991). However, the data for *Z. marina* presented by Duarte (1991) suggest that the amount of light needed to sustain this species at depth is almost double that for seagrasses in general (Table 8-4). Duarte (1991) also suggested that the world-wide relationship between K_d and the maximum seagrass colonization depth (all species) was linear, and that it could be simply calculated as

$$Z_c = \frac{1.86}{K_d} \tag{8.2}$$

where Z_c is the maximum colonization depth (m).

Duarte (1991) also noted that this result was similar to the results obtained for *Z. marina* (Z_c = $1.53/K_d$ and $Z_c = 1.62/K_d$) on the Atlantic Coast of the U.S. (Dennison, 1987) and within Danish estuaries (Nielsen et al., 1989). The trends in Z_c and K_d for *Z. marina* in the Yaquina Estuary are consistent with and appear to extend Duarte's (1991) relationship into more turbid waters of the Yaquina Estuary (Figure 8-21). These K_d values were converted (Equation 8.1) to the percent of surface irradiance at Z_c which are presented in Table 8-4. Also included in Table 8-4 are the minimum light requirements recommended for the growth and survival of submerged aquatic vegetation (SAV) in Chesapeake Bay (Batiuk et al., 2000).

Although the mean percent of surface irradiance needed to maintain *Z. marina* in the Yaquina Estuary appears to be lower than mean literature values, individual measurements within these literature values are highly variable with mean values of the present study within the range of those previously reported (Table 8-4).

Table 8-4. Comparison of percent of surface irradiance needed to maintain *Z. marina* at its maximum colonization depth from published data and from Yaquina Estuary data.

SOURCE	MEAN	STANDARD ERROR OF MEAN	N	MAX	MIN
Duarte, 1991	20.5	2.1	29	43.9	4.7
Current Study	13.4	1.9	64	85.8	1.5
Batiuk et al., 2000	15[a]				
[a] Value is not a mean but is based on an analysis of literature and on an evaluation of monitoring and modeling research.					

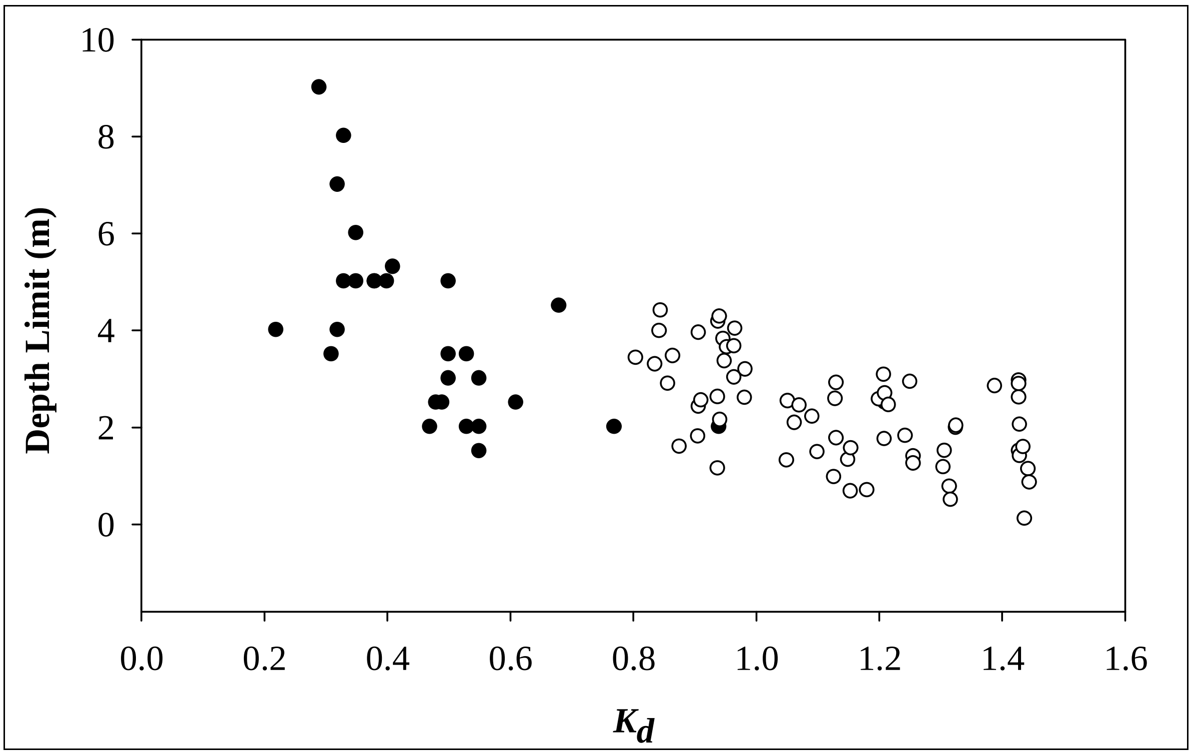

Figure 8-21. Relationship between *Z. marina* maximum depth limit (m) and K_d. (●) Data from Duarte (1991) and the Yaquina Estuary (o) from the present study.

8.3.4 *Zostera marina* Light Criteria

Minimum light requirements for maintaining and restoring seagrass have been proposed for Chesapeake Bay (Batiuk et al., 2000) and Puget Sound (Thom et al., 1998). Chesapeake Bay light criteria values were empirically estimated by measuring the maximum depth of seagrass annually and associated K_d values monthly (Dennison et al., 1993). For Chesapeake Bay, proposed water column light requirements vary by estuarine salinity classification with higher light requirements suggested for polyhaline and mesohaline (>22% of surface irradiance) regions than for fresh and oligohaline regions (>15% of surface irradiance). These proposed criteria were modified (Batiuk et al., 2000) by the amount of light absorbed by epiphytes encrusting seagrass leaf surfaces, which reduced these criteria values to 9 and 15 %, respectively. These proposed criteria are also applicable only to the seagrass growing season (typically spring though fall).

In contrast, light requirements for *Z. marina* in Puget Sound, Sequim Bay, Willapa Bay, and Coos Bay were reported as integrated light intensity levels (Thom et al., 1998; Thom et al., 2008). These were estimated using maximum seagrass depth measures, K_d values and production-irradiance (P vs. I) relationships. Based on this methodology, Thom et al. (2008) suggested that the minimum light requirements to maintain *Z. marina* is ~300 µmoles photons m^{-2} s^{-1} for at least three hours daily or ~3 moles photons m^{-2} d^{-1} during May through September. However, growth would be light limited during the spring and summer at light levels < 7 moles photons m^{-2} d^{-1}. Thom et al. (1998) went on to suggest that for *Z. marina* to minimally persist would require mid-day minimum irradiance values at the maximum depth limit to be approximately 150 µmoles photons m^{-2} s^{-1} during the year. Assuming that mid-day surface irradiance is in the range of 1000-2000 µmoles photons m^{-2} s^{-1}, this corresponds to approximately 15-30% of surface irradiance which is consistent with other published criteria values and the present study (Table 8-4). At a lower margin *Z. marina* site in the Yaquina Estuary, the mean daily irradiance value was approximately 3.8 moles photons m^{-2} d^{-1} (Kaldy and Lee, 2007). Although this value exceeds the Thom et al. (1998) criterion, irradiance values were highly variable, ranging from 0.5 to 7 moles photons m^{-2} d^{-1}, with extended periods of apparently inadequate lighting at depth from October to December (Kaldy and Lee, 2007). However, even during these periods of apparently inadequate irradiance, Yaquina Estuary *Z. marina* continues to grow (Boese et al., 2005; Kaldy and Lee, 2007). This continued growth and survival under conditions of negative carbon balance is most likely maintained by the utilization of rhizome carbohydrate reserves (Cabello-Pasini et al., 2002).

8.3.5 Yaquina Estuary *Zostera marina* Light Criteria

While it is tempting to directly apply the existing light criteria values to PNW estuaries, there are several additional factors that need to be considered. Chesapeake Bay, and the estuaries from which Duarte (1991) and Thom et al. (1998) derived their relationships are generally less turbid (mean K_d ~ 0.5 m^{-1}) than the Yaquina (mean K_d ~ 1.1 m^{-1}) and the other estuaries surveyed in the present study. *Zostera marina* has been shown to adapt to lower winter irradiance by increasing chlorophyll content (Zimmerman et al., 1995). Although we are not aware of any study that documents an analogous response to turbidity, a similar response to chronically more turbid water might allow for deeper colonization.

Temperature is a possible confounding factor. The range of near-surface temperatures within Chesapeake Bay, Puget Sound, and in the estuaries used in Duarte's (1991) review are likely greater than those observed within the Yaquina (Boese et al., 2005) due to the latter's twice daily flushing with cold ocean water. Increased respiration rates due to higher summer temperatures in these other estuaries would potentially need to be offset by increased irradiance for plants not only to maintain themselves but to store carbohydrates in rhizomes which could then be used to maintain the plant during the winter when irradiances may be less than optimal (Zimmerman et al., 1995; Burke et al., 1996; Zimmerman and Alberte, 1996). Therefore, it is possible that *Z. marina* in PNW coastal estuaries may require less spring and summer irradiance to perform the same function because of the generally cooler water temperatures in these systems.

Our study of epiphytes growing on *Z. marina* leaves in the Yaquina Estuary revealed a reduction in the amount of epiphyte biomass in the riverine segment (lower salinity region). With the exception of 2004 there appeared to be a seasonal pattern in epiphyte biomass such that the greatest biomass occurred in the summer and fall. This accumulation of epiphytes reduced the amount of light reaching leaf surfaces by about 60% in the oceanic segment of the Yaquina Estuary where the greatest densities of *Z. marina* occur. At present we are not sure how the epiphyte load and its impact on light availability compares to that found in other estuaries, but such variation will need to be considered in future efforts to derive water column light criteria for *Z. marina*.

CHAPTER 9:
EMERGING PACIFIC NORTHWEST PARADIGM

Henry Lee II and Cheryl Brown

9.0 Overview of the Pacific Northwest Environment and Regulatory Implications

In an effort to see the "forest" from the "trees" presented in the previous eight chapters, we offer a synopsis of the environmental conditions and nutrients dynamics in PNW coastal estuaries. This overview is based on our long-term research in the Yaquina Estuary, the classification study presented here, and a growing body of literature on the dynamics of PNW estuaries. Some of the environmental conditions presented here will not apply to every PNW estuary, in particular the estuaries with restricted tidal flushing. Nor does this synopsis apply to Puget Sound, which would require a separate analysis. Additionally, in most cases we had to extrapolate from a limited number of estuaries. Nonetheless, we believe these regional elements form the basis of an emerging "Pacific Northwest Paradigm." We then evaluate how these regional characteristics affect the classification of PNW estuaries as well as their potential regulatory implications.

9.1 Environmental Conditions

Climate and River Flow

The PNW has a mild Mediterranean climate with a wet winter and dry summer. River flows reflect this seasonal precipitation pattern.

Average annual rainfall is on the order of 140 to 350 cm y^{-1}, with higher rainfall in the northern portion of the PNW.

Watersheds and Land Cover

Watersheds are primarily forested, with low population densities and percent impervious surfaces.

Logging is common in many coastal watersheds but this alteration is not well captured in the standard land cover data.

Estuarine Inventory and Characteristics

There are 103 estuaries in the PNW. Most of these are small, with 73 estuaries less than 1 km^2 in size. Thirteen estuaries are larger than 10 km^2, of which Grays Harbor and the Willapa Estuary in Washington and the Columbia River Estuary are larger than 100 km^2.

PNW estuaries are characterized by large intertidal zones, which on average equal approximately 50% of the estuarine area.

PNW estuaries are classified as mesotidal, with tidal ranges of about 2 meters.

In general, PNW estuaries have high flushing with short residence times. Flushing is higher in the winter due to increased river flow. In the summer, flushing is driven by tidal exchange.

The PNW estuaries can be classified into seven classes (see Section 9.2), with the drowned river mouth estuaries sub-divided into tide-dominated, moderately river-dominated, and highly river-dominated systems.

Nutrient Concentrations and Loadings

Total nitrogen loading into PNW estuaries is high, often equivalent to loadings in eutrophic systems.

Dissolved inorganic phosphorous (DIP) concentrations are high, and on a national scale are rated as “fair”.

Differences in the extent of flushing and freshwater inflow suggest that estuary classes vary in their relative vulnerability to terrestrial nutrient loading in the following order (from most to least vulnerable):

lagoon≥blind estuary>tide-dominated river mouth>river-dominated river mouth≥coves/harbors

Nutrient Sources

Seasonal coastal upwelling from approximately April to October is the major nutrient source to the near-coastal region. The intensity of upwelling is highly variable over scales of days to decades.

Advection of oceanic water into the lower estuary is the major nutrient (nitrate and phosphate) source for the lower estuarine segments of PNW estuaries during the summer growing season. This portion of the estuary is referred to as the “ocean-dominated” segment of the estuary.

Terrestrially derived nutrients are the dominant nitrogen source for the upper estuary portions of PNW estuaries during the summer growing season and for most of the estuary during the winter. This portion of the estuary is referred to as the “river-dominated” segment of the estuary.

Nitrogen fixation associated with red alder, a native tree, may be an important terrestrial nitrogen source. Red alder populations may increase after fire, logging, and other disturbances but for the purpose of this analysis they are considered a “natural” nutrient source.

Point sources, such as sewage discharges may be locally important nitrogen sources but do not appear to be major sources for PNW estuaries in general.

Phytoplankton Concentrations and Sources

Chlorophyll *a* concentrations in PNW estuaries are generally low to moderate, and were rated as "good" on a national scale.

Advected oceanic phytoplankton is a major source of phytoplankton in the lower portion of the estuaries.

Ocean-dominated estuaries appear to have higher chlorophyll *a* levels than river-dominated estuaries.

Grazing by the benthic communities and by burrowing shrimp in particular, may substantially reduce phytoplankton concentrations and help to ameliorate enrichment effects.

