

An Approach to Developing Nutrient Criteria for Pacific Northwest Estuaries: A CASE STUDY OF YAQUINA ESTUARY, OREGON



SCIENCE

An Approach to Developing Nutrient Criteria for Pacific Northwest Estuaries:
A Case Study of Yaquina Estuary, Oregon

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Preface

Disclaimer

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Executive Summary

A proposed approach that could be used by the State to develop nutrient criteria for the Yaquina Estuary, Oregon is presented. The approach is based on a synthesis of research results derived from field sampling at multiple temporal and spatial scales, assembling data to construct historical trends in water quality parameters, and a variety of modeling approaches.

Yaquina Estuary is a small, drowned, river valley estuary located along the central Oregon coast. Approximately 48% of the estuarine area is intertidal. The designated uses within the Yaquina Estuary and River include aquatic life harvesting (shellfish growing and fishing), agricultural (livestock watering), municipal (public water supply), recreation (water contact recreation), ecological (resident fish and aquatic life, salmonid spawning and rearing, anadromous fish passage) and aesthetics.

Spatial and temporal variability in water quality indicators were assessed for multiple water quality parameters, including nitrogen, phosphorus, chlorophyll *a*, dissolved oxygen, total suspended solids, and water column light attenuation. Spatial scales examined included variation within the Yaquina Estuary, as well as comparison of some parameters to short term studies of six additional Oregon estuaries, and comparison to a single sampling of 14 additional Oregon estuaries conducted by the US EPA National Coastal Assessment program. Green macroalgal occurrence was evaluated to determine whether this was an appropriate indicator for nutrient responses within the Yaquina Estuary. Lower depth limits for the seagrass (*Zostera marina*) were determined in order to estimate the minimum light requirements for sustaining seagrass. Field results were used to confirm output from a Seagrass Stressor-Response Model.

Because there were limited data for applying the reference condition approach for the class of estuaries similar to the Yaquina Estuary, we used *in situ* observations within Yaquina Estuary as a basis for determining an Estuarine Reference Condition. Cumulative distribution functions (CDFs) were produced for water quality variables for the Yaquina Estuary and compared to CDFs for other Oregon estuaries using two independent data sets. Key percentiles (25th, 50th, 75th) for water quality parameters were used as inputs to a Seagrass Stressor-Response Model to determine whether particular percentile values would be adequately protective of seagrass within the Yaquina Estuary.

The Yaquina Estuary has strong seasonal variation in the magnitude of nutrient loading and in the dominant nutrient sources. Response variables (particularly, chlorophyll *a* and dissolved oxygen) exhibited similar patterns of seasonal variation. During the wet season (November-April), riverine nitrogen inputs dominate, whereas during the dry season (May – October) oceanic nitrogen sources dominate. Riverine inputs are primarily related to the presence of nitrogen-fixing red alder (*Alnus rubra*) trees in the watershed. There are also strong zonal differences in nutrient levels, response variables, and dominant nutrient sources within the Yaquina Estuary. In the lower estuary (Zone 1), water quality conditions are strongly influenced by ocean conditions, while in the upper portions of the estuary (Zone 2), watershed and point source inputs increase in importance.

We suggest that criteria be developed for wet and dry seasons to address the strong seasonal variation in nutrient loads and sources. Dry season criteria (May-October) are most important, since during the wet season, there appears to be little utilization of nutrients within the estuary, and chlorophyll *a* levels are low and the dissolved oxygen concentrations are high. Thus, it is not clear that wet season criteria are needed within the Yaquina Estuary. We suggest that separate criteria be developed for Zones 1 and 2, with dry season criteria for Zone 2 a first priority. The high degree of ocean-estuary coupling found for Zone 1 within the Yaquina Estuary with associated short-term variability in water quality parameters suggests that monitoring for compliance with nutrient criteria in this region may be problematic. During the dry season, phosphate, nitrate, chlorophyll *a*, and dissolved oxygen levels in Zone 1 are primarily determined by ocean conditions and separation of oceanic from anthropogenic inputs would require, at the least, continuous monitoring capability, and may require additional techniques.

Use of Total Suspended Solids (TSS) as a water quality criterion may not be practical due to inconsistent spatial and temporal patterns relative to that of adjacent Oregon estuaries. Macroalgal biomass response within Yaquina Estuary appears to be primarily driven by oceanic nitrogen input, and thus does not appear to be useful as an indicator of cultural eutrophication.

Based on weight of scientific evidence, we conclude that Yaquina Estuary is not exhibiting symptoms of cultural eutrophication. Thus, following the recommendations in U.S. EPA (2001), median values could be used as criteria for most water quality parameters. The Seagrass Stress-Response Model confirmed that the median percentile for water clarity (k_d) would be protective of the existing eelgrass (*Zostera marina*) habitat in the Yaquina Estuary.

Modeling results were consistent with analysis of seagrass depth limits which indicated that the median k_d provided for persistence of seagrass at depths within the two estuarine zones that were comparable to current depth distributions.

The current Oregon DO criterion of 6.5 mg l^{-1} should be adequately protective of estuarine resources, but is closer to the 25th percentile value rather than the median value for DO data in Zone 2. Recent DO measurements demonstrate that hypoxic water is imported into the estuary from the coastal shelf during the dry season. As a result, exceedances of the DO criterion should be expected particularly in Zone 1. The current Oregon chlorophyll *a* criterion of $15 \text{ } \mu\text{g l}^{-1}$ is approximately 3 times greater than the median value for Zone 2. The chlorophyll *a* criterion is determined as a 3-month average, and if chlorophyll *a* levels were to approach the present criterion for such a time period, significant trophic shifts in the estuary would be likely. Thus, the current chlorophyll *a* criterion may not prevent some impacts on designated use.

Potential dry season criteria for the Yaquina Estuary based on median values for all parameters except for DO.		
Parameter (units)	Zone 1	Zone 2
DIN (μM)	14	14
Phosphate (μM)	1.3	0.6
Chlorophyll <i>a</i> ($\mu\text{g l}^{-1}$)	3	5
Water Clarity (m^{-1})	0.8	1.5
Dissolved Oxygen (mg l^{-1})	6.5	

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1. Introduction

1.1 Purpose of This Case Study

The Office of Science and Technology (OST), Office of Water, U.S. EPA provides guidance to the States and tribes for developing nutrient criteria for estuarine and coastal waters. The Office of Research and Development, National Health and Environmental Effects Laboratory (NHEERL) has been conducting research to support improvements to the scientific basis for estuarine nutrient criteria for over 5 years under the NHEERL Aquatic Stressors Research Program. Parallel research efforts have been on going at the Western (WED), Gulf (GED) and Atlantic Ecology Divisions (AED). To support the OST criteria effort, NHEERL scientists have synthesized the research results of field sampling, trend analyses, and modeling approaches to produce nutrient criteria case studies for Yaquina Estuary, OR and Pensacola Bay, FL. Each case study describes one or more approaches that may be used for establishing nutrient criteria and offers specific recommendations for the particular system. Here we describe a recommended approach for developing nutrient criteria values for the Yaquina Estuary.

1.2 Nutrient Criteria Objective

The Nutrient Criteria Technical Guidance Manual: Estuarine and Coastal Marine Waters (U.S. EPA, 2001) provides a detailed summary description of the nutrient criteria development process in Section 1.4 of the manual. This guidance defines two objectives for establishment of numeric nutrient criteria:

To reduce the anthropogenic component of nutrient overenrichment to levels that restore beneficial uses (i.e. described as designated uses by the CWA), or to prevent nutrient pollution in the first place.

Quantitative, long term data on the status of eutrophication in most Oregon estuarine systems is limited (Bricker et al., 1999). The EPA National Coastal Assessment (U.S. EPA, 2004a) sampled Oregon estuaries for a variety of water quality indicators in 1999-2000, and concluded that there was little evidence of eutrophication effects in Oregon estuaries. Additional qualitative and quantitative assessments of Oregon estuaries by WED generally support the conclusions of the NCA report, but also suggest that in limited regions under certain circumstances, water quality problems may arise. Thus, the principle objective in developing nutrient criteria for the

Yaquina Estuary is to prevent future degradation of estuarine water quality and accompanying loss of beneficial uses from the system.

1.3 Designated Uses of Yaquina River and Estuary, Impairments and Assessments

The Yaquina River and Estuary have many designated uses, including aquatic life harvesting (shellfish growing and fishing), agricultural (livestock watering), municipal (public water supply), recreation (water contact recreation), ecological (resident fish and aquatic life, salmonid spawning and rearing, anadromous fish passage) and aesthetics (Table 1.1). Causes for impairment listings in the Yaquina Estuary and River include pathogens, thermal modifications, diminished biologic integrity, and organic enrichment/low dissolved oxygen (Table 1.1). Most of the impairments occur in the Yaquina River, with the exception of fecal coliform impairment which occurs in the lower portion of the estuary.

The Oregon Department of Environmental Quality (ODEQ) assessed the water quality in the Oregon Mid Coast Basin, which includes the Yaquina River during 1986-1995 (Cude, 1995). The following is an excerpt from this assessment.

Nitrate nitrogen is the primary limiting factor on water quality throughout the Mid Coast basin. High levels of nitrates accompanied by increases in total phosphates, total solids, and biochemical oxygen demand, appear during periods of heavy precipitation.

Nutrient-rich erosion products deposited during storm events place a high demand on available dissolved oxygen in the water. These products may be naturally occurring, but are more likely the result of non-point source pollution.

As part of this ODEQ study, an Oregon Water Quality Index (OWQI) that incorporates temperature, dissolved oxygen, biochemical oxygen demand, pH, total solids, ammonia and nitrate nitrogen, total phosphorous and fecal coliforms was developed. Based on this index, water quality for the Yaquina River (at Rivermile 24.9) was categorized as poor during the fall, winter, and spring, and good during the summer. The water quality in the Yaquina estuary was reassessed in 2006 using data from water years 1996-2005 (Mrazik, 2006). In this more recent assessment, the Yaquina River was assessed as having good condition throughout the year.

In the National Estuarine Eutrophication Assessment (Bricker et al., 1999), the Yaquina Estuary was placed in the low category of eutrophication status based on a qualitative assessment that it exhibited few symptoms of eutrophication. However, conditions were expected to worsen

by 2020, primarily as a result of increasing population pressures. The confidence levels for the assessment of the eutrophic conditions in the Oregon region were low due to paucity of data (Bricker et al., 1999).

Table 1.1 Designated uses and water quality attainments for Yaquina Estuary/River. (Source: U.S. EPA National Assessment Database, 305(b) Lists/Assessment Unit Information Year 2002; http://www.epa.gov/waters/305b/index.html)				
Rivermile	State Designated Use	Attainment Status	Threatened	Basis of Impairment Classification
0 – 6.3	Shellfish Growing	Not Supporting	No	Fecal Coliform
5.1-15.4	Shellfish Growing	Not Supporting	No	Fecal Coliform
6.3-14.2	Shellfish Growing	Fully Supporting	No	NA
15.4 – 27.6	Resident Fish and Aquatic Life	Fully Supporting	No	NA
	Salmonid Fish Spawning			
	Water Contact Recreation			
	Salmonid Fish Rearing	Not Supporting	No	Thermal Modifications
	Anadromous Fish Passage			
27.6 – 42	Resident Fish and Aquatic Life	Partial Supporting	No	Biologic Integrity ¹
27.6–57.5	Aesthetics	Fully Supporting	No	NA
	Fishing			
	Livestock Watering			
	Resident Fish and Aquatic Life			
	Water Contact Recreation			
	Water Supply			
	Salmonid Fish Rearing	Not Supporting	No	Organic Enrichment/Low Dissolved Oxygen
	Salmonid Fish Spawning			
	Anadromous Fish Passage			

¹Aquatic communities (primarily macroinvertebrates) which are ≤60% of the expected reference community for multimetric and multivariate model scores are considered impaired.

1.4 Oregon Estuarine Water Quality Criteria

Water quality criteria standards are developed to protect beneficial uses. The State of Oregon presently has numeric water quality criteria for chlorophyll *a* and dissolved oxygen for estuarine waters (Table 1.2). The dissolved oxygen (DO) criterion for estuarine waters is primarily based on the freshwater literature and on salmon and trout requirements (ODEQ, 1995). This DO criterion is relatively high compared to DO criterion for other estuaries, such as the Chesapeake Bay (U.S. EPA, 2003). In addition to the criteria in Table 1.2, the state wide narrative criteria states that:

“where a less stringent natural condition of a water of the State exceeds the numeric criteria” ... “the natural condition supersedes the numeric criteria and becomes the standard for that water body.”

In the narrative criteria, the natural condition refers to non-anthropogenic conditions. A review of the DO criterion (ODEQ, 1995) found that the 6.5 mg l⁻¹ may be difficult to achieve in Oregon estuaries during the summer due to natural background conditions. If it is not achievable due to natural background conditions, then the background conditions become the criteria.

Table 1.2 Selected water quality criteria for Oregon estuaries.		
Parameter	Estuarine Criterion	Water Quality Limited Determination
Chlorophyll <i>a</i>	15 µg l ⁻¹	Average based on minimum of 3 samples collected over any 3 consecutive months at a minimum of one representative location exceeds criterion ¹
Dissolved Oxygen	6.5 mg l ⁻¹	Greater than 10% of samples exceed the criterion and a minimum of at least 2 exceedances of the criterion for the time period of interest. A minimum of 5 representative data points per site collected on separate days per applicable time period. Daily means of continuous data represents 1 data point. ²
Note: ¹ Criterion applies to river and estuaries; ² Estuarine waters defined as those with conductivity > 200 µS cm ⁻¹ for dissolved oxygen criterion. Other dissolved oxygen criterion applies to freshwater region.		

1.5 Summary of Yaquina Case Study Approach

In Chapter 2, we provide a description of the watershed and estuary, including a description of landuse in the watershed, and a brief history of anthropogenic activities in the estuary and watershed. Chapter 3 presents a summary of the nitrogen inputs to Yaquina Estuary with discussion of the seasonality and magnitude of natural and anthropogenic sources. A

summary of data used in this report and analysis techniques are presented in Chapter 4. Chapter 5 provides information on spatial and temporal patterns in water quality data, including important factors influencing water quality distributions. Seasonal, zonal, and long-term trends in causal (nitrogen and phosphorous) and response (chlorophyll *a*, dissolved oxygen, and water clarity) variables are presented in Chapters 6-9. These chapters also include cumulative distribution functions for causal and response variables and comparison of water quality conditions in Yaquina Estuary to those in other Oregon estuaries. In addition, comparisons of observations from Yaquina Estuary to existing State of Oregon chlorophyll *a* and dissolved oxygen criteria are summarized in Chapters 7 and 8. Chapter 10 examines the usefulness of macroalgal biomass as a response variable for the Yaquina Estuary, including descriptions of seasonal, interannual, and zonal patterns in macroalgal biomass and factors which influence its distribution. Chapter 11 provides a description of the distribution, variability, and factors influencing *Zostera marina* habitat in the estuary as well as light requirements for this species. Chapter 12 provides a demonstration of using a mechanistic stress-response model for *Z. marina* to assess whether specific water clarity percentiles are protective of existing habitat. A summary of the results of this study and recommendations are provided in Chapter 13.

2. Description of Study Area

2.1 Physical Characteristics of the Estuary

Yaquina Estuary is a small, drowned, river valley estuary located along the central Oregon coast (latitude = 44.62°N, longitude = 124.02° W) of the United States (Figure 2.1) with an estuarine surface area of 19 km² and a watershed area of 650 km² (Figure 2.2; Lee et al., 2006). Approximately 48% of the estuarine area is intertidal. This estuary experiences mixed semidiurnal tides and is mesotidal with a mean tidal range of approximately 1.9 m and a tidal prism volume of 2.4 x 10⁷ m³ (Shirzad et al., 1988). Yaquina Estuary has jetties that extend into the Pacific to the 10-m depth contour. Due to the small volume of the estuary (25 x 10⁶ m³ at Mean Lower Low Water (MLLW)) and the strong tidal forcing, there is close coupling between the estuary and the coastal ocean. Approximately 70% of the volume of the estuary is exchanged with the coastal ocean during each tidal cycle (Karentz and McIntire, 1977).

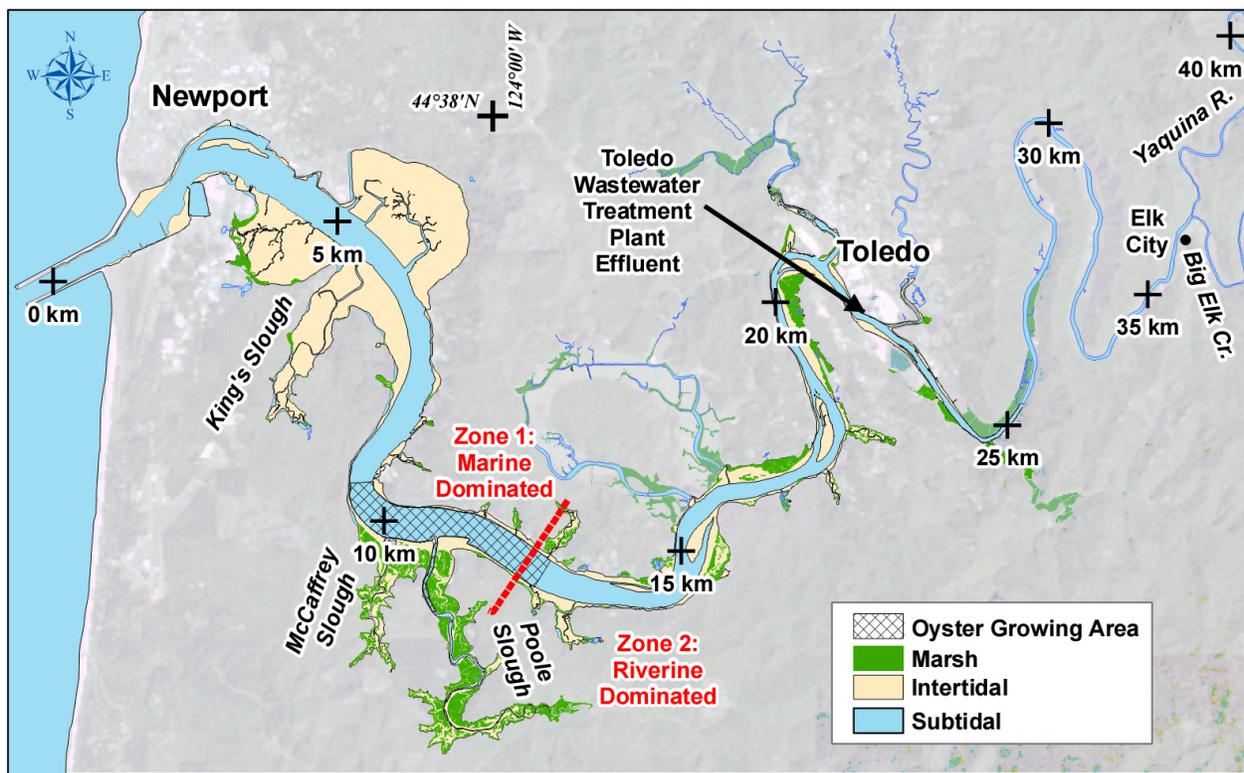


Figure 2.1. Location map of Yaquina Estuary. The estuary is divided into “marine dominated” (Zone 1) and “riverine dominated” (Zone 2) segments (Lee et al., 2006) based on the relative proportion of oceanic-derived nutrients versus terrestrially-derived nutrients.

Yaquina Estuary receives freshwater inflow primarily from two tributaries, the Yaquina River and Big Elk Creek, which have similarly sized drainage areas and contribute approximately equally to freshwater inflow (Figure 2.2; State Water Resources Board, 1965). The long-term median freshwater input to Yaquina Estuary is $7.5 \text{ m}^3 \text{ s}^{-1}$. There is a strong seasonal pattern in freshwater input to the Yaquina Estuary (Figure 2.3). During the months of November through April, the Oregon coast receives high precipitation and the estuary is river dominated. Beginning in May and continuing through October, there is a decline in the riverine freshwater inflow and the estuary switches from riverine to marine dominance. For this document, we defined the wet season (November – April) as months when the median monthly discharge exceeds the long-term median annual discharge and the dry season (May- October) as months when the median monthly discharge is less than the long-term median. The estuary is well mixed under low flow conditions, and partially- to well- mixed during winter high inflow conditions (Burt and McAlister, 1959; Kulm and Byrne, 1966). The flushing time of the estuary during the dry season varies from 1 day near the mouth to 9 days in the upstream portions (Choi, 1975).

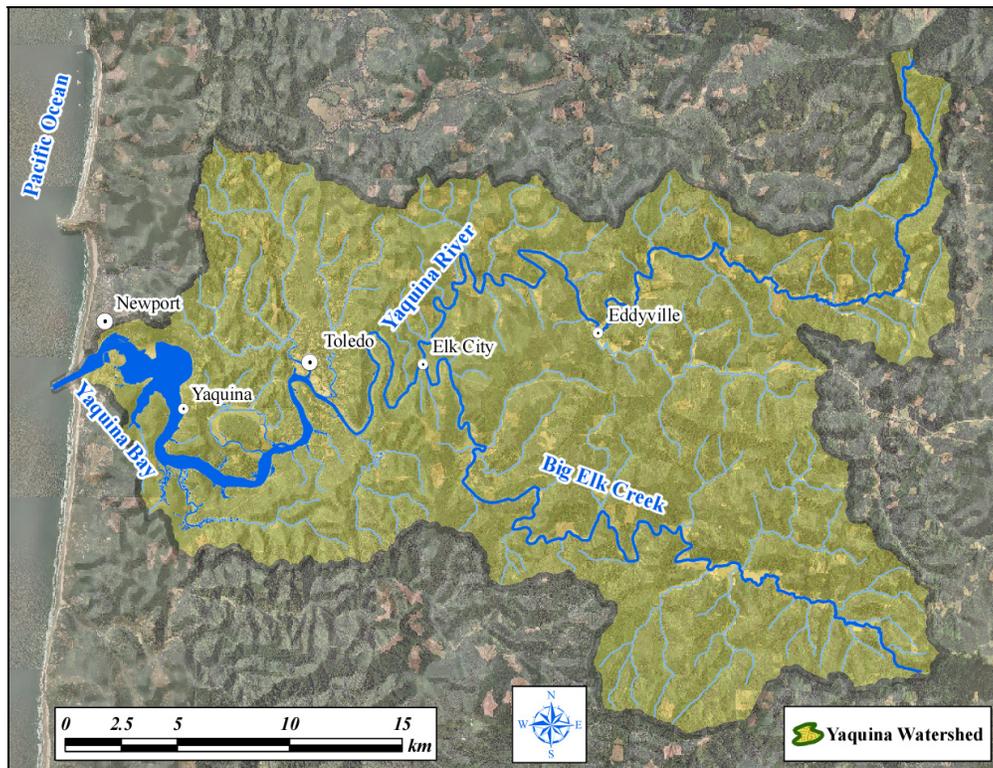


Figure 2.2 Map of watershed of the Yaquina Estuary, showing the two primary tributaries (Yaquina River and Big Elk Creek).

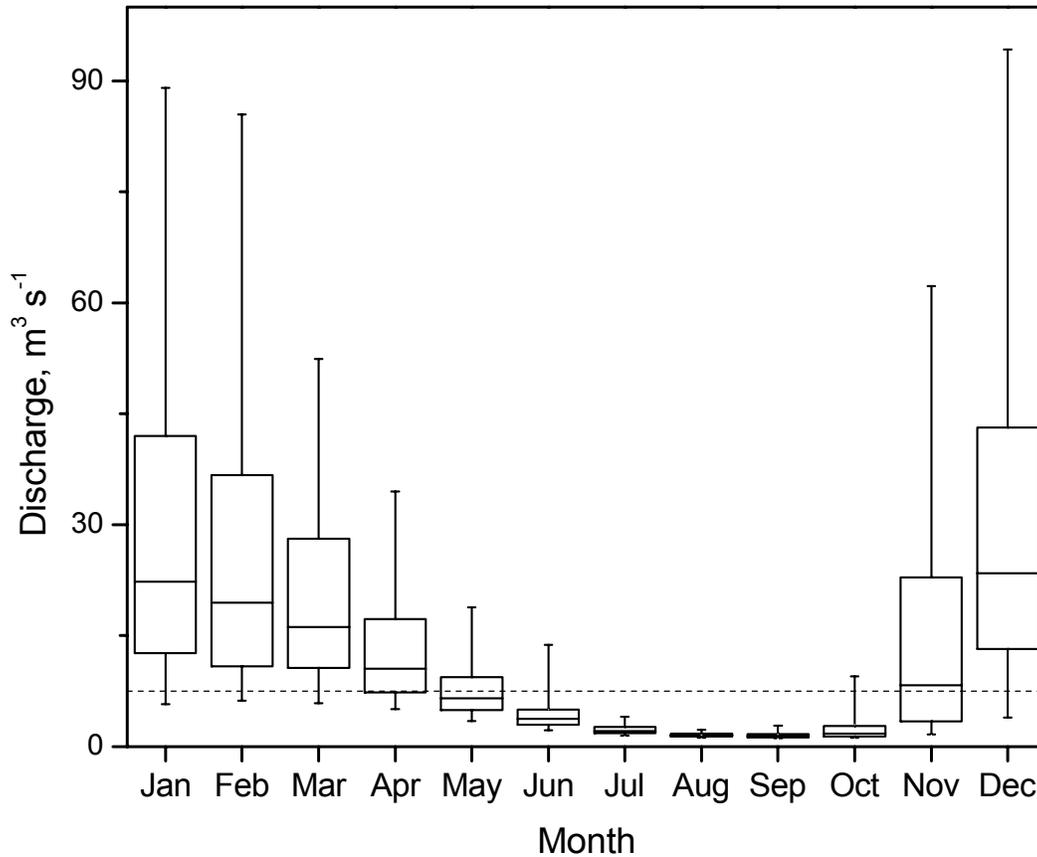


Figure 2.3 Monthly discharge statistics (Yaquina River + Big Elk Creek) calculated using data from 1972-2002 Chitwood gauge (corrected for Big Elk Creek using relationship from Brown and Ozretich, in review). In the plot, the boxes represent the 25th and 75th percentiles, the whiskers represent the 5th and 95th percentiles, and the horizontal line is the median. The dashed line indicates 30-year median discharge.

Estuaries in the Pacific Northwest (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 (NO_3^- and PO_4^{3-}) water to the surface. The upwelling season typically commences in April and continues through September, approximately coinciding with the dry season. 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 United States (e.g., de Angelis and Gordon, 1985; Roegner and Shanks, 2001; Roegner et al., 2002; Colbert and McManus, 2003; Brown and Ozretich, in review).

2.2 Biotic Characteristics

PNW estuaries, including Yaquina Estuary, are highly productive ecosystems, supporting several hundred species of macrophytes, macroinvertebrates, fish, birds, marine and terrestrial mammals. The Yaquina estuarine ecosystem contains six major habitats, three of which are defined by the presence of an ecosystem-engineering species: tidal channels (including water column and subtidal unvegetated sediments), eelgrass beds (lower intertidal and shallow subtidal sediments dominated by *Zostera marina*), mud shrimp beds (mid- to lower intertidal muddy sediments dominated by *Upogebia pugettensis*), ghost-shrimp beds (lower- to upper intertidal sediments dominated by *Neotrypaea californiensis*), unvegetated intertidal sediments, and tidal marshes. Each habitat supports different floral and faunal communities (Seliskar and Gallagher, 1983; Simenstad 1983; Phillips, 1984; Ferraro and Cole, 2006).

One hundred twenty-eight species of macroalgae (Kjeldsen, 1967) and three species of seagrass have been recorded in Yaquina Estuary. The species diversity and biomass of macroalgae is greatest near the mouth of the estuary and decreases up river (Kjeldsen, 1967). From late spring to early fall, green macroalgae (principally *Enteromorpha* spp. [6 spp.], *Ulva* spp. [6 spp.], and *Cheatomorpha* spp. [2 spp.]) form extensive intertidal and shallow subtidal mats in the lower portion of the estuary, but are largely absent upstream of Poole Slough (about 11 km from mouth; Figure 2.4). Above Toledo, macroalgae diversity declines to <5 spp. and biomass is negligible (Kjeldsen, 1967; WED unpublished data). Large meadows and long patches of the native seagrass, *Zostera marina* (eelgrass), occur on the intertidal flats and along channel edges in the lower estuary. From Poole Slough upriver to Toledo, eelgrass occurs sporadically in the shallow subtidal, with the largest patches occurring near the Toledo public boat launch, which is about 18 km from the mouth of the estuary (Figure 2.4). The introduced seagrass, *Z. japonica*, occurs in the upper-to-mid intertidal zone from the lower estuary to Toledo, occasionally forming large beds; its abundance is increasing and it may eventually compete for space with the native eelgrass. Widgeon grass (*Ruppia maritima*), the third seagrass

species, occurs in small, isolated patches, but is uncommon relative to the other seagrasses (Bayer, 1996).

Recent surveys identified over 168 species of macroinvertebrates in Yaquina Estuary, with diversity and biomass highest in the lower estuary and lowest in the upper estuary (WED unpublished data). Polychaetes are the most numerous macroinvertebrate taxa, but ghost and mud shrimp (*Neotrypaea californiensis* and *Upogebia pugettensis*) dominate the infaunal biomass (WED unpublished data). Bioturbation, bioirrigation, and feeding activities of these shrimps accelerate carbon and nutrient cycling within the estuary, and enhance the flux of dissolved nitrogen from sediments to the water column (DeWitt et al., 2004). Deposit and filter feeders are the most abundant benthic consumers, with filter feeders, primarily mud shrimp, dominating in the lower estuary. As mud shrimp abundance declines up-estuary, deposit feeders become more abundant.

Five species of bivalves (cockle [*Clinocardium nuttali*], soft-shell clam [*Mya arenaria*], littleneck clam [*Venerupis staminea*], gaper clam [*Tresus capax*], and butter clam [*Saxidomus giganteus*]) are harvested recreationally, primarily in the lower portions of Yaquina Estuary. Although commercial harvest of these species is currently allowed, there have been no significant landings since the mid-1990's. Prior to that time, commercial landings, varied between 1,000 and 8,000 lbs. per year (P.M. Vance, Oregon Department of Fish and Wildlife (ODFW), personal communication). Non-native Pacific oysters (*Crassostrea gigas*) are grown commercially on 519 acres of leased tidelands in the middle reach of Yaquina Estuary, near McCaffrey and Poole Sloughs, with an annual production of 15,028 bushels valued at \$594,000 (Oregon Department of Agriculture 2000-2005). This equates to 45% of the Oregon commercial oyster production on state-owned tidelands.

At least 62 species of finfish and epibenthic crustaceans occur in Yaquina Estuary, with the highest diversity and abundance found in the lower estuary, and reduced diversity and abundance upriver (DeBen et al., 1990). Fish and crustacean abundance and diversity is highest during summer and lowest in winter. Estuary-wide, English sole [*Parophrys vetulus*], Pacific snake blenny [*Lumpenus sagitta*], and shiner sea perch [*Cymatogaster aggregata*] are the three most abundant fishes, and sand shrimp [*Crangon* spp.], dungeness crabs [*Cancer magister*], and mysids [*Neomysis mercedis*] are the three most abundant epibenthic crustaceans (DeBen et al., 1990). Of these, dungeness crabs have the greatest economic value, supplying recruits to the

offshore commercial crab fishery (Armstrong et al., 2003) and adults to the within-estuary recreational fishery. Annually, approximately 75,000 dungeness crabs are harvested in the recreational fishery in Yaquina Estuary (P.M. Vance, ODFW, personal communication). Fifteen species of fish account for >90% of the fish caught recreationally for food or bait from Yaquina Estuary (PSMFC, 2006). Statewide, lower-estuary recreational finfish fishing contributes \$18.8 million to Oregon's economy (The Research Group 2005). Estuary-specific estimates for the recreational fishery's value are not available, but recreational salmon fishing in Newport-area estuaries (i.e., predominantly Yaquina Estuary) contributes \$4.05 million to the State economy (The Research Group, 2005). The Yaquina watershed and estuary support breeding populations of five salmonid species (chinook salmon [*Oncorhynchus tshawytscha*], coho salmon [*O. kisutch*], chum salmon [*O. keta*], steelhead [*O. mykiss*], and cutthroat trout [*O. clarki clarki*]), including the southern-most population of chum salmon in North America (Bob Buckman, ODFW, personal communication). Coho salmon are being considered for special conservation status because of reduced population size in Yaquina and other Oregon mid-coast estuaries (ODFW, 2006). The only commercial finfish fishery within the estuary is for Pacific herring (*Clupea pallasii pallasii*), whose ovaries and roe are marketed to Asia, with an annual average (1979-2006) landing of 153,300 lbs, valued at \$76,300 (Keith Matteson, ODFW, personal communication).