Submerged Aquatic Vegetation

The native *Zostera marina* is the dominant species of submerged aquatic vegetation (SAV) in PNW estuaries in the lower intertidal and shallow subtidal.

The areal extent of *Z. marina* varies substantially among estuaries. In general, the largest SAV beds are in the tide-dominated and bar-built estuaries.

The lower depth limit to which *Z. marina* grows decreases in the upper estuary segments, which correlates with a general decrease in up estuary water clarity. There are differences among estuaries in the lower limit to which *Z. marina* grows, which are also likely related to differences in water clarity but additional factors, especially current velocities may contribute to these differences.

The non-native *Zostera japonica* has become abundant in a number of Pacific Coast estuaries over the last 20 to 50 years, and its populations appear to be expanding. In several estuaries, it covers a greater percentage of the intertidal zone than the native *Z. marina*.

The research to date suggests that *Z. japonica* may have both beneficial and detrimental impacts on *Z. marina*, estuarine food webs, nutrient dynamics, and maintenance of ecological services.

Benthic Macroalgae

Seasonal blooms of benthic macroalgae are a natural phenomenon in many of the PNW estuaries and are important primary producers in the estuaries larger than about 3.0 km^2.

Benthic macroalgae cover substantial portions of the intertidal in many estuaries, and reach standing stocks associated with adverse impacts on seagrasses in Chesapeake Bay. It is not known to what extent, if any, benthic macroalgae impacts SAV in the PNW.

The primary source of nitrogen for the intertidal benthic macroalgae blooms is natural, primarily consisting of oceanic water transported into the estuary.

Benthic Communities and Burrowing Shrimp

The burrowing shrimp, *Neotrypaea californiensis* and *Upogebia pugettensis*, are two frequently occurring ecological engineering species in moderate- to large-sized PNW estuaries. These species affect sediment structure, benthic community composition, and benthic nutrient fluxes through intense bioturbation and irrigation of the sediment.

Neotrypaea californiensis extends further up estuary than *U. pugettensis.* Additionally, *U. pugettensis* is either less abundant or absent in river-dominated estuaries.

Exposure of SAV and Other Resources to Oceanic vs. Riverine Nutrients

A fundamental question in evaluating vulnerability to anthropogenic nutrient loading is whether a species is primarily exposed to oceanic or riverine nutrients during the summer growing season. Populations that are primarily exposed to oceanic nutrients would have a relatively low vulnerability to terrestrially derived nutrient enrichments. Conversely, species that have a large proportion of its population in the riverine segment would be exposed to any increases in terrestrially derived nutrients, and hence have a higher vulnerability to nutrient loading from watersheds.

The majority of the *Z. marina*, benthic macroalgae, *N. californiensis,* and *U. pugettensis* populations occurred in the oceanic segments. While there were some differences among estuaries, this basic pattern held for both river- and tide-dominated estuaries and for large and small estuaries. Thus, the bulk of the populations of these species are primarily exposed to a natural nutrient source during the summer growing season.

The exception to this pattern was the nonindigenous *Z. japonica* which was abundant in both the oceanic and riverine segments. As a result of this distribution, *Z. japonica* populations are more likely to be exposed to anthropogenic nutrient enrichments. This increases their vulnerability to high anthropogenic nutrient loading though moderate nutrient enrichment could potentially stimulate their growth and establishment.

Expressions of Eutrophic Conditions

Dissolved oxygen concentrations in PNW estuaries are generally above levels indicative of eutrophic conditions. However, at times hypoxic shelf water can be advected into the lower portion of PNW estuaries (Brown et al., 2007).

The current levels of benthic macroalgae appear to be natural and not an indication of eutrophication.

The high nutrient loadings of PNW estuaries appear to be due to advection of high nutrient ocean water into the lower estuary and the effects of the nitrogen-fixing red alder in the watershed.

Based on these observations, there is little evidence of cultural eutrophication at the estuarine scale in PNW estuaries.

There are some observations suggesting that eutrophication may be occurring in localized areas of some estuaries. In particular, phytoplankton blooms and reduced dissolved oxygen may be occurring in sloughs with reduced flushing in the riverine segments of some estuaries.

While not presently showing eutrophic conditions over large areas, increases in nutrient loading from anthropogenic sources could potentially result in harmful algal blooms and depressed oxygen in the upper estuary segments of these systems. However, more detailed studies will be required to accurately assess the vulnerability of the upper estuary segment and to predict the nutrient concentrations or loadings at which such deleterious impacts would be expected.

Diagnostics of Eutrophication

With further development, the lower depth limit of *Z. marina* could potentially be used as an indicator for assessing estuarine condition in the PNW.

Increases in macroalgae especially in the riverine segment could be used as an indicator of nutrient enrichment, though there are a number of sampling issues.

Analysis of stable isotopes ($\delta^{15}N$) in the benthic macroalgae would help determine whether ocean-derived or terrestrially-derived nitrogen was the major nutrient source associated with macroalgal blooms.

9.2 Classification of Pacific Northwest Estuaries

Based on the importance of upwelling, large tidal ranges and flushing, wet/dry seasons, and other environmental characteristics of the PNW, we suggest that a regional classification system that captures these drivers is required to group the estuaries in an ecologically meaningful manner. The classification schemes we presented are specifically for the PNW, and as with any classification scheme, should be considered hypotheses until it is demonstrated that they adequately predict types of biotic assemblages, nutrient concentrations/loadings, impacts, or vulnerability.

Pacific Northwest estuaries break out into seven types based on geomorphology, ocean exchange, and river influence: coastal lagoons, blind estuaries, tidal coastal creeks, tidally restricted coastal creeks, marine harbors/coves, bar-built estuaries, and drowned river mouth estuaries. Using metrics normalizing total precipitation in the watershed to either estuarine volume or area, the drowned river mouth estuaries are further divided into tide-dominated river mouth estuaries and river-dominated river mouth estuaries.

Small PNW estuaries <1 km^2 are an important resource for salmon. Many of these systems were tentatively classified as "tidally restricted coastal creeks" but additional information is needed to evaluate the accuracy of this grouping.

Classification of estuaries by the absolute or relative areas of wetland classes from the National Wetland Inventory (NWI) offers the advantage of grouping estuaries with similar biotic habitats, which presumably integrates a wide range of environmental conditions. In the same way, classification of estuaries by similarity in land cover in the associated watersheds generates insights into potential anthropogenic loadings.

Six pairs of estuaries showed a high degree of similarity based on the intersection of the clustering by NWI wetland habitat type and the clustering by land cover patterns. These six estuary pairs - Grays Harbor and Willapa, Yaquina and Alsea, Coquille River and Siuslaw River, Rogue River and Klamath River, Quillayute River and Smith River, and Hoh River and Queets River – could potentially form the basis for developing management frameworks in the PNW.

9.3 Regulatory Implications

The primary regulatory implication of our results is that the development of national estuarine nutrient criteria that do not take into account the naturally high nutrient concentrations resulting from upwelling are likely to result in numerous false non attainments of nutrient criteria in PNW coastal estuaries (Brown et al., 2007). These false non attainments are most likely to occur in the lower estuary during the summer (Brown et al., 2007).

Another implication is that the development of total daily maximum loads (TMDLs) for nutrients need to take into account the intrusions of high nutrient ocean water into estuaries during a portion of the year. The influence of nitrogen fixation by red alder also needs to be evaluated in the development of TMDLs.

Our results strongly suggest that a regional approach is required both for the development of nutrient criteria and in the formulation of TMDL strategies.

9.4 Future Directions

This report describes a pilot effort at classifying PNW estuaries with regards to landscape attributes and their susceptibility to nutrient enrichment. In addition to the seven target estuaries described in this report, we have continued the sampling effort and to-date have sampled an additional 8 estuaries (Coquille, Humboldt, Klamath, Necanicum, Netarts, Siletz, Sixes, Yachats) using a modification of the methods described in this report. As these and any new data are analyzed, the classifications and models presented here will be tested and refined as needed.

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APPENDIX A: MAPS OF THE DISTRIBUTION OF *ZOSTERA MARINA* BASED ON AERIAL SURVEYS

The maps in this appendix show the areal extent of the intertidal and shallow subtidal *Zostera marina* in seven target estuaries. Distributions were mapped using aerial photography combined with ground survey results (see Chapter 6). The superimposed boxes show the extent of orthophotography used in mapping.

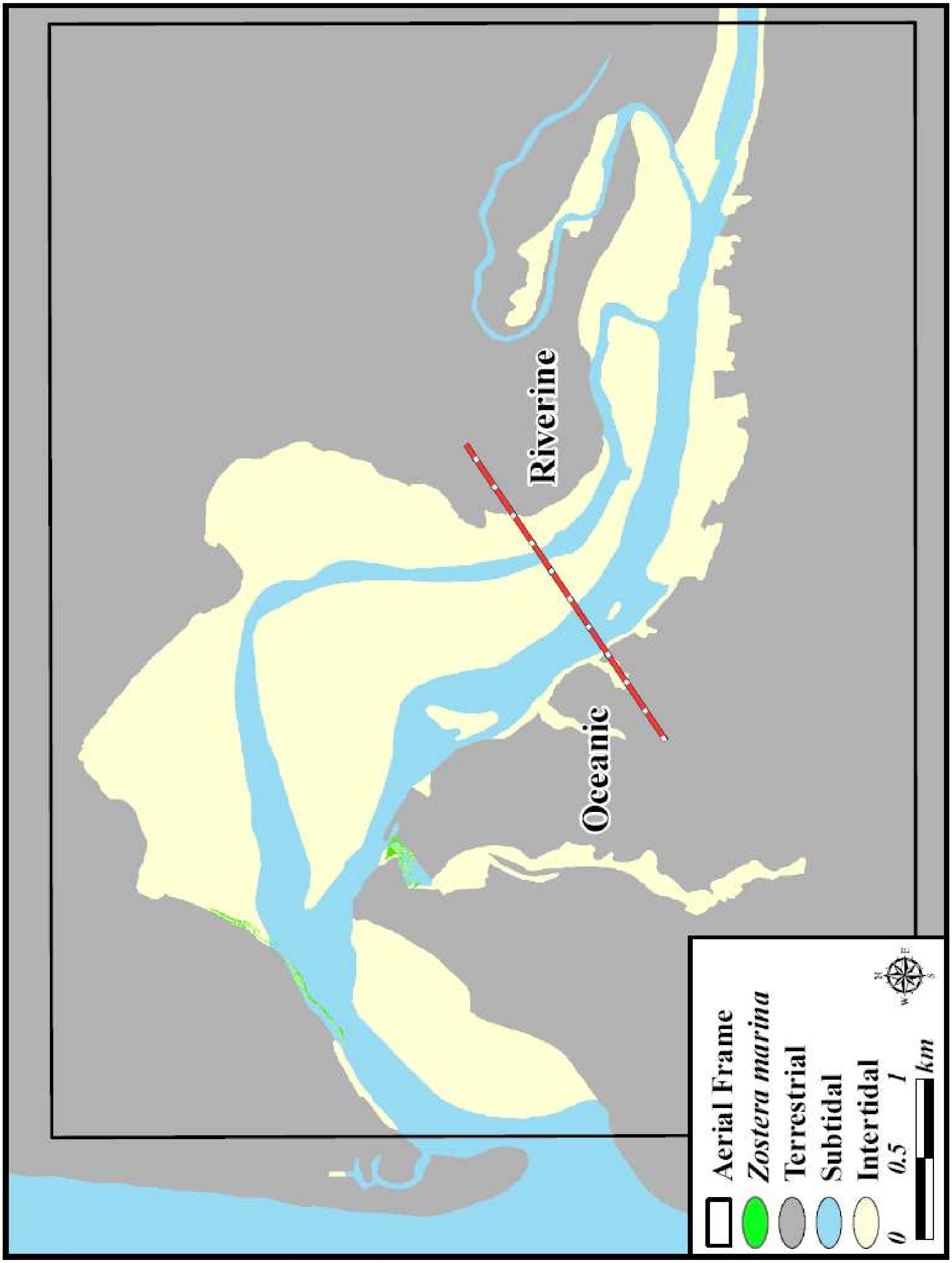

Figure A-1. Distribution of intertidal and shallow subtidal *Z. marina* in the Alsea Estuary.

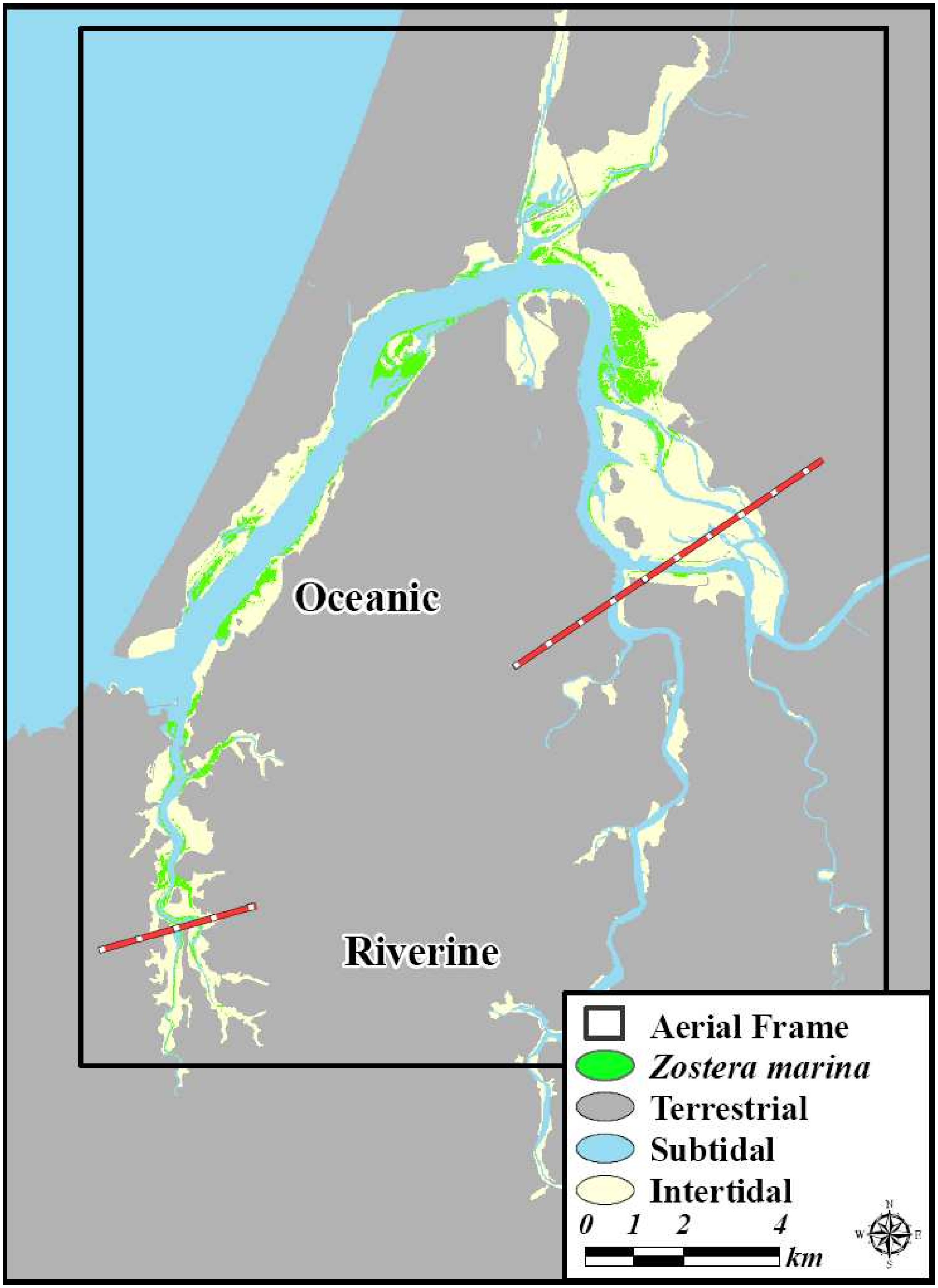

Figure A-2. Distribution of intertidal and shallow subtidal *Z. marina* in the Coos Estuary.

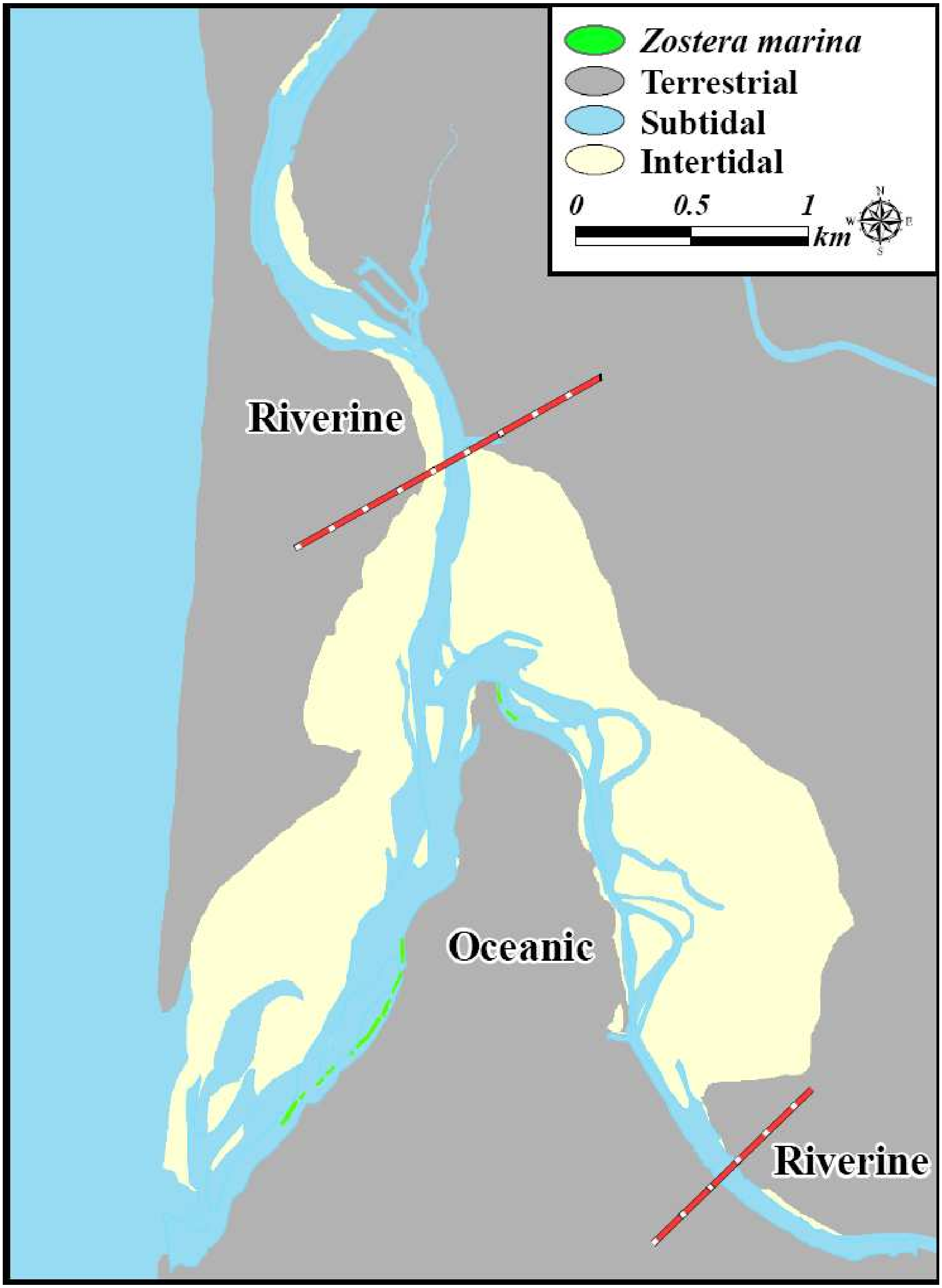

Figure A-3. Distribution of intertidal and shallow subtidal *Z. marina* in the Nestucca Estuary.

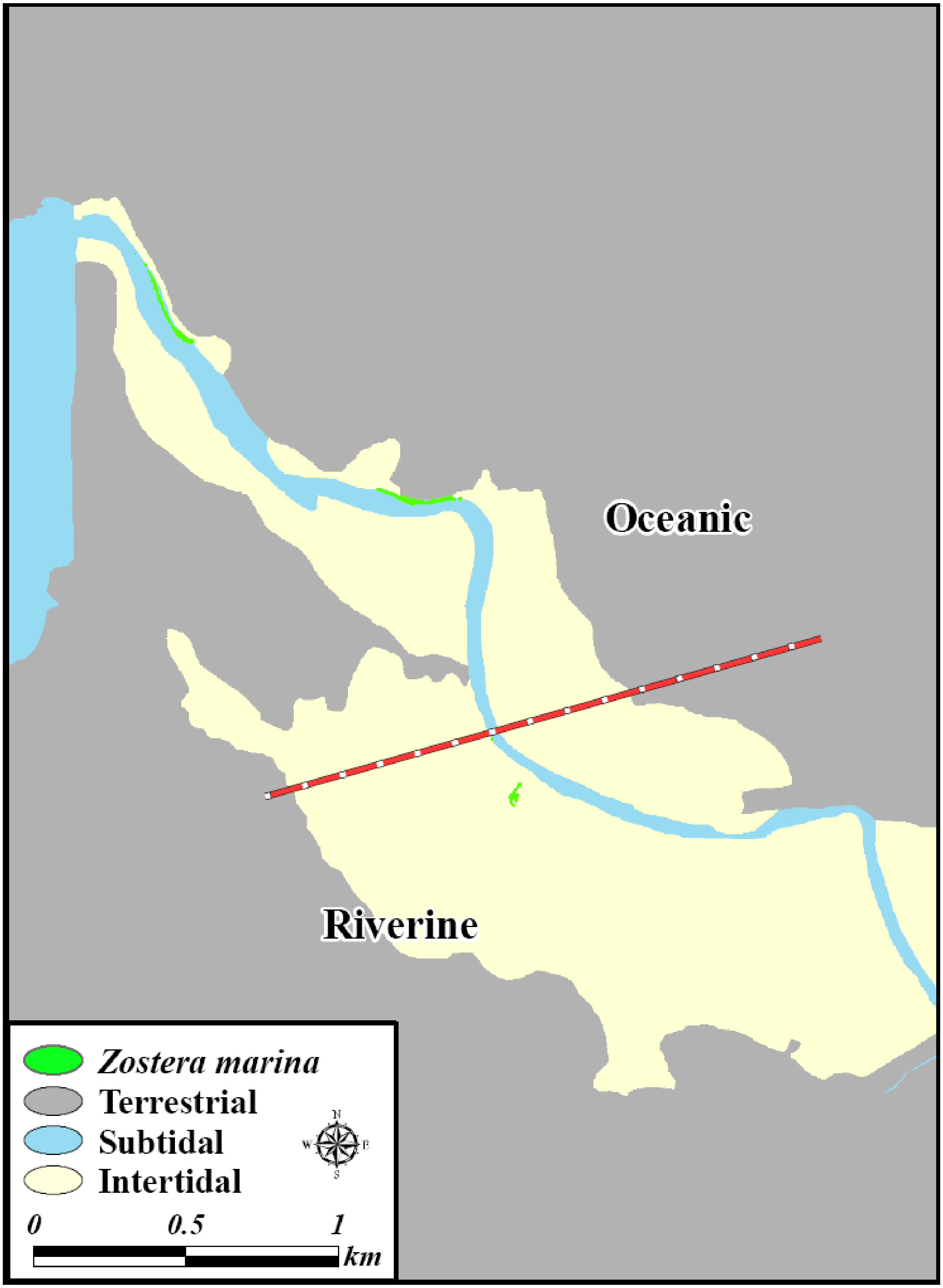

Figure A-4. Distribution of intertidal and shallow subtidal *Z. marina* in the Salmon River Estuary.

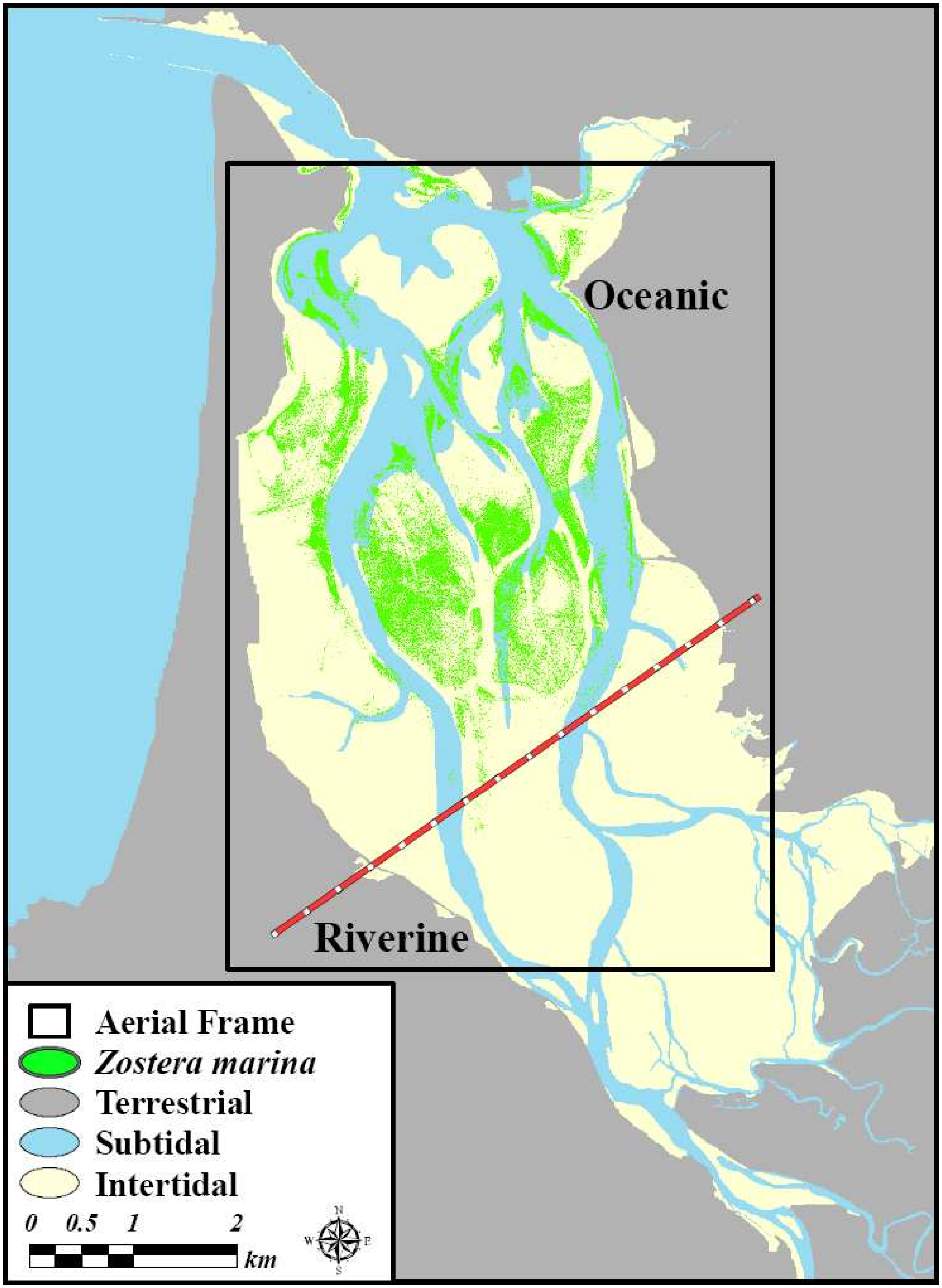

Figure A-5. Distribution of intertidal and shallow subtidal *Z. marina* in the Tillamook Estuary.