Thousands of birds live in or migrate through Yaquina Estuary, which is designated as a Continental Important Bird Area (IBA) by the American Bird Conservancy and as a State IBA by the National Audubon Society. Two hundred-sixteen species of birds have been observed during 1994-2006 Christmas Bird Count surveys (National Audubon Society, 2002). Sixty-seven species of waterbirds were censused during 1993-1994 in the estuary, of which 41 were year-round or seasonal residents; maximum diversity and abundance occurred in December, and was at minimum in June (Merrifield, 1998).

Small populations of Pacific harbor seals (*Phoca vitulina richardsi*) and California sealions (*Zalophus californianus*) are present year-round in Yaquina Estuary, feeding on fish and crabs in the lower estuary (Orr et al., 2004; Brown et al., 2005). Killer whales (*Orcinus orca*), gray whales (*Eschrichtius robustus*), and harbor porpoises (*Phocoena phocoena*) occasionally and briefly enter the lower estuary. Other common mammals in the tidal portions of Yaquina Estuary include river otter (*Lutra Canadensis*), raccoons (*Procyon lotor*), muskrat (*Ondatra*

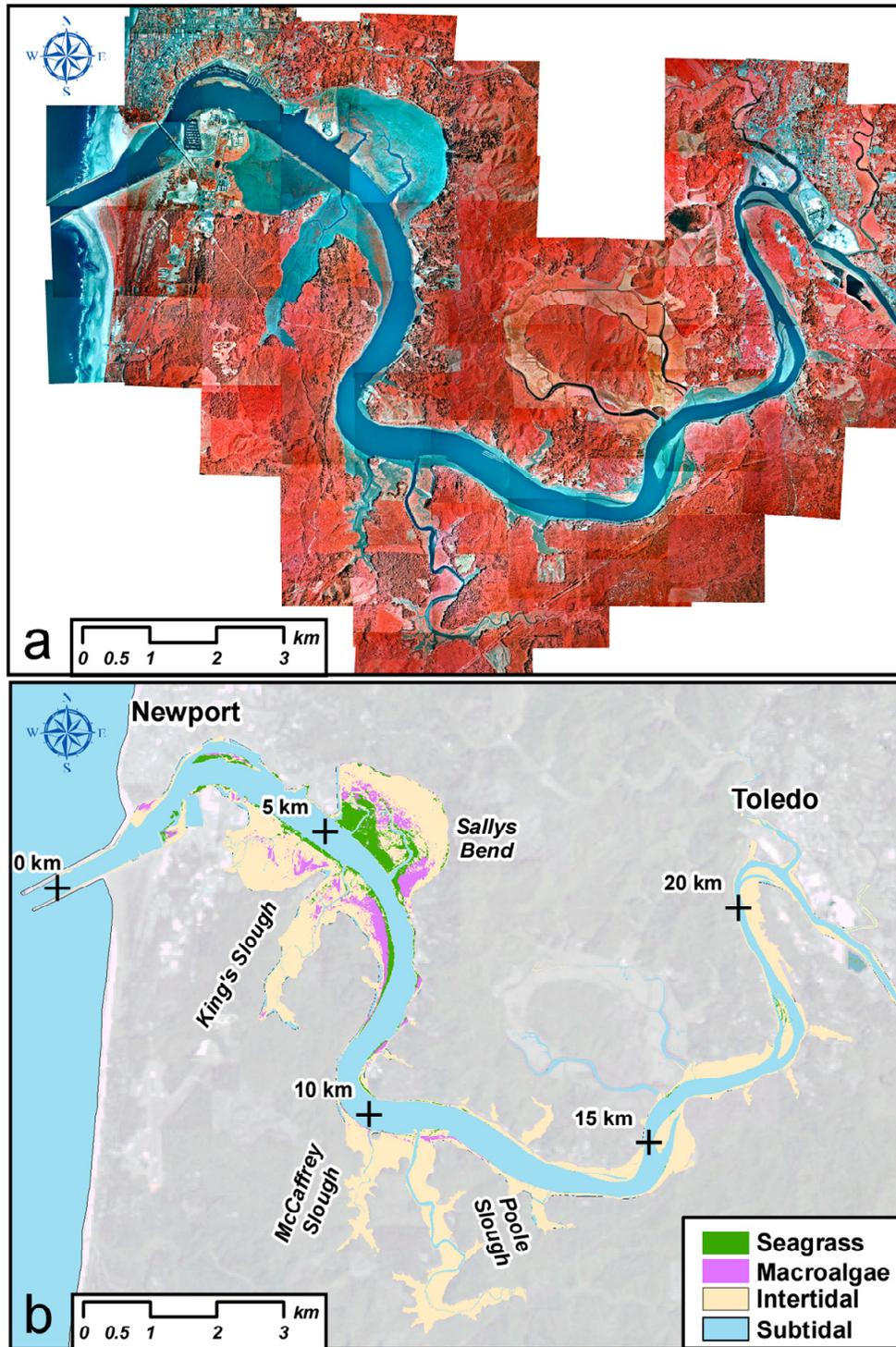


Figure 2.4 a) False-color, infrared aerial photography mosaic of Yaquina Estuary (taken in 1997) and b) map of intertidal seagrass and macroalgae in Yaquina Estuary classified from image analysis (WED unpublished imagery).

zibethicus), and nutria (*Myocastor coypus*), particularly in low-salinity tidal marshes (USFWS, 1968). River otter hunt for fish in tidal channels, raccoons forage for molluscs and crustaceans on tide flats, whereas muskrats and nutria feed on marsh plants (Howerton, 1984). Over 70 other mammals are reported from Lincoln County watersheds, many of which use wetland and terrestrial habitats bordering the estuary (National Wildlife Federation eNature ZipGuides website: <http://www.enature.com/zipguides/>).

2.3 *Land Use and History of Anthropogenic Modifications*

The Yaquina watershed covers an area of 650 km², tapering towards the mouth of the Yaquina Estuary but extending about 35 km inland. The watershed contains the city of Toledo, however, most of the city of Newport with the exception of the “Bay Front” lies outside of the watershed boundaries (Figure 2.2). The total population in the Yaquina watershed in 2000 was approximately 7970 or 12.3 persons per km² (source: Lee et al., 2006). The population density in the Yaquina watershed is similar to other PNW estuarine watersheds (mean = 15 persons per km², Lee et al., 2006), and is much lower than the national average for the coastal region of the United States (mean = 116 persons per km²; Crossett, et al., 2004). The population trend in the Yaquina watershed differs from many coastal watersheds in the United States in that the population in the Yaquina watershed declined by 4.8% from 1990 to 2000. Population changes during the interval of 1980 to 2000 in PNW coastal watersheds (excluding Puget Sound and Columbia River) are among some of the lowest in the United States (Crossett, et al., 2004). Utilization of the Yaquina Bay Front increases substantially with the influx of tourists during the summer.

Historical population data on a watershed basis are not available before 1990 because the census block data are not available in a GIS format to allow proration of the population by watershed boundaries. However, it is possible to track the historical population changes in the cities of Newport and Toledo (Figure 2.5). The major population center in the Yaquina watershed is the city of Toledo, which accounted for 44% of the population in the watershed in 2000. Toledo has experienced low growth, increasing by 12% from 1960 to 2005. In contrast, Newport has grown steadily, increasing by 84% from 1960 to 2005. While most of the city of Newport lies outside of the Yaquina watershed, the increase in population, as well as an increase

in tourism over this period, reflects an increase in utilization of the Yaquina Estuary through recreational boating, recreational fishing, and utilization of bay-side restaurants and facilities.

The Yaquina watershed is heavily forested with deciduous, evergreen and shrub land use classes constituting 85% of the watershed (Lee et al., 2006 based on NOAA 2001 C-CAP data (www.csc.noaa.gov/crs/lca/ccap.html)). Grasslands constitute 6% of the watershed while high and low intensity development combined only constitute 0.5% of the watershed. While developed areas constitute a small percentage of the watershed, they are increasing with the high residential and low residential land use classes increasing by 4.5% and 6.8%, respectively, from 1995 to 2001. Reflecting the low extent of development, the percent impervious surface is only 2.4% (Lee et al., 2006). As is typical of coastal watersheds in the PNW, the Yaquina watershed is “rugged” with a median slope of 29.7 percent (16.5 degrees).

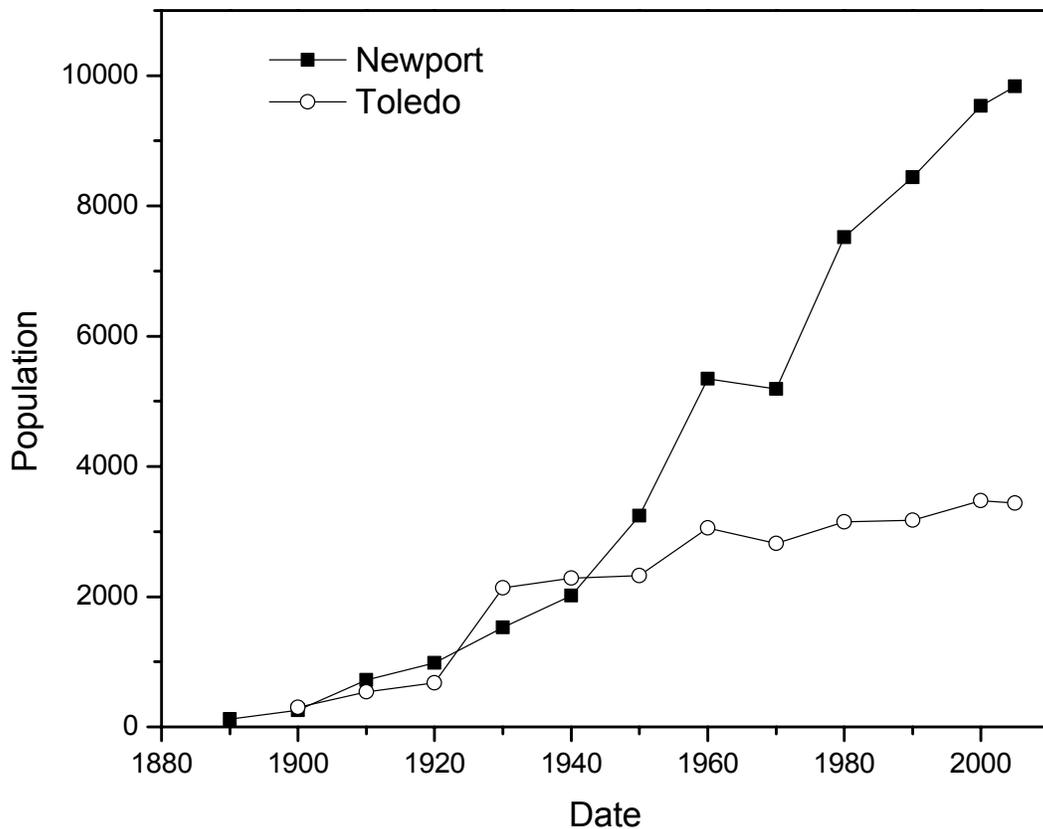


Figure 2.5 Populations in the cities of Toledo and Newport, Oregon from 1890 to 2005.

Although primarily forested and showing little “urban footprint”, the Yaquina watershed has been impacted by a variety of disturbances during the last century, in particular fires and logging. The largest fire occurred in 1853 when the “Yaquina Burn” consumed 1942 km² of coastal forest from near Corvallis to Yaquina Estuary. Logging of the coast range began in the mid-1800’s and extensive logging of Sitka spruce occurred along the coast during World War I, much of it centered near Toledo

(www.ohs.org/education/oregonhistory/narratives/subtopic.cfm?subtopic_ID=76). Logging has continued within the Yaquina watershed to the present, and forest lands are the dominant land use within the watershed, accounting for 566 km² (90%) of the land zoning area in the Yaquina Basin (area = 639 km²; Garono and Brophy, 2001). While the intensity of logging varies with economic trends and the age and marketability of the standing timber, it is not uncommon to see patches of clear cut forest within the Yaquina watershed.

In addition to the direct effects of logging on erosion and water quality, rafting of logs can potentially affect freshwater and estuarine habitats by physical disturbance, altering flow regimes, and accumulation of wood and bark debris which in turn can smother the benthos and result in low dissolved oxygen and/or elevated H₂S (Sedell et al., 1991). During the early 1900s until the 1980s, the estuaries and streams of the PNW were used for the transport and storage of logs (Sedell and Duval, 1985). Logs have been rafted in the Yaquina since at least 1920, with a substantial increase after the construction of the Georgia Pacific West mill in 1957 in Toledo (Figure 2.6). Peak abundance of rafted logs occurred in 1962, and log rafts declined through the early 1980s with the increase in environmental regulation and changes in markets (Figure 2.6); Sedell and Duval, 1985). In addition to the bark debris, accumulation of sawdust has also been observed in the estuary (Kulm and Byrne, 1966).

In addition to log rafting, three other sources of biological oxygen demand (BOD) in the Yaquina Estuary are sewage from municipal discharges, industrial discharges, and non-point inputs, in particular from septic systems. As was common for the period, untreated sewage and industrial waste from Toledo and the Newport bay front were discharged directly into the Yaquina Estuary in the 19th century and the first half of the 20th century. Sufficient untreated sewage and other wastes were discharged that they represented a potential health hazard for the oysters grown in the bay in the first quarter of the 20th century (Fasten, 1931).

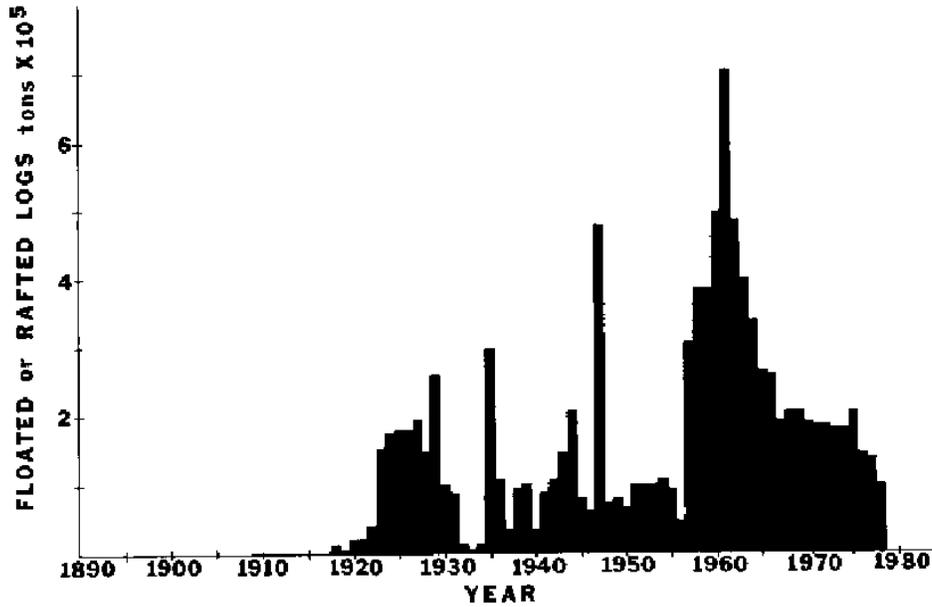


Figure 2.6 Number of logs floated or rafted on the Yaquina River from 1918 to 1978 (reproduced from Sedell and Duval, 1985).

A combined sewage discharge with a pump station was constructed for Newport in the mid-1950s, which eliminated the direct discharge of sewage from Newport into Yaquina Estuary (Lee Ritzman, City of Newport, personal communication). A municipal sewage system with primary treatment and an offshore discharge was constructed in Newport in 1964, which has since been upgraded to secondary treatment. A combined stormwater/sewage system that discharged raw sewage into the Yaquina River was constructed in Toledo in 1926, and then upgraded in 1954 to a primary treatment facility to handle the municipal waste from the city of Toledo (T. McFetridge, ODEQ, personal communication). This facility, which discharges into the Yaquina Estuary (about 22 km from the mouth of the estuary), was upgraded to secondary treatment in 1981. In the late 1980's and early 1990's, the City of Toledo made improvements to their stormwater collection system, reducing the bypassing of the treatment plant during high flow periods. In 1996, the Toledo plant had a discharge of 0.979 million gallons per day (MGD) with a design capacity of 3.5 MGD (www.epa.gov/OW-OWM.html/mtb/cwns/1996report2/or.htm).

In addition to the Toledo municipal discharge, a number of houses along the Yaquina Estuary and River have on-site septic systems. The primary environmental impact of these septic systems appears to be microbial contamination which primarily affects the oyster industry in Yaquina Estuary. The lower portion of the Yaquina Estuary is impaired for shellfish growing

due to fecal coliform (Table 1.1). Due to concern for microbial contamination associated with human and animal waste, a survey of residential septic systems was conducted during 1985-1986. Septic systems for 160 residences adjacent to the Yaquina Estuary were surveyed and it was found that approximately 17% of the residences surveyed had marginal septic systems and 16% had failing systems. The failing systems identified have since been corrected (Bill Zekan, Lincoln County Oregon, Planning and Development, personal communication).

There are three types of industrial discharges into Yaquina Bay/River. Six seafood processing plants discharge waste into Yaquina Bay (<http://www.deq.state.or.us/wq/sisdata/sisdata.asp>), all of which are classified as “minor” by the ODEQ. Though relatively small discharges, two of the companies have been fined by ODEQ for violating their permits. The Yaquina Bay Fruit Processors also discharges brine waste into Yaquina Estuary. The third type of discharge is waste from the Georgia Pacific West kraft pulp and linerboard mill. The mill went into production in 1957 with the primary discharge through an ocean outfall offshore of Newport. There is an emergency overflow outfall (located about 21 km from the mouth of the estuary) that discharges directly into the Yaquina Estuary; however, this outfall has discharged only ten times from 1999 to 2004, with a maximum discharge of 0.24 MGD. The discharges typically occur during heavy rain events for short time periods (less than 24 hours).

2.4 *Classification of the Yaquina Estuary*

Classification has been proposed as an important tool for developing nutrient criteria for estuarine systems (e.g., U.S. EPA 2001). Classification of estuaries in terms of their susceptibility to nutrient enrichment is theoretically highly desirable because of the large number of estuaries in the United States and limited resources, which make it unfeasible to develop nutrient criteria on a case by case basis for each individual system. Numerous types of estuarine classifications have been developed or proposed, including ones based on geomorphology, physical and hydrodynamic factors, and susceptibility to nutrient enrichment (Kurtz et al., 2006). A key aspect of the use of any classification system for setting nutrient criteria is that estuaries within the same class respond similarly to nutrients, which is a step that must be validated and has not yet been accomplished for national scale estuarine classifications in the U.S.

Several estuarine classifications have included Yaquina Estuary. Bottom et al. (1979) classified Yaquina Estuary as a “Drowned River Valley” and partially mixed estuary. NOAA classified Yaquina Estuary as “River Dominated” with “Straits and Terminal Bay.” Quinn et al. (1991) classified estuaries along the west coast of the United States based on their susceptibility to nutrient pollution. In this study, they classified the Yaquina Estuary as in the high category for dissolved concentration potential (DCP) and in the low category for particle retention efficiency. They estimated that the nutrient concentration for nitrogen and phosphorous would be in the medium class based on DCP and estimates of nutrient loadings. Additionally, Quinn et al. (1991) estimated that Yaquina Estuary would require > 20% increase in nutrient loading to change the concentration from medium to high class. Burgess et al. (2004) classified estuaries in the U.S. based on a statistical cluster analysis of physical and hydrologic factors. They classified Yaquina Estuary as a “Medium Area, Low Volume, Shallow and Mixed Salinity” estuary.

2.5 *Conceptual Model for Yaquina Estuary*

Figure 2.7 illustrates some of the major drivers influencing causal (nutrients) and response (chlorophyll *a*, water clarity and dissolved oxygen) variables within the Yaquina Estuary, which will be presented in this case study. Nutrient, chlorophyll *a*, and dissolved oxygen conditions in the lower portion of the estuary are strongly influenced by ocean conditions due to close coupling between the shelf and the estuary resulting from strong tidal forcing. The watershed is primarily forested, and riverine inputs are related to the presence of nitrogen-fixing red alder (*Alnus rubra*) trees in the watershed. Seagrasses occur at shallower depths in the upper portions of the estuary than they do in the lower estuary, which we believe is related to increased turbidity upriver and the resulting light limitation. Dense macroalgal blooms occur in the lower portion of the estuary, but they appear to be fueled by oceanic nitrogen inputs rather than being a response to anthropogenic nutrient enrichment.

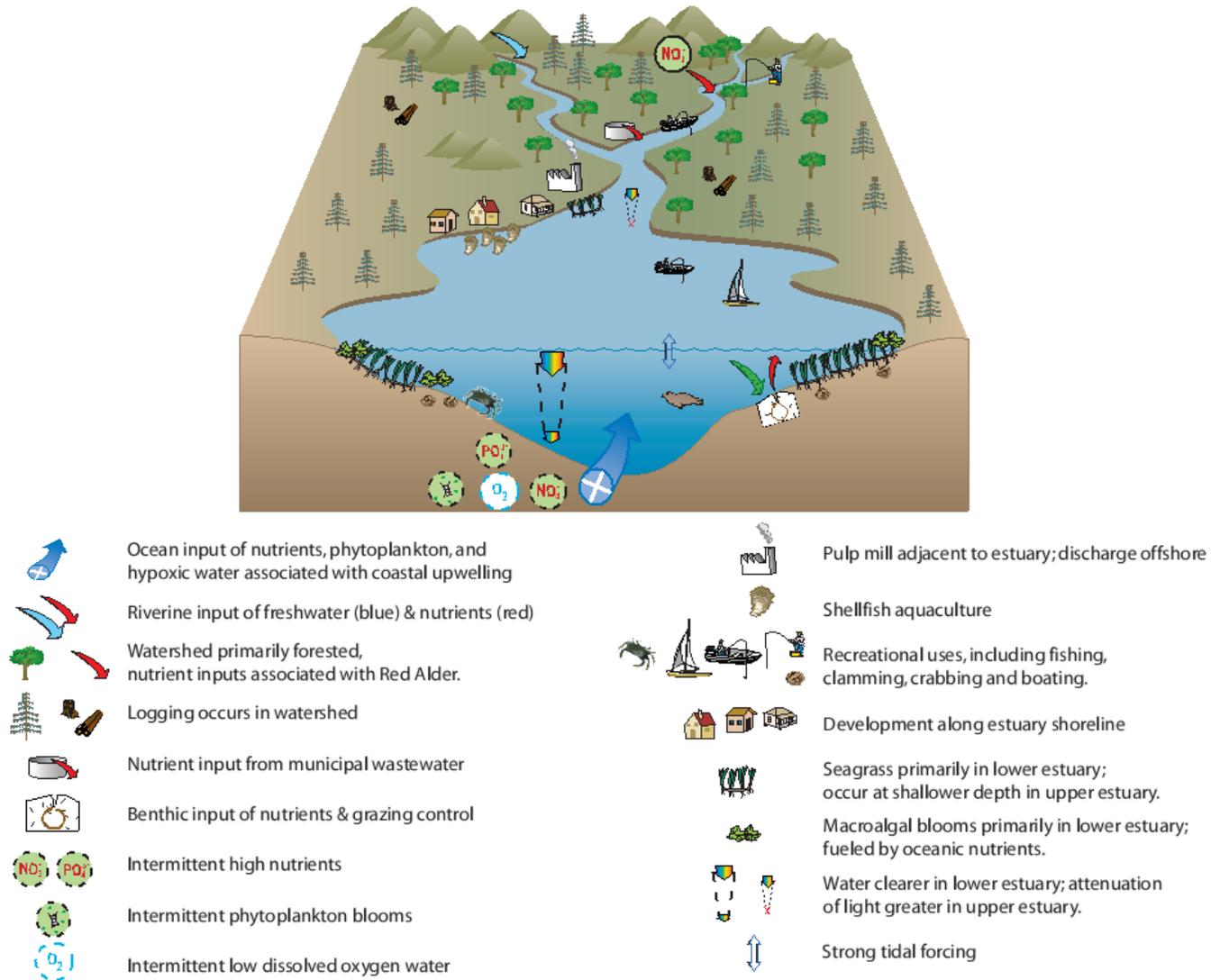


Figure 2.7 Conceptual model of factors influencing nutrient and response variables in the Yaquina Estuary. Iconography from the University of Maryland Center for Environmental Science Integration and Application Network, <http://ian.umces.edu>.

3. Description of Sources/Sinks of Nutrients

3.1 Background

In most estuaries, the major sources of nitrogen are atmospheric deposition, agricultural nitrogen fixation, fertilizer runoff, animal feeding operations runoff, and in heavily populated areas point source inputs associated with wastewater treatment facilities (WWTF) (Driscoll et al., 2003; Howarth et al., 2002; Boyer et al., 2002). For many PNW estuaries (with the exception of Puget Sound), there is relatively low population density in the watersheds and low atmospheric nitrogen deposition. The watersheds are predominantly forested, resulting in low nitrogen inputs associated with fertilizer and agriculture nitrogen fixation. Upwelling provides a major source of nutrients to estuaries adjacent to coastal upwelling regions, such as the PNW (e.g., Hickey and Banas, 2003 and Brown and Ozretich, in review). Low intensity landuse and coastal upwelling result in a significant difference in dominant sources of nutrients to PNW estuaries compared to estuaries elsewhere in the U.S.

In a recent review, Tappin (2002) found that the input of nitrogen to temperate and tropical estuaries from the ocean is poorly quantified. It is important to quantify the contribution of oceanic input to nutrient loading in order to determine background conditions for estuaries that are adjacent to upwelling regions and to distinguish natural variability from anthropogenic inputs. We also do not know how susceptible estuaries subjected to large oceanic inputs of nutrients (dissolved inorganic nitrogen and phosphorous) are to future changes in anthropogenic inputs of nutrients. Addressing issues associated with ocean input of nutrients is critical in the process of developing nutrient criteria for estuaries in the PNW region.

3.2 Nitrogen Loading to Yaquina Estuary

Brown and Ozretich (in review) compared the sources of nutrients to Yaquina Estuary during the wet and dry seasons (Table 3.1). There are large seasonal differences in the sources of nitrogen 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 dissolved inorganic nitrogen ($DIN = NO_2^- + NO_3^- + NH_4^+$) from the sediments into the water column is the second largest source of DIN. Atmospheric deposition of inorganic nitrogen along the central Oregon coast is among the lowest in the United States.

Average annual deposition during 1980-2002 was $0.6 \text{ kg N ha}^{-1} \text{ y}^{-1}$ (NADP, 2003). Atmospheric deposition of nitrogen is a minor component of nutrient input to Yaquina Estuary with direct deposition on the estuary only representing 0.05% of the nitrogen input to the estuary.

Atmospheric deposition on the watershed is a small source (8%) compared to the watershed input associated with nitrogen-fixing red alder trees in the watershed (Brown and Ozretich, in review). Annual input of nitrogen from WWTF effluent is estimated to be 0.4% of the total nitrogen input to the estuary. A NOAA study of estuarine susceptibility to nutrients (Quinn et al., 1991) estimated point source loading to Yaquina Estuary as about an order of magnitude higher than our estimates.

3.2.1 *Watershed*

There is approximately an order of magnitude difference in the 30-year average daily riverine nitrogen input to Yaquina Estuary between the wet and dry seasons. In addition, there are considerable interannual differences in riverine nitrogen input, with wet season riverine nitrogen input varying from $6.5 \times 10^4 \text{ mol N d}^{-1}$ to $5.2 \times 10^5 \text{ mol N d}^{-1}$, and dry season riverine nitrogen input ranging from $1.1 \times 10^4 \text{ mol N d}^{-1}$ to $6.3 \times 10^4 \text{ mol N d}^{-1}$ (Brown and Ozretich, in review). During the wet season, riverine input is the largest source of DIN to the estuary, contributing approximately 78% of the input, while 91% of the annual riverine nitrogen input is delivered during the wet season. Our estimates of riverine nitrogen loading (Table 3.1) are similar to Quinn et al. (1991) whose estimate of non-point loadings are 7% higher than our estimate of annual riverine loading. Sigleo and Frick (2007) estimated that the annual riverine nitrate (NO_3^-) input to Yaquina varied from $2.4 \times 10^5 \text{ mol N d}^{-1}$ to $5.2 \times 10^4 \text{ mol N d}^{-1}$ during a drought year.

Oregon Coast Range streams have high NO_3^- concentrations relative to other forested watersheds in the PNW (Compton et al., 2003; Wigington et al., 1998). Wigington et al. (1998) hypothesized that forest vegetation, in particular the presence of red alder, is the primary factor determining stream NO_3^- levels in the Oregon Coast Range. Red alder is a native tree species in the PNW that colonizes areas disturbed by fires, logging and landslides. Red alder have symbiotic N_2 fixing bacteria that can fix $50\text{-}200 \text{ kg N ha}^{-1} \text{ y}^{-1}$ in pure stands (Binkley et al., 1994). Compton et al. (2003) found a significant relationship between alder cover and stream NO_3^- concentration in the Salmon River watershed, which is about 45 km north of Yaquina

Estuary. 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. In the Yaquina Estuary watershed, 23% of the watershed is vegetated with red alder (Brown and Ozretich, in review). Brown and Ozretich (in review) estimated that > 80% of the riverine nitrogen loading to Yaquina Estuary is related to red alder cover. Thus, riverine nutrient loading in the PNW is influenced by forest species composition.

3.2.2 *Ocean Input*

Brown and Ozretich (in review) estimated oceanic input of DIN to the Yaquina Estuary during the dry season of 2002 and 2003. The oceanic input of DIN was calculated using the time-series of flood tide input of DIN multiplied by the volume of water entering the inlet during each tidal cycle. The volume of water entering the inlet was calculated using a two-dimensional, laterally averaged hydrodynamic and water quality model (described in Brown and Ozretich, in review). Daily water samples were collected during flood tide approximately 0.5 m below the surface at a station about 3.7 km from the mouth of the estuary. These samples were analyzed for dissolved inorganic nutrients ($\text{NO}_3^- + \text{NO}_2^-$, NH_4^+ , PO_4^{3-} and Si(OH)_4). During the dry season of 2002, the amount of DIN entering the estuary from the ocean during each flood tide varied from 8.8×10^3 mol N to 6.7×10^5 mol N with a mean value of 2.4×10^5 mol N, and the mean daily flood tide input of DIN was 4.7×10^5 mol N d^{-1} . During the 2003 dry season, the mean oceanic input of DIN is 3.7×10^5 mol N d^{-1} or 21% less than 2002 dry season. Sigleo et al. (2005) calculated the flood tide input of NO_3^- to Yaquina Estuary during August of 2000 to be 13×10^5 mol N d^{-1} , which is about triple our estimate. However, these ocean input numbers were calculated using a constant flood tide NO_3^- of 30 μM .

Table 3.1 Comparison of the magnitude of nitrogen sources during wet and dry seasons for Yaquina Estuary, Oregon. Benthic flux measurements not available (NA) for wet season.