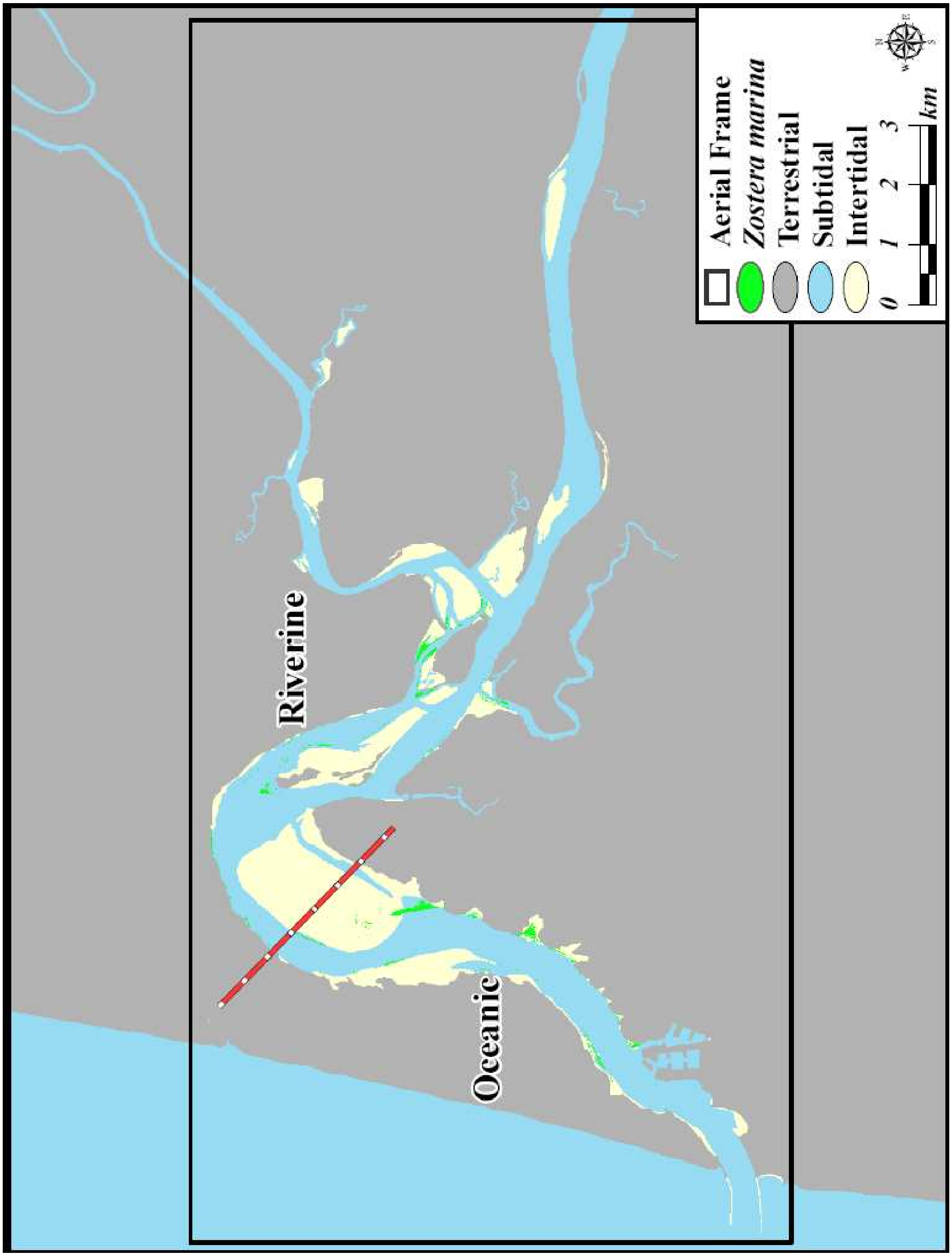

Figure A-6. Distribution of intertidal and shallow subtidal *Z. marina* in the Umpqua River Estuary.

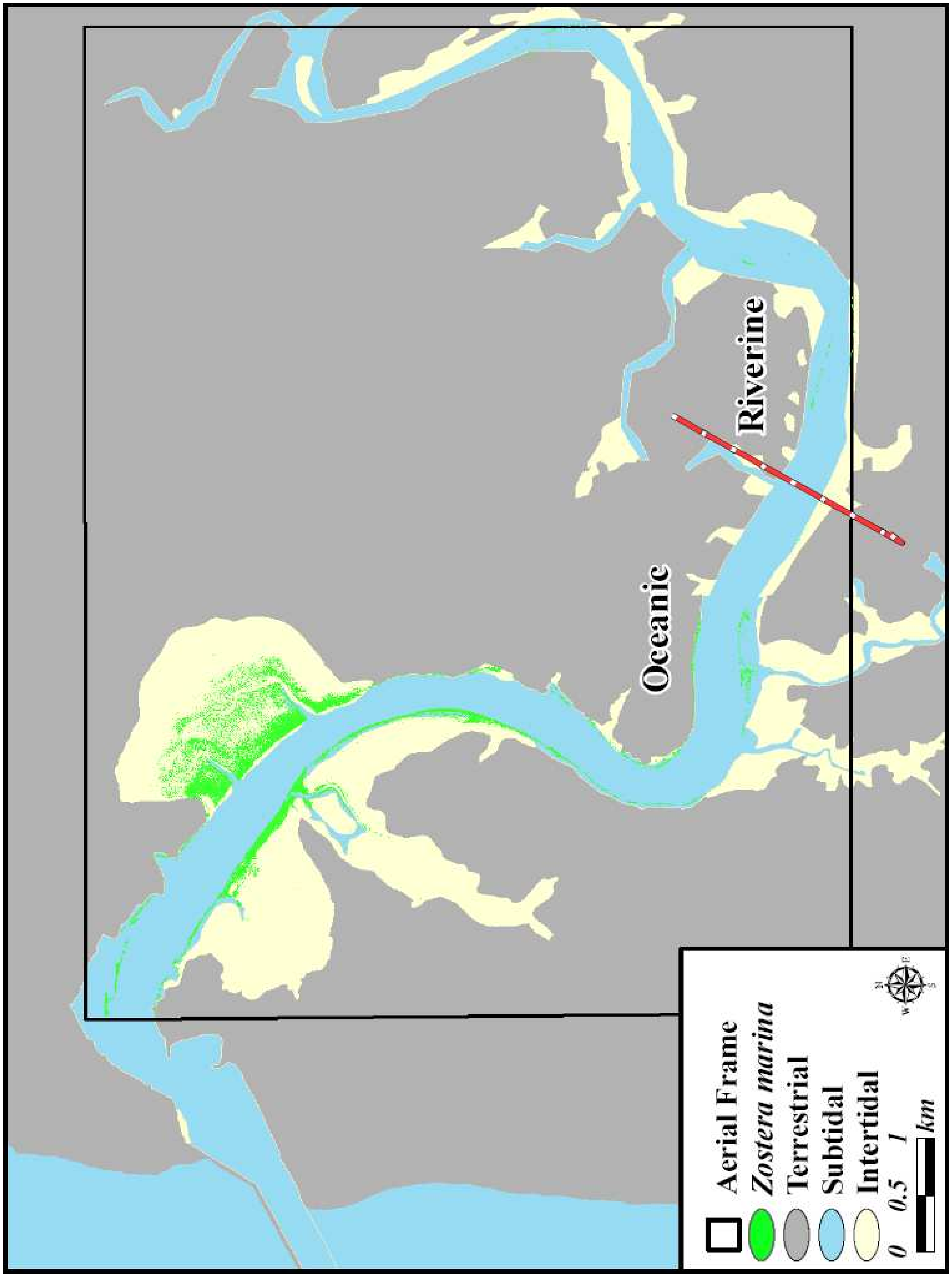

Figure A-7. Distribution of intertidal and shallow subtidal *Z. marina* in the Yaquina Estuary.

APPENDIX B: DETAILED DESCRIPTION OF QUALITY ASSURANCE PROCEDURES

B.1 Introduction

The quality assurance/quality control (QA/QC) program for this study is defined by the "Western Ecology Division Data Quality Management Plan (QMP) (US EPA, 2006). Measurement Quality Objectives (MQO's) establish the data user's requirements for precision and accuracy. The Measurement Quality Objectives for each parameter in this study are presented in Table B-1. Quality control measures were incorporated to assure data reliability and comparability and are described in the QMP plan. All contributing research was performed in compliance with an approved Quality Assurance Project Plan (QAPP). In addition, Standard Operating Procedures (SOP's) were followed to standardize routine data collection, processing and analysis for specific parameters. All procedural documents and QA/QC plans are approved by the WED Quality Assurance Manager.

Standard QMP protocols include routine instrument calibrations, measures of analytical accuracy and precision (e.g., analysis of standard reference materials, spiked samples, and field and laboratory replicates), overall data, range checks on the various types of data, cross-checks between original data sheets (field or lab) and the various computer-entered datasets, and participation in intercalibration exercises. Additionally, QA/QC included ensuring field and laboratory personnel were properly trained and experienced. Specific QA procedures are detailed in the following sections relative to each data parameter.

Accuracy and precision are indicators of MQO's and were established from considerations of instrument manufacturer's specifications, scientific experience, and/or historical data. Measures of systematic error (accuracy) and of random error (precision) are used to evaluate the quality of the results. Accuracy is a measure of how close measured values are to true values. In this appendix, accuracy is calculated using the following equation:

$$\text{Accuracy (\%)} = (1.0 - ((\sum(|V_t - V_m|)/ n)/ V_t) * 100$$

where V_t is the true or standard value, V_m is the measured values, and n is the number of measured values. Precision is an indication of the similarity of repeated analyses or sampling. Precision is calculated with the following equation:

$$\text{Precision (\%)} = (1.0-(\text{SD} / \overline{X})) * 100$$

where SD is the standard deviation, and $\overline{X}$ is the mean.

Table B-1. Measurement quality objectives for data collected by Western Ecology Division.					
Parameter	units	Expected range	Accuracy	Precision	SOP(s) or other
Seabird SBE-19 instrumentation					
Conductivity	mS cm^{-1}	0-100	± 0.5% of reading	0.01 mS cm^{-1}	SOP FSP.03, Manual
Salinity	PSU	0 – 35 PSU	± 0.5% of reading or 0.1 PSU	0.01 PSU	SOP FSP.03, Manual
Temperature	°C	0 - 25 °C	± 0.15°C	0.01°C	SOP FSP.03, Manual
Sea Point Turbidity	NTU	0-125 NTU	± 2% of reading or 2 NTU	0.1 NTU	SOP FSP.03, Manual
Depth	meters	0-15 m	± 0.018 m	0.001 m	SOP FSP.03, Manual
YSI 6600 Multiparameter Datasonde					
Conductivity	mS cm^{-1}	0-100 mS cm^{-1}	± 0.5% of reading or ± 0.001 mS cm^{-1}	0.01 mS cm^{-1}	SOP IOP.09, YSI manual
Salinity	PSU	0 – 35 PSU	± 1 % or 0.1 PSU	0.1 PSU	SOPIOP.09, YSI manual
Temperature	°C	0 to 25 °C	± 0.15°C	0.01°C	SOP IOP.09, YSI manual
Turbidity	NTU	0 to 1,000 NTU	±2% of reading or 0.3 NTU	0.1 NTU	SOP IOP.09, YSI manual
Depth	meters	0-15 m	± 0.018 m	0.001 m	SOP IOP.09, YSI manual
Dissolved Oxygen	mg l^{-1} or % Saturation	0-20 mg l^{-1} or 0-200% saturation	± 0.2 mg l^{-1} or ±2% of reading	0.01 mg l^{-1}	SOP IOP.09, YSI manual
Star ODDI Mini-CTD					
Salinity	PSU	0-40 PSU	± 0.75 PSU	0.02 PSU	Manufacturer Specifications
Temperature	°C	-1°C to +40°C	±-0.1°C	0.032°C	Manufacturer Specifications
Depth	meters	100 m	±0.4% of range	0.03% of full scale	Manufacturer Specifications
Discrete Water Samples					
Dissolved NO_3^-+NO_2^-	µM	0 - 100	±5%	5%	MSIAL UCSB, 2005
Dissolved NH_4^+	µM	0 - 5	±5%	5%	MSIAL UCSB, 2005
Dissolved PO_4^{3-}	µM	0 - 3	±5%	5%	MSIAL UCSB, 2005
Dissolved $Si(OH)_4$	µM	1 - 100	±10%	10%	MSIAL UCSB, 2005

Table B-1. Measurement quality objectives for data collected by Western Ecology Division.

Parameter	units	Expected range	Accuracy	Precision	SOP(s) or other
Total Suspended Solids	mg l^{-1}	1 - 150	15%	15%	WRS 14B.2
Water Column chlorophyll *a*	$\mu g\ l^{-1}$	0-200	±15%	±15%	MES SOP06.rev0
***Zostera marina* and Macroalgae Data**					
Seagrass shoot density	shoots m^{-2}	1-1500 m^{-2}	± 2 shoots 0.25 m^{-2}	1 shoot per 0.25 m^2 quad	QAPP02.01
Z. marina shoot length	mm	5-1200 mm	± 2 mm	± 1 mm	QAPP 04.01
Z. marina shoot width	mm	1-7 mm	± 1 mm	± 1 mm	QAPP 04.01
Z. marina shoot dry wt.	mg	0-1500 mg	± 1 %	± 0.1 mg	QAPP 04.01
Z. marina lower limit depth	cm	0-10 m MLLW	≤60 cm	N/A	Dynamac 2005 Work Plan
Epiphyte biomass dry wt.	mg	0-1500 mg	± 1 %	± 0.1 mg	QAPP 04.01
% Plant cover (Visual)	% area	0-100%	85%	±15%	QAPP 1.04
Point intercept (Frequency of Occurrence) SAV	# of intercepts	0-25	10%	11%	Dynamac 2005 Work Plan
Algal volume (wet)	ml	0-20,000	100 ml	50 ml	QAPP 01.06, Robbins and Boese 2002
Macroalgae nitrogen isotope ratio ($\delta^{15}N$)	‰	-2 to +25 ‰	0.5 ‰	0.5 ‰	QAPP02.01; QAPP 01.02
Burrowing Shrimp					
Shrimp Hole Density	holes m^{-2}	0-1500 m^{-2}	± 10%	30%	QAPP 2000.01
Physical Data					
Estuary bathymetry	ft	-40 to +10 MLLW	± 0.5	N/A	Survey contract
Sediment waves crests	m	0.1 – 5 m	± 0.1	± 2 %	Dynamac 2005 Work Plan
Sediment Properties					
Sediment total organic carbon and nitrogen content	Percent of dry wt	0-12%	± 10% of standard	± 10%	QAPP 2000.01
Sediment grain size distribution	Percent of dry wt	0-100% (0.4 µm – 948µm)	± 5% of dry wt	10%	QAPP 2000.01

Table B-2. Quality calibration and control checks for instruments and parameters.

Instrument	Calibration procedure	Quality Control Check	Frequency	Acceptance criteria	Action if values are unacceptable
Seabird CTD Conductivity	SeaBird factory calibration	Solution of known conductivity	If drift is suspected	MQO	Factory return
Seabird CTD Temperature	SeaBird factory calibration	Place in water bath with NIST traceable thermometer	If drift is suspected	MQO	Factory return
Seabird CTD Depth (pressure)	SeaBird factory calibration	In air reading compared to Fortin type Hg barometer (Nat. Wea. Serv. type)	If drift is suspected	MQO	Factory return
Seapoint turbidity	Seapoint factory calibration	1. Laboratory 3-levels of spherical particle solutions 2. Field total suspended solids vs. reading	1. annually 2. quarterly to monthly	Linear R^2>0.95; slope ±25% of initial	Clean and re-test
YSI Multiparameter Sondes (6600 models)	Factory calibrated for depth; calibrations performed to manufacturer's specifications for remaining parameters	All parameters checked against factory standards upon retrieval from deployment	Immediately prior to deployment and upon retrieval	Performed to factory specification, methods and standards	Clean and re-calibrate or return to factory
YSI Hand-held meters	Force to temperature corrected standard	All parameters checked against factory standards	Monthly	Performed to factory specification	Clean and re-calibrate or return to factory
CMT-DGPS	Self-calibrates with satellites with every use	Visit type 1 USGS reference site with multiple readings	Annually	MQO	Check post processing values re-test
LiCor LI-193SA sensor	Li-Cor factory calibration performed every 2 years	Solar noon clear sky exposure	Bi-annually	MQO	Return to factory
Balance (5-place)	Factory representative adjusts on annual basis	Check with standard weights-Class 'S'	Before each session	Within specific class tolerance	Contact balance maintenance personnel
Turner 10-AU Fluorometer	2-point calibration using solid secondary standard	Check solid secondary standard reading (low and high settings) for instrument drift.	At start, middle and end of every run	If high setting of secondary standard reading differs by ± 1% from true value	Re-calibrate and re-run samples

Table B-2. Quality calibration and control checks for instruments and parameters.

Microbalance	Check calibration with internal weights	Readings of 5 class ‘S’ masses that cover expected range	Before each session	± 2 mg	Clean weighing pan and catch plate, recalibrate; have serviced
Isotope Ratio Mass Spectrometer	Background and reference gas stability	Zero enrichment test	Before each session	± 0.07‰ precision; < 0.05‰ offset from zero	Re-focus mass spec source, repeat zero enrichment test; service if necessary
	Working standards calibrated to NIST or equivalent certified standards	Nitrogen and carbon isotope ratios of working standards	24 QC working standards per run of 72 samples	± 0.2 ‰ precision within a run	Repack combustion and reduction columns, reanalyze samples; service if necessary
Mettler AT250 balance	Annual calibration by contractor (Quality Control Services, Inc.)	Check accuracy with ‘S’ class weights	With each use	Contractor determination	Check cell holder alignment, optic cleanliness, repeat
Coulter LS 100 Q	Beckman-Coulter factory calibrated	Analyze grains of known size. Instrument compares background with factory established reference	1-2 times a year Every sample	t-test comparing results; Visual evaluation of curve differences	Realign laser beam to adjust to offsets and clean diffraction cell; Factory service call.
Carlo Erba LS 100 Q 1108 elemental analyzer	Annual calibration by factory technician	Run standards of known TOC	With each run	MQO	Dismantle and clean diffraction cell or request company technician visit

B.2 Water Quality CTD Profiles and Grab Samples

Water quality cruises were conducted in each of the target estuaries at both high and low tides. Depending on the depth, either a Seabird SBE 19 CTD or YSI sonde 6600 was used to measure water quality parameters. For deep stations (>1.5 m) water quality parameters were measured continuously through the water column and processed into 0.25-m depth intervals. At shallower stations, readings were recorded at the surface, mid, and bottom. Variables measured by the CTD and associated instruments include depth, temperature, conductivity, turbidity, photosynthetically active radiation (PAR), and *in situ* fluorescence, and calculated variables include salinity and density. In addition to the PAR sensor attached to the Seabird unit, PAR readings were also taken in the air with a flat sensor and at 15 cm below the surface for profile interpretation. Dissolved oxygen was measured at discrete depths with a YSI sonde or a DO meter attached to the frame of the Seabird CTD.

For each water quality profile, the CTD was lowered to the bottom and a water sample collected from mid-depth during the up-cast. This water sample was analyzed for chlorophyll *a*, total suspended solids (TSS), and dissolved inorganic nutrients. Near-bottom conditions were measured at 0.5 m above the bottom. Data were collected every second and binned with 'Seasoft' software into 0.25 m discrete intervals. The YSI sonde was calibrated prior to use following the manufacturer's specifications. Light attenuation coefficients (*k*) were calculated for the water column on the downcast. Prior to analysis, the data were reviewed to eliminate any false reading caused by reflection from the aluminum boat. Care was taken so that the PAR sensor on the CTD was not in the shadow of the boat.

The water column was sampled at each site for dissolved inorganic nutrients ($Si(OH)_4$, NO_3^- +NO_2^-, NH_4^+, and PO_4^{3-}), chlorophyll *a* concentration, and total suspended solids (TSS). Water column samples were collected and prepared per MES SOP09.rev 0 (2003). Water quality samples were filtered and processed on board the boat or upon return to the laboratory or hotel within several hours of collection. Water samples were stored in coolers on ice prior to filtration.

A performance-based approach was used for evaluating the quality of the chemical analysis. Depending upon the compound, laboratory practices included 1) continuous laboratory evaluation through the use of Certified Reference Materials (CRMs) and/or Laboratory Control Materials (LCMs), 2) laboratory spiked sample matrices, 3) laboratory reagent blanks, 4) calibration standards, and 5) laboratory and field replicates.

B.2.1 Chlorophyll *a* and Total Suspended Solids

Water samples for chlorophyll *a* and total suspended solids (TSS) analyses were collected in duplicate and filtered on board (if possible) or upon return to the hotel or laboratory. Typically, the samples were filtered within 1-2 hours of sample collection. For TSS analysis, 1-liter of unfiltered seawater was collected at mid-depth as described above, filtered on board the boat and further processed according to SOP WRS 14B.2. The complete procedure for sample processing and analysis of chlorophyll *a* samples is detailed in WED SOP06.rev 0. The samples were stored in the freezer until analyses. Samples were extracted in 90% acetone solution. The fluorometer was calibrated with a 10-AU solid secondary standard and "blanked" with freshly prepared 90% acetone solution prior to each sample set analyzed. The high setting of the solid secondary standard was used in calibration and the low setting was used as a quality control check after calibration. During analyses, the solid secondary standard and 90% acetone blank were checked midway through and at the end of a sample set to verify that the fluorometer performance had not changed. If the solid secondary standard high setting differed from true values by ±1%, the instrument was re-calibrated and the previous half-set of samples were reanalyzed.

The solid secondary standard was calibrated to chlorophyll *a* concentrations using fresh chlorophyll *a* standards provided by the manufacturer (Turner Designs). The chlorophyll *a* standards supplied by the manufacturer (Turner Designs) consisted of a high concentration (181 $\mu g\ l^{-1}$) and a low concentration (18.2 $\mu g\ l^{-1}$). The solid secondary standard was calibrated with newly purchased standards in 2002. During 2006, the solid secondary standard was checked using additional set of chlorophyll *a* standards. This quality control check revealed that the accuracy was 99.2%. To assess the accuracy of the chlorophyll *a* measurements, we compared the known value of the solid chlorophyll *a* standard (low setting) to the actual measured values of the solid standard. Accuracy analysis was performed for all chlorophyll *a* data used in this report. The accuracy and precision for the chlorophyll *a* data reported in this study are estimated to be 99.7 %.

B.2.2 Nutrient Data

Nutrient analysis for nitrate+nitrite, ammonium, phosphate and silicate was performed by the Marine Science Institute Analytical Laboratory (MSIAL) of University of California at Santa Barbara. Nutrient analysis was carried out with a Lachat Instruments Model QuickChem 8000 Flow Injection Analyzer. Data quality indicators include representative calibration data, reagent blanks, replicate analyses and percent recovery analyses of spiked and control samples. Acceptable levels for these parameters are detailed in the MSIAL QA guidelines and provide means of monitoring data quality (MSIAL Quality Assurance Manual 2005). In addition to these internal QA checks, samples obtained from the National Institute of Sampling and Technology and other producers of certified reference material were analyzed periodically to audit performance. Deionized (DI) water blanks and sea water blanks (low-nutrient natural sea water, aged to allow nutrient values to drop to near-zero levels) were also run. An independently prepared "control" solution containing an intermediate concentration of each of the nutrients was also run. The chemistries used in determining the various nutrient species on this instrument have been developed by the manufacturer to have little or no salt effect, so the analytical response is the same for fresh, DI water samples, and standards as for salt water samples and standards. Saltwater samples, however, exhibit a refractive index-related response in the flow-through detector, so the sea water blank was used to adjust the measurement timing parameters to compensate for the refractive index effect. Instrument calibration was checked at the beginning of a sample-batch run, at the end of the run, and periodically during the run. Each calibration sample was analyzed in duplicate, and the resulting data were used to establish calibration curves for each nutrient species. If the mean of the two replicates of any standard differed from the known concentration of that standard by more than ten percent or more than one-half the concentration of the lowest standard, whichever, was greater, the calibration for that species was considered invalid and the calibration run was repeated. (See MSIAL Quality Assurance Manual 2005 for more details). The precision of the replicates and the accuracy of the standards for the data used in this report are presented in Table B-3.

Table B-3. Accuracy and precision for nutrient analyses.			
	PO_4^{3-} (%)	NO_3^-+NO_2^- (%)	NH_4^+ (%)
Precision	98.0	99.5	96.8
Accuracy	99.1	98.5	98.2

B.3 Handheld YSI Meters

The handheld multiparameter Yellow Springs Instruments (YSI) meters were checked on a monthly basis and prior to use as needed with the manufacturer's (YSI) conductivity standard. To check the dissolved oxygen (DO) reading, the DO probe was placed in a 100% saturated environment for several minutes. If the percent saturation value differed by more than 2% from 100% saturation, the electrodes were cleaned, a new KCl solution was applied and the membrane was replaced. A second DO reading was taken after the DO probe was serviced to ensure that it was operating properly. All QA/QC checks and calibrations were recorded in a database (Microsoft® Office Access 2003).

B.4 YSI Multiparameter Sonde

The data presented in this report were collected with YSI 6600 Multiparameter Sondes. The sondes were calibrated prior to deployment following the manufacturer's recommendations. Conductivity was calibrated with a one-point calibration using standards with conductivity values closest to the expected salinity range (50 mS cm^{-1} for high salinity stations, 10 mS cm^{-1} for mesohaline and 1 mS cm^{-1} for low salinity stations). Turbidity was calibrated with a two-point calibration; using reverse osmosis water (RO) followed by a 123 NTU YSI standard solution. The DO sensor was calibrated for DO in air at sea level using the saturated air in water method. The DO anode was cleaned and fresh KCl solution added prior to applying a new membrane film. The probes were set in a calibration cup with a small amount of water for maximum water vapor saturation for 15 minutes before the calibration reading was taken. The barometric pressure was determined from either a mercury barometer or from a YSI 650 or 556 hand-held meter. Temperature cannot be calibrated but its performance was checked. The datasondes were set in a flow through seawater bath in the laboratory for multiple readings immediately before deployment and upon return to the lab. The temperature and salinity of the water bath were cross-checked using an independent YSI handheld unit (YSI 650 or 556). All QA/QC calibration data and ancillary metadata are recorded in an Access database. The calibration accuracy for conductivity and turbidity was defined as the accuracy of the probe in a standard solution. Post deployment accuracy was defined as the accuracy of the probes after they are retrieved from the field and were tested against the known standard solutions. A database was used to calculate the pre- and post-deployment accuracy of the readings when compared to the known standards (Table B-4).