Source	Nitrogen Input (mol N d ⁻¹)		
	Wet Season	Dry Season	Annual Average
River	2.7 x 10 ⁵	2.5 x 10 ⁴	1.4 x 10 ⁵
Ocean	3.0 x 10 ⁴	3.7-4.7 x 10 ⁵	2.3 x 10 ⁵
Wastewater	1.7 x 10 ³	1.5 x 10 ³	1.6 x 10 ³
Benthic Flux ¹	NA	4.3 x 10 ⁴	NA
Atmospheric Deposition ²			
On Estuary	2.2 x 10 ²	1.2 x 10 ²	1.7 x 10 ²
On Watershed	1.1 x 10 ⁴	6.0 x 10 ³	8.5 x 10 ³

Source: ¹DeWitt et al. (2004); ²NADP (2003)

3.2.3 Benthic Processes

Intertidal and subtidal sediments can be sources and sinks for nutrients and organic matter, with the direction and magnitude of fluxes determined by infaunal invertebrates, benthic primary producers, and microbial communities living on or in the estuarine benthos. (See Appendix A for additional details on benthic processes).

Five studies of benthic nutrient flux have been conducted in Pacific estuaries north of San Francisco, however the reported benthic flux data in four of the studies (i.e., Dollar et al. 1991; Garber et al. 1992; Thom et al., 1994; Larned, 2003) may not accurately estimate estuary-scale nutrient fluxes in Yaquina Estuary because they do not account for the presence of thalassinid burrowing shrimp. The presence of burrowing shrimp can result in the water inside of the benthic flux chamber being exchanged with water outside of the chamber via shrimp burrows (e.g., Hughes et al., 2000), which violates the requirement that benthic chambers be closed microcosms (Forja and Gomez-Parra, 1998).

To avoid this problem, DeWitt et al. (2004) inserted 1-m deep core barrels into sediments at their study sites, and fit benthic chambers to the tops of the core barrels to isolate water, sediments, shrimp and burrows inside the chamber from the outside world. DeWitt et al. (2004) demonstrated that DIN efflux was strongly affected by both burrowing shrimp species and population density (Appendix A). Integrated over the whole estuary, net DIN efflux for intertidal habitats in Yaquina Estuary was 4.3 x 10⁴ mol N d⁻¹ from the benthos to the water column (DeWitt et al., 2004). (Additional details on composition of estimated DIN efflux provided in Appendix A)

3.3 Zonation Based Upon Nitrogen Sources

We divided the estuary into two zones, one of which is dominated by ocean input (Zone 1) and the other which is more influenced by watershed and point source inputs (Zone 2, Figure 2.1). We used a transport model combined with natural abundance stable isotopes ($\delta^{15}\text{N}$) of green macroalgae to identify the dominant nitrogen sources within the estuary as a function of time and location for two years (2003 and 2004). 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. For more details on this analysis, see Chapter 5 of Lee et al. (2006).

Model simulations combined with $\delta^{15}\text{N}$ of green macroalgae suggest that during the wet season, riverine nitrogen sources dominate throughout the estuary, which is consistent with our comparison of nutrient loadings presented in Section 3.2. During the dry season, ocean nitrogen sources dominate in Zone 1, comprising between 53 – 87% of DIN (depending upon location within the zone), whereas riverine and WWTF inputs contribute 12-40% and 2-8%, respectively. In Zone 2, riverine nitrogen sources dominate contributing between 56-92% of DIN (depending upon location). WWTF contribution to water column DIN is maximal during the month of August.

During the dry season, oceanic input of nitrogen propagates up estuary as the freshwater inflow declines. This can be seen in simulation results from 2004 (Figure 3.1) which show that Station N1 is ocean dominated (fraction ≥ 0.5) during the entire dry season (May – September), while Station N2 is river dominated during May and ocean dominated from June – September. At Stations N3, N4, and N5 ocean inputs increase in importance from May – August, but never dominate. There is interannual variability in the position of the line demarking the oceanic and riverine dominated zones. The exact location of this line varies with ocean conditions (e.g., El Niño, La Niña conditions) as well as freshwater inflow. To be conservative, we placed the line demarking the two zones at the most seaward location found in our analysis (see Figure 2.1 for location). Analysis of salinity data reveals that the demarcation of the two zones corresponds to a dry season median salinity of 26.

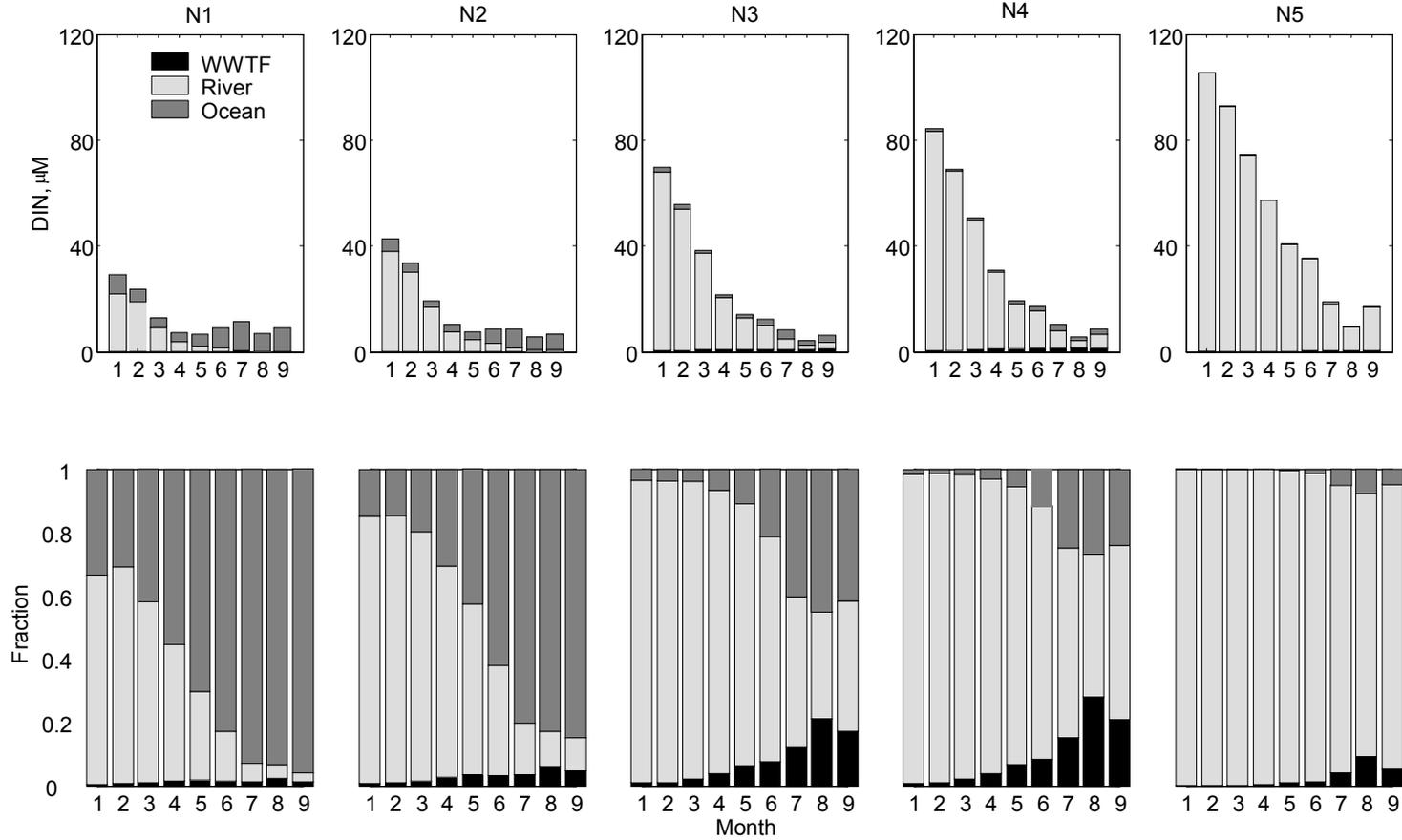


Figure 3.1 Modeled contribution of WWTF effluent, riverine, and oceanic sources to DIN at 5 locations in the estuary during January – September of 2004. Stations N1 and N2 are located in Zone 1 about 3.7 and 10.1 km from the mouth of the estuary, respectively. Stations N3, N4, and N5 are located in Zone 2 about 15.6, 18.4, and 26.0 km from the mouth of the estuary.

4. Data Sources and Methods

We assembled causal (nutrient) and response variables (chlorophyll *a*, dissolved oxygen, water clarity, total suspended solids, macroalgae biomass, and submerged aquatic vegetation distribution) and physical data (temperature and salinity) at three spatial scales. Water quality data from Yaquina Estuary was compared to water quality data collected during the dry season for a set of seven Oregon estuaries for the purpose of estuarine classification (Section 4.2), and from a random sampling of all Oregon estuaries conducted as part of the EPA National Coastal Assessment (NCA) (Section 4.3). Historical data were assembled to assess whether there have been any long term trends in causal or response variables. For the trend analyses, we parsed the data into zones and seasons to minimize bias associated with differences in sampling (temporal or spatial). The zones are presented in Figure 2.1 and discussed in Section 3.3, while the seasons are defined in Section 2.1. For details on the methods used and the quality assurance/quality control (QA/QC) of data used in this study see Appendix B.

4.1 Yaquina Estuary Data

4.1.1 Recent Data

Data assembled for the Yaquina Estuary included recent (1998-2006) water quality cruises conducted by the Western Ecology Division, U.S. EPA. The sampling frequency and number of stations depended upon the year and month (Table 4.1). 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), *in situ* fluorescence (WETStar Chlorophyll Fluorometer, WET Labs, Philomath, Oregon), and photosynthetically active radiation (PAR; PAR LI-193 underwater irradiance sensor, Lincoln, Nebraska) were measured. 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. For the cruises conducted in 2006, dissolved oxygen was measured at surface, mid-depth, and bottom using a YSI multiparameter sonde (YSI 6600 EDS, YSI Inc., Yellow Springs, OH). At each station, water samples were collected, which were analyzed for dissolved inorganic nutrients ($\text{NO}_3^- + \text{NO}_2^-$, NH_4^+ , PO_4^{3-} and Si(OH)_4). During the 2006 cruises, additional water samples were collected and analyzed for total nitrogen (TN) and total phosphorous (TP). Water samples were collected for chlorophyll *a* analysis at each cruise location quarterly during 2002 and 2003 (surface

samples), and monthly during 2006 (mid-depth samples). Additional water samples were collected for total suspended solids (TSS) analysis.

Light attenuation coefficients (k_d) were determined for each station as the slope of the regression of $\ln(\text{PAR})$ vs. depth for the 1.00 m to 3.75 m depth intervals. Many of the light profiles measured during the cruises were conducted during flood tides; therefore, the light attenuation coefficients may be biased toward clearer flood tide conditions. In addition to the cruise data, PAR was monitored continuously with 15 minute averages recorded at five locations in the estuary (WED unpublished data). Three of these sites were in Zone 1 (located 3.7, 3.9, and 9.0 km from the mouth of the estuary) and two were in Zone 2 (located 18.4 and 16 km from the mouth of the estuary). Measurements at these sites were taken nearly continuously from 1999 through 2003 using two PAR sensors placed 0.75 m apart in depth, which were used to calculate light attenuation coefficients. The sensors were cleaned at one to two week intervals. For the analyses presented in this document, we used attenuation coefficients measured at local noon time and within 4.5 days of cleaning for the continuous data set.

Additional high temporal resolution data were collected at the riverine and oceanic boundaries to quantify the oceanic and riverine inputs of dissolved inorganic nutrients and chlorophyll *a* to the estuary. Continuous data (including water temperature, salinity, dissolved oxygen, and *in situ* fluorescence at 15-min intervals) from YSI multiparameter sondes (YSI 6600 EDS, YSI Inc., Yellow Springs, OH) were available at approximately six locations in the estuaries (with the exact number of locations depending upon the year and month).

4.1.2 Additional Data Sources

A summary of historic data compiled for the Yaquina Estuary is provided in Table 4.2. In addition to the sources listed in Table 4.2, data were obtained from the Oregon Department of Environmental Quality Laboratory Analytical Storage and Retrieval (LASAR) database (<http://deq12.deq.state.or.us/lasar2/>), which included data for 27 sampling locations and spanned the time interval of 1960-2005.

There was a gap in the data for causal and response variables during the interval of 1984-1997. The majority of the nutrient data was in the form of dissolved inorganic nutrients rather than total nitrogen or phosphorous. Most of the data compiled was collected at fixed sampling locations, rather than through probabilistic sampling. All the data were collected along the main channel of the estuary, and did not extend into the sloughs. The locations of stations sampled extended from the mouth to the

tidal fresh portion of the estuary. The estuary narrows upstream of about 25 km from the mouth and there was limited sampling upstream of this region. There were limited historic data for water clarity (secchi depth) and as a result we were unable to assess trends in water clarity. All of the chlorophyll *a* data compiled were obtained using spectrophotometric or fluorometric methods.

Table 4.1 Sampling frequency for cruise data collected by U.S. EPA from 1998-2006.				
Year	Month	Sampling Frequency	# of Sampling Locations	Distance from mouth of Estuary (km)
1998	Jun, Jul, Sept, Nov	Once a Month	35	2-21
1999	Jan-Dec	Once a Month	Varied 5-35	2-21
2000	Jan-Dec	Once a Month	Varied 5-36	2-21
2001	Mar, Apr, Aug-Oct	Once a Month	Varied 5-36	2-21
	May-Jul	Twice a Month	Varied 12-34	2-21
2002	Apr-Sept	Weekly	12	2-21
	Jan, Feb, Mar, Oct	Twice a Month	12	2-21
	Nov, Dec	Once a Month	12	2-21
2003	Apr-Sept	Weekly	12	2-26
	Jan, Feb, Dec	Once a Month	12	2-26
	Mar, Oct	Twice a Month	12	2-26
2004	April-September	Twice a Month	12	3-26
2006	February-December	Once a Month	12	3-35

Table 4.2 Summary of historic data compiled include source of the data, temporal and spatial sampling frequency of the data set and parameters measured.					
Source	Time Interval	Number of Sampling Events	# of Stations	Dist. from mouth (km)	Parameter
Matson (1964)	11/62 – 1/64	30	4	2-81	Salinity, Water temp, DO, PO ₄ ³⁻ , Si(OH) ₄
Gibson and Snow (1967)	6/66 – 11/67	30	6	7-16	Salinity, Water temp, DO
De Ben et al. (1990)	3/67 – 11/68	42	10	2-26	Salinity, Water temp, DO
Gibson (1974)	4/68 – 3/70 8/71 – 1/72	23 11	4 2	4-15 15-16	Salinity, Water temp, DO PO ₄ ³⁻ , Si(OH) ₄ , NO ₃ ⁻ + NO ₂ ⁻
Johnson (1980)	6/73 – 10/74	80	5	9-16	Chlorophyll <i>a</i>
Amspoker (1977)	12/73 – 8/74	4	6	3-36	NO ₃ ⁻ + NO ₂ ⁻ , Total Phosphates, Si(OH) ₄
Karentz (1975)	7/74 – 4/75	21	4	3-19	Chlorophyll <i>a</i>
Karentz and McIntire (1977)	5/74 – 5/75	12	4	3-19	Salinity, Water temp, NO ₃ ⁻ + NO ₂ ⁻ , PO ₄ ³⁻ , Si(OH) ₄ , Chlorophyll <i>a</i>
WED unpublished	7/76- 12/77	9	16	3-42	Salinity, Water temp, PO ₄ ³⁻ , NH ₃ , NO ₂ ⁻ , NO ₃ ⁻ , TN, TP, TSS
Frey (1977)	2/77 – 6/77	8	3	3-35	Salinity, Water temp, PO ₄ ³⁻ , Si(OH) ₄ , NO ₃ ⁻ + NO ₂ ⁻ , Chlorophyll <i>a</i>
Butler (1986)	6/83 – 8/85	12	7-16	2-35	Salinity, Water temp, DO, NO ₂ ⁻ , NO ₃ ⁻ , NH ₄ ⁺ , PO ₄ ³⁻ , Si(OH) ₄
Arnold et al. (1992)	4/86- 3/87	149	1	11	Salinity, Water temp, TSS

4.2 *Oregon Estuarine Classification Study*

As part of an effort to classify estuaries by the susceptibility of their submerged aquatic vegetation and food webs to nutrients, WED surveyed seven Oregon estuaries during the dry seasons of 2004 and 2005 (Lee et al., 2006). The estuaries sampled have regional drivers and landuse characteristics similar to the Yaquina Estuary. Their watersheds were primarily forested (66-86%) with low land development (high and low intensity development $\leq 1\%$), and low human population densities (4 – 25 individuals km^{-2} ; Lee et al, 2006). The estuaries sampled (Asea, Nestucca, Yaquina, Salmon River, Coos, Umpqua River and Tillamook) vary in size from 2 to 55 km^2 , and from river dominated to ocean dominated. As is typical of many PNW estuaries, they have extensive intertidal zones with the percentage of intertidal area ranging from 32 to 87% of total estuarine area.

Water quality data together with measurements of the natural abundance stable isotope ratio for nitrogen ($\delta^{15}\text{N}$) of green macroalgae data were collected to evaluate current water quality conditions. These data were also used to divide each estuary into oceanic and riverine dominated zones (in terms of nitrogen sources). The sampling consisted of high tide and low tide cruises and of short-term deployments of water quality datasondes. During each cruise between 10 and 17 stations were sampled in each estuary, depending upon the size of the estuary, and the stations extended from the mouth of the estuary to the fresh water portions of the estuary for all systems except Coos Estuary (lowest salinity in Coos was 14 psu). For more details on the methods used and the data collected, see Lee et al. (2006).

4.3 *National Coastal Assessment (NCA)*

As a part of the NCA, the Environmental Monitoring and Assessment Program (EMAP) assessed the condition of estuarine resources of Oregon based on a range of indicators of environmental quality, including water quality indicators (chlorophyll *a*, nutrients, and dissolved oxygen). The study utilized a stratified random sampling design and sampled over two years (1999-2000). The NCA Oregon estuary data set was obtained during the summer, and thus corresponds to the Yaquina Estuary “dry season.” The NCA data set allows comparison of Yaquina Estuary values for water quality parameters (e.g. median DO) to values for the same parameter across the set of all Oregon estuaries.

Details of the sampling program and results of the Oregon NCA assessment are provided in Nelson et al. (2004). Briefly, the Oregon 1999 sampling design consisted of 50 sites distributed among

14 estuaries of the State. An additional 30 sites were sampled in Tillamook Bay to assess condition of this National Estuary Program system, and Tillamook Bay was thus not included in the sample selection for the other 50 sites. Tributary estuaries of the Columbia River that are located within Oregon were included in the 1999 sampling effort, while the main channel area was not sampled until 2000. In 1999, estuaries were divided into four strata based on size, and approximately equal sampling effort was placed in each stratum, to insure sampling across the entire estuarine size spectrum. The Oregon 2000 study included only the main channel area of the Columbia River, and was split into two strata, the lower, saline portion and the upper, freshwater portion, with 20 and 30 sites sampled, respectively. Additional samples were obtained in the WA tributary estuaries of the Columbia River in 1999, but were not included in the data presented in this section. A total of 128 out of the 130 target stations were successfully sampled for water quality indicators.

4.4 *Percentile Approach*

Previous assessments of water quality conditions in PNW estuaries were hindered by the limited availability of water quality data for estuaries in the region, particularly in Oregon (Bricker et al., 1999). Since there were limited data for applying the reference condition approach for the class of estuaries similar to the Yaquina Estuary, we used *in situ* observations within Yaquina Estuary as a basis for the Estuarine Reference Condition (as recommended by U.S. EPA, 2001). To accomplish this, we produced cumulative distribution functions (CDFs) for the Yaquina Estuary and compared those to CDFs of other Oregon estuaries using two independent data sets (Classification Study and NCA for Oregon estuaries). The NCA and Classification data sets were sampled at different temporal and spatial scales. The Classification data set used in our analyses included samples from six estuaries, with 10-17 stations sampled per estuary during both flood and ebb tidal conditions. Data from the Yaquina Estuary collected as part of the Classification Study were not included in the computation of percentiles for this data set. The NCA data set sampled 14 Oregon estuaries. The number of stations in each estuary was randomly determined within an estuarine size stratum. Timing of sample collection with respect to tidal stage was random. The number of sampling locations per estuarine system in the NCA data set ranged from 1 (Alsea and Yachats) to 67 (Columbia).

In Appendix C, we present various classifications of the estuaries in the Classification Study and NCA data sets based upon geomorphology, susceptibility to nutrient pollution, and statistical clustering of physical and hydrologic variables. The number of classes of estuaries (or types) depends

upon the scale of the classification system as well as the classification system utilized (see Table C.1). The estuaries sampled in the Classification Study and NCA data set fell into a limited number of estuary classes (2-4); however, there was not a consistent pattern in the grouping of estuaries within a class among the different classifications. One exception was the Columbia River Estuary, which consistently was placed in a separate class for classifications based on geomorphology, susceptibility to nutrient pollution, and statistical clustering.

Cumulative distribution functions (CDFs) were produced for each of the three data sets (Yaquina, Classification, and NCA). For this analysis, the data from the three data sets were divided into marine and riverine dominated regions (Zones 1 and 2, respectively). In addition, the Yaquina data set was further divided into wet and dry seasons. Only recent data (1998-2006) were used in creating the CDFs for Yaquina Estuary. The CDFs produced for the Yaquina and Classification data sets represent percentiles associated with the number of samples (i.e., not weighted by percentage of estuarine area). The NCA program typically computes CDFs using the appropriate sampling area weightings, which are based on areas of sampling strata determined from GIS (US EPA, 2004a). This allows estimation of the areal extent of Oregon's estuaries associated with any value of an indicator variable. However, for the present study, estimates of percentiles for NCA data sorted by salinity zone were produced without use of area weightings and represent percentiles associated with the number of samples. This was done for consistency among data sets, and because area estimates of salinity zones were not available for all Oregon estuaries. An additional set of CDFs were produced for the NCA data set excluding the Columbia Estuary, which differs from the other Oregon estuaries in size, geomorphology, and other factors.

4.5 *Statistical Analysis*

Due to the non-normal distribution of the data, non-parametric statistical tests were used for all analyses. The Mann-Whitney Rank Sum test was used to determine whether there were significant differences in median values between zones or seasons. The Kruskal-Wallis one way analysis of variance on ranks was used to test whether there were significant differences in the median values between the Yaquina (dry season only), Classification Study, and NCA data sets. If there were significant differences ($p < 0.05$), then Dunn's test was used for pairwise multiple comparisons. For all tests, p values less than 0.05 were considered significant.

To assess whether there were temporal trends in water quality measures, the data were divided into zones and seasons to minimize biases associated with differences in sampling (spatial and temporal). For chlorophyll *a*, there was insufficient wet season data available, so trend analysis was only performed for the dry season. A Mann Kendall trend test was used to test whether there were significant trends within a zone and a season. If there were significant seasonal patterns within a zone, then the Seasonal Kendall test was used to determine if there was a significant increasing or decreasing trend. The Seasonal Kendall test performs the Mann Kendall test for each season and then combines the results of these into one overall test for whether there is a consistent monotonic trend over time (Helsel et al., 2006). For the Seasonal Kendall test all of the data (within a zone and season) was used. For the Mann Kendall trend test, there can only be one observation for each date, so multiple observations (either multiple stations or sampling events) on a single day were averaged. For all trend tests, p values less than 0.05 were considered significant. In addition to the trend analysis, we divided the data into historical and recent groups and tested whether there were significant differences in median values using the Mann-Whitney Rank Sum test.

Mann-Whitney Rank Sum test and Kruskal-Wallis one way ANOVA were performed using SigmaStat software package (version 3.5, Systat Software, Inc., San Jose, CA), while trend analysis (Mann Kendall and Seasonal Kendall) were performed using a Windows Program written by the U.S. Geological Survey (Helsel et al., 2006).

5. Spatial and temporal patterns in water quality parameters in the Yaquina Estuary

For the analyses in this report, we divided the year into two seasons (wet and dry) and divided the estuary into two zones. There are significant seasonal and spatial patterns in the water quality data resulting from differences in sources, transport, and losses.

5.1 Salinity

There are strong seasonal differences in salinity within the Yaquina Estuary driven by differences in freshwater inflow (Figure 5.1 and Figure 2.3). During the dry season, Zone 1 is marine dominated with mean salinity of 26 psu at the boundary demarking Zones 1 and 2. Salt penetrates about 35 km into the estuary during periods of minimal freshwater inflow.

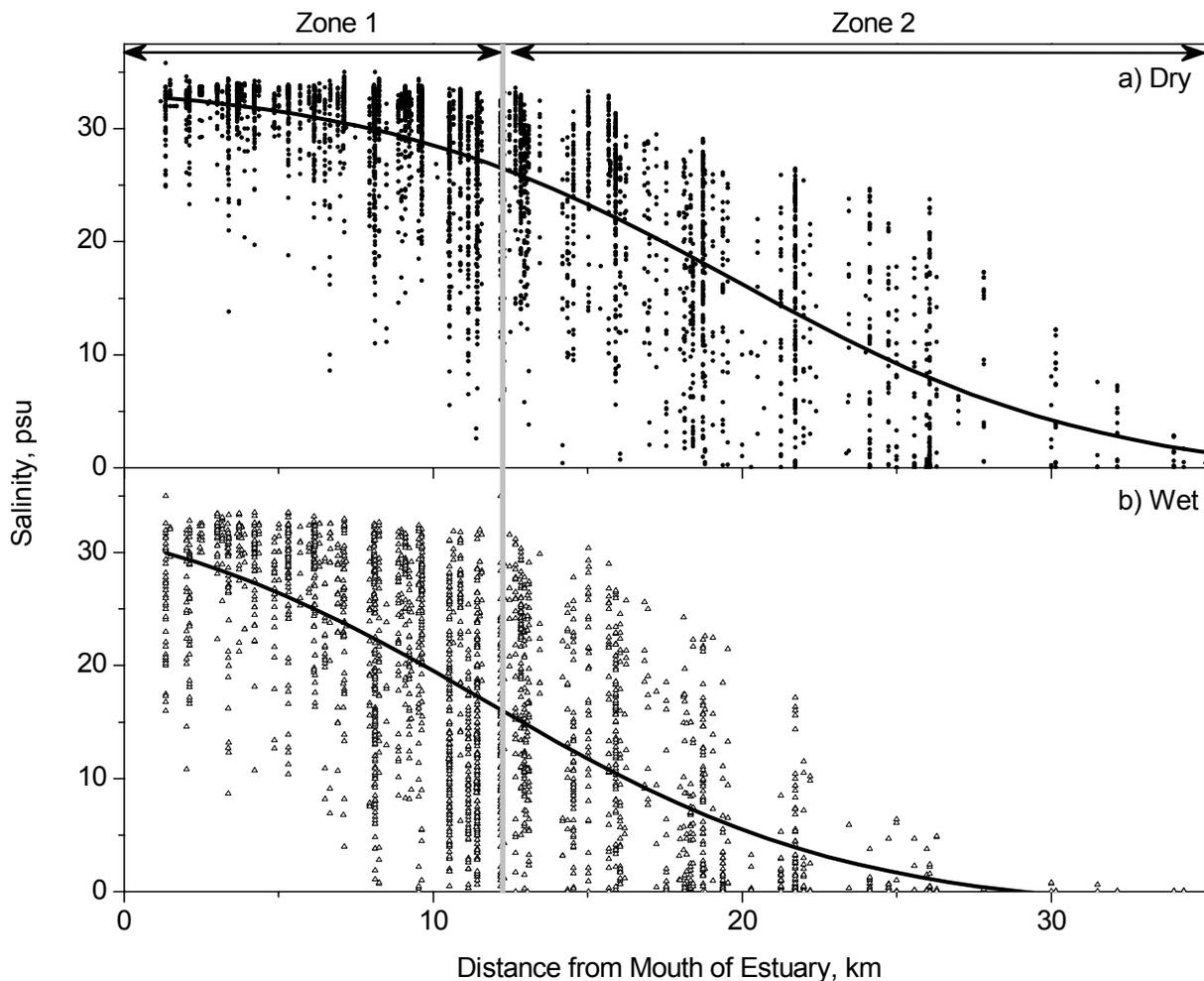


Figure 5.1 Salinity versus distance from mouth of the estuary during the a) dry and b) wet seasons.

The gray lines demark the two zones

5.2 *Nutrients*

During the wet season, NO_3^- is the primary form of DIN in the estuary (median of 88% of DIN, $n = 873$). There is little utilization of dissolved inorganic nutrients by phytoplankton within the estuary during the wet season due to short residence time (high freshwater inflow) and low solar irradiance. The average incident photosynthetically active radiation (PAR) varies from $15 \text{ mol quanta m}^{-2} \text{ d}^{-1}$ during the wet season to $38 \text{ mol quanta m}^{-2} \text{ d}^{-1}$ during the dry season. Mixing diagrams (property salinity plots) are often used to infer biogeochemical cycling occurring within estuaries (e.g., internal sources and sinks). Mixing diagrams of DIN for wet season cruises exhibit conservative mixing behavior, indicating river inputs are the primary nitrogen source and that there is little utilization within the estuary during this time. Minimal utilization of nutrients is also evident in the low chlorophyll *a* levels observed during the wet season (see Chapter 7).

The dry season coincides with the growth season and with upwelling on the shelf. As discussed in Chapter 3, nutrient rich water associated with coastal upwelling is advected into Yaquina Estuary during flood tides. During the dry season, high levels of DIN and PO_4^{3-} enter the estuary about two days after upwelling conditions (Brown and Ozretich, in review). Median concentrations of oceanic NO_3^- and PO_4^{3-} entering the estuary during the dry season are $8.6 \mu\text{M}$ and $1.3 \mu\text{M}$, respectively ($n = 830$). The maximal nutrient concentrations ($\text{NO}_3^- = 31.5 \mu\text{M}$ and $\text{PO}_4^{3-} = 2.9 \mu\text{M}$) entering the Yaquina Estuary during upwelling periods are similar to those found in other upwelling regions (Dugdale, 1985) and elsewhere on the Oregon shelf (Corwith and Wheeler, 2002).

During the dry season, NO_3^- is the primary form of DIN (median of 75%, $n = 2028$), while NO_2^- is a minor component only composing 2% of DIN. There is a mid-estuary minimum in mean dry season $\text{NO}_3^- + \text{NO}_2^-$ (with a mean of $7 \mu\text{M}$, Figure 5.2) suggesting that the estuary receives NO_3^- from both the ocean and the river. For mixing diagrams to be useful in identifying the importance of internal processes (e.g., biological uptake) steady state conditions need to apply. Due to the temporal variability of the ocean end member, it is not appropriate to use mixing diagrams to determine the role of internal estuarine processes (i.e., biological uptake) in the formation of this mid estuary minimum. The primary source of PO_4^{3-} to the system is the ocean and there is a steady decline in PO_4^{3-} with distance into the estuary (Figure 5.3). The oceanic signal in NO_3^- and PO_4^{3-} propagates approximately 13 km up the estuary (Brown and Ozretich, in review).