Table B-4. Precision and accuracy for YSI multiparameter sondes. *The post-calibration data were not recorded for conductivity.					
	Temperature (%)	Salinity (%)	Conductivity (%)	Turbidity (%)	Dissolved Oxygen (%)
Calibration Accuracy Pre-deployment	98.1	98.6	99.5	100.0	99.7
Post Deployment Accuracy (includes the effect of biofouling)	92.8	96.5	N/A*	91.9	98.5

B.5 Stable Isotope Data

Five replicate macroalgae samples were collected from each sampling location in each of the estuaries. Algal material was collected from hard substrates to eliminate contamination from any additional nutrient sources other than the water column. Samplers wore sterile lab gloves while collecting to prevent contamination. Each algae sample was washed thoroughly in Milli-Q water, frozen and lyophilized. The dried material was ground into a fine powder for isotope analysis. Grinding mortar and pestles were thoroughly rinsed with acetone and allowed to completely dry between samples (QAPP 02.01, 2002).

The EPA Integrated Stable Isotope Research Facility (ISIRF) analyzed the macroalgae samples for $\delta^{15}N$ according to SOP CL-6 (1999). Samples were combusted using a Carla Erba elemental analyzer (model # 1108) equipped with a 4 meter poraplot Q gas chromatograph column directly coupled to an isotope ratio mass spectrometer operating in a continuous flow mode (Delta S, Finnigan MAT, San Jose, CA, USA). This continuous flow mode also provides a direct measurement of nitrogen content. Protocol and methods for operation of the mass spectrometer are all based on published approaches that have been verified through multiple approach analyses and inter-lab comparisons. A concentration, calibration and reference standard were run at the beginning, mid and end of each run. Additionally, a spike, concentration standard, and blank were run every ten samples. Standard material included NBS tomato leaves (1573) for concentration standard; NBS spinach (1570) for reference standard; NIST Corn Stalk for calibration standard and spike recovery. Results for precision and accuracy for stable isotope data used in this study are presented in Table B-5.

Table B-5. Precision and accuracy of $\delta^{15}N$ data.	
	$\delta^{15}N$ (%)
Precision	99.1 (n=23)
Accuracy	96.6 (n=63)

B.6 Total Organic Carbon and Nitrogen of Sediment Sample
At all intertidal stations, a small sediment core was collected for total organic carbon and nitrogen (TOC/N) analysis. Sediment cores were stored on ice until they could be frozen upon return to the lab. One ml of 10% Reagent Grade HCl in Milli-Q water was added to the sediment vial prior to analysis. The acid was mixed thoroughly into the sample using a glass rod. The sediment samples were then left at room temperature overnight and checked in the morning to confirm that pH was below 2. Sediment samples were then dried overnight in a 70º C oven with the last hour at 105º C. After drying the samples were homogenized on the roller mill for 24 hours, sub-sampled and pelletized. Samples were analyzed using a Carlo Erba EA 1108 elemental analyzer. Work was performed by Dynamac Corporation on-site at the WED laboratory in Corvallis. Blanks, unknowns and three different standards were analyzed at the beginning of the analytical run and after every tenth sample. Every 10th experimental sample was also analyzed in duplicate and a 3rd aliquot was spiked with the EuroVector Soil 3 SRM. Results are reported as weight %.

The empirically determined limit of quantitation (LOQs) for carbon and nitrogen are 0.38 and 0.08 wt. %, respectively. The empirically determined limit of detection (LODs) for carbon and nitrogen are 0.08 and 0.02 wt. %, respectively. These LOQs and LODs values are based on sediment samples weighing approximately 30 mg. If the C or N concentration of an experimental sample is below the detection limit, the value is flagged but reported. If this condition is true for duplicate samples, the DQOs for precision are ignored. The actual calculated value for precision is reported. Acceptable precision and accuracy values for both C and N are > 90% and < 110% for the BCR 101 spruce needle, EuroVector Soil 3, and MESS-2 marine sediment SRMs. Acceptable precision and accuracy values for both C and N are > 90% and < 110% for acetanilide. All samples met the DQOs and the r^2 values for the acetanilide standard curves ranged from 0.9991 to 0.9998 for N and from 0.9994 and 0.9998 for C. Results for precision and accuracy for TOC/N data used in this study are presented in Table B-6.

Table B-6. Precision and accuracy of TOC/N sediment analysis.

	Nitrogen	Carbon
Precision (duplicates, %)	97.5	95.2
Spike Recovery (Accuracy, %)	98.4	101.7

B.7 Particle Size Analysis (PSA)
Sediment aliquots were analyzed for particle size by laser diffraction on a Beckman Coulter LS 100 Q series. Aliquots of sediment samples introduced into the LS100Q sample chamber were well-mixed in water by a high speed, reversing mixer to assure representativeness. For each sample, two aliquots were analyzed, with a minimum of two runs of each. The final value was an average of all replicates. The precision calculations were run on the sum of all size fractions in each sediment size class; clay: 0.38 – 3.86 µm and silt: 3.86 – 63.41 µm. The data were reported as percent fines which were calculated as the sum of the silt and clay size fractions. Control standards of nominally 15 µm garnet and 500 µm glass particles were purchased from the manufacturer. They are provided with a certification of their average size and the standard deviations of the size distributions. A certified Coulter technician evaluated the accuracy of the instrument using these standard materials on seven service calls between 2001 and 2005

(Table B-7). During operation, the instrument performs a number of self-checks. One of these is a measure of the background reading prior to introduction of every sample, and this background measurement is compared to a reference background reading file established at the factory. The Coulter LS100Q software is programmed to automatically check and align its laser beam and to measure the sensor offsets at startup and every 60 minutes thereafter.

Table B-7. Precision and accuracy of sediment particle size analysis.

	Silt	Clay
Precision (duplicates) %	96.2	95.0
Accuracy (control recovery) %	98.6	

B.8 Intertidal Probabilistic Sampling

QA/QC information for the probabilistic intertidal sampling was collected by comparing repeat measurements taken by four individuals from the same quadrat. Either seven or fourteen 2.5 m^2 quadrats were laid out in intertidal habitats spanning the natural range of seagrass and macroalgae densities (low to high densities). Two smaller 0.25 m^2 quadrats were laid out within the larger 2.5 m^2 quadrats in the same arrangement as in the actual field sampling and additional measurements were collected. Four individuals independently recorded data from the same plots and quadrats without discussing their estimates. After all quadrats were examined the data were analyzed for precision. The results of the independent analysis are presented in Table B-8.

Table B-8. Data comparability between four field crew members for intertidal probabilistic sampling.

Measure (units)	Precision (%)	Avg. STDEV between reported values	Avg. Max Difference between reported readings	# of Quads (n)
Number of Seagrass Shoots (Number shoots per quadrat)	82.3	5.8	5.5	14
Point Intercept Seagrass (Number of intercepts)	78.6	7.6	2.7	14
Point Intercept Macroalgae (Number of intercepts)	98.1	0.1	0.1	12
# Leaves *Z. marina* per shoot (Number per shoot)	88.4	0.7	1.3	7
Longest *Z. marina* Leaf Length (cm)	94.2	1.5	3.0	7
Z. marina Leaf Width (mm)	92.2	0.4	0.7	7
Sediment Wave Crests (cm)	91.6	0.9	1.7	7

Percent cover data were collected for five class categories (i.e., 0 [absent], 1-10%, 11-40%, 41-70%, 71-100%), therefore, precision could not be calculated as with other numeric data. However, in comparing the individual data assessments from four field crew participating in an exercise to evaluate consistency between individuals the following statements can be made about the categorical data. Percent Cover Green Macroalgae: Out of the 14 plots evaluated there was no more than one class difference between the four independent assessments in ten of the plots with four plots having 100% agreement. Percent Cover Seagrass: Out of 14 plots evaluated there was no more than one class discrepancy between the four evaluations at eight of the plots with six plots having 100% agreement. Percent Cover Other Algae: Out of seven plots examined there was 100% agreement in six plots with only one of the four data collectors reporting one size class difference in the eighth plot.

In assessing sediment color, several individuals combined the categories to encompass color variations in the surface sediment so calculating a precision value was not straightforward; however, for five of the seven plots all four crew included one or more of the same colors in their assessment. In the other two plots, three of the four individuals agreed 75% of the time. Sediment firmness was more subjective. In three of the seven plots all four crew agreed 100% of the time, in two plots they agreed 75% of the time and in two other plots the four crew was split between two different firmness values.

Macroalgae biomass was determined by the volumetric cylinder method (Robbins and Boese 2002). At each site all macroalgae were removed from each quadrat and measured volumetrically in-situ using a 4000-ml plastic cylinder. The algae were pressed with a plunger to remove excess water before recording the algal level in ml. This quick field method was determined to be an accurate surrogate for biomass dry weight determination with a linear relationship yielding an r^2 value of 0.78 to 0.88 for macroalgae species (Robbins and Boese, 2002).

B.9 Aerial Mapping of *Z. marina*

The remote sensing procedure used in this study to map the intertidal and shallow subtidal distribution of *Z. marina* utilizes aerial photography with false-color near-infrared (color infrared, CIR) film. This allows an aerial survey to be conducted during daylight low tides (and clear or uniform overcast weather) when a large majority of the *Z. marina* distribution in PNW estuaries is exposed or visible. The CIR film provides substantially better spectral resolution of exposed intertidal vegetation than true color film (Young et al., 1999). To map perennial *Z. marina* habitat, the surveys were conducted in late spring or early summer before the summer bloom of benthic green macroalgae that can interfere with the classification of *Z. marina*. Photoscales utilized ranged from about 1:6,000 to 1:20,000. The aerial photographs were digitally scanned and georectified while correcting for terrain and camera distortions to produced digital orthophotos. The spatial accuracy of the photomap for this estuary (photoscale: 1:10,000) was assessed by comparing 14 Root Mean Square Error (RMSE) offset values for positions of photovisible objects obtained from the photomap, referenced to published National Geodetic Survey (NGS) positions. The mean offset was 0.72 m ± 0.27 m (95% CI; Young et al., in press). The digital orthophotos were classified into *Z. marina* (defined as >10% cover) and bare substrate habitats (defined as ≤ 10% cover). On-the-ground resolution of 0.25 m was obtained in this process. A hybrid technique using both unsupervised and supervised classification steps has been developed for this habitat mapping project (Clinton et al., 2007). The technique requires training data from ground truth surveys, with station positioning accomplished by a differential-corrected global positioning system (GPS). The RMSE of GPS positions obtained at an NGS first-order monument in Yaquina Estuary was 0.62 m.

The ground survey technique provides percent cover measurements or estimates from quadrat placement or 35 mm snapshots within each stratum being classified, preferably augmented by GPS traces of sections of the intertidal meadow upper margins, as training data used in the image classification process. Another part of the ground survey employed a detailed procedure based upon the recommendations of Congalton and Green (1999) to provide accuracy assessment data from randomly positioned stations within each stratum (Young et al., in press). The results were analyzed with the aid of an error matrix (also know as "confusion matrix"). This is a square array of numbers set out in columns and rows that expresses the number of sample units assigned to a particular category in one classification relative to the number of sample units assigned to a particular category in another classification (Congalton and Green, 1999). Here, the number of stations (units) classified as *Z. marina* or bare substrate habitat based on the ground survey measurements (listed in the table's "column total" cells) are considered to be correct (i.e. reference data), while the values listed in the individual table cells are the corresponding number of stations in each class generated from the remotely sensed data. The quotient (as a percentage), obtained by dividing the total number of correctly classified sample units in a given category (e.g. *Z. marina* stratum) by the total number as indicated by the reference data, is termed "producer's accuracy". Similarly, the quotient obtained by dividing the total number of correct units in the given category by the total number of units classified as belonging to that category (listed in the table's "row total" cells) is termed "user's accuracy". A third accuracy value, obtained by dividing the sum of the "table diagonal" units by the total number of sample units in the matrix, is termed "overall accuracy" (Congalton and Green, 1999). To assess the effectiveness of this classification method, values for the Kappa index KHAT (and their estimated 95% confidence intervals) for these matrices also were calculated as a measure that assesses improvement over chance (Fielding and Bell, 1997). Results were obtained in the spring of 2004 from 51 randomly positioned stations within intertidal *Z. marina* meadows and 28 randomly positioned stations within bare substrate strata of Yaquina Estuary**.**

Based upon a comparison of results from the image classification with those from the ground survey (taken as the reference), application of the classical error matrix analysis yielded an overall accuracy for Yaquina Estuary of 97%, with a Kappa Index value of 0.94 ± 0.002 (Table B-9). The high index value attained indicates excellent agreement (Landis and Koch, 1977). The investigators attribute this very high accuracy level to the extensive training data provided via GPS mapping of the intertidal *Z. marina* meadow margins (Young et al., in press). For corresponding classifications of intertidal *Z. marina* and bare substrate strata in Tillamook, Umpqua and Coos estuaries, obtained without meadow margin mapping for training of the image analyst, overall accuracy values were 86%, 89% and 83%, respectively. In contrast, in Alsea Bay where extensive *Z. marina* meadow margin mapping was conducted, an overall accuracy of 100% was obtained (Table B-9). Due to the subtidal habitat of seagrass in Nestucca and Salmon River estuaries, seagrass patches were outlined using either a boat or by foot with a GPS unit. A "best fit" line was created from three replicate traces.