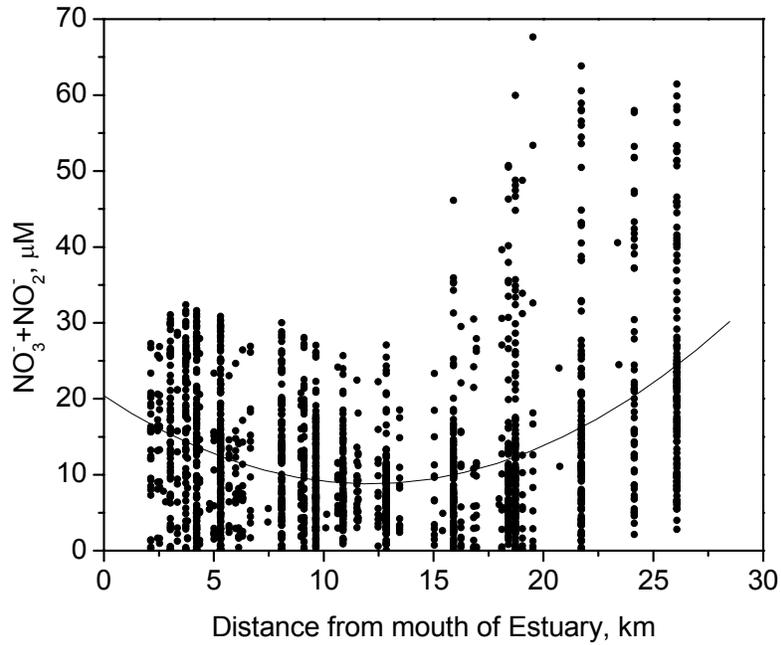


Figure 5.2 Spatial variation in dry season $\text{NO}_3^- + \text{NO}_2^-$. The line indicates a 2nd order polynomial fit to the data ($\text{NO}_3^- + \text{NO}_2^- = 20.4 - 1.9 * \text{distance} + 0.079 * \text{distance}^2$, $r^2 = 0.15$, $p < 0.001$)

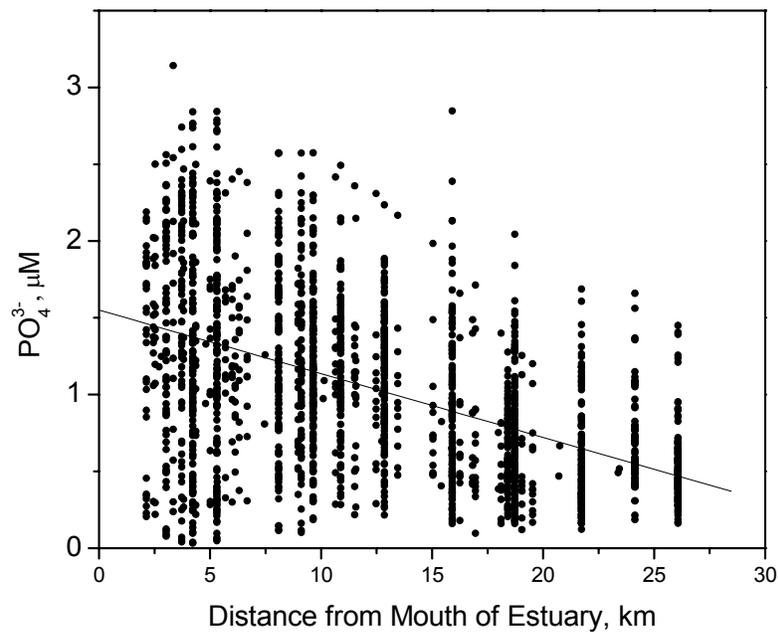


Figure 5.3 Spatial variation in dry season PO_4^{3-} . The line indicates linear regression ($\text{PO}_4^{3-} = 1.55 - 0.041 * \text{distance}$, $r^2 = 0.24$, $p < 0.0001$).

5.3 Chlorophyll *a*

Previous studies have demonstrated that chlorophyll *a* is advected into estuaries along the Oregon and Washington coasts from the coastal ocean during the dry season (Roegner and Shanks, 2001; Roegner et al., 2002). Brown and Ozretich (in review) found similar results for Yaquina Estuary. In Yaquina Estuary, peak chlorophyll *a* concentrations imported from the coastal ocean during the dry season reach $50 \mu\text{g l}^{-1}$ with a median value of $4 \mu\text{g l}^{-1}$ ($n=181$). The input of phytoplankton to the estuary lags upwelling favorable winds by approximately 6 days, suggesting that it takes this amount of time for phytoplankton to utilize the recently upwelled nitrogen and be transported across the shelf into the estuary (Brown and Ozretich, in review).

Figure 5.4 shows the import of chlorophyll *a* from the ocean, as indicated by the fact that high chlorophyll *a* occurs at high salinities. The oceanic signal attenuates more rapidly for chlorophyll *a* compared to NO_3^- and PO_4^{3-} . The statistically significant relationship between oceanic chlorophyll *a* concentrations and within estuary chlorophyll *a* is only evident up to about 11 km into the estuary (Brown and Ozretich, in review). The more rapid decline in the ocean signal in chlorophyll *a* is probably the result of benthic grazing on oceanic phytoplankton. Oyster aquaculture is present in Yaquina Estuary in the region 10-15 km from the mouth (Figure 2.1) and in the lower estuary there are tidal flats that have high densities of filter-feeding burrowing shrimp (DeWitt et al., 2004, see Section 3.2.3). Data from an *in situ* fluorometer (located 3.7 km from the mouth of the estuary) indicate that there is an import of oceanic chlorophyll *a* to the estuary and that a 60% reduction in chlorophyll *a* occurs between successive flood and ebb tides. Flood tide chlorophyll *a* values (median = $14 \mu\text{g l}^{-1}$) were significantly higher than ebb tide values (median = $9 \mu\text{g l}^{-1}$; Mann Whitney Rank Sum, $p<0.001$, $n = 53$). The import of chlorophyll *a* to Zone 1 is consistent with the findings of Karentz and McIntire (1977) that during the spring through fall seasons marine diatom genera dominated in the lower estuary (stations 3.4 and 6.7 km from the mouth of the estuary), while freshwater and brackish taxa dominated in the upper estuary (stations located 12.3 and 18.8 km from the mouth). Phytoplankton blooms occur in the tidal fresh portion of the estuary as indicated by the high chlorophyll *a* values at low salinities (Figure 5.4).

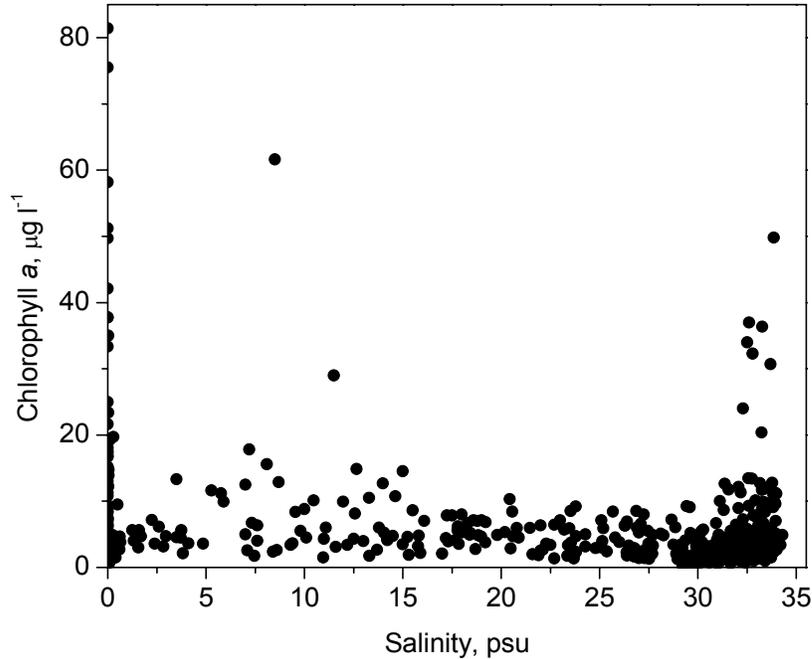


Figure 5.4 Dry season chlorophyll *a* versus salinity (all stations from 1974-2006) showing high chlorophyll *a* at high salinities, demonstrating the oceanic import of chlorophyll *a* from the coastal ocean into the Yaquina Estuary. Plot also shows the high chlorophyll *a* in the tidal fresh portion of the estuary.

5.4 Nutrient Limitation and Primary Productivity

Potential for nutrient limitation of phytoplankton is often estimated by examining the ratio of dissolved inorganic nutrients relative to the Redfield ratio (16 mol N: 1 mol P) and comparing the ambient dissolved inorganic nutrient concentrations to phytoplankton half saturation constants for nutrient uptake (e.g., Eyre, 2000). Typically, if the N:P ratio of the water column falls below 10:1 then phytoplankton may be limited by nitrogen, and if the ratio is greater than 20:1 there is the potential for phosphorous limitation (Boynton et al., 1982). In addition, if the ambient water column concentrations are less than the half saturation constants for nutrient uptake then we assume that the phytoplankton may be nutrient limited. Typical half saturation constants for DIN and DIP are 1.0 - 2.0 μM and 0.1 - 0.5 μM , respectively.

The median N:P ratio during the dry season is approximately 12:1, suggesting that nitrogen will be depleted prior to phosphorous for the majority of the estuary. There is evidence of phosphorous limitation in the upper portions of the estuary (17- 27 km from mouth) with the N:P ratio reaching as high as 260:1. In only 12% of the estuarine sampling events was the N:P ratio greater than 20 and DIP

less than 0.5 μM , suggesting the potential for phosphorous limitation. During the dry season, the median DIN concentration is 14 μM (n=2028), and 95% of the time the DIN concentration is greater than 2 μM (typical half saturation constant for phytoplankton). In only 5% of the estuarine sampling events was the N:P ratio less than 10 and DIN less than 2 μM . This suggests that although the N:P ratio often falls below 16:1, the estuary is not usually limited by either nitrogen or phosphorous. This is supported by assimilation ratio data (primary production : chlorophyll *a*) of Johnson (1980) that was collected during the dry season at a station about 16 km from the mouth of the estuary (Figure 2.1). Johnson's data showed 77% of the time there were sufficient nutrients for planktonic primary production, while 15% of the time there was borderline nutrient deficiency and 8% of the time there was evidence of nutrient depletion.

Specht (1975) conducted algal bioassays at six locations in Yaquina Estuary during 1972-1975 to examine the potential for nitrogen and phosphorous limitation. These experiments suggested that the upper portion of the estuary (26 km from mouth to tidal fresh) was predominantly phosphorous limited, while in the lower estuary, the system is nitrogen limited during the dry season and phosphorous limited during the wet season.

There is limited water column primary productivity data for Yaquina Estuary. Water column primary production (at a station 14 km from the mouth of the estuary) during the dry season ranged from 0.25-2.8 $\text{g C m}^{-2} \text{d}^{-1}$ with mean of 0.9 $\text{g C m}^{-2} \text{d}^{-1}$ (Johnson, 1980). For comparison, primary productivity associated with benthic microalgae in the lower portion of the estuary (Zone 1) ranged from 125-325 $\text{g C m}^{-2} \text{y}^{-1}$ (depending upon the location and elevation; Riznyk and Phinney, 1972). Davis (1981) measured net primary production during the dry season in the lower portion of the estuary of 46 $\text{g C m}^{-2} \text{d}^{-1}$ and 0.26 $\text{g C m}^{-2} \text{d}^{-1}$ for green macroalgae and benthic microalgae, respectively. Net primary production for *Zostera marina* and *Z. japonica* in the lower portion of the estuary was 181 and 130 $\text{g C m}^{-2} \text{y}^{-1}$, respectively (Kaldy, 2006ab).

Based on the existing primary productivity data, Yaquina Estuary can be characterized as mesotrophic. Water column planktonic primary production is a minor component of the total primary productivity, which is dominated by benthic primary producers (macroalgae, microalgae and seagrasses). This is consistent with the findings of Valiela et al. (2000b) that for systems with moderate and high nitrogen loading, macroalgae is the dominant primary producer in short residence time estuaries (≤ 3 days), while phytoplankton dominate in systems with relatively long residence times (≥ 45 days).

5.5 *Relationships between TN and TP and Chlorophyll a*

Relationships between causal and response variables are useful for demonstrating the relationship between nutrient loading and biological effects. Several studies have found relationships between nutrients (nitrogen and phosphorous) and chlorophyll *a* in estuaries (e.g., Monbet, 1992; Smith, 2006; Dettmann and Kurtz, 2006). During 2006, we conducted monthly cruises of the Yaquina Estuary to examine if similar relationships were present. The cruises included 12 sampling stations extending from the mouth of the estuary to the tidal fresh region. During the dry season, the Yaquina Estuary receives nitrogen from both the riverine and oceanic sources, resulting in a curvilinear relationship in total nitrogen (TN) versus distance, while the ocean is the main source of phosphorous (TP) to the estuary (Figure 5.5). There is also a curvilinear pattern in the chlorophyll *a* versus distance resulting from oceanic input of chlorophyll *a* from the ocean (Figure 5.6). Relationships between dry season nutrients (TN and TP) and chlorophyll *a* are driven by ocean input (rather than a response to watershed nutrient sources) as evident by the significant trends of increasing chlorophyll *a* with increasing nutrients (TN and TP) in Zone 1 but not in Zone 2 (Figure 5.7 and Figure 5.8). Based on these findings, we feel that these types of relationships would not be useful for developing nutrient criteria for the Yaquina Estuary.

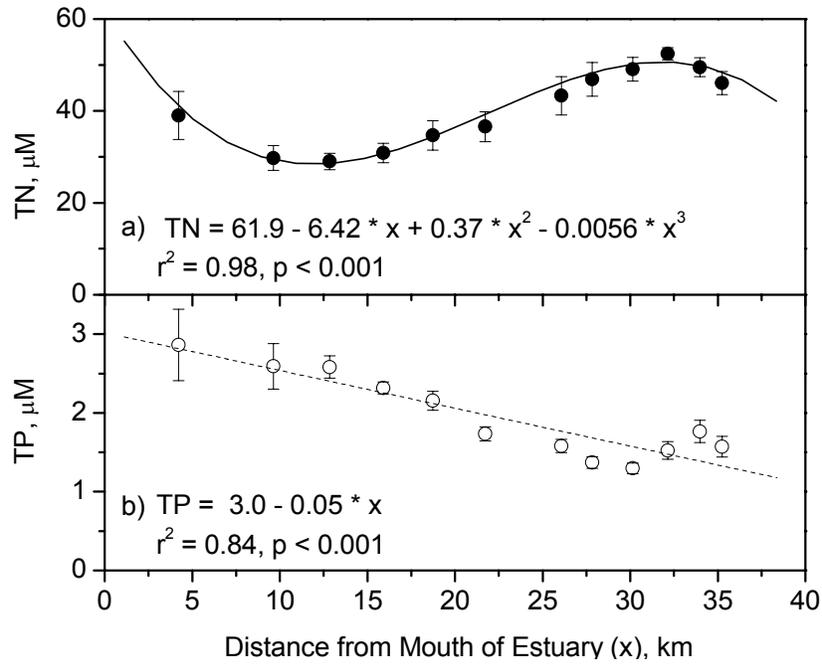


Figure 5.5 Mean 2006 dry season a) total nitrogen (TN) and b) total phosphorous (TP) versus distance from mouth of estuary with error bars representing standard errors (n=12). Solid and dashed lines represent a 3rd order polynomial and linear fit to TN and TP data, respectively.

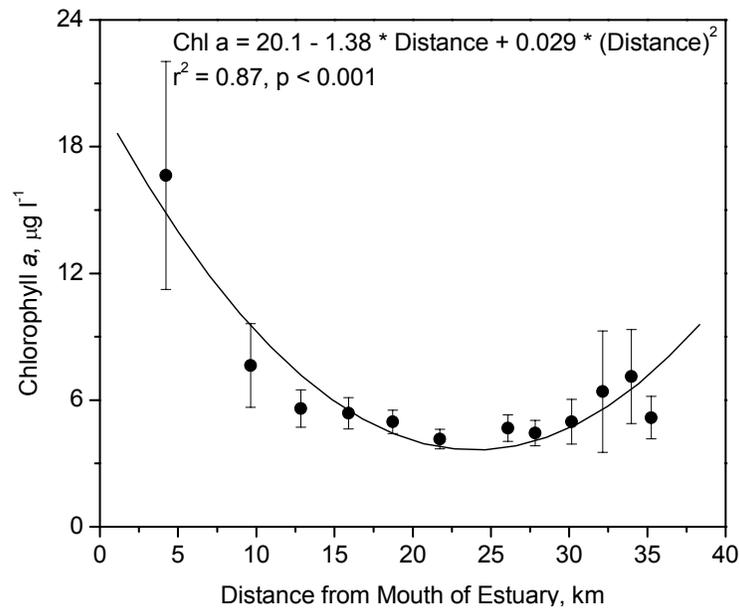


Figure 5.6 Mean dry season (2006) chlorophyll *a* versus distance from mouth of estuary with error bars representing standard error (n = 12) and line representing 2nd order polynomial fit to the data.

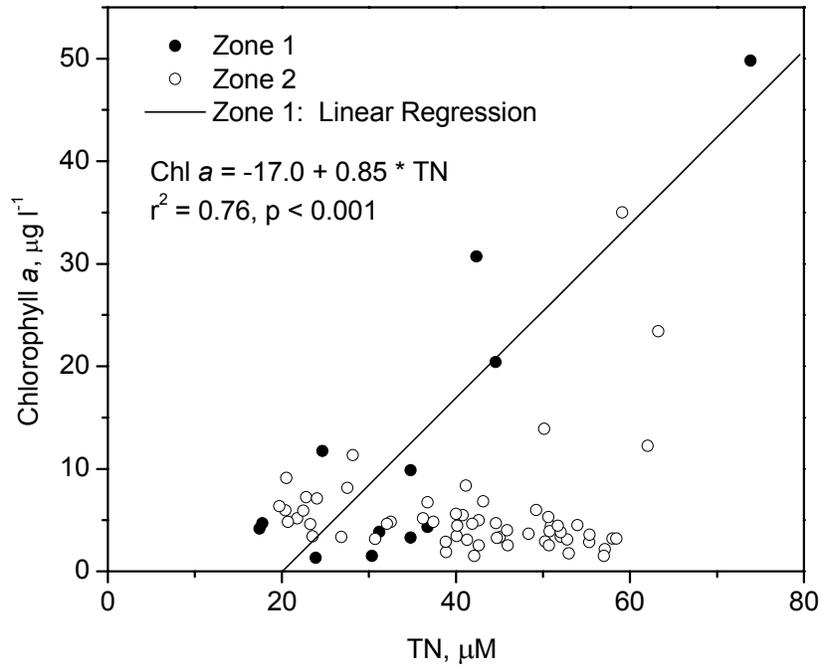


Figure 5.7 Total nitrogen (TN) versus chlorophyll *a* for the dry season (2006) with data divided by zones and solid line showing significant regression for Zone 1. There is not a significant relationship for Zone 2.

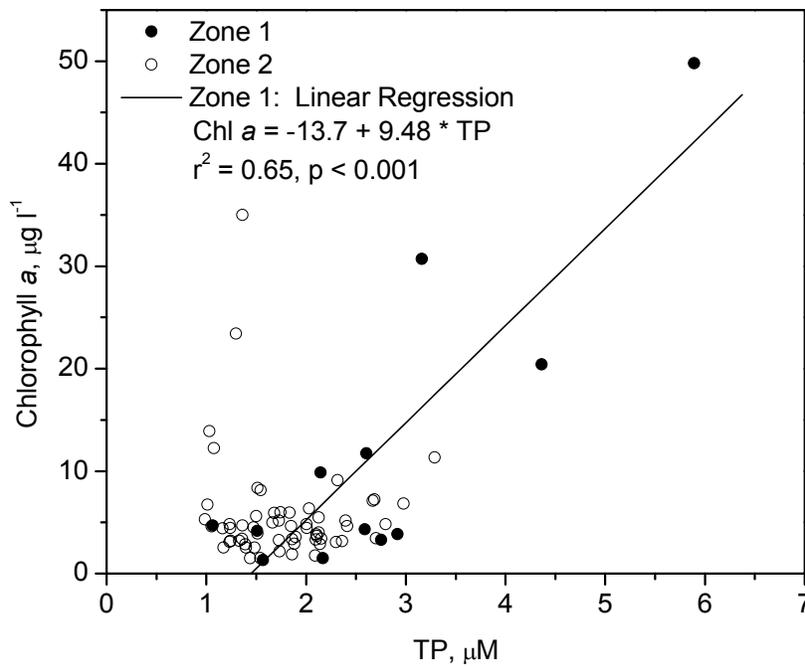


Figure 5.8 Total phosphorous (TP) versus chlorophyll *a* during the dry season (2006) with data divided by zones and solid line showing significant regression for Zone 1. There is not a significant relationship for Zone 2.

6. Nitrogen and Phosphorus as Water Quality Criteria

6.1 Seasonal, Zonal, and Long-term Trends in N and P

There are seasonal differences in water column nutrients within the estuary. DIN levels are significantly higher during the wet season (median = 21.1 μM , n = 874) than during the dry season (dry season median = 13.9 μM , n = 2028; calculated using data from 1998-2006 combining Zones 1 and 2; Mann-Whitney Rank Sum Test, $p < 0.05$). In contrast, PO_4^{3-} levels during the dry season (median = 0.97 μM , n = 2029) are almost twice as high as those during the wet season (median = 0.52 μM , n = 873; calculated using data from 1998-2006 combining Zones 1 and 2; Mann-Whitney Rank Sum Test, $p < 0.05$). During both the dry and wet seasons, Zone 1 has significantly higher PO_4^{3-} concentrations than Zone 2, reflecting the ocean input of phosphorous (Mann-Whitney Rank Sum Test, $p < 0.05$). The ocean input of PO_4^{3-} dominates during the dry season with Zone 1 PO_4^{3-} levels (median = 1.25 μM , n = 1114) twice that of those in Zone 2 (median = 0.64 μM , n = 915). During the dry season, there are no significant differences in DIN levels between Zones 1 and 2 (Mann-Whitney Rank Sum Test, $p > 0.05$). In contrast, during the wet season, DIN levels in Zone 2 (median = 55.2 μM , n = 354) are significantly higher than those in Zone 1 (median = 11.6 μM , n = 520), reflecting the dominance of riverine inputs (Mann-Whitney Rank Sum Test, $p < 0.05$).

Because there are limited historic NH_4^+ data, we were unable to assess whether there are any long-term trends in DIN; however, we do have sufficient historical data to examine trends in $\text{NO}_3^- + \text{NO}_2^-$ (Figures 6.1-6.4), the major component of DIN, and PO_4^{3-} (Figures 6.5-6.8). Recent (1998-2006) dry season $\text{NO}_3^- + \text{NO}_2^-$ and PO_4^{3-} in Zone 1 are significantly higher than historical data (Table 6.1 and Table 6.2; Mann-Whitney Rank Sum, $p < 0.001$). In contrast, historical dry season $\text{NO}_3^- + \text{NO}_2^-$ and PO_4^{3-} in Zone 2 are significantly higher than recent data (Table 6.1 and Table 6.2). Peak wet season $\text{NO}_3^- + \text{NO}_2^-$ concentrations in Yaquina Estuary are similar to wet season NO_3^- observed in Oregon Coast Range streams (peak NO_3^- of 172 μM ; Wigington et al., 1998). The relatively high $\text{NO}_3^- + \text{NO}_2^-$ concentrations that occur in the historic data from Zone 1 dry season (Figure 6.1) are related to an anomalous freshwater inflow event in June 1984 (peak flow of 634 cfs compared to long-term mean for June of 81 cfs; calculated using data from Chitwood gauge on Yaquina River). Historical wet season $\text{NO}_3^- + \text{NO}_2^-$ levels are significantly higher than recent observations in Zones 1 and 2 (Table 6.1;

Mann-Whitney Rank Sum, $p < 0.001$). Wet season PO_4^{3-} in Zone 1 was significantly higher in the historical data set compared to recent, while in Zone 2 there was no difference between recent and historic PO_4^{3-} levels (Table 6.2; Mann-Whitney Rank Sum, $p > 0.05$).

There were no significant trends in $\text{NO}_3^- + \text{NO}_2^-$ within Zones 1 or 2 during either season (Figures 6.1 and 6.2; determined using the Seasonal Kendall and Mann Kendall tests). In Zone 1, there was a significant increasing trend in PO_4^{3-} during the dry season (Figure 6.3a) and a significant decreasing trend in PO_4^{3-} during the wet season (Figure 6.4a), while in Zone 2 there were not significant trends during either the wet or dry season (Figures 6.3b and 6.4b). Due to the opposing seasonal trends in Zone 1, there was not a significant trend in PO_4^{3-} using the Seasonal Kendall test.

Caution needs to be used in interpreting the trends and the differences in historic and recent median $\text{NO}_3^- + \text{NO}_2^-$ and PO_4^{3-} levels (Tables 6.1 and 6.2) due to differences in sampling frequencies. There are considerably more recent data (more stations and higher sampling frequency) than historic data, particularly in Zone 1 during the dry season (Table 6.1 and Table 6.2). Nutrient ($\text{NO}_3^- + \text{NO}_2^-$ and PO_4^{3-}) inputs associated with oceanic sources are highly variable depending upon the wind forcing and respond rapidly to changes in wind forcing. In addition, there is considerable interannual variability in oceanic input to estuaries (Brown and Ozretich, in review) associated with variability in upwelling (Corwith and Wheeler, 2002; Wheeler et al., 2003). During the dry season, the recent nutrient data ($\text{NO}_3^- + \text{NO}_2^-$ and PO_4^{3-}) are consistently higher than the historic data in Zone 1 (Tables 6.1 and 6.2), possibly reflecting either differences in ocean conditions or better characterization of ocean input due to increased sampling frequency in recent data. In contrast, the historic nutrient ($\text{NO}_3^- + \text{NO}_2^-$ and PO_4^{3-}) median levels are higher than recent data in Zone 2 (Tables 6.1 and 6.2). Since the pattern in Zone 2 is opposite to that in Zone 1, this suggests that differences in ocean input are not driving this difference in Zone 2. Caution is needed in interpreting these differences since the trend analysis revealed that there were no significant trends in nutrients ($\text{NO}_3^- + \text{NO}_2^-$ and PO_4^{3-}) in Zone 2. The differences in water column nutrients during the last 30-40 years, although some are statistically significant, do not indicate a major change in nutrient loading (as inferred by nutrient concentrations) as experienced in other estuarine and coastal systems (e.g., Cloern 2001; Soetaert et al., 2006).

Table 6.1 Comparison of historic and recent $\text{NO}_3^- + \text{NO}_2^-$ (μM) concentrations in the Yaquina Estuary. There are statistically significant differences in median concentrations between historic and recent data for all zones and both seasons (Mann-Whitney Rank Sum, $p < 0.001$).				
	Historic		Recent	
	Median	Time Interval (Sample Size)	Median	Time Interval (Sample Size)
Zone 1				
Dry	6.5	1974-1984 (157)	10.0	1998-2006 (1127)
Wet	19.8	1974-1984 (65)	8.5	1998-2004 (520)
Zone 2				
Dry	14.0	1971-1984 (247)	9.8	1998-2004 (919)
Wet	69.6	1971-1984 (148)	52.6	1998-2004 (354)

Table 6.2 Comparison of historic and recent PO_4^{3-} (μM) concentrations in the Yaquina Estuary. There are statistically significant differences in median concentrations between historic and recent data for all zones and both seasons with the exception of Zone 2 wet season (Mann-Whitney Rank Sum, $p < 0.05$).				
	Historic		Recent	
	Median	Time Interval (Sample Size)	Median	Time Interval (Sample Size)
Zone 1				
Dry	1.01	1963-1984 (223)	1.25	1998-2006 (1126)
Wet	0.83	1962 – 1984 (129)	0.59	1998-2004 (519)
Zone 2				
Dry	0.74	1963-1984 (308)	0.65	1998-2004 (919)
Wet	0.54	1962-1984 (212)	0.49	1998-2004 (354)

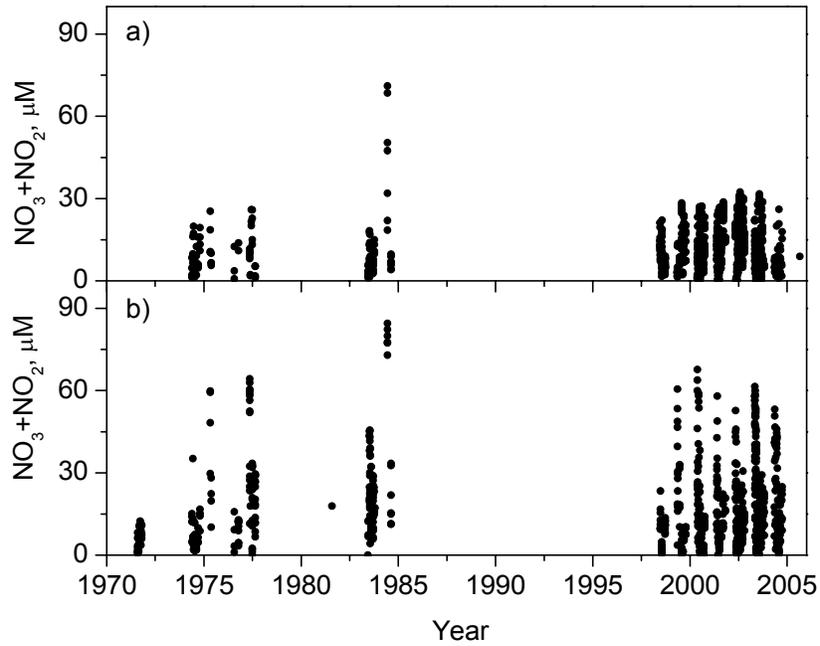


Figure 6.1 Comparison of historic and recent $\text{NO}_3^- + \text{NO}_2^-$ during the dry season in a) Zone 1 and b) Zone 2.

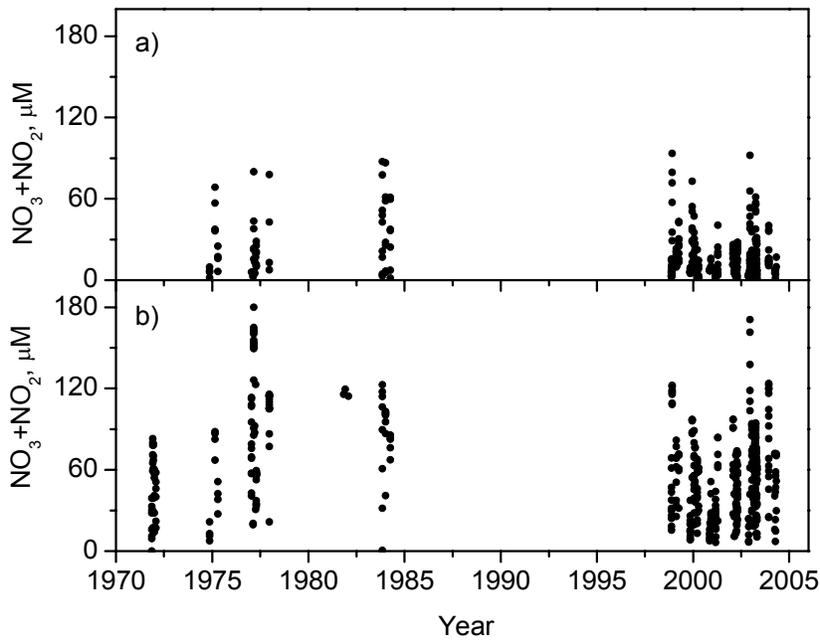


Figure 6.2 Comparison of historic and recent $\text{NO}_3^- + \text{NO}_2^-$ during the wet season in a) Zone 1 and b) Zone 2.

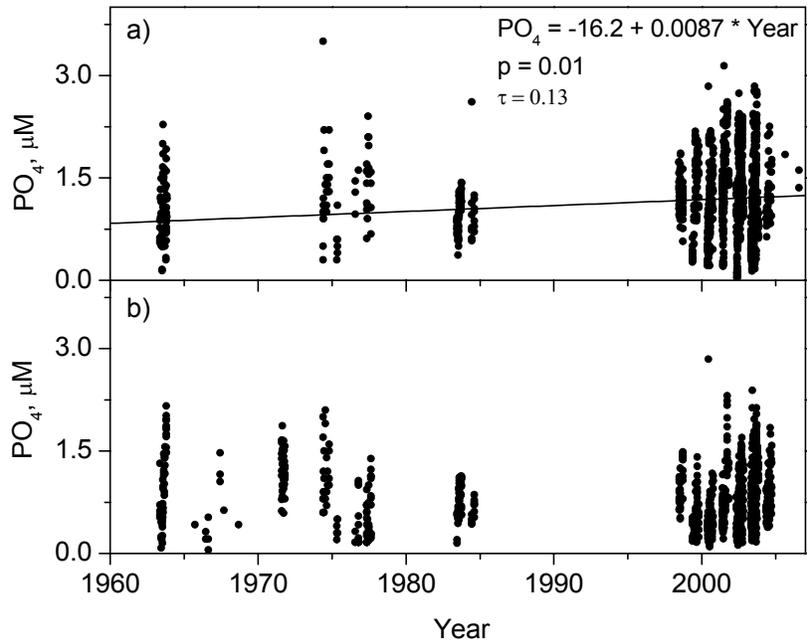


Figure 6.3 Comparison of historic and recent PO_4^{3-} during the dry season in a) Zone 1 and b) Zone 2. The line in the upper panel shows a significant increasing trend in Zone 1 (Mann Kendall, $p = 0.01$).

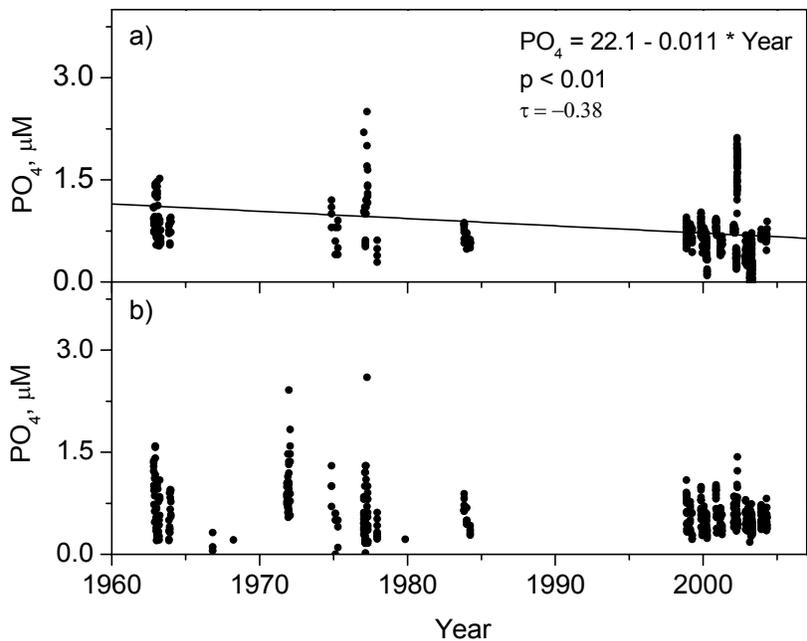


Figure 6.4 Comparison of historic and recent PO_4^{3-} during the wet season in a) Zone 1 and b) Zone 2. The line in the upper panel shows a significant decreasing trend in Zone 1 (Mann Kendall, $p < 0.01$).

6.2 Percentile Approach for Nitrogen and Phosphorous

There were significant differences in median DIN and PO_4^{3-} values between the Yaquina (dry season), Classification Study, and NCA datasets for both Zones 1 and 2 (Kruskal-Wallis one way ANOVA on ranks, $p < 0.001$). The dry season DIN concentrations observed for Yaquina Estuary (Zones 1 and 2) were significantly higher than those observed in the NCA data set both with and without the Columbia included (Table 6.3, Dunn's method for pairwise comparison, $p < 0.05$); however, there was not a significant difference between dry season DIN levels in the Yaquina Estuary (Zones 1 and 2) and those observed in the Classification data set (Table 6.3). The PO_4^{3-} levels were significantly higher in the Yaquina Estuary (Zones 1 and 2) than those observed in the Classification and NCA data sets (Table 6.4, Dunn's method for pairwise comparisons, $p < 0.05$). The higher DIN and PO_4^{3-} levels in Zone 1 in the Yaquina Estuary as compared to the NCA data set is probably an artifact of sampling (both time of sampling and differences in the number of samples). In Zone 1, water column nutrients are dependent upon ocean conditions at the time of sampling. Inspection of the sampling dates during the 1999 NCA field effort, reveals that 82% of the estuaries were sampled during a time period of low nutrient conditions in the coastal ocean (determined using flood tide water temperature at Yaquina Estuary and a relationship generated between flood tide water temperature and $\text{NO}_3^- + \text{NO}_2^-$, for details see Lee et al., 2006).

Zone 2 dry season DIN levels in the Yaquina Estuary are comparable to values for streams measured in Level III Ecoregion No. 1 - Coast Range (summer median $\text{NO}_3^- + \text{NO}_2^- = 12 \mu\text{M}$; U.S. EPA, 2000). The PO_4^{3-} levels in Yaquina Estuary, particularly during the dry season and in Zone 1, are higher than Level III Ecoregion No. 1 - Coast Range values for streams (median = 0.28- 0.60 μM ; U.S. EPA, 2000), due to the input of PO_4^{3-} from oceanic sources. Wet season DIN concentrations in Zone 2 of Yaquina Estuary are similar to wet season NO_3^- observed in Oregon Coast Range streams (median $\text{NO}_3^- = 56 \mu\text{M}$; Wigington et al., 1998) and streams data for Level III Ecoregion No. 1 - Coast Range (winter median $\text{NO}_3^- + \text{NO}_2^-$ of 37 μM ; U.S. EPA, 2000). Higher PO_4^{3-} levels in Zone 1 compared to Zone 2 are present in the NCA and classification data sets, demonstrating that oceanic input of PO_4^{3-} occurs at a regional scale. The DIN and PO_4^{3-} levels in Oregon estuaries would be considered to be medium levels using criteria from Bricker et al. (2003); however, based on analysis of sources (see Section 3.2) we believe that the high DIN and PO_4^{3-} levels are associated with natural

sources (i.e., red alder in the watershed and oceanic input) rather than anthropogenic sources. Systems in the PNW appear to have relatively high background levels of DIN and DIP compared to other estuaries in the U.S. (U.S. EPA, 2004a).

Table 6.3 Percentiles for DIN (μM) calculated using Yaquina (1998-2006), Classification (2004-2005), and NCA Oregon estuaries (1999-2000) data sets. NCA and Classification values are for dry season only, while Yaquina data include values for dry and wet seasons.				
Data set	Percentiles for DIN (μM)			Sample Size
	25 th	50 th	75 th	
Zone 1				
Yaquina				
Dry	8.3	14.1	20.4	1113
Wet	6.8	11.6	19.1	520
Classification	5.7	11.1	18.6	68
NCA	6.0	8.6	11.8	36
NCA excluding Columbia	5.8	8.4	11.8	33
Zone 2				
Yaquina				
Dry	7.3	13.7	23.1	915
Wet	30.7	55.2	73.5	354
Classification	8.3	14.0	36.9	88
NCA	7.0	9.4	13.2	89
NCA excluding Columbia	4.8	7.2	11.9	27

Table 6.4 Percentiles for PO_4^{3-} (μM) using Yaquina (1998-2006), Classification (2004-2005), and NCA Oregon estuaries (1999-2000) data sets. NCA and Classification values are for dry season only, while Yaquina include dry and wet season values.				
Data set	Percentiles for PO_4^{3-} (μM)			Sample Size
	25 th	50 th	75 th	
Zone 1				
Yaquina				
Dry	0.88	1.25	1.69	1114
Wet	0.39	0.59	0.77	519
Classification	0.62	0.89	1.20	68
NCA	0.76	0.95	1.15	36
NCA excluding Columbia	0.76	1.00	1.16	33
Zone 2				
Yaquina				
Dry	0.43	0.64	0.99	915
Wet	0.41	0.49	0.62	354
Classification	0.33	0.45	0.75	88
NCA	0.35	0.52	0.71	89
NCA excluding Columbia	0.18	0.33	0.73	27

7. Chlorophyll *a* as a Water Quality Response Measure

7.1 Seasonal, Zonal, and Long-Term Trends in Chlorophyll *a*

Chlorophyll *a* is often used as a surrogate for phytoplankton biomass and as an indicator of trophic status in eutrophication assessments (Bricker et al., 1999). There were seasonal differences in water column chlorophyll *a* in the estuary. Peak chlorophyll *a* levels occurred during the months of June to August (Figure 7.1). Seasonal chlorophyll *a* patterns were likely related to light limitation and flushing (as discussed in Section 5.2). The median wet season chlorophyll *a* was $1.6 \mu\text{g l}^{-1}$ ($n = 293$), while during the dry season the median increased to $4.9 \mu\text{g l}^{-1}$ ($n = 1205$). Dry season chlorophyll *a* levels in Zone 2 (Median = $5.3 \mu\text{g l}^{-1}$, $n = 229$) were significantly higher than those in Zone 1 (Median = $3.3 \mu\text{g l}^{-1}$, $n = 347$; Mann Whitney Rank Sum, $p < 0.001$). During the late spring, there have been recurrent non-toxic red tide blooms of *Myrionecta rubra* in the vicinity of Toledo. During the dry season, chlorophyll *a* concentrations occasionally reached $15 \mu\text{g l}^{-1}$ in the vicinity of Toledo (8% of the recent observations). In the tidal fresh portion of the estuary, there were recurrent algal blooms during June and July, with chlorophyll *a* concentrations reaching $80 \mu\text{g l}^{-1}$.

There are limited historical data to assess long-term trends in chlorophyll *a*. Comparison of historic (1973-1983) and recent (2000-2006) chlorophyll *a* levels during the dry season reveal that there has been a decline in median chlorophyll *a* levels in both zones (Figure 7.2); although these declines are statistically significant (Mann Whitney Rank Sum, $p \leq 0.001$) they do not indicate a shift in trophic status of the estuary. There was a statistically significant decreasing trend in dry season chlorophyll *a* in Zone 1 (Mann Kendall, $p < 0.001$), while in Zone 2 there was no significant trend. The changes in chlorophyll *a* that occurred in the Yaquina Estuary are small in magnitude ($1 \mu\text{g l}^{-1}$) compared to changes that have occurred in other estuaries (Cloern, 2001; Harding and Perry, 1997). For example, in Chesapeake Bay chlorophyll *a* levels increased 5- to 10-fold in the lower portion of the estuary during the interval of 1950-1994 (Harding and Perry, 1997). A statistically significant zonal difference in chlorophyll *a* levels (Zone 2 higher than Zone 1) is present in the historic data as well as the recent data (Mann Whitney Rank Sum, $p \leq 0.001$). Peak chlorophyll *a* levels in Zone 1 appear to be higher in the recent data compared to the historic; however, this is probably an artifact of sampling frequency. Blooms imported into Zone 1 from the coastal ocean are episodic in nature, reflecting the variability in wind forcing. Peak chlorophyll *a* levels in Zone 2 are similar for the historic and recent data.

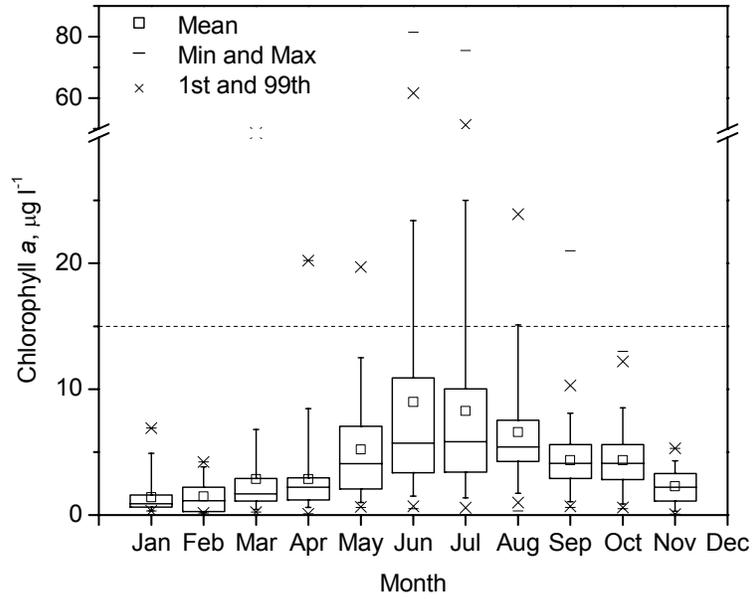


Figure 7.1. Box plot of monthly chlorophyll *a* data from the Yaquina Estuary (all stations from 1973-2006). The dashed line indicates the Oregon estuarine chlorophyll *a* criterion. The boxes represent the 25th and 75th percentiles, the whiskers represent the 5th and 95th percentiles, and the horizontal line is the median.

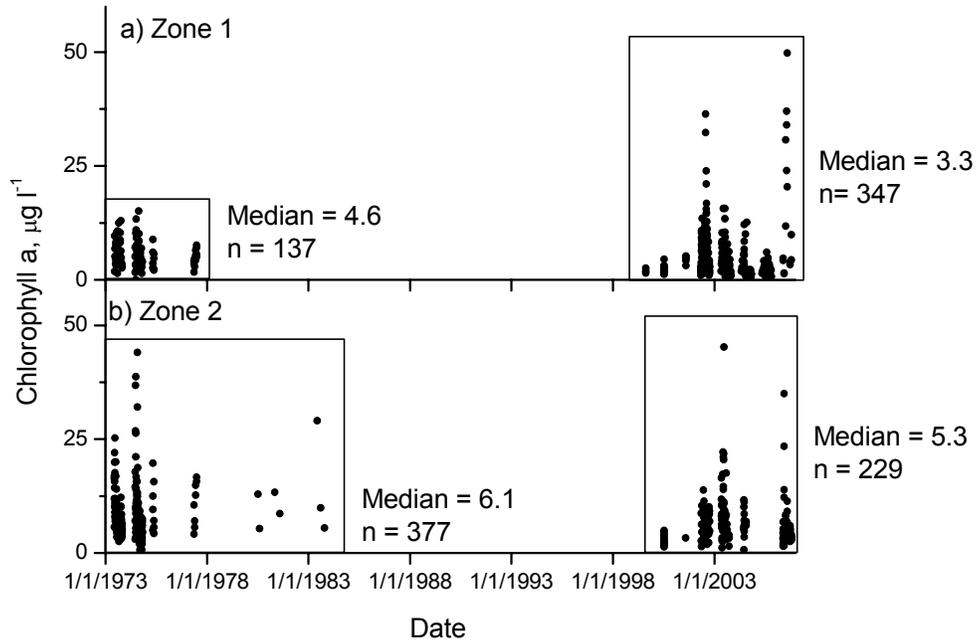


Figure 7.2 Comparison of historic and recent dry season chlorophyll *a* for a) Zone 1 and b) Zone 2 in the Yaquina Estuary. The boxes indicate the time interval for the historic and recent median calculations.

7.2 Percentile Approach for Chlorophyll *a*

The chlorophyll *a* levels in Oregon estuaries, including Yaquina Estuary, are relatively low with median values of 2 – 5 $\mu\text{g l}^{-1}$ (Table 7.1). These chlorophyll *a* levels would be in the ‘low’ category when used as an indicator of eutrophication (Bricker et al., 1999) and in the ‘good’ category using the West Coast criteria for water quality parameters from the National Coastal Condition Report (US EPA, 2004a). The Oregon chlorophyll *a* criterion of 15 $\mu\text{g l}^{-1}$ is exceeded 4% of the time during the dry season in Zones 1 and 2. At Elk City (tidal fresh part of the estuary) the 15 $\mu\text{g l}^{-1}$ criterion is exceeded 28% of the time during the dry season (WED unpublished data; collected during 2002 and 2003). There was a significant difference in median chlorophyll *a* between the Yaquina (dry season), Classification Study, and NCA datasets for both Zones 1 and 2 (Kruskal-Wallis one way ANOVA on ranks, $p < 0.001$; Table 7.1). Dry season chlorophyll *a* levels in both zones of the Yaquina Estuary are significantly higher than those found in the other Oregon estuaries sampled in the NCA and Classification datasets (Dunn’s method for pairwise comparison, $p < 0.05$). Although chlorophyll *a* levels in the Yaquina Estuary are significantly higher than for the other Oregon estuaries sampled, they are ‘low’ compared to many other U.S. estuaries (US EPA, 2004a).

Table 7.1 Percentiles for chlorophyll <i>a</i> ($\mu\text{g l}^{-1}$) calculated using Yaquina (1998-2006), Classification (2004-2005), and NCA Oregon estuaries (1999-2000) data sets. NCA and Classification values are for dry season only, while Yaquina include dry and wet season values.				
Data set	Percentiles for Chlorophyll <i>a</i> ($\mu\text{g l}^{-1}$)			Sample Size
	25 th	50 th	75 th	
Zone 1				
Yaquina				
Dry	2.2	3.3	5.7	347
Wet	0.6	1.1	1.7	95
Classification	1.0	2.0	3.8	68
NCA	1.5	2.1	3.8	36
NCA excluding Columbia	1.5	2.0	3.2	33
Zone 2				
Yaquina				
Dry	3.8	5.3	7.9	229
Wet	0.4	0.9	2.5	46
Classification	0.8	1.6	2.5	78
NCA	2.0	3.3	4.9	89
NCA excluding Columbia	1.4	1.8	2.6	27

8. Dissolved Oxygen as a Water Quality Response Measure

Dissolved oxygen (DO) is an important water quality metric because of its effects on the well-being of estuarine resident and transitory organisms. Salmon and trout are particularly esteemed fishes in the PNW and traverse the estuaries during upstream and downstream migrations. The dissolved oxygen criterion for Oregon's estuaries and streams focuses on the oxygen concentration needed for these fish because of their socioeconomic importance and their requirement for comparatively high oxygen levels. As a result, "salmon and trout rearing and migration" is a common designated use for Oregon coastal estuaries, including the Yaquina Estuary and River.

Two species of salmon that are of particular importance in the Yaquina are the coho salmon (*Onchorhynchus kisutch*) and steelhead trout (*O. mykiss*). The reduced size of Oregon coastal coho populations have been a cause of particular concern. Chinook salmon (*O. tshawytscha*) spawn and occur as juveniles in brackish waters, and are also present in the Yaquina River. The cutthroat trout (*O. clarki*) is a fourth important salmonid species found in the Yaquina system, and a portion of this population also follows the salmon life history of migrating to the sea, where it grows to adulthood before returning to the natal stream to spawn. The timing of salmonid migrations through the estuaries varies by species, and is influenced by local conditions and hydrology. However, adults generally enter the estuary in the fall and progress upstream to freshwater spawning streams. The juvenile outmigrants, termed "smolts," typically move downstream during the months of March to June.

8.1 Seasonal, Zonal and Long-term Trends in Dissolved Oxygen

There are strong seasonal patterns in dissolved oxygen within the Yaquina Estuary (Figure 8.1). Oxygen levels (expressed as both mg l^{-1} and % saturation) in the estuary are comparatively stable during the wet season, but show a decline during the dry season. The wet season dissolved oxygen have an overall mean value of 9.7 mg l^{-1} ($n = 869$) dissolved oxygen. The dry season data were fitted using a nonlinear least squares procedure to have a descending cosine curve that begins at the wet season value of 9.7 mg l^{-1} , declines to a value of 5.8 mg l^{-1} on August 2, and then returns to the wet season value. Zones 1 and 2 appear to follow the same pattern. Subsequent data analyses used the deviations from this modeled seasonal pattern (the solid line in Figure 8.1), so that the overall seasonal changes in oxygen concentration and differences in sampling would not confound more detailed analyses. All dissolved oxygen values used in the analyses were collected during daylight hours. Therefore, diel cycling of oxygen values due to plant photosynthesis and respiration are not

represented in the data. Nighttime respiration can significantly reduce water column oxygen levels below daytime levels.

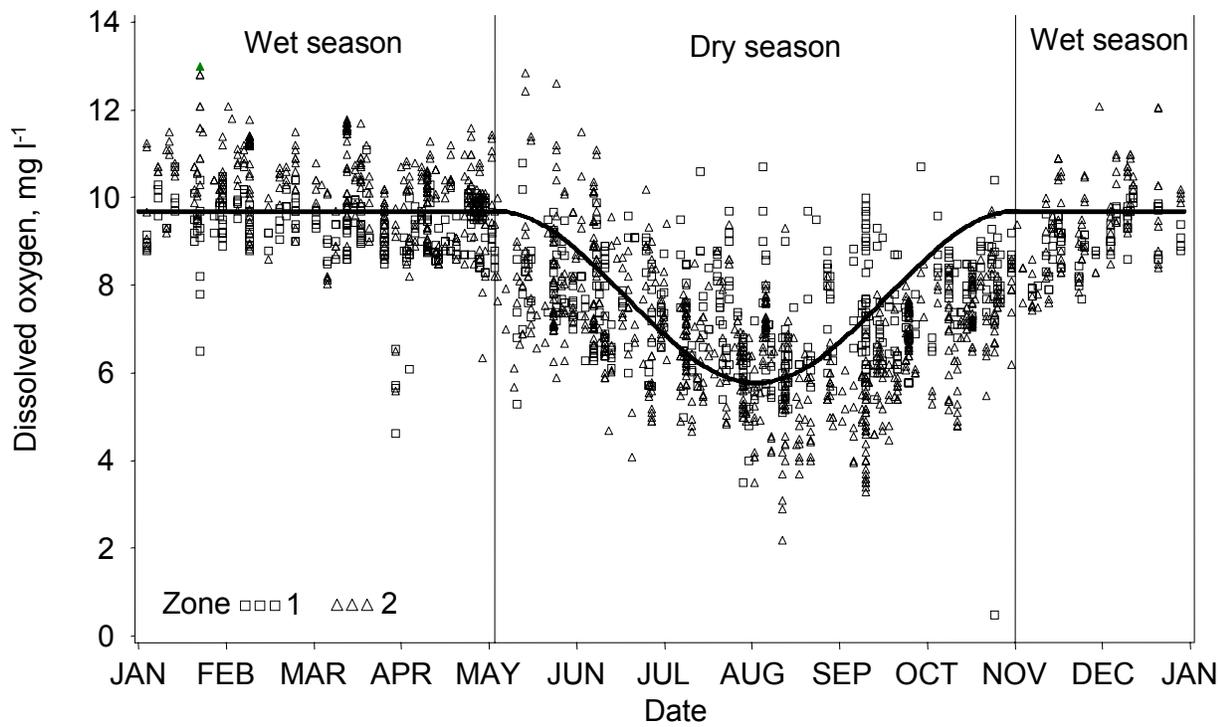


Figure 8.1 Seasonal pattern in dissolved oxygen at all locations and all years in the Yaquina Estuary and River with squares and triangles representing samples from Zones 1 and 2, respectively. Solid line is nonlinear least-squares fit to data, which was modeled as a constant during wet season and a cosine function of date during the dry season.

During the interval of 1960-1984, there was a significant trend of increasing DO in Zone 2 during both the dry and wet seasons (Figure 8.2b; Mann Kendall, $p < 0.05$). In addition, there was a significant seasonal trend in Zone 2 (Seasonal Kendall, $p < 0.05$). Similar significant trends were found regardless of whether dissolved oxygen was expressed as non-transformed, residual, or percent saturation. A report by the Federal Water Pollution Control Administration (1966) stated that the water quality in the lower portion of the Yaquina basin was “adversely affected by existing and man-made conditions,” including “inadequately treated wastes from municipalities and industries” that placed “an excessive demand on oxygen resources of Yaquina Bay during annual periods of low streamflow.” In 1956, the City of Toledo upgraded their wastewater treatment facility to primary

treatment (prior to this raw sewage was discharged into the estuary), and in 1981 it was upgraded to secondary treatment.

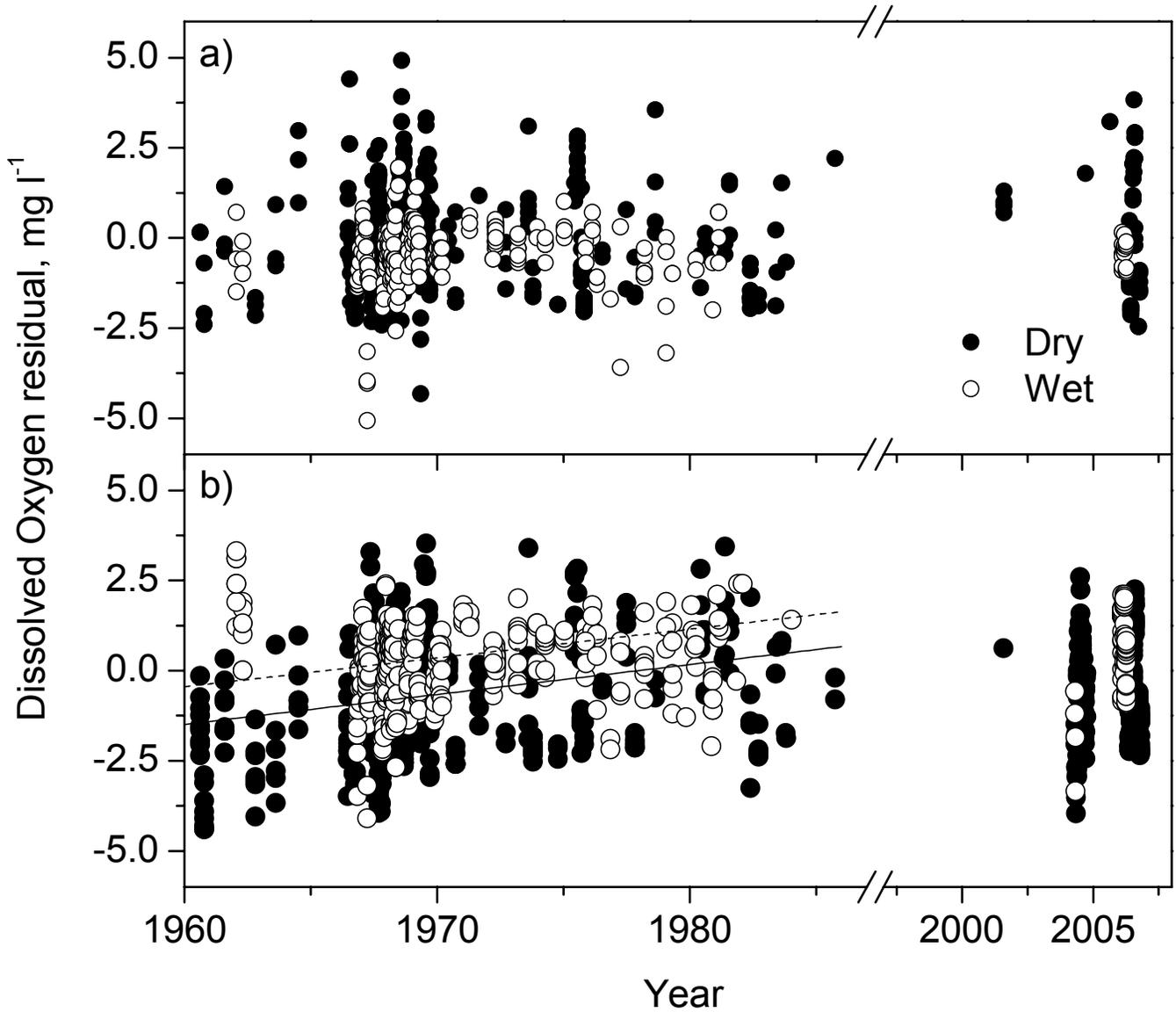


Figure 8.2 Interannual trend in residual dissolved oxygen values during 1960 to 1986 for a) Zone 1 and b) Zone 2. Zone 2 regressions are significant at the $p < 0.05$ level, while Zone 1 regressions are not. The solid and dashed lines represent the significant dry and wet season trends, respectively (Mann Kendall, $p < 0.05$). Data from recent years are also shown for comparison, but were not included in regression computations.

There was also a decline of log rafting in the Yaquina Estuary from 1962 through the 1980s (Seddell and Duval, 1985). One effect on the water column of bark debris associated with log rafts is increased biochemical oxygen demand (Seddell and Duval, 1985). Due to the multiple stressors on the Yaquina Estuary during this time period, there is no way to determine the cause of the observed trend in DO levels in Zone 2. Recent (2002-2006) DO levels in Zone 2 are similar to DO levels during the mid 1980's, suggesting that there has been no recent changes in DO levels. In contrast, there were no significant trends in dry or wet season DO in Zone 1 (Figure 8.2a), suggesting that the trend in historic DO levels in Zone 2 was not a result of differences in ocean conditions.

Since 2002 there has been an increase in the incidence of hypoxic events on the Oregon shelf (Grantham et al., 2004), which have the potential to influence DO levels within the estuary (particularly Zone 1). DO data collected 3.7 km from mouth of the estuary (using a YSI datasonde deployed at a mean depth of 1 m below the surface; WED, unpublished data) demonstrate that there is import of hypoxic shelf water into Yaquina Estuary during flood tides. A time series of DO and salinity measured during July 9-19, 2002, coinciding with a documented hypoxic event on the Oregon shelf off of Newport, Oregon (Grantham et al., 2004), clearly shows import of hypoxic shelf water to the estuary (Figure 8.3a). Minimum DO levels occurred during maximum salinities, demonstrating that the hypoxic water was imported into the estuary during flood tides. In addition, minimum DO levels occur during minimum water temperatures (~ 9 deg C), which is indicative of recently upwelled water. This trend of increasing DO with increasing temperature is opposite the trends of solubility, suggesting that differences in solubility are not causing the observed variability in DO levels. During this 10-day interval, minimum DO levels were 0.42 mg l⁻¹. The intervals of low DO conditions were relatively short, with DO levels increasing to 6-8 mg l⁻¹ during ebb tides. The DO versus salinity plot (Figure 8.3b) shows that low DO levels occurred at high salinities (> 33 psu). A plot of dissolved oxygen expressed as percentage of saturation versus salinity had a similar pattern to that presented in Figure 8.3b, demonstrating that differences in solubility of dissolved oxygen are not the cause of the variability.

The import of hypoxic shelf water into Oregon estuaries is not a recent phenomena. Gibson (1974) found low dissolved oxygen (5 mg l⁻¹) in the lower Yaquina Estuary during July 1968, which he attributed to coastal upwelling. Callaway observed the intrusion of low dissolved oxygen (< 2 mg l⁻¹) into the Umpqua Estuary (as cited in Percy et al., 1974). The NCA data set is also suggestive of import of low dissolved oxygen at a regional scale as indicated by lower DO values in Zone 1

compared to Zone 2. In a review of dissolved oxygen conditions in Oregon estuaries (ODEQ, 1995), the opposite spatial pattern was found, with minimum DO levels occurring near the upper end of salt water intrusion and higher concentrations associated with inflow of ocean water. They also stated that greater frequency of low DO would be expected if sampling occurred near the upper extent of salt water intrusion. Our results demonstrate that this may not be the case.