Table B-9. Image classification accuracy assessment.				
		Z. marina	Bare Substrate	row total
Tillamook Estuary				
Image Class (Predicted)	*Z. marina*	88	13	101
	Bare Substrate	12	68	80
	column total	100	81	181
Producer's Accuracy		88 %	84 %	
User's Accuracy		87 %	85 %	
Overall Accuracy:	86 %			
Kappa Index KHAT*:	0.72 ± 0.003			
Yaquina Estuary				
Image Class (Predicted)	*Z. marina*	50	1	51
	Bare Substrate	1	27	28
	column total	51	28	79
Producer's Accuracy		98 %	96 %	
User's Accuracy		98 %	96 %	
Overall Accuracy:	97 %			
Kappa Index KHAT*:	0.94 ± 0.002			
Alsea Estuary				
Image Class (Predicted)	*Z. marina*	44	0	44
	Bare Substrate	0	60	60
	column total	44	60	104
Producer's Accuracy		100 %	100 %	
User's Accuracy		100 %	100 %	
Overall Accuracy:	100 %			
Kappa Index KHAT*:	1.0			
Umpqua Estuary				
Image Class (Predicted)	*Z. marina*	37	10	47
	Bare Substrate	4	81	85
	column total	41	91	132
Producer's Accuracy		90 %	89 %	
User's Accuracy		79 %	95 %	
Overall Accuracy:	89 %			
Kappa Index KHAT*:	0.76 ± 0.004			
Coos Estuary				
Image Class (Predicted)	*Z. marina*	28	9	37
	Bare Substrate	10	68	78
	column total	38	77	115
Producer's Accuracy		74 %	88 %	
User's Accuracy		76 %	87 %	
Overall Accuracy:	83 %			
Kappa Index KHAT*:	0.62 ± 0.005			
* 95% confidence interval assuming KHAT is normally distributed				

B.10 *Zostera marina* Lower Depth Limit

Lower depth margin of *Zostera marina* was determined by georeferencing the position where the deepest seagrass was encountered. Transects were randomly selected in distinct *Z. marina* beds (identified from aerial photography) and were approached either by boat or foot depending on the water depth. Sampling was conducted on the lowest tides possible to increase the accuracy of locating plants growing at the lowest depth limit. In the deeper systems, an underwater video camera was mounted on a long PVC pole linked to a video monitor on deck. When the first *Z. marina* patch was seen on the monitor the pole was quickly thrust into the sediment to stop the momentum of the boat. A GPS reading was taken and a lead line was used to record the depth to the closest centimeter. In shallower waters, seagrass blades were clearly apparent on the water surface and could easily be approached by foot. Personnel walked along the transect from shore to the deepest *Z. marina* patch, recorded a GPS location, measured the depth with a lead line.

B.10.1 Tidal Corrections for Lower Depth Limits

Tidal corrections were applied to account for variations in tide elevation at time of lower limit depth observations, with corrected lower depth limits expressed as depth below mean lower low water (MLLW). Tidal predictions were used to make these tidal corrections. Tidal predictions are least accurate during storms and extreme low and high tides. Review of weather and tidal conditions during time periods when the lower depth limit was measured, indicated that conditions were relatively calm during the sampling and not collected during extreme tides. Tidal heights relative to MLLW were calculated using WXTIDE 32 for each depth site using the time of data collection and the nearest tide prediction location. The largest source of error in the tidal corrections resulted from not having predicted tides available for all locations along the longitudinal axis of the estuary and having to use the closest tide prediction station. To estimate the error associated with the tidal correction, differences in tidal corrections between the two stations that are located upstream and downstream of the observations were calculated

B.10.1.1 Yaquina

The difference between predicted and observed tidal heights at the South Beach tide gauge was less than 0.15 m on all data collection days. Tidal predictions that take into account variations in tidal amplitude and phase lags were available for four locations (South Beach, Yaquina, Winant, and Toledo) in the Yaquina Estuary (http://co-ops.nos.noaa.gov/tides05/tab2wc1b.html#132). The maximum distance along the river at any given depth station from a tide station was 4.5 km. The error associated with the tidal correction is estimated to be 0.1-0.2 m. The error associated with the tidal correction increases with distance from the mouth of the estuary, being a minimum of 0.1 m in the lower estuary and as high as 0.2 m near Toledo.

B.10.1.2 Tillamook

The maximum difference between predicted and observed tidal heights at the Garibaldi tide gauge was 0.3 m on data collection days. Tidal predictions that take into account variations in tidal amplitude and phase lags were available at five locations (Barview, Garibaldi, Miami Cove, Bay City and Hoquarten Slough) in the Tillamook Estuary (http://co-ops.nos.noaa.gov/tides05/tab2wc1b.html#132). The maximum distance along the river any given depth station from a tide station was ~ 3.6 km. The error associated with the tidal correction is estimated to be 0.1- 0.3 m. The error associated with the tidal correction increases with distance from the mouth of the estuary, being a minimum of 0.1 m in the lower estuary and as high as 0.3 m between Bay City and Hoquarten Slough.

B.10.1.3 Nestucca

The difference between predicted and observed tidal heights at the Nestucca tide gauge on data collection days is unknown. The tidal predictions for Nestucca used parameters for variations in tidal amplitude and phase lags from a tidal reference station in Crescent City, CA for a single station at the bay entrance available at: http://co-ops.nos.noaa.gov/tides04/tab2wc1b.html#132. The maximum distance along the river at any given depth station from the tide station was ~ 4.7 km. The error associated with the tidal correction increases with distance from the mouth of the estuary. To estimate the error associated with the tidal correction, we compared the maximum distance above to the maximum distances from tide prediction stations in Yaquina and Tillamook Bays and estimate the error to be between 0.1 m and 0.3 m.

B.10.1.4 Salmon

The difference between predicted and observed tidal heights at the Salmon River on data collection days is unknown. No tidal prediction stations are available in the Salmon estuary. The nearest tidal prediction station is at Taft in the Siletz Estuary which is 11.1 km to 12.5 km from the sample locations in the Salmon Estuary. The tidal predictions for Taft use parameters for variations in amplitude and phase lags of tides from a tidal reference station (Crescent City, CA) for a single station at the bay entrance available at: http://co-ops.nos.noaa.gov/tides04/tab2wc1b.html#132 . The error associated with the tidal correction is unknown.

B.10.1.5 Alsea

The difference between predicted and observed tidal heights at Alsea Bay on data collection days is unknown. The tidal predictions for Alsea used parameters for variations in tidal amplitude and phase lags from a reference station at Crescent City, CA for two single stations available at: http://co-ops.nos.noaa.gov/tides04/tab2wc1b.html#132. The maximum distance along the river at any given depth station from the tide station was ~ 1.2 km. To estimate the error associated with the tidal correction, we compared the predicted tide lag between the Waldport, Alsea Bay station and the Drift Creek, Alsea River station (which are ~7.6 km apart) during hours of upstream data collection. The error associated with time lag in the predicted tides for the data collected is less than 0.1 m.

B.10.1.6 Umpqua

The difference between predicted and observed tidal heights at Umpqua River on data collection days is unknown. The tidal predictions for Umpqua used parameters for variations in tidal amplitude and phase lags from the tidal reference station in Humboldt Bay, North Spit, CA for three stations available at: http://co-ops.nos.noaa.gov/tides04/tab2wc1b.html#132. The maximum distance along the river at any given depth station from the tide station was ~5.5 km. To estimate the error associated with the tidal correction, we compared the predicted tide lag between the Umpqua River, Entrance station, the Umpqua River, Gardiner station and the Umpqua River, Reedsport station (which are ~10.5 km and ~ 4.0 km apart) during hours of data collection. The error associated with time lag in the predicted tides for the data collected is less than 0.37 m.

B.10.1.7 Coos

The maximum difference between predicted and observed tidal heights at the Charleston, Coos Bay tide station was 0.1 m. Historic tide data for the Charleston, OR station is available at: (http://tidesandcurrents.noaa.gov/geo.shtml?location=9432780). The tidal predictions for Coos use parameters for variations in amplitude and phase lags of tides from a tidal reference station within the estuary (Charleston, OR) and for two more stations, Empire and Coos Bay available at: http://co-ops.nos.noaa.gov/tides04/tab2wc1b.html#132. The maximum distance along the river at any given depth station from the tide station was ~10 km. The error associated with time lag in the predicted tides for the data collected is unknown because ~70% of the depth stations are upstream of the tide stations.

B.11 Field Locations and Distance Upriver Calculations

The geoposition of each station was collected in the field with a Global Positioning System (GPS) and differentially corrected in post processing with data from the nearest National Geodetic Survey (NGS) Continuously Operating Reference Station (CORS). The GPS data collection device (CMT March II) published post-processed differentially corrected two dimension root mean square error (spatial accuracy) ranges from 1 m to 5 m.

The distance from the mouth of each estuary to each station was calculated using GIS mapping software ArcMap. For each estuary, National Hydrologic Dataset (NHD) center line features extending the length of the rivers from the ocean shore (disregarding jetties) were converted into route files. The ArcToolBox linear referencing tool, “LocateFeaturesAlongRoutes”, was used to calculate the distances in meters for each station from the ocean shore to the nearest point along the centerline route feature. The distance from each station to the nearest point along the centerline route was calculated simultaneously by the geoprocessing tool.

B.12 Estuarine and Watershed Landscape Analysis

To capture the diversity of estuary types on the Pacific Coast, a national standard, the National Wetland Inventory (NWI) developed by the U. S. Fish & Wildlife Service (U.S. Fish Wild. Ser., 2002; http://www.fws.gov/nwi) was used as the criterion for defining estuaries. The NWI classifies aquatic habitat types using a hierarchical set of attributes based on Cowardin et al. (1979), which includes marine, estuarine, riverine, palustrine, and lacustrine areas with further subdivisions for tide height, substrate type and the presence of broad classes of wetland plants (e.g., emergent vs. aquatic bed). In an effort to inventory all estuaries, an estuary was defined as any coastal water body that contained an NWI estuarine polygon.

B.12.1 Estuary Inventory

The estuary watershed geospatial layer generated for NOAA’s Coastal Assessment Framework (ftp://sposerver.nos.noaa.gov/datasets/CADS/GIS_Files/ShapeFiles/caf/) was used as the initial source to identify the larger Pacific Coast estuaries. However, this layer was not sufficiently detailed to represent the moderate to small estuaries. To develop a complete inventory of coastal estuaries, estuary water bodies were identified from previously mapped estuarine polygons from the NWI. Additionally, visual inspection of coastal sites either in the field or from aerial photographs was used to identify any additional coastal water body that appeared to have “estuarine” characteristics but that did not have a NWI estuarine polygon. Geospatial data from the NWI were obtained from digital databases and from on-site digitization of paper maps not otherwise available in digital formats. Sites classified by NWI as estuarine, and in some cases, marine, palustrine or lacustrine, were inserted into the estuarine/watershed GIS layer, retaining the NWI classification system attributes. In some cases, adjustments were made to the attribute coding of water bodies to reflect judgments that these were part of an estuarine system.

B.12.2 Watershed Delineation
The Sixth Field hydrologic unit code (HUC) subwatershed geospatial layer created by the Forest Service from 1:24,000 scale USGS maps, digital elevation models and other data sources (http://www.reo.gov/gis/projects/watersheds/REOHUCv1_3.htm) was used as a primary reference. In Oregon, the Forest Service and Oregon State University have produced a watershed layer refined to the 7th field HUC boundary lines for most of coastal Oregon (http://www.fsl.orst.edu/clams/cfsl0233.html) north of the Rogue River.

In all states, final refinements to the drainage boundaries between coastal and estuarine basins were often based on review of the hydrologic drainage patterns derived from digital elevation data (10-meter resolution in Oregon) and from USGS 1:24,000 scale quadrangle maps.

The watersheds delineated in this project capture the entire drainage area for each of the estuaries. By delineating the entire watershed, these watershed areas are equivalent to the sum of NOAA's Estuarine Drainage Area (EDA, portion of watershed that empties directly into the estuary and is affected by tides) and Fluvial Drainage Area (FDA, portion of an estuary's watershed upstream of the EDA boundary; see http://coastalgeospatial.noaa.gov).

B.12.3 Land Cover Sources
The estuary watersheds and CDAs were used as clipping boundaries for several land use/land cover datasets that are available for the Pacific coastal region at this time. The National Land Cover Data (NLCD, http://www.mrlc.gov) represents land cover circa 1992 and its extent is nationwide. This dataset was clipped to the watershed boundary for each estuary or CDA and is complete for all watersheds except for the interior Columbia River basin in Canada, and the Tijuana River estuary in Mexico. The 1992 NLCD data contains 21 classes of land cover (http://erg.usgs.gov/isb/pubs/factsheets/fs10800.pdf) with some modification in the 2001 NLCD data (http://www.epa.gov/mrlc/definitions.html#1992). The area of each land cover class in square kilometers and as a percentage of the watershed was calculated and entered into an Access database.

Two additional land cover datasets have been created by the NOAA's Coastal Services Center (C-CAP, http://www.csc.noaa.gov/crs/lca/ccap.html) program. The more recent data were derived from late 2000 and 2001 Landsat TM (thematic mapper) imagery. NOAA also produced a layer from imagery collected circa 1995-1996 and the earlier dataset was used to generate a land cover change layer. As the NOAA data were generated primarily to investigate land cover effects on coastal resources, the Pacific coast datasets did not completely cover the entire drainage areas of the largest three estuary watersheds. The interior portion of the San Francisco Bay estuary and the eastern portion of the Klamath River drainage were not included in the NOAA datasets. The lower Columbia River basin downstream of Bonneville Dam is included, but much of the interior Columbia River basin is not within the extent of the NOAA datasets. The 2001 NOAA data are based on 22 land cover classes (http://www.csc.noaa.gov/crs/lca/oldscheme.html), which are not exactly the same as those used in the NLCD.

In January 2007, the Multi-Resolution Land Characteristics Consortium (MRLC, http://www.mrlc.gov) released a new national land cover data, NLCD 2001. The areas classified by the NOAA C-Cap program were incorporated into the 2001 release. Procedures used in the development of the 2001 land cover data layer are presented in Homer et al. (2004). The land cover in NLCD 2001 is based on 30-meter resolution data derived from Landsat imagery and uses 21 classes that are a modified version of the land classes used in the 1992 NCLD analysis (see http://www.epa.gov/mrlc/definitions.html#1992). The 2001 data were analyzed by zone, with the analysis of the Pacific Coast watersheds utilizing data from eight zones. The bounding coordinates of these eight zones are given in Table B-10.

Table B-10. Bounding coordinates for the eight 2001 MRLC zones containing Pacific Coast watersheds.