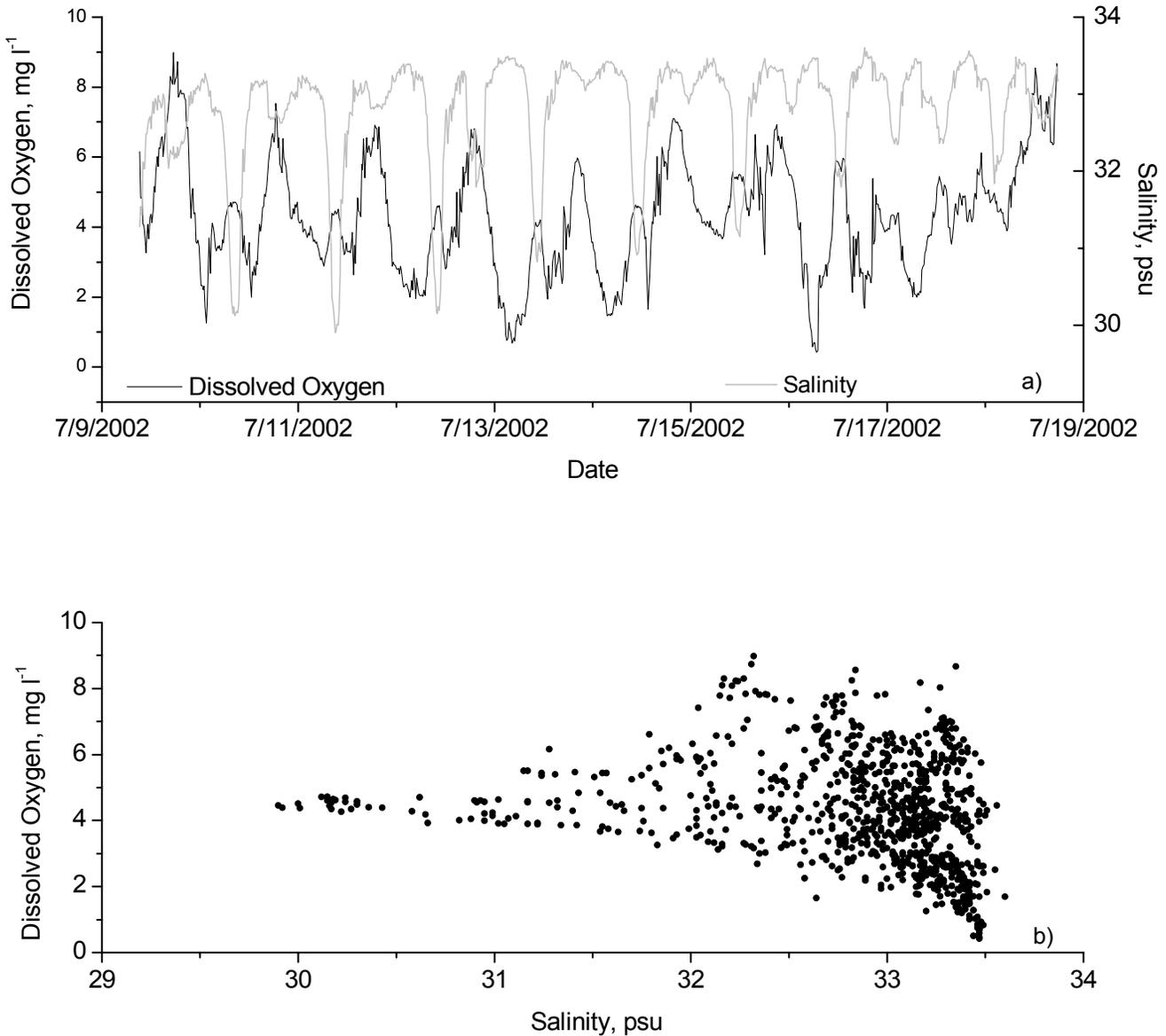


Figure 8.3 a) Time-series of dissolved oxygen and salinity and b) salinity versus dissolved oxygen showing import of hypoxic ocean water at a station 3.7 km from mouth of estuary.

8.2 *Percentile Approach for Dissolved Oxygen*

Dry season DO levels in the Yaquina Estuary are comparable to those found in other Oregon estuaries (Table 8.1) and are relatively high compared to other estuaries in the U.S. (U.S. EPA, 2004a). There was not a significant difference in dissolved oxygen levels in Zone 1 of the Yaquina Estuary (dry season using discrete samples) and Zone 1 of the other estuaries sampled in the Classification and NCA data sets (Kruskal-Wallis one way ANOVA on ranks, $p > 0.05$). The dissolved oxygen levels in Zone 2 of the Yaquina Estuary (dry season using discrete samples) are significantly lower than those in Zone 2 of the estuaries sampled in the Classification and NCA (including Columbia) datasets (Dunn's Method for pairwise comparison, $p < 0.05$). In the Yaquina Estuary during the dry season, the DO levels do not meet the Oregon criterion of 6.5 mg l^{-1} for 25% and 19% of the time in Zones 1 and 2, respectively (using discrete samples). There was not a statistically significant difference in dry season median DO levels between Zones 1 and 2 (calculated using recent discrete data; Mann-Whitney Rank Sum, $p > 0.05$). During the wet season, DO conditions do not appear to be a cause for concern.

There is considerable temporal variability in DO levels, which is not well captured in discrete point measurements. Continuous data are valuable in that they provide insight into the processes influencing observations such as the import of hypoxic water (Section 8.1), and they can allow evaluation of DO levels during both day and night conditions. Continuous data were available from datasondes deployed at two locations (Zone 1 - 3.7 km from the estuary mouth, Zone 2 - 18 km from the estuary mouth) in the estuary. The data were inspected to ensure that biofouling was not influencing observations, and only data from the first 7 days of each deployment was included in the analysis. In Table 8.1 we present the percentiles of the discrete and continuous data for the dry season for comparison; however, we did not perform formal statistical analyses due to the large difference in sample size. Median DO levels are lower for the continuous data compared to the discrete data, particularly in Zone 1

Using the continuous data, we examined how often the State of Oregon DO criterion was not met during May-October of 2006. Dissolved oxygen levels fell below the 6.5 mg l^{-1} criterion 37% and 28% of the time in Zones 1 and 2, respectively. The frequencies that observations fall below the criterion in the two zones are comparable but slightly higher than those calculated from the discrete samples (Table 8.1). A plot of salinity versus DO for the 2006 datasonde data has a pattern of low DO at high salinities (similar to that presented in Figure 8.3b) for the station in Zone 1; however, this pattern is absent in the datasonde data from the Zone 2 station. Based on these patterns, the below

criterion observations in Zone 1 are probably related to the oceanic import of hypoxic water, but possibly not in Zone 2. In the continuous data, there is not significant difference in dry season DO levels (median =7.0 mg l⁻¹) between the two locations (Mann Whitney Rank Sum, p>0.05). However, median DO levels expressed as percentage of saturation are significantly lower in Zone 1 than in Zone 2 (Mann Whitney Rank Sum, p<0.001). The ODEQ (1995) review of the state DO criterion notes that in some bays, the 6.5 mg l⁻¹ criterion may not always be achievable due to natural background conditions. This conclusion is consistent with the analyses in this section.

Table 8.1 Percentiles for dissolved oxygen (mg l ⁻¹) calculated Yaquina (1998-2006), Classification (2004-2005), and NCA Oregon Estuaries (1999-2000) data sets. NCA and Classification values are for dry season only, while Yaquina include dry and wet season values.				
Data set	Percentiles for Dissolved Oxygen (mg l ⁻¹)			Sample Size
	25 th	50 th	75 th	
Zone 1				
Yaquina				
Dry				
Discrete	6.5	7.8	8.5	46
Continuous	5.9	7.0	8.0	2856
Wet	9.0	9.4	9.5	36
Classification	6.7	7.3	8.3	37
NCA	7.2	7.8	8.4	35
NCA excluding Columbia	7.3	8.0	8.5	32
Zone 2				
Yaquina				
Dry				
Discrete	6.7	7.2	7.6	259
Continuous	6.4	6.9	7.5	2862
Wet	10.2	11.0	11.6	184
Classification	6.9	8.1	9.3	53
NCA	7.6	8.5	9.0	88
NCA excluding Columbia	6.7	7.3	8.3	26

9. Water Clarity (k_d) and Turbidity as Water Quality Response Measures

9.1 Seasonal and Zonal Patterns in Water Clarity and Turbidity

A turbidity maximum occurs about 15 km from the mouth of the Yaquina Estuary (Figure 9.1). The water is relatively clear throughout the year in the lower estuary due to the input of ocean water, which is evident in the low turbidities and light attenuations near the mouth (0 – 5 km; Figures 9.1 and 9.2). Turbidity tends to decrease upriver of the turbidity maximum (Figure 9.1). There is a significant increase in light attenuation with distance from the mouth of the estuary during both the wet and dry seasons (Figure 9.2). *In situ* light attenuation measured at fixed stations up estuary from the turbidity maximum at 16 and 19 km from the mouth was generally greater than for fixed sites at 4 and 9 km from the mouth (Figure 9.3). This difference is not as clearly seen in the turbidity data set (Figure 9.1).

There was not a significant difference in dry and wet season total suspended solids (TSS) with median values of 7.8 and 8.9 mg l⁻¹, respectively (Zones 1 and 2 combined; Mann Whitney Rank Sum, $p > 0.05$). TSS levels in Zone 2 (Median = 11.7 mg l⁻¹, $n = 119$) were significantly higher than those in Zone 1 (Median = 6.5 mg l⁻¹, $n = 158$; Mann Whitney Rank Sum, $p < 0.001$). Light attenuation is positively correlated with turbidity ($r^2 = 0.70$, $n = 1400$), but not with chlorophyll *a*.

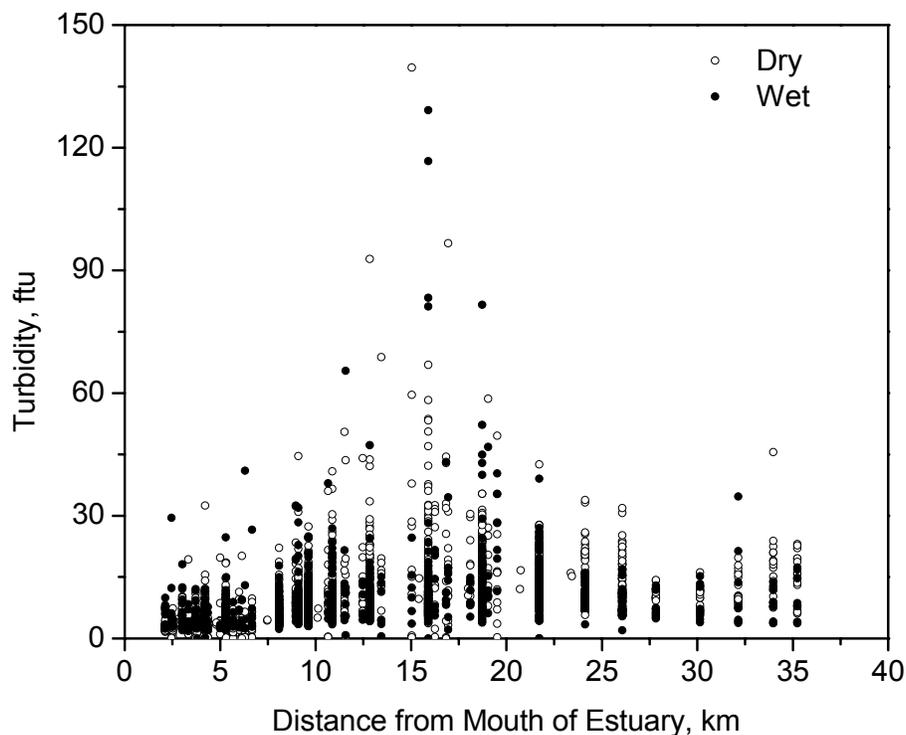


Figure 9.1. Spatial variation in turbidity during wet and dry seasons (1998-2006).

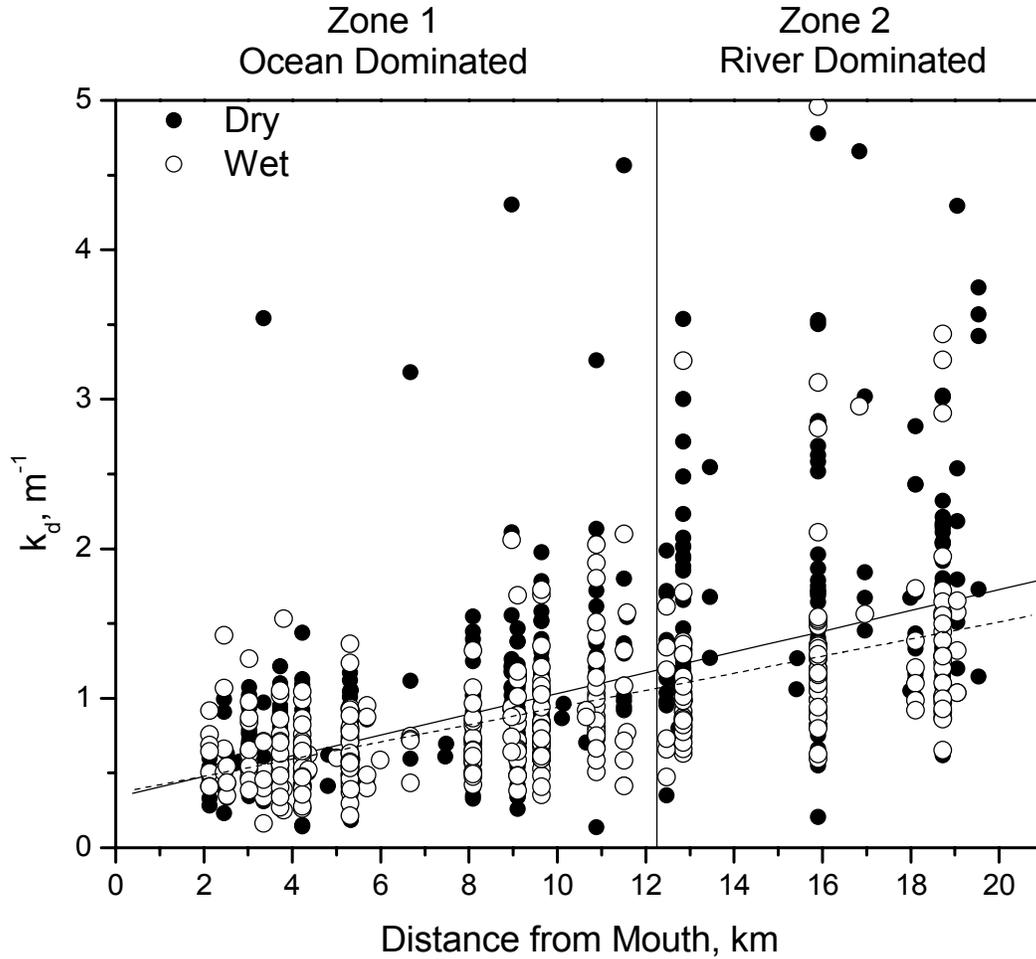


Figure 9.2 Light attenuation coefficients (k_d) versus distance from the mouth of the estuary from cruise data (years 1998 to 2006), with filled and open symbols representing dry and wet seasons, respectively. Solid line - dry season regression ($k_d = 0.34 + [0.069 * \text{Distance}]$, $r^2 = 0.33$, $p < 0.001$). Dashed line - wet season regression ($k_d = 0.36 + [0.057 * \text{Distance}]$, $r^2 = 0.30$, $p < 0.001$). While dry and wet season light attenuation are significantly correlated with distance, their regression coefficients are not different. Combining both seasonal data sets yields ($k_d = 0.34 + [0.066 * \text{Distance}]$, $r^2 = 0.32$, $p < 0.001$).

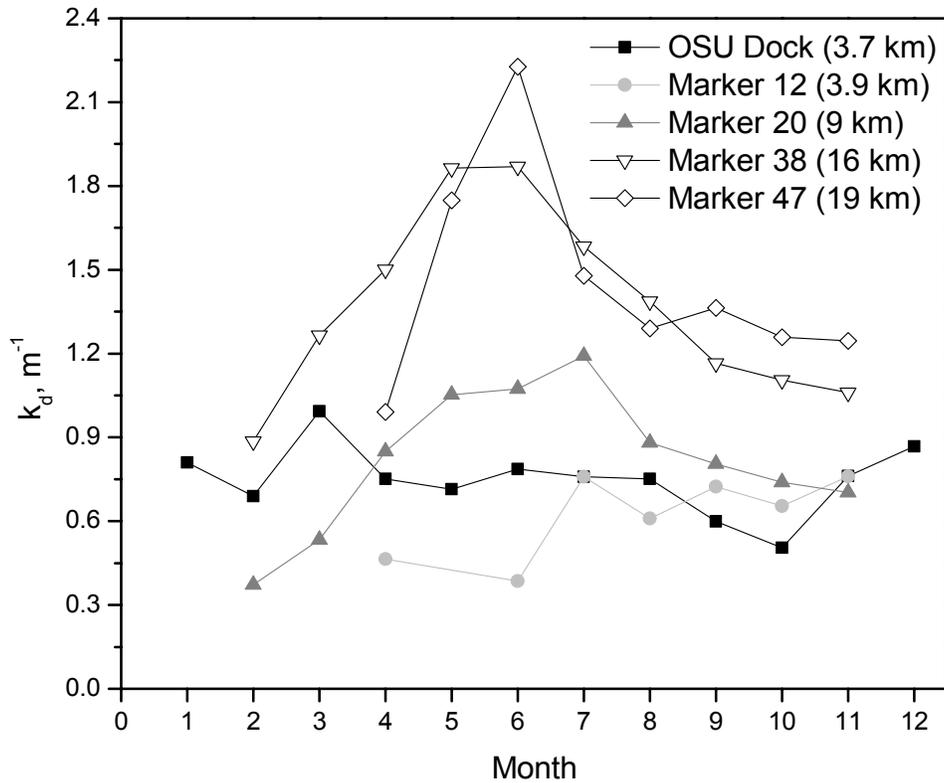


Figure 9.3. Median monthly light attenuation coefficients from the continuous data set at 5 locations in Yaquina Estuary (1999-2003).

9.2 Percentile Approach for Water Clarity and TSS

Due to methodological differences (NCA data set) and missing data (Classification data set), we calculated CDFs for light attenuation for only the Yaquina Estuary data sets. During the wet season, there is not a significant difference between light attenuation coefficients (k_d) computed from the continuous and cruise data within each zone (Table 9.1; Mann Whitney Rank Sum, $p > 0.05$). During the dry season, the light attenuation coefficients computed using the continuous data are significantly higher than those computed from the cruise data (Mann Whitney Rank Sum, $p < 0.001$), and this difference is greater in Zone 2 (median about 16% higher for continuous) than in Zone 1 (median about 8% higher). Median light attenuation coefficients within Zone 1 during the dry season were significantly higher (5-8% for continuous and cruise data, respectively) than those from the wet season (Mann Whitney Rank Sum, $p < 0.05$). Within Zone 2, median light attenuation coefficients were significantly higher (10-27% for cruise and continuous data, respectively) in the dry season than in the wet season (Mann Whitney Rank Sum, $p < 0.001$).

For the Yaquina Estuary, median values of TSS were similar within Zones between wet and dry seasons, but Zone 1 median values were approximately half that of Zone 2 values (Table 9.2). This pattern is consistent with the presence of a turbidity maximum at 14 km up the estuary, within Zone 2. There was a significant difference in median TSS between the Yaquina (dry season), Classification Study, and NCA datasets for both Zones 1 and 2 (Kruskal-Wallis one way ANOVA on ranks, $p < 0.001$; Table 9.2). Comparison of the Yaquina Estuary data to that from both the Classification and NCA data sets showed some differences in zonal patterns. In Zone 1, there was not a significant difference between median TSS levels between the Classification and Yaquina data sets, but the Zone 2 median value in the Classification data set was significantly lower (66%) than the value for the Yaquina (Dunn's method for pairwise comparison, $p < 0.05$). NCA data for TSS showed still a different pattern across the region, where the median value was significantly higher (23%) than for the Yaquina in Zone 1, while the median value was significantly lower (50%) than that for the Yaquina in Zone 2 (Table 9.2; Dunn's method for pairwise comparison, $p < 0.05$). This pattern from the NCA data was present regardless of whether samples from the Columbia River Estuary were included.

The NCA study used a probability based sampling within the dry season that was generally random with respect to tidal stage. Much of the sampling from the Yaquina Estuary for TSS was from cruises during flooding tides. It is not clear whether the zonal pattern differences observed were the result of methodology differences or that the Yaquina Estuary is somehow different in its spatial pattern for TSS.

Table 9.1 Percentiles for light attenuation coefficient k_d (m^{-1}) calculated using continuous (1999-2003) and cruise (1998-2006) data sets from the Yaquina Estuary for dry and wet seasons.				
Data set	Percentiles for Light Attenuation Coefficient k_d (m^{-1})			Sample Size
	25 th	50 th	75 th	
Zone 1				
Yaquina				
Dry				
Continuous	0.62	0.78	1.00	678
Cruise	0.56	0.72	0.94	541
Wet				
Continuous	0.55	0.74	0.89	505
Cruise	0.53	0.66	0.87	248
Zone 2				
Yaquina				
Dry				
Continuous	1.14	1.53	2.26	376
Cruise	1.09	1.32	1.72	439
Wet				
Continuous	0.97	1.20	1.54	247
Cruise	0.95	1.20	1.54	178

Table 9.2 Percentiles for TSS ($mg\ l^{-1}$) calculated using Yaquina (1998-2004), Classification (2004-2005), and NCA (1999-2000) data sets. NCA and Classification values are for dry season only, while both dry and wet season values are provided for the Yaquina Estuary.				
Data set	Percentiles for TSS ($mg\ l^{-1}$)			Sample Size
	25 th	50 th	75 th	
Zone 1				
Yaquina				
Dry	3.0	6.3	9.6	102
Wet	3.7	6.6	12.2	56
Classification	4.6	6.7	10.8	66
NCA	10.4	14.0	16.0	36
NCA excluding Columbia	11.0	14.0	16.1	33
Zone 2				
Yaquina				
Dry	6.9	11.2	19.6	83
Wet	6.6	12.3	34.7	35
Classification	1.4	3.8	8.2	83
NCA	4.0	6.0	10.4	91
NCA excluding Columbia	5.0	9.0	11.8	27

10. Macroalgal Biomass as a Water Quality Response Measure

10.1 Introduction

Excessive algal growth is one of the major symptoms of eutrophication in coastal estuaries (Bricker et al., 1999). Three of the principal classes of algae are phytoplankton, epiphytic algae, and macroalgae. In PNW coastal estuaries, epiphytic algae (attached to other organisms) and macroalgae (seaweed) generally are considered to be of greater concern than is excessive growth of phytoplankton, which is rapidly transported out of the estuaries by tidal exchange. In this section, we report on the macroalgae issue as it relates to the question of eutrophication in Yaquina Estuary.

10.2 Approach

Beginning in 1997, numerous studies involving macroalgae have been conducted in Yaquina Estuary by WED. These include aerial photomapping surveys in 1997 and 1998, and intensive ground surveys of percent cover and biomass during 1998-2004. A listing of the individual studies conducted, and the analytical approaches utilized here, are presented in Appendix B.

10.3 Results and Discussion

10.3.1 Annual Variation: 1997 - 1998

The aerial distributions of benthic green macroalgae documented in the aerial photography of July 23, 1997 and August 10, 1998 indicate a substantial increase in coverage in 1998 (Figure 10.1). Part of the increase very probably is due to the fact that the 1998 aerial photographs were taken two and one half weeks later than were those in 1997. However, based on seasonal percent cover distributions obtained in 1999-2000 (Figure 10.2), an increase in cover of only about 15% would be expected. In contrast, the benthic macroalgal cover of bare substrate on August 10, 1998 was approximately 250 % that on July 23, 1997 (Fig. 10.1).

An empirical model has been developed that uses flood tide water temperatures to predict $\text{NO}_3^- + \text{NO}_2^-$ concentrations in coastal ocean water entering Yaquina Estuary during flood tides (Brown and Ozretich, in review). The average concentrations predicted by this model for two-month intervals preceding the aerial surveys of 1997 and 1998 are 2.1 and 6.0 μM , respectively. This difference in average $\text{NO}_3^- + \text{NO}_2^-$ concentration is assumed to be the result of the 1997 El Niño condition that

suppressed normal upwelling of nutrient-rich subsurface water that year (Corwith and Wheeler, 2002). The ~ 300 % increase in the modeled nutrient concentration agrees well with the ~ 200 % net increase in intertidal macroalgal cover between 1997 and 1998 surveys.

10.3.2 Seasonal Variation: 1999-2000

Monthly averages (\pm 1 std. error) for percent cover and biomass of benthic green macroalgae were measured for six sites (Appendix B) in Zone 1 during 1999-2000 (Figure 10.2). Maximum values occurred in September - October for both percent cover (~ 50 %) and biomass (~ 200 gdw m⁻²), with rapid declines in November. Between December 1999 and May 2000 the respective averages were below 5 % and 5 gdw m⁻². In Zone 1 more than 95% of the intertidal cover and biomass accumulation for benthic green macroalgae occurred during the dry season.

Macroalgal composition was assessed in 2001, and consisted of taxa most closely resembling *Ulva linza*: ~60%; *U. fenestrata*: ~30%; *U. flexuosa*: ~10%; *U. intestinalis*: <5%; (WED unpublished data). The seasonality and peak biomasses are consistent with historical data sets from the lower portion of Yaquina Estuary. Davis (1981) observed mean biomass of 400-500 gdw m⁻² for green macroalgae during June to September, 1980 and Garber et al. (1992) observed green macroalgae biomass of 185-370 gdw m⁻² during June to October of 1984 and 1985. These comparisons suggest that there was no increase in the frequency or intensity of macroalgal blooms within Yaquina Estuary over this 20 year period.

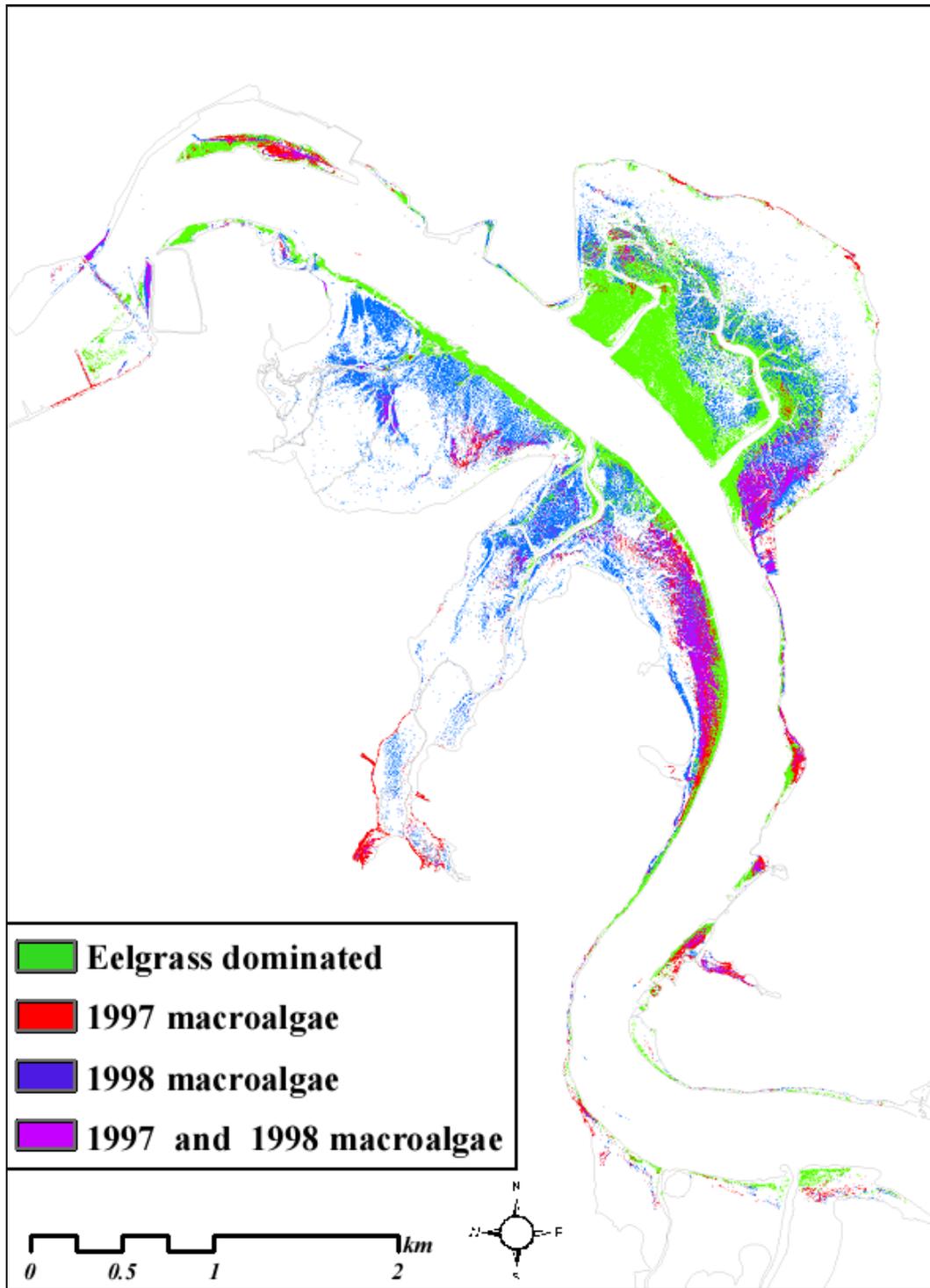


Figure 10.1 Photomap of intertidal vegetation in Yaquina Estuary from aerial surveys of July 23, 1997 and August 10, 1998 illustrating interannual differences of benthic green macroalgae cover.

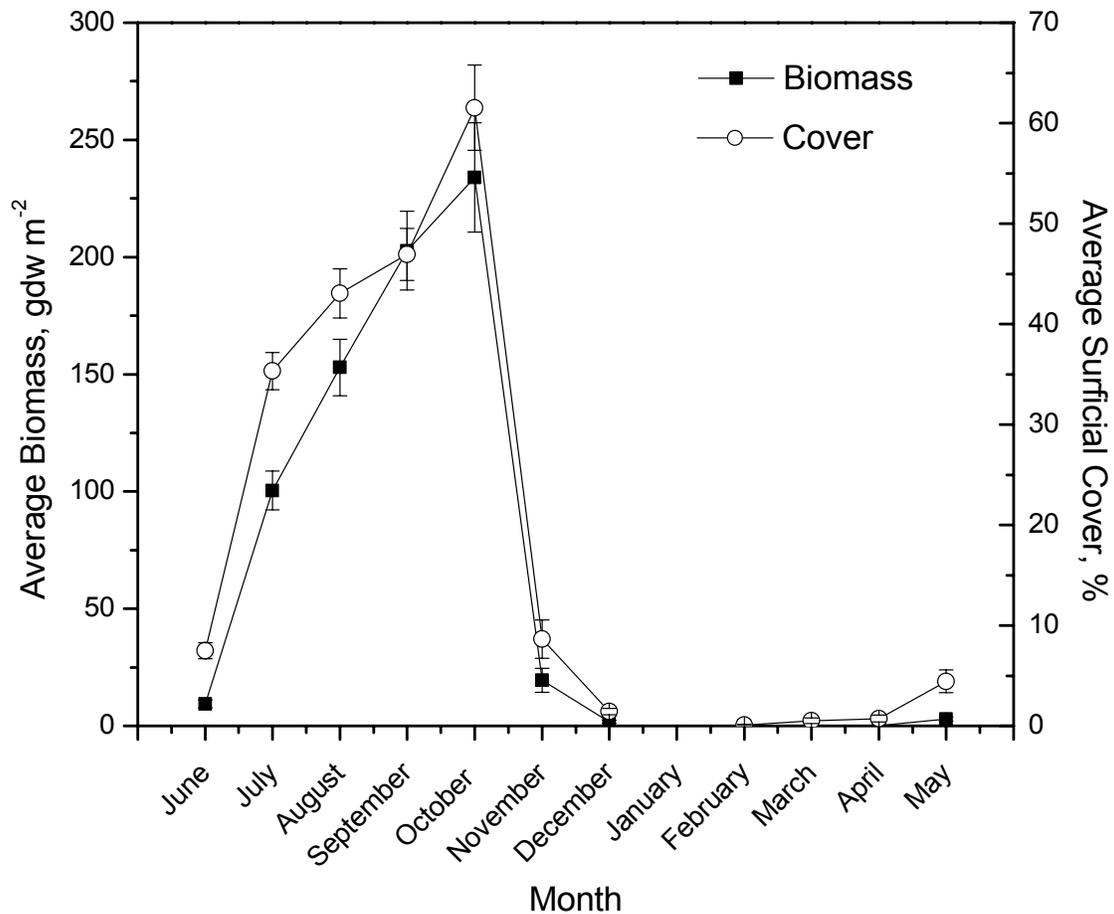


Figure 10.2 Average percent cover and biomass values (± 1 std. err.) of benthic green macroalgae in the Yaquina Estuary between June 1999 and May 2000. The values are averages for six sites within Zone 1.

10.4 Percentile Approach

CDFs for both dry and wet season were calculated for benthic green macroalgal biomass data from 1998-2004 within Zone 1 of Yaquina Estuary. In addition, CDFs were calculated for data from the Classification surveys conducted during the dry season in Zones 1 and 2 of six other Oregon estuaries (Table 10.1).