MRLC Zone	Longitude, degrees		Latitude, degrees	
	West	East	North	South
1	-125.274529	-118.634037	49.66092	45.14402
2	-126.118507	-121.051106	46.59432	41.63202
3	-125.110557	-121.036959	42.36827	37.2476
4	-122.777682	-115.735672	39.03779	31.62669
5	-123.051273	-118.480769	41.04824	34.4985
6	-122.184297	-117.601669	41.35061	34.52835
7	-124.458076	-119.43724	45.86137	40.46927
12	-121.009858	-113.825176	42.42003	36.26734

B.12.4 Land Cover Patterns and Watershed Characteristics

Land cover data from the 1992 and 2001 MRLC NLCD data and from the NOAA 1995 and 2001 data were used to calculate the area and percentage of the watershed for each of the 21 (NLCD) or 22 (NOAA) land cover classes. Accuracy of the 1992 NLCD data by EPA region is presented at http://landcover.usgs.gov/accuracy/index.php. Based on this analysis, users were cautioned about applying the data to highly localized studies, such as over a small watershed. Accuracy of the 1992 NLCD can be found at http://www.epa.gov/mrlc/accuracy.html while discussion of the 2001 NLCD data accuracy can be found at http://www.epa.gov/mrlc/accuracy-2001.html. The overall accuracy by MRLC zone ranged from about 86% to 98% (Table B-11)

These datasets were used to generate estimates of the area of impervious surfaces in each watershed using default coefficients from the Analytical Tools Interface for Landscape Assessments (ATtILA) software (U.S. EPA, 2004c). The MRLC 2001 impervious surface layer, which represents an estimate of developed impervious surface per pixel by percent imperviousness, was clipped and summarized for each watershed. Overall accuracies for the impervious surfaces from the 2001 MRLC data range from 83 to 91 percent (Homer et al., 2004; Yang et al., 2003), and represent a higher resolution estimate of impervious surfaces than available from ATtILA. Estimates of nitrogen and phosphorus loadings from land cover were calculated from the watershed land cover data using coefficients from the ATtILA program.

Estimates of slope were calculated for each watershed from slope surfaces generated from 10-meter (Oregon) digital elevation models (DEMS). Mean slope by percent and by degrees for land surfaces above the mean high water level were calculated and all slope values were exported to an Access database. The 30 meter DEMS were obtained from the National Elevation Dataset (NED, http://ned.usgs.gov), a seamless mosaic of the best elevation data. The 10-meter elevation data for Oregon was obtained by the USDA Forest Service for the Coastal Landscape Analysis and Modeling Study project (CLAMS, http://www.fsl.orst.edu/clams) from USGS drainage enforced digital elevation models.

Table B-11. Overall accuracy of NLCD 2001 land cover analysis. Zones refer to the MRLC zones containing Pacific Coast estuarine watersheds.

MRLC Zone	Overall Accuracy (%)
1	86.1
2	86.1
3	88.0
4	88.0
5	85.2
6	88.3
7	98.9
12	84.2

B.12.5 Population Density

Human population estimates from the 1990 and 2000 censuses (http://www.census.gov/) were generated for each drainage unit. Area weighted estimates of total population by census block were summed for each drainage and population density (individuals km^{-2}) was calculated from the total drainage population estimate.

B.12.6 Climate

Estimates of air temperature and precipitation were made for each watershed from several sources of long-term climate data. Annual mean temperature and mean precipitation were obtained from the 1 kilometer resolution climate summaries from 1980 to 1997 from the Daily Surface Weather and Climatological Summaries (DAYMET; http://www.daymet.org). Annual mean, annual mean minimum, annual mean maximum temperature and mean annual precipitation were also obtained from the 800 meter resolution PRISM climate model for the period 1971-2000 (http://www.climatesource.com). Comparisons were made between the two climate datasets and diverging values were examined.

B.12.7 National Wetland Inventory

As part of the development of the sampling frame for the Western Coastal EMAP 2002 survey, the available National Wetlands Inventory digital layers for coastal estuaries in Oregon, Washington and northern California were collected by EPA's onsite GIS staff in 2001. Some significant areas of Oregon estuaries were not included in the existing NWI digital dataset and paper maps were obtained for these areas and scanned. The scanned maps were registered and were used for "heads up" digitizing of the missing features.

In July 2005, all of the available NWI datasets for the Pacific coast were retrieved from the National Wetlands Inventory website. Digital maps were downloaded from the NWI website, mostly individual ESRI shapefiles of 1:24,000 scale quadrangle maps. The zipped versions of the shapefiles were created by NWI in the spring of 2003, according to the file creation dates. In some cases where shapefiles for specific quadrangles were missing, older ArcInfo coverages were available and retrieved from the NWI download website. All of these datasets are what the current NWI mapping team calls the "static" data. Where the circa "2005" data differed from the final version of the EPA NWI aggregation, the most recent data from the NWI website superseded the previous data for inclusion in the estuary geospatial layer. The NWI targeted mapping unit for the scale of photography most often used in the Pacific estuary classification varied from 1 to 3 acres.

Currently the NWI Mapping project is transitioning to the ESRI geodatabase format with wetlands data being available in larger seamless blocks (1:100,000 scale) or as selected views from an online mapping tool. The latest data for the Pacific Northwest (as of mid August 2006) shows some alterations to the previous wetlands in portions of the Oregon coast. These changes were captured in a revision of the Oregon estuary watershed geospatial layer in December 2007.

Descriptions for parsing the NWI attribute field codes to a habitat type were obtained from the NWI documentation and from queries to the online Map Codes Search form (http://www.fws.gov/wetlands/Data/webatx/atx.html). Some of the polygons in the NWI data selected for inclusion in the estuary classification spatial layer have codes that are considered obsolete under the current system. These will be kept in the database until revised geospatial data are available from the NWI. The coding used in the wetlands classification system used is based upon the Cowardin et al. (1979).

B.12.8 Estuarine Characteristics

Strahler stream order, which represents a hierarchical view of stream complexity based on the number of tributary junctions in a stream system from headwaters to mouth, was recorded for the largest order stream entering an estuary. A summarization of all the stream orders entering each estuary has been completed for the Pacific Northwest and will be completed for all the Pacific Coast estuaries. Order was recorded for the streams included in the National Hydrological Dataset (NHD; http://nhd.usgs.gov/), which corresponds to streams on the 1:100,000K map series and often is complete with streams mapped at the 1:24,000 level. Mouth width for each estuary was estimated using aerial photography from Terraserver (http://www.terraserver.com) and other imagery sources.

B.13 Other Data Sources

Historical salinity and nutrient data were obtained from the Oregon Department of Environmental Quality web-based database. Data from this site were previously reviewed internally by DEQ and assigned a data quality grade. Only data with grades of A and A+ were used as part of this report.

B.14 Hydrodynamic and Nutrient Source Model

All field and laboratory data used in the development of hydrodynamic and nutrient source model were collected in accordance with WED's SOPs and QAPPs. A two-dimensional, laterally averaged hydrodynamic and water quality model (Cole and Wells, 2000) was used to simulate the transport of riverine, oceanic and wastewater treatment facility (WWTF) effluent dissolved inorganic nitrogen (DIN) sources. This model is well suited for long-narrow estuaries, such as Yaquina Bay, where there are minimal lateral variations in water column properties. U.S. EPA (2001) suggested that this model may be useful in the estuarine nutrient criteria development and has been used in developing estuarine Total Maximum Daily Loads (TMDLS).

In the model simulations presented in this study, Yaquina Estuary was represented by 325 longitudinal segments spaced approximately 100-m apart with each longitudinal segment having 1-m vertical layers. The model domain extended about 37 km from the tidal fresh portion of the estuary at Elk City, Oregon to the mouth of the estuary. Model simulations were performed for the interval January 1 to October 1 of 2003 and 2004 and included tidal and wind forcing as well as freshwater inflow. Parameters simulated included water surface elevation, salinity, water temperature, and DIN.

Model calibration is the process of determining model parameters that are appropriate for the specific study location and time interval being simulated. The model used in this study was calibrated through adjustment of friction coefficient, eddy viscosity, and eddy diffusivity. To assess the model performance at simulating the hydrodynamics, we compared simulated and observed water level variations at two locations in the estuary and salinity and water temperature at four locations utilizing data from the YSI datasondes. Since the datasondes used at these stations were not leveled in we could only compare relative water level fluctuations, not absolute water level (referenced to MLLW). In addition, temperature and salinity from the CTD cruises were compared to simulated values. The model was assessed by calculating the root mean square error between observed and predicted variables.

Each nitrogen source, riverine (N_{river}), oceanic (N_{ocean}), and WWTF effluent (N_{wwtf}), was modeled as a separate component. The nitrogen sources were modeled as

$$\frac{dN}{dt} = transport - \mu N$$

where N is the DIN source and μ is a loss/uptake rate. The same value of μ was used for all three nitrogen sources and the value of μ was determined by fitting total modeled DIN ($N_{ocean}+N_{river}+N_{wwtf}$) to observations of DIN within the estuary. The best fit to observations was found with $\mu = 0.1\ d^{-1}$. Simulations were also performed with no uptake ($\mu = 0$) which is equivalent to conservative transport of the sources. The results from the transport model were used to mix the three nitrogen sources using the following equation

$$\delta_M = f_R\delta_R + f_O\delta_O + f_W\delta_W$$
$$f_R + f_W + f_O = 1$$

where f_R, f_W, and f_O are the fractions of riverine, wastewater treatment facility, and oceanic DIN, respectively, and δ_R, δ_W, and δ_R are the isotopic end members for riverine, wastewater treatment facility effluent, and oceanic sources, respectively. Estimates of the oceanic and riverine end members were obtained by examination of the observed isotope ratios at the stations located near the mouth of the estuary and in the riverine portion of the estuary and comparison to the literature. The initial estimate for the WWTF end member (δ_W = 15-22‰) was determined from the literature. To arrive at the final end member isotope ratios, model simulations were performed varying each end member over the range estimated from the data and literature. The final isotope ratio of end members for the three sources (δ_R=2‰, δ_W=20‰, and δ_O=8.4‰) was determined from the best fit (minimum root mean square error, RMSE) between predicted and observed isotope ratio at the five isotope sampling stations during 2003 and 2004. The final oceanic end member selected is consistent with marine end members for the west coast of the United States (Fry et al., 2001). While the riverine end member is consistent with the isotope ratio expected for nitrogen associated with red alder (leaf tissue ranges between -3 and -0.5‰; Hobbie et al., 2000; Tjepkema et al., 2000; Cloern et al., 2002).

B.15 Quality Assurance Project and Quality Management Plans (QAPPs) Used in This Study

QAPP 2000.01. Changes in the Abundance and Distribution of Estuarine Keystone Species in Response to Multiple Abiotic Stressors. T.H. DeWitt, EPA, 2000.

QAPP 01.02. Modeling of Landscape Change Effects on Estuarine Trophodynamics: an Optimization Approach Using Inverse and Forward Modeling. P. Eldridge, EPA, 2001.

QAPP 01.04. Assessment of the Spatial and Temporal Distribution of Submersed Aquatic Vegetation and Benthic Amphipods within the Intertidal Zone of Yaquina Bay Estuary, Oregon via Color Infrared Aerial Photography. D.R. Young, EPA, 2001.

QAPP 01.06. Upper Margin Expansion: Influences on Seagrass, *Zostera marina* L. B.L. Boese, EPA, 2001.

QAPP 02.01. Autecological studies of marine macrophytes including the sea grasses *Zostera marina* and *Z. japonica* in Yaquina Bay, Oregon. J. Kaldy, EPA, 2002.

QAPP 04.01. Seagrass Research - Epigrowth Light Attenuation Task: Estimation of spatial and temporal variation in light attenuation due to epigrowth on *Zostera marina* in Yaquina Bay. W. Nelson, EPA, 2004.

QAPP-06.01. Development of the Pacific Coast Ecosystem Information System (PCEIS). H. Lee II and D. Reusser, 2006.

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B.16 Standard Operating Procedures (SOPs) Used in This Study

CL -6. V.2. Standard Operating Procedures for Stable Isotope Ratio Mass Spectrometer Analysis of Organic Material. EPA. 1999.

GPEP SOP 3.01 **–** TERA SOP. Carbon/Nitrogen Elemental Analysis. Rick King, Dynamac Corporation. 1989.

MES EP01.rev 0. Draft. Standard Operating Procedure for Collecting and Processing *Zostera marina* and Associated Epiphytes for Light Attenuation Measurements. Dynamac Corporation. 2004.

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MES SOP02.rev 0. Standard Operating Procedures for Weighing Food Web Samples and Submitting them to ISIRF for Stable Isotope Analysis. K. Rodecap, Dynamac Corporation. 2002.

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SOPCMT.02, Sampling and Preparation for TOC Analysis, Sercy, Kathleen, et al, US EPA, Western Ecology Division. 1991.

SOPIOP.09. Operating Procedure For YSI Series 6 Multiparameter Water Quality Meters, Model #s 6000UPG and 6600, 6600EDS. D.T. Specht. EPA. 2004.

SOPFSP.01. Use of The Seabird Seacat (SBE-19) CTD Package. R. J. Ozretich. EPA. 1999.

WRS 14B.2. Standard Operating Procedure for the Determination of Total Suspended Solids (Non- Filterable Residue). Dynamac Corporation. 2005.

SOPPMP.04. Measurement of sediment grain size distribution using a laser diffraction particle size analyzer. James H. Power. EPA. 2007.

SOP-NHEERL/WED/PCEB/PC/2006-01-r1. Standard Operating Procedures for producing digital aerial photomaps of estuarine intertidal ecosystems using color infrared film, classifying eelgrass and non-vegetated habitats, and assessing the accuracy of the classifications. Clinton, P.J. and D.R. Young. EPA. 2006.