Table 10.1 Percentiles for benthic green macroalgae biomass (gdw m ⁻²) for Yaquina Estuary (1998 – 2004, Zone 1 only) and the Classification data set (2004 - 2005).					
Data set	Percentiles for Macroalgae Biomass			Percentile for 100 gdw m ⁻²	Sample Size
	25 th	50 th	75 th		
Zone 1 Yaquina					
Dry	0	34.9	189.8	62.7 %	4432
Wet	0	0	2.9	95.1 %	2142
Classification	0	0	11.6	92.9 %	351
Zone 2 Classification	0	0	0	99.6	231

Median Zone 1 dry season macroalgal biomass between 1999-2004 in Yaquina Estuary (Table 10.1) was less than the mean value (83 gdw m⁻²) measured from six band transects in 1999-2000 (Figure 10.2), and considerably higher than the median value from the Classification study. In the Yaquina Estuary, biomass exceeded 100 gdw m⁻² for 20% of the intertidal area, compared with only 1-6% of intertidal area for the six estuaries of the Classification Study. The Classification Study found that >98% of benthic green macroalgae occurred in the ocean dominated Zone 1 of the Yaquina Estuary (Lee et al., 2006). The reasons for the higher algal biomass found in the Yaquina Estuary compared to other Oregon systems studied is not clear, and makes extrapolation of information to the rest of the Oregon coast difficult.

10.5 Comparisons with Findings from Other Regions

Literature review demonstrates that there is a wide range of macroalgal densities that cause, or are correlated with, negative effects on estuarine organisms (Appendix D, Table D.2). Water temperatures reported in the reviewed literature ranged from 9 to 20 °C versus 8-18 °C for Zone 1 of Yaquina Estuary, and were thus reasonably similar. Approximately one-third of the studies reported negative ecological effects from macroalgae for percent cover values of ≥50% and biomass densities of ≥200 gdw m⁻². In the Yaquina Estuary, ~27% of the intertidal zone exceeded 50% cover, and ~10% had macroalgal biomass exceeding 200 gdw m⁻² (Lee et al., 2006). We also note that this density (200 gdw m⁻²) is twice the threshold accepted for damage by macroalgae to seagrass in Chesapeake Bay (Bricker et al., 2003).

Literature values for macroalgal impacts suggest that during the dry season, the accumulation of benthic green macroalgae could have a negative effect on the abundance of some infaunal invertebrates (while possibly enhancing epifauna), and on certain other fauna (e.g., juvenile flatfish, shorebirds). However, the preponderance of green macroalgae occurs in the marine dominated Zone 1 of Yaquina Estuary during the dry season. Results from stable isotope studies (Section 3.3 and Lee et al., 2006) provide strong support for the conclusion that benthic green macroalgae in Zone 1 of the Yaquina Estuary derive most of their nutrients from tidal influx of near shore marine waters. Summer green macroalgal blooms thus appear to be a natural response of the estuarine system. Thus, at present, the occurrence of benthic green macroalgae does not appear to be a useful indicator of eutrophication in Yaquina Estuary.

11. Submerged Aquatic Vegetation (SAV) as a Management Objective (Designated Use)

11.1 Background

The NHEERL Aquatic Stressors Framework (U.S. EPA, 2002) defines loss of submerged aquatic vegetation (SAV) as a major assessment endpoint for nutrient effects research. Seagrass, the dominant marine SAV, provides a critical three-dimensional structure often used by commercially and ecologically important species as a refuge from predation, and simulates estuarine biogeochemical cycling through trapping and recycling of seston and leaf material in sediments. Seagrasses also influence water quality and clarity by attenuating current velocity, promoting sediment deposition, and removing nutrients (N and P) from the water column. Thus, seagrass habitats function in a way that improves the quality of coastal and estuarine ecosystems. Sustaining seagrasses has become an important priority for federal agencies, the States, and tribes.

Eelgrass (*Zostera marina*), the principal seagrass in PNW estuaries (Phillips, 1984), is a rooted, flowering plant, which is present in many temperate estuaries world wide (den Hartog, 1970). Eelgrass meadows serve as a nursery ground for juveniles of commercially important species such as Pacific herring (*Clupea pallasii*) and as a refuge for juvenile salmonids (Griffin, 1997; Simenstad and Wissmar, 1985; Levings, 1990; den Hartog, 1977). Eelgrass meadows are significant sites of primary production and eelgrass shoots can be utilized directly for food by some waterfowl such as the western black brant (*Branta bernicula*) (Griffin, 1997; Kentula and McIntire, 1986), and indirectly by many species via consumption of detritus (Thayer et al., 1975). Eelgrass roots stabilize the sediment (Thayer et al., 1975) and the presence of eelgrass dampens wave energy which may serve to reduce erosion and to enhance larval settlement (Orth, 1992). Because of these characteristics, species abundances in eelgrass patches are usually greater than in other estuarine habitats (Everett et al., 1995). In recognition of the importance of seagrass beds, EPA Region III has proposed a “Shallow-water Bay Grass Designated Use” for Chesapeake Bay to insure adequate protection of living resources.

Anthropogenic nutrient additions have been suggested by many authors as a cause for the dramatic decline in seagrasses world wide and for *Z. marina* in particular on Atlantic Coasts (Short et al., 1995; Valiela et al 2000a; Hauxwell et al., 2003). The principal effect of excess nutrients is to reduce light available at leaf surfaces via enhanced macroalgal and leaf epiphyte production and by increasing the water column light attenuation coefficient (k_d) through the stimulation of the production of phytoplankton (Hauxwell et al, 2001; Madden and Kemp, 1996).

The Yaquina Estuary contains ~98.5 hectares of eelgrass which covers approximately 5% of the total area of the estuary (Figure 11.1). This eelgrass is in three zones consisting of: 1) a permanent bed of perennials in the lower intertidal and subtidal¹ (below Mean Lower Low Water, MLLW), 2) an intertidal transition zone (0.0 m to 0.5 m above MLLW) consisting of perennial patches and annual shoots; and 3) an upper intertidal zone (0.5 m to 1.5 m above MLLW) consisting of only annual shoots (Bayer, 1979).

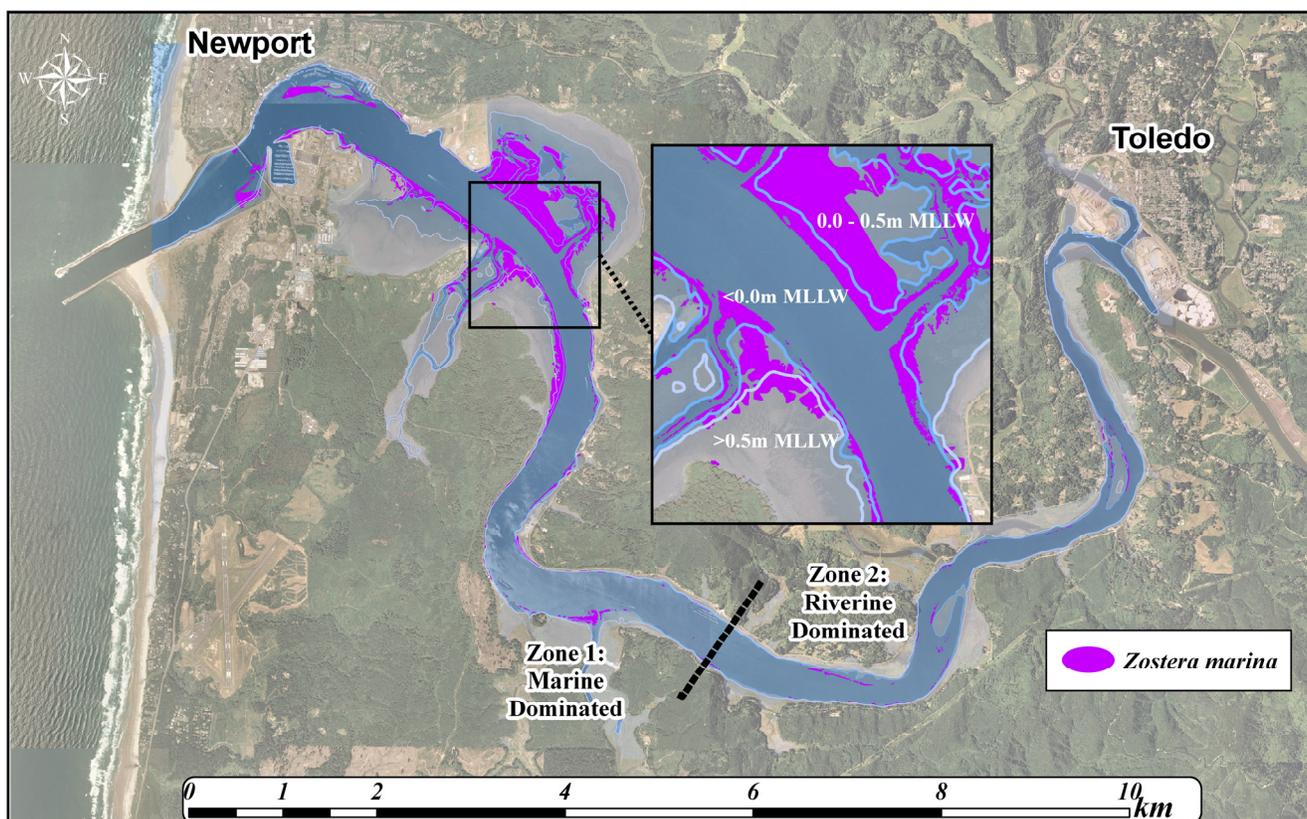


Figure 11.1 Spatial distribution of Yaquina Estuary eelgrass.

¹ Below -1.0 m MLLW

11.2 Spatial Seagrass Patterns

The spatial distribution of *Z. marina* was determined within the Yaquina Estuary from 1997 to the present utilizing aerial photographs and false-color near-infrared (color infrared, CIR) film (Young et al., 1999). Details on how this analysis was accomplished are presented in Lee et al. (2006) and Appendix B.

Permanent bed perennial shoots make up the vast majority (90%) of the eelgrass population (Boese and Robbins, in prep.), and almost all of this eelgrass is in the intertidal zone in both the ocean (Figure 11.2) and river dominated (Figure 11.3) estuarine portions. Details on the methods used to generate Figures 11.2 and 11.3 are presented in Appendix B. The portions of the graphs corresponding to depths deeper than -1.5 m (MLLW) may have errors due to limitations in mapping methods and bathymetric modeling. Most (97%) of the *Z. marina* in the Yaquina Estuary is located in ocean dominated estuarine portions (Figure 11.1), which is illustrated by the differences in y-axis scales in Figure 11.2 and Figure 11.3. Although the distribution suggests an effect of salinity on eelgrass distribution, *Z. marina* appears to be able to tolerate a wide range of salinities (Nelson, 2005). *Z. marina* also appears able to survive short-term exposures to fresh water, however, net leaf photosynthesis decreases in waters with salinities below 5 and totally ceases in completely fresh water (Hellblom and Björk, 1999; Biebl and McRoy, 1971). Within the Yaquina Estuary, Kentula and DeWitt (2003) found that salinity appeared to be a statistically significant factor in controlling the within estuary distribution of *Z. marina*, even though the reported mean summer and winter salinity ranged from 25 to 33, which are well within published tolerance limits for *Z. marina* (Nelson, 2005). The results of the Kentula and DeWitt (2003) study were complicated by changes in light attenuation and temperature that tended to co-vary with salinity. Results may have been further complicated since the bathymetry of the Yaquina Estuary changes with distance from the estuary's mouth such that the amount of suitable area in the optimal depth range for seagrass growth becomes limited in upriver estuarine segments (Lee et al., 2006).

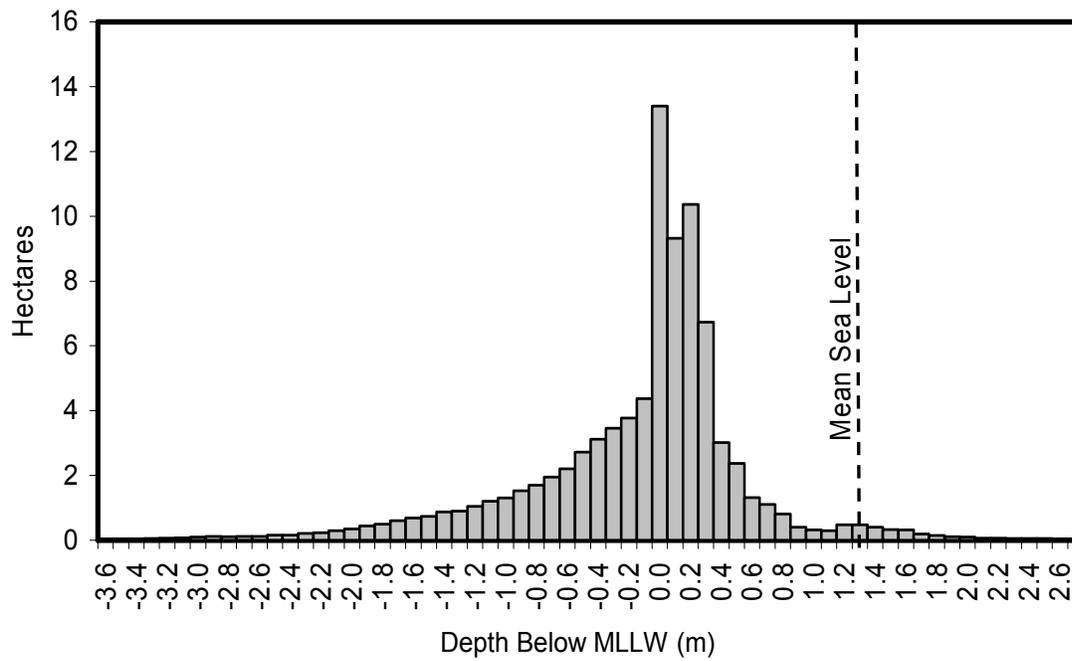


Figure 11.2 *Z. marina* depth distribution in the marine dominated portion (Zone 1) of Yaquina Estuary.

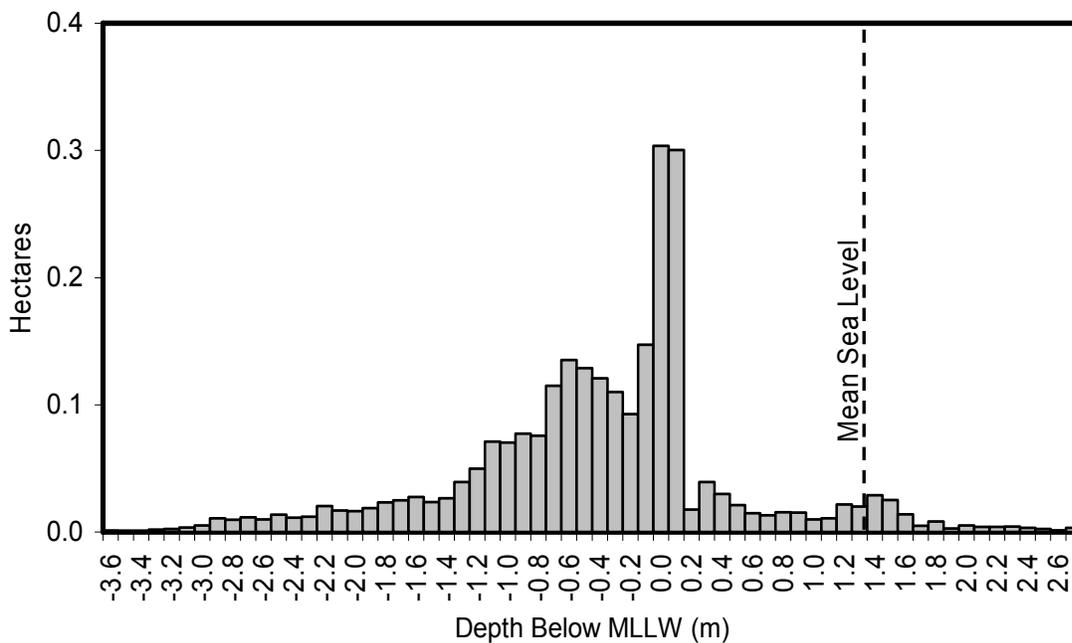


Figure 11.3 *Z. marina* depth distribution in the river dominated portion (Zone 2) of Yaquina Estuary.

11.3 Temporal Seagrass Patterns

Aerial photos of the Yaquina Estuary suggest that there is little year to year variability in *Z. marina* coverage from 1997 to the present. For example, Figure 11.4 shows details of the spatial distribution of a large *Z. marina* meadow and a narrow fringing bed, both of which are in a portion of the ocean dominated area of the estuary. Although there are apparent differences in these seagrass coverages across years, most of these differences are likely within classification error limits resulting from differences in ambient lighting conditions, the presence/absence of small amounts macroalgae, and subtle differences in how photos were interpreted. For the fringing seagrass bed in Figure 11.4, there is a possible relationship between seagrass temporal variability and the formation of intertidal drainage channels. Fringing seagrass beds which grow on steeply sloped sites are often less aggregated and tend to form into elongated and complicated shapes (Fonseca et al., 1983; Fonseca and Kenworthy, 1987; Frederiksen et al., 2004) thus providing more bed edges where erosion may be more effective in dislodging shoots. Erosion and strong physical disturbance events have often been observed in these marginal seagrass areas of the Yaquina Estuary where tidal drainage channel changes and storm events have either eroded *Z. marina* bed margins or deposited large woody debris on top of them (Boese and Robbins, in prep.). Episodic events such as these have been implicated in other studies (e.g. Krause-Jensen et al., 2003) as factors which alter shallow water seagrass populations. Thus, it is likely that the *Z. marina* losses observed at this marginal seagrass habitat area of the Yaquina Estuary were due to natural rather than anthropogenic stressors. Overall the result of our aerial surveys, when coupled with ancillary published (Boese et al., 2003; Young et al., 1999) and WED unpublished data, indicate that over the past decade the spatial distribution of *Z. marina* within the Yaquina Estuary has been stable.

The oldest known spatial coverage data for *Z. marina* in the Yaquina Estuary were published in the Oregon Estuary Plan Book (Cortright et al., 1987). This coverage was based on aerial photographs that were taken in the mid 1970's. A comparison of this historical coverage (Figure 11.5) to the present *Z. marina* distribution (Figure 11.1 and Figure 11.4) suggests an overall loss of seagrass in the Yaquina Estuary. However, there is no indication of what was meant by "seagrass bed" in terms of percent cover criteria that were used to delineate areas where *Z. marina* was present or absent (Cortright et al., 1987). In general, where seagrass habitat is shown on the Oregon Plan Book map, some seagrass is found in that general location either in recent photographs or has been observed as less than 10% cover during recent ground truthing surveys. Considering the differences in

methodologies, there appears to be no gross differences in the spatial distributions of *Z. marina* within the Yaquina Estuary over the last thirty years.

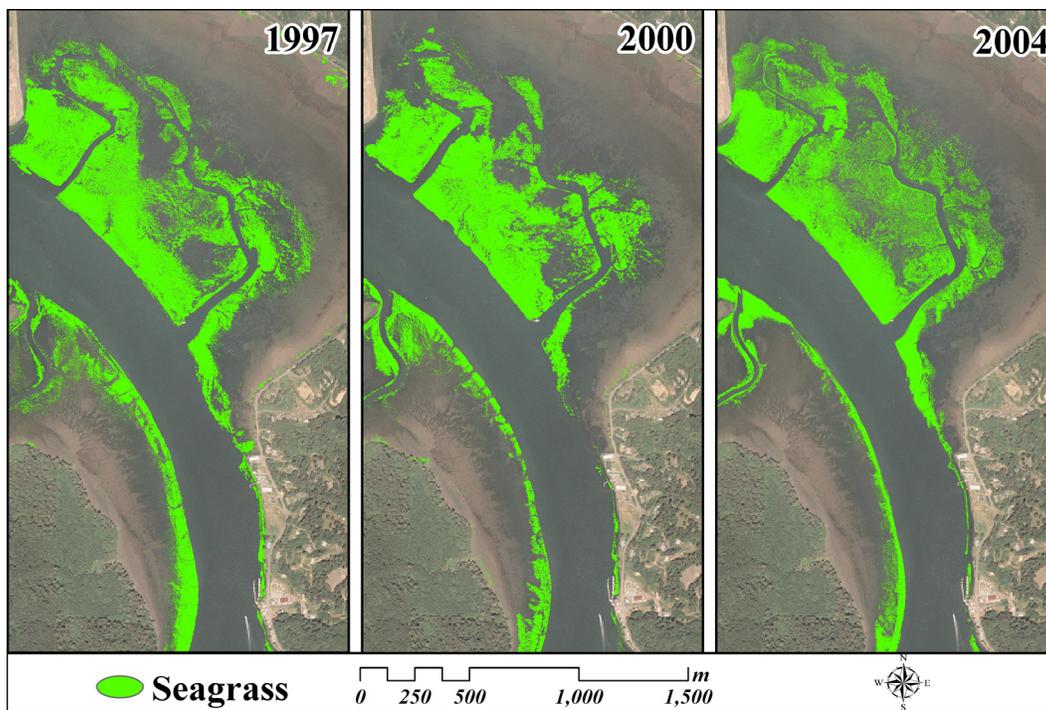


Figure 11.4 Comparison of the spatial distribution of *Z. marina* in a portion of the Yaquina Estuary (see inset on Figure 11.1) from 1997, 2000, and 2004. Figure shows a large contiguous meadow (north of channel) and a narrow fringing bed (south of channel).

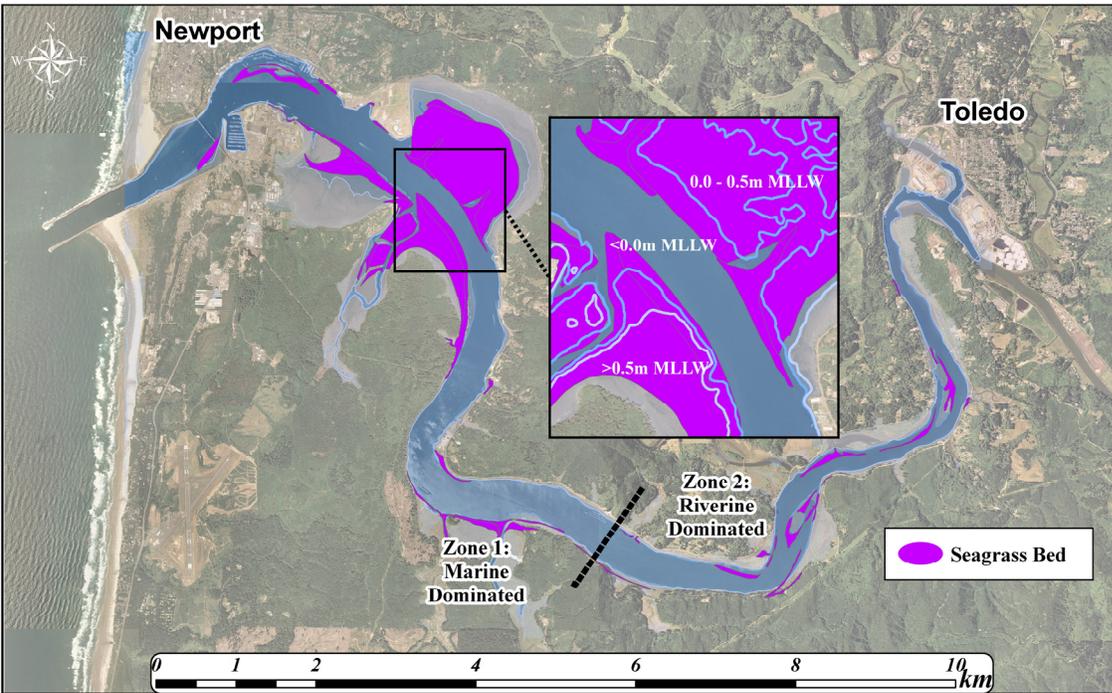


Figure 11.5 Historical distribution (mid 1970's) of *Z. marina* from the Oregon Estuary Plan Book (Cortright et al., 1987).

11.4 Water Clarity and Seagrass Lower Depth Limit

11.4.1 Background

The depth distribution of seagrasses has been shown to be dependent upon light penetration, with coastal seagrasses in general extending to depths receiving, on average, ~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 depth to which seagrass grows might be used as an integrative water quality assessment measure (Dennison et al., 1993), and has been suggested for use as a monitoring tool (Sewell et al., 2001; Virnstein et al., 2002). Additionally, understanding the minimal light requirements for seagrasses is necessary for preservation of existing seagrass meadows and for restoration purposes (Batiuk et al., 2000; Dennison et al., 1993; Fonseca et al., 1998).

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 U.S. Pacific Coast is problematic due to differences in tidal amplitude that tend to narrow the depth range of

seagrasses (Koch and Beer, 1996) and due to other differences including lower temperature ranges and faster estuarine flushing rates. Criteria for Chesapeake Bay *Z. marina* were derived for a spring through fall growing season (Batiuk et al., 2000), when carbohydrates are accumulated and used to maintain plants during the winter when they cannot maintain a positive carbon balance (Zimmerman et al., 1989). In contrast, for *Z. marina* in the Yaquina Estuary, 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).

11.4.2 Methods

During the summers of 2004 and 2005 the lower depth limit of *Z. marina* was determined at 64 randomly selected locations in Yaquina Estuary by underwater video and direct visual observation techniques (for methods see Appendix B). These data were then compared to calculated light attenuation coefficient (k_d) values that were measured at or near the same locations during a series of sampling cruises (See Section 4.1.1).

11.4.3 Relationship between Lower Margin and Water Clarity

Figure 11.6 shows the relationship between the lower depth limits of *Z. marina* and the distance from the mouth of the Yaquina Estuary. Although this relationship shows a great deal of variability, the lower depth limit appears to be greater toward the estuary's mouth and in the ocean dominated estuarine areas. There also appears to be a difference in this depth-distance relationship depending upon whether it is determined in the ocean or river dominated sections of the estuary, as illustrated by the lack of a significant linear relationship when the data from the ocean dominated section of the estuary are excluded (Figure 11.6).

The reduction in the *Z. marina* lower depth limit is consistent with a reduction in mean water clarity, which was also linearly related to distance from the mouth (Figure 9.2). We derived an additional relationship between the lower limits for *Z. marina* and the estimated k_d values as follows.

$$\text{Depth} = 4.4 - (1.79 * k_d), r^2 = 0.32. \quad (11.1)$$

where Depth = m below Mean Sea Level (MSL)² and k_d is the light attenuation coefficient (m^{-1}) computed from Figure 9.2 (for wet and dry seasons combined) at the location of the lower limit

² For the Yaquina Estuary MSL = MLLW + 1.39 m

observation.

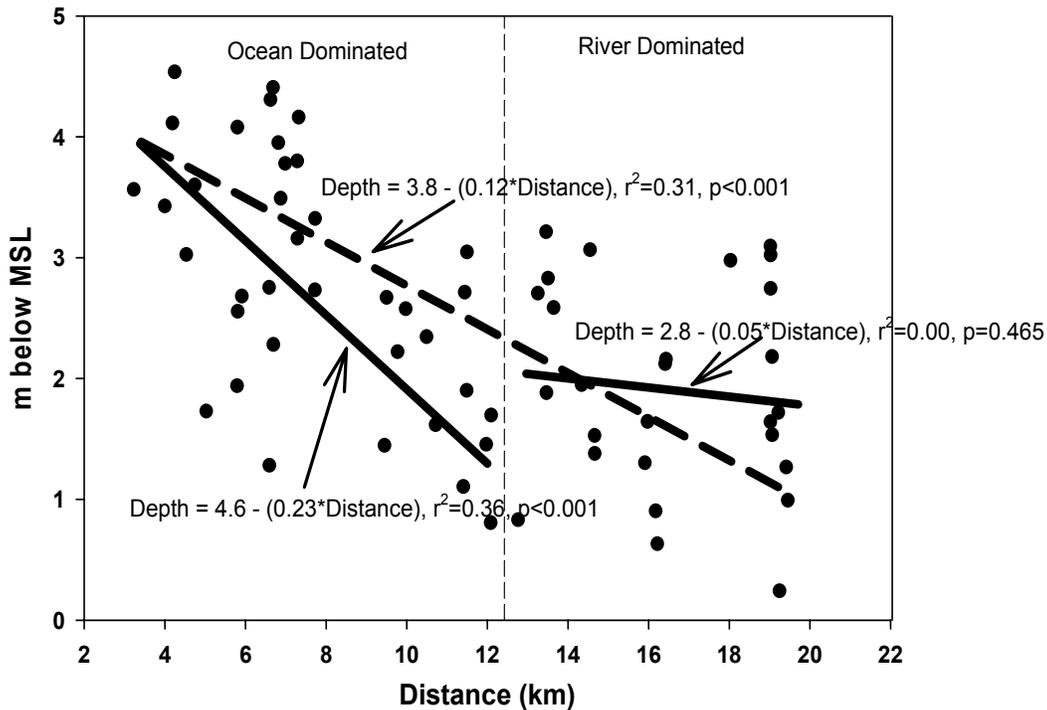


Figure 11.6 Relationship between distance from mouth of the estuary and the lower depth limit (below Mean Sea Level) for *Z. marina*. The three regression lines are for Zone 1, Zone 2, and the entire estuary.

The maximum depth to which a seagrass grows is dependant upon water clarity which is often presented in the literature as the fraction of surface irradiance found at the maximum seagrass colonization depth (Duarte, 1991). This value is calculated from k_d as

$$\frac{I}{I_0} = e^{-k_d z} \quad (11.2)$$

where I is the irradiance at depth, I_0 is the surface irradiance, k_d is light attenuation coefficient, and z is depth (m below MSL). Using this equation, the amount of surface irradiance reaching the observed lower depth limits for *Z. marina* within the Yaquina Estuary was estimated. These values ranged from 7 to 68% with a mean \pm standard deviation of 12.6 ± 1.9 % ($n = 64$). Standard deviation was determined using the propagation of error associated with estimating by linear regression from multiple k_d values at a given distance from the mouth of the estuary (see ANOVA with regression in Sokal and Rohlf, 1981).

The trend for deeper depth limits 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) suggested that the amount of light needed to sustain this species at depth is almost double that for seagrasses in general (Table 11.1). Duarte (1991) went on to note 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:

$$Z_c = \frac{1.86}{k_d} \quad (11.3)$$

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* ($Z_c = 1.62/k_d$ and $Z_c = 1.53/k_d$) on the Atlantic Coast of the U.S. (Dennison, 1987) and within Danish estuaries (Nielsen et al., 1989), respectively. The trends in Z_c and k_d for *Z. marina* in the Yaquina Estuary are consistent with Duarte's (1991) relationship (Figure 11.7) even though the waters in the Yaquina Estuary were more turbid than those reported by Duarte (1991). These k_d values from Duarte (1991) were converted (Equation 11.2) to the percent of surface irradiance at Z_c to generate the values which are presented in Table 11.1. Also included in Table 11.1 are the minimum light requirements recommended for the growth and survival of SAV in Chesapeake Bay (Batiuk et al., 2000).

Although the mean percent of surface irradiance needed to maintain *Z. marina* in the Yaquina Estuary is lower than literature values, they are within the range of published values. Literature values were either determined in waters which were considerably less turbid than those of the Yaquina Estuary (Duarte, 1991) or derived from a synthesis of literature values and area specific research (Batiuk et al., 2000). The *Z. marina* values published by Duarte (1991) rely heavily on Danish studies, especially Nielsen et al. (1989), which account for 20 of the 29 literature values shown in Figure 11.7.

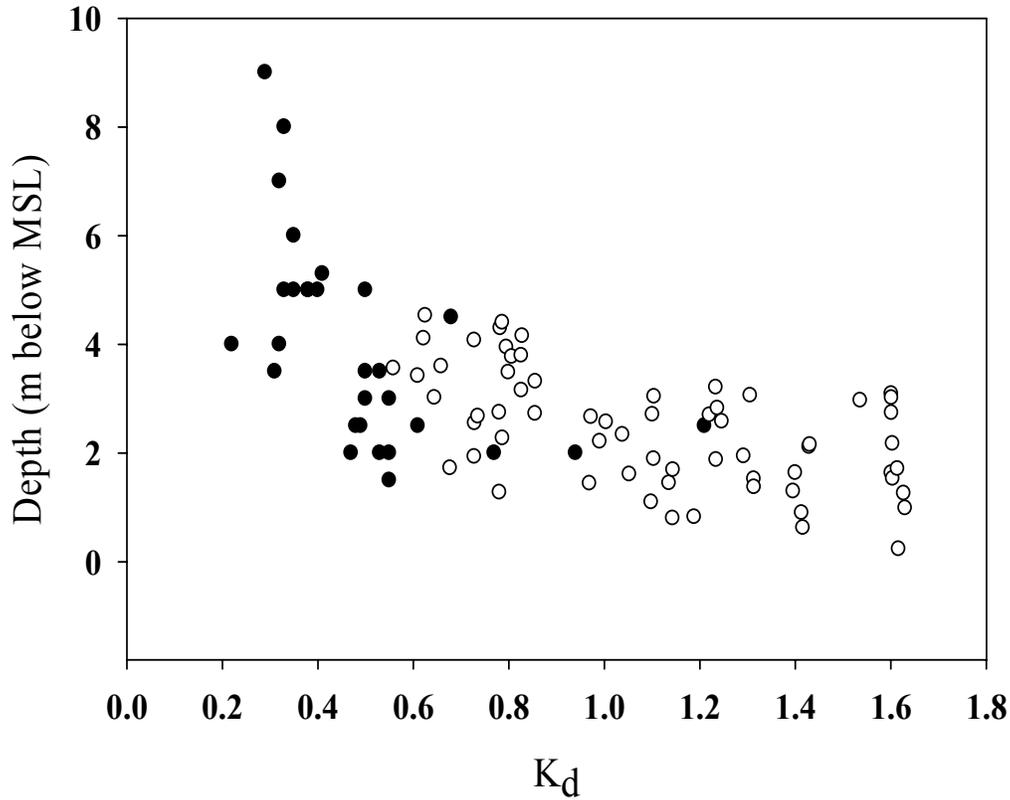


Figure 11.7 Relationship between *Z. marina* maximum depth limit (m below MSL) and k_d . Filled circles represent data from Duarte (1991) and hollow circles represent data from the Yaquina Estuary.

Table 11.1 Comparison of mean and range of percent of water column surface irradiance needed to maintain *Z. marina* at its colonization depth from published data and from Yaquina Estuary data. SE = standard error.

Source	Mean	N	Range	
			Max	Min
Duarte (1991)	20.5	29	43.9	4.7
Current Study	12.6	64	68.3	7.2
Batiuk et al. (2000)	22 ^a			

^aValue is not a mean but according to the authors is based on an analysis of literature and on an evaluation of monitoring and modeling research.

11.5 Epiphyte Patterns and Impact on Z. marina

The Chesapeake Bay water quality criteria for shallow water bay grass includes values for percent of ambient light reaching a plant through the water column, and a value for percent of light at the leaf, after attenuation by epiphytes. A study of epiphytes growing on *Z. marina* leaves was conducted within the Yaquina Estuary from 2000 through 2004 at six stations distributed between 3.5 and 17 km upriver from the mouth of the Yaquina Estuary. Methodological details are presented in Appendix B.

11.5.1 Spatial and Temporal Patterns in Epiphytes

In the Yaquina Estuary, there was a general annual pattern in 2000 through 2003 in which epiphyte biomass increased in the spring to a maximum in the summer and fall. This statistically significant parabolic relationship was most clearly seen on the older, external seagrass blades within a shoot (Figure 11.8). For unknown reasons, this yearly pattern was not observed in the 2004 samples.

In the Yaquina Estuary, epiphyte biomass per unit surface area of seagrass leaves was higher in Zone 1 (ocean dominated) than in Zone 2 (river dominated) in both wet and dry seasons (Figure 11.9). However, 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 zones (Lee et al., 2006).

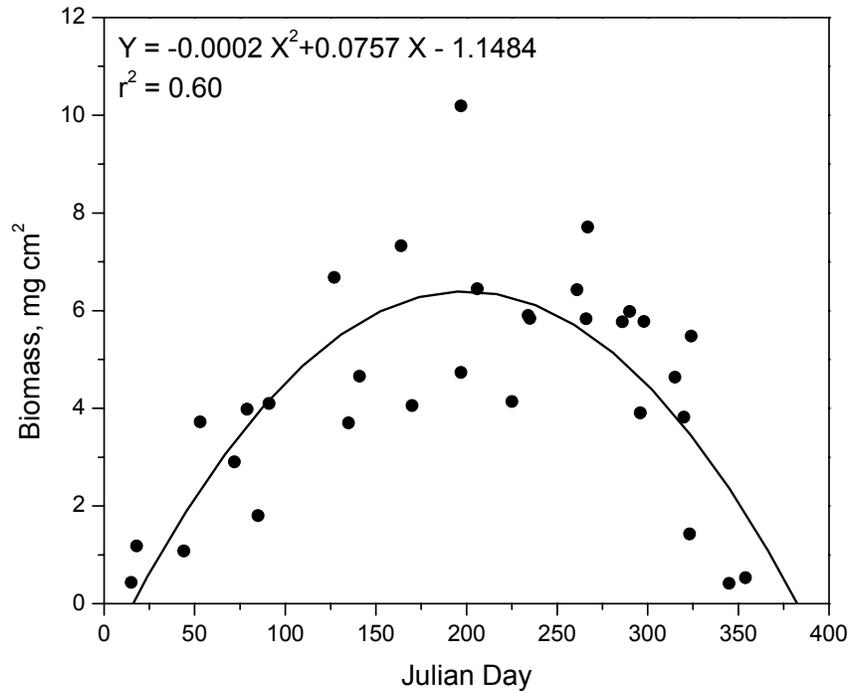


Figure 11.8 Temporal relationship of epiphytic biomass per unit leaf area on *Z. marina* external leaves in the Yaquina Estuary, 2000-2003.

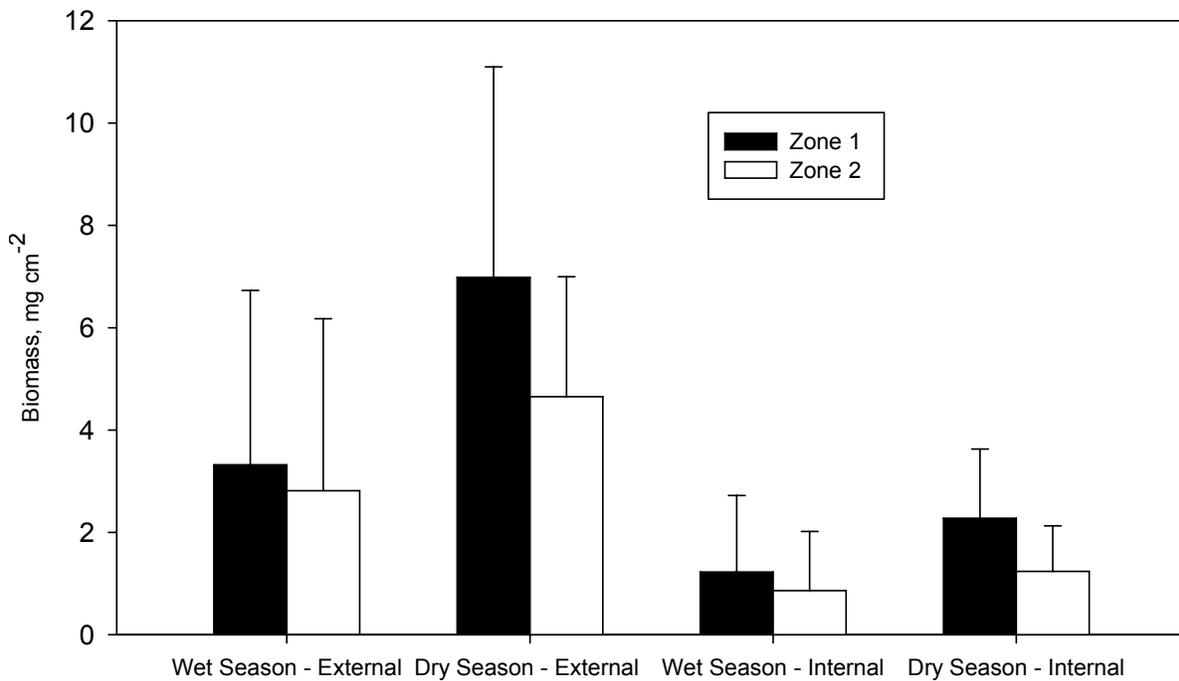


Figure 11.9 Epiphyte biomass per unit leaf area on old (external) and young (internal) *Z. marina* leaves by season (wet or dry) and salinity zone in the Yaquina Estuary.

There was a significant positive linear relationship (Figure 11.10) between percent light reduction and log+1 transformed biomass data, for external and internal blades combined. The linear regression relationship overestimates light reduction for the low epiphyte biomass samples.

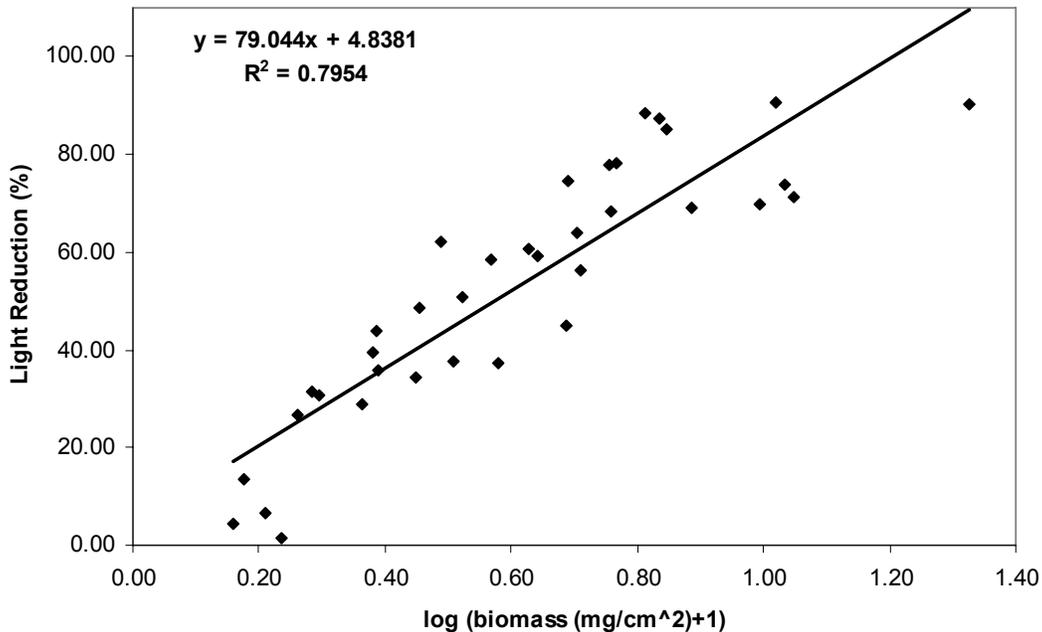


Figure 11.10 Linear regression relationship between the percent of light reduction to log(x+1) transformed epiphyte biomass per unit *Z. marina* surface area.

Epiphytes reduced the amount of light reaching the surface of *Z. marina* leaves. The monthly range of variation in light reduction was high, ranging from 4 - 91% for external leaves, and 2 - 62 % for internal leaves. The range in mean light reduction for a plant was estimated as 3 - 76 %, with an overall mean estimated light reduction of 53% (n=18, SE=4.6). As a result of the spatial differences in epiphyte biomass within the two salinity zones in the estuary, average light reduction for a plant was higher in Zone 1 (61%, n=72, SE=3.4) than in Zone 2 (37%, n=44, SE=4.5) as a result of the more heavily fouled external blades.

11.6 *Zostera marina* Light Requirements

Minimum light requirements for maintaining and restoring SAV 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 SAV 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 zones (>22% of surface irradiance) than for tidal fresh and oligohaline zones (>13% of surface irradiance). These zonal differences are in part due to the different species of SAV which are typically found in the different salinity zones (Batiuk, 1992). It is also important to recognize that the Chesapeake Bay values are designed to be protective of multiple SAV species, and not just *Z. marina*. The zonal irradiance values were based upon previously published k_d values of 2.0 m^{-1} for tidal fresh and oligohaline sections and 1.5 m^{-1} for polyhaline and mesohaline sections (Batiuk et al., 1992). The proposed irradiance criteria were adjusted for the amount of light absorbed by epiphytes encrusting SAV leaf surfaces and reported as the Percent of Light at the Leam surface or PPL. These minimum PPL values were 9 and 15 % respectively for the two salinity groupings (Batiuk et al. 2000). These proposed criteria were also applicable only to the SAV growing season (typically spring through fall).

In contrast, light requirements for *Z. marina* in Puget Sound were reported as integrated light intensity levels (Thom et al., 1998). 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., (1998) suggested that to maintain the greatest densities of *Z. marina*, $\sim 300 \mu\text{moles m}^{-2} \text{s}^{-1}$ (3 moles $\text{m}^{-2} \text{d}^{-1}$) were required for at least three hours daily during the growing season. 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 $150 \mu\text{moles m}^{-2} \text{s}^{-1}$ during the year. These same values are also suggested as minimum requirements for outer coast PNW estuaries like Willapa Bay and Coos Bay (R. Thom, Pacific Northwest Environmental Laboratory, pers. comm.). Assuming that mid-day surface irradiance is in the range of 1000-2000 $\mu\text{moles m}^{-2} \text{sec}^{-1}$, the minimum light requirement corresponds to approximately 15-30% of surface irradiance, which is consistent with other published criteria values and with the present study (Table 11.1). Additional verification of these minimum light requirements within the Yaquina Estuary is currently in progress (WED unpublished data). The mean daily irradiance value was approximately 3.8 moles $\text{m}^{-2} \text{d}^{-1}$ at a single lower margin site for *Z. marina* in the Yaquina Estuary (WED unpublished data). Although this value exceeds the Thom et al. (1998) criteria, irradiance values were highly variable, ranging from 0.5 to 7 moles $\text{m}^{-2} \text{d}^{-1}$, with extended periods of apparently inadequate lighting at depth from October to December (WED unpublished

data). However, even during these periods of apparently inadequate irradiance, *Z. marina* in the Yaquina Estuary continued to grow (Boese et al., 2005).

While it is tempting to directly apply the existing light criteria values to PNW estuaries like the Yaquina, there are several additional factors that need to be considered. The estuaries from which Duarte (1991) and Thom et al., (1998) derived their relationships are generally less turbid (mean $k_d \sim 0.5 \text{ m}^{-1}$) than the Yaquina (see Table 9.1). *Z. 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 Estuary (Boese et al., 2005) due to the latter's twice daily flushing with cold ocean water. Increased respiration rates due to higher summer temperatures 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 eelgrass in the Yaquina Estuary may require less spring and summer irradiance to perform the same function because of the generally cooler waters of these systems.

Additionally, the Yaquina Estuary is mesotidal. Koch and Beer (1996) found that greater tidal amplitude reduced the range of water depths that *Z. marina* colonized in Long Island Sound. Due to increased tidal amplitudes and turbidity, *Z. marina* growing in western Long Island Sound was limited to a 1 m depth range compared to the 4 m range observed in eastern Long Island Sound (Koch and Beer, 1996). Plants that are forced into a narrower depth range by these factors are likely to be more vulnerable to stressors such as storm events which may have contributed to the historic losses of *Z. marina* meadows. Thus, to assure seagrass survival in mesotidal and macrotidal estuaries, it may be prudent to establish more restrictive water clarity requirements in those estuaries.

Our study of epiphytes growing on *Z. marina* leaves in the Yaquina Estuary revealed a reduction in the amount of epiphyte biomass in upriver, lower salinity areas. 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, when ambient light levels are highest. The accumulation of epiphytes

was estimated to reduce the amount of light reaching leaf surfaces by an average of about 60% in the ocean dominated portion of the Yaquina Estuary (Zone 1). At present we are not sure how epiphyte load and its impact on light availability to eelgrass leaves 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*. Epiphyte light reduction will be incorporated in future versions of the seagrass stress-response model described in Chapter 12.

Although the effects of tides, temperature and epiphytes constitute current uncertainties in estimating minimum light requirements for seagrass in Yaquina Estuary, general conclusions can be made. Maximum depth of colonization of eelgrass in Yaquina Estuary suggests that a mean of 12.7% of surface illumination is required for persistence of seagrass at the deepest edge of the bed. Applying the median light extinction coefficient (k_d) for Zone 1 (0.8 m^{-1} , Chapter 9) to Equation 11.2 yields an estimate of percent of surface illumination at depths of 1, 2, and 3 m of 36, 20 and 9 %, respectively. This suggests that the use of the median k_d as a criterion in Zone 1 would allow persistence of eelgrass to a depth between 2-3 m. Use of the median k_d for Zone 2 (1.5 m^{-1}) in Equation 11.2, yields estimates of percent surface illumination at depths of 1 and 2 m of 22 and 5%, respectively. This suggests that the use of the median k_d as a criterion in Zone 2 would allow persistence of eelgrass to a depth between 1-2 m. These results are generally consistent both with empirical data on bathymetric distribution of eelgrass within the Yaquina Estuary and with the conclusions generated by use of the Stressor-Response Model (see Chapter 12).

12. Stress-Response Approach for Protection of SAV

12.1 *Introduction*

A previous report summarizes EPA research to develop mechanistic modeling approaches for examining the sensitivity of seagrasses to nutrient stressors (Kaldy and Eldridge, 2006). Here we use the mechanistic Seagrass Stressor-Response Model (SRM) developed by Kaldy and Eldridge (2006) in a heuristic fashion to assess the protective capacity of the Percentile approach (see Section 4.4).

The SRM used the 25th, median, and 75th percentile results from the cumulative distribution function (CDF's) developed in Sections 6.2 and 9.2 to determine if these potential criteria are protective of seagrass distribution and biomass in Yaquina Estuary. Our approach was to use the quartile values from the CDF's as inputs to the seagrass SRM with the objective of testing which values maintained seagrass at present depth distributions and which values resulted in decline of seagrass. These evaluations will provide guidance to aid in the selection of water clarity criteria that are protective of seagrass habitat in PNW estuaries. The response variables of the SRM model were seagrass biomass and carbohydrate content.

The SRM is composed of a set of mechanistic models that can be run in a variety of configurations depending on the study or management goals. The advantage of this approach is that, unlike the regression model approach, we can examine the direct and indirect effects of particular environmental conditions. Full model details and validation description are provided by Kaldy and Eldridge (2006).

12.2 *Description of Model*

The seagrass SRM was developed through an integrative effort that used a variety of data sources such as field studies and manipulative experiments, published literature, and existing and new models. A detailed description of the SRM development, calibration and validation is provided by Kaldy and Eldridge (2006). Briefly, the SRM is composed of an Allocation Model, a Plant Productivity Model and a Sediment Diagenetic Model. The Allocation Model integrates field data and provides estimates of carbon, nitrogen and phosphorus fluxes between plant components and the environment. These flux rates are then used to parameterize the Plant Productivity Model. The seagrass Plant Productivity model predicts above-ground biomass, carbohydrate reserves and plant growth in response to nutrients (both water-column and sediment porewater), salinity and underwater

light, and it provides boundary conditions for the Sediment Diagenetic Model. The Sediment Diagenetic Model provides estimates of the inorganic chemical environment in the root zone of the plant. The build-up or depletion of particular compounds in the sediments may have positive or negative effects on seagrass health and production. These models can be run independently or can be coupled together and run as the full SRM. Only the plant model configuration was used in the current analysis as there were no sediment geochemical data for the upper Yaquina Estuary (Zone 2). Calibration data for the plant model are shown in Appendix Figure E.1. Field and mesocosm experiments were used to validate model predictions (Kaldy and Eldridge, 2006).

The SRM has been used to examine seagrass response to a number of environmental variables including nutrients, canopy level irradiance, water turbidity, and organic matter input to sediments. The SRM can be used to assess the effectiveness of proposed nutrient loading criteria designed to be protective of seagrass. Assessment of the protective capacity of a particular water quality criterion was based on evaluation of trends in modeled seagrass biomass and carbohydrate for each depth interval. A downward trajectory in simulated biomass indicates that the water quality criterion was not protective at that depth. We also looked at the clustering of model outputs to assess breakpoints among the depth contours. Large differences in biomass or carbohydrate concentration between contours provides an approximation of the depth where conditions become inhospitable.

Models are simplifications of observed processes; as such they are subject to a number of simplifying assumptions and caveats. Further these models are being revised to include new types of calibration data that presumably will produce more accurate predictions. For example, the SRM does not include the effects of irradiance attenuation due to epiphytes, algae, self-shading or surface reflectance. As a result, the current simulations represent the “best case scenario” for underwater light. The model does include the effects of turbidity, nutrients, and salinity (Appendix Figure E.2). Furthermore, seagrass physiology was assumed to be similar between Zones 1 and 2. For presentation purposes, an upper margin for seagrass distribution of 0.2 m above mean lower low water (MLLW) was used; however, there are areas throughout the bay where the upper limit can not easily be defined by a single bathymetric level as a result of differential effects of desiccation, erosion, and sediment deposition.

12.3 Model Simulations and Input Data

The SRM was used in a heuristic fashion with idealized input data. Composite temperature and salinity time series were generated using YSI datasonde data from two stations, one located in each zone (distances from mouth of the estuary of 3.7 and 17.9 km). The composite time series represented average conditions from 1999-2003. For the solar irradiance, a composite incident photosynthetically active radiation (PAR) time series was generated using data collected by WED during 1999-2003 (Appendix Figure E.2). This PAR time series represents average incident light conditions (I_o) in the study area at 15 minute intervals for the year. The underwater light environment used in the model includes daily variations in surface irradiance with the addition of tidal variations in water surface elevation, and zonal and seasonal differences in water clarity (k). The underwater light environment was simulated for each zone as

$$I(z_{i=1,\dots,n}, j, t) = I_o(t) e^{-k_{jS}(z_i + h)}$$

where j represents the zone, z_i represents the depth (relative to MLLW), h is tidal variation in water surface elevation, and k_{jS} is the diffuse light attenuation coefficient for the specific zone (j) and season (S), which were obtained from the percentile analysis calculated using the continuous Yaquina Estuary data set (Table 9.1). Tidal variations in water surface elevation (h) were incorporated using hourly water level data from a tide gauge in Zone 1 (<http://tidesandcurrents.noaa.gov/>, Station 9435380 South Beach).

Case	Zone	Dry Season		Wet Season	
		k_d, m^{-1}	DIN, μM	k_d, m^{-1}	DIN, μM
1 (Median)	1	0.78	14.1	0.74	11.6
	2	1.53	13.7	1.20	55.0
2 (25 th %)	1	0.62	8.3	0.55	6.8
	2	1.14	7.3	0.97	30.7
3 (75 th %)	1	1.00	20.4	0.89	19.1
	2	2.26	23.1	1.54	73.5

Simulations were run for 3 cases, the median, 25th, and the 75th percentiles as representative of different levels of potential protective criteria for Yaquina Estuary. For these analyses, the estuary was divided spatially into a lower (Zone 1) and upper (Zone 2) region and temporally into a wet and dry season (Table 12.1). The transition from wet to dry season conditions for both k_d and DIN

concentrations was done as a step-function (Table 12.1). In these model simulations, water column DIN was used in nutrient uptake kinetics for the seagrass plant, but does not include indirect nutrient effects such as epiphyte, macroalgal or phytoplankton blooms. Additionally, simulations were conducted for parameters at a series of depths ranging from 0 m relative to mean lower low water (MLLW) down to a maximum depth of 5 m below MLLW in Zone 1 and 0 to 2.5 m below MLLW in Zone 2. The expanded depth range in Zone 1 was used because of lower light attenuation in this region. Additionally, this range encompasses the known depth distribution of *Z. marina* in Yaquina Estuary and allows for the expansion of the seagrass into deeper waters.

12.4 Results

As described above, the individual model simulation runs differed by depth, irradiance attenuation and DIN (Table 12.1), and temperature and salinity (Appendix Figure E.2). Temperature and salinity were also different between Zones 1 and 2, with a greater range in each variable occurring in Zone 2 as a result of seasonal heating and cooling. The model incorporated functions that increased photosynthesis and metabolism with temperature. The larger range in water temperature in Zone 2 affected seagrass physiology, while the relatively stable water temperatures in Zone 1 had a minimal impact (Appendix Figure E.2). Salinity had no influence on seagrass biomass or production in Zone 1 but affected production in Zone 2 during winter months.

Model results indicated that the median values would maintain the existing distribution of seagrass within Yaquina Estuary (Figure 12.1). Current maps indicate that seagrass covers approximately 0.97 km⁻² in Zone 1 and 0.013 km⁻² in Zone 2. In Zone 1 (lower estuary), seagrass would be protected to a depth of about 2 m below MLLW. The median values are representative of present conditions. In Zone 2 (upper estuary), the median criteria would protect seagrass to a depth of about 0.5 m below MLLW. Model simulations indicated that criteria based on the 25th percentile were the most protective (Table 12.2), permitting seagrass survival to depth of 3 m below MLLW in Zone 1 and 1 m below MLLW in Zone 2 (Figure 12.2). In contrast, model simulations using criteria generated from the 75th percentile were the least protective of seagrass (Table 12.2). The 75th percentile was protective of seagrass to about 2 m below MLLW in Zone 1 but, only maintained seagrass at a depth of 0 m MLLW in Zone 2 (Figure 12.3). Model simulations for each case and zone are provided in Appendix Figures E.3-E.8.

Table 12.2 Summary of the protective capacity of different potential criteria derived from Yaquina Estuary percentile data. Depth (below MLLW) to which these potential criteria permit long-term seagrass persistence and a narrative description for each case and zone. Percent change is relative to the median case.

Case	Zone	Protective Capacity		
		Depth, m	% change	
1 (Median)	1	>2	0	Present Condition
	2	>0.5	0	
2 (25 th %)	1	>3	+38	Most Protective
	2	>1	+41	
3 (75 th %)	1	>2	0	Least Protective
	2	0	-48	

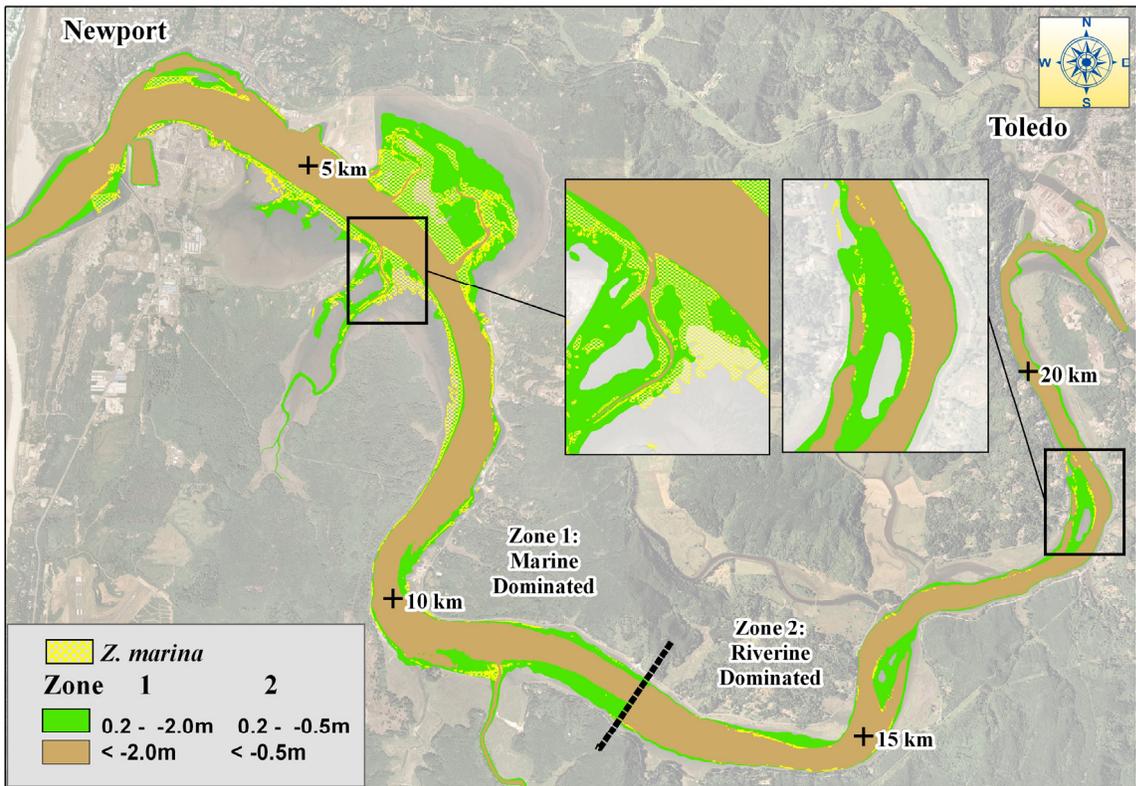


Figure 12.1 Observed eelgrass distribution (yellow cross-hatch) and simulated eelgrass (green) depth distribution based on the median case. Brown regions are unsuitable for eelgrass survival. Inset boxes show that the median case should maintain current eelgrass distribution in both Zones 1 and 2.

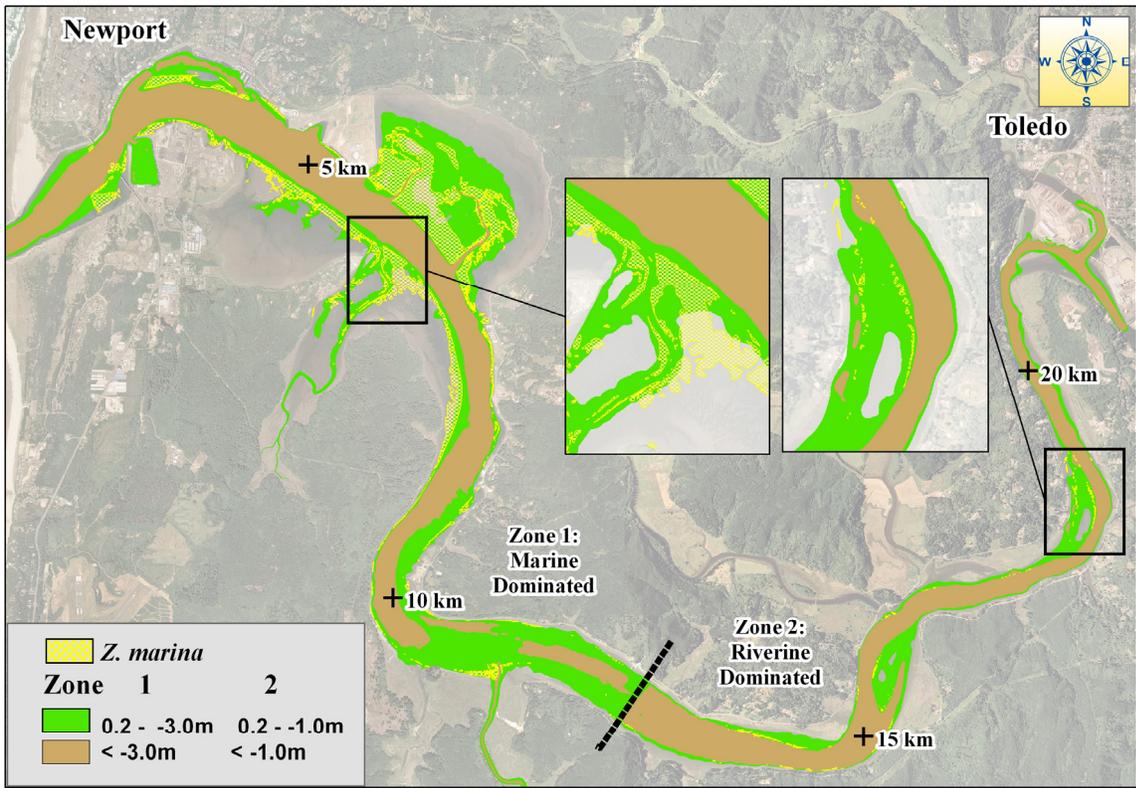


Figure 12.2 Observed eelgrass distribution (yellow cross-hatch) and simulated eelgrass (green) depth distribution based on the 25th percentile case. Brown regions are unsuitable for eelgrass survival. Inset boxes show that the 25th percentile case should permit expansion at the lower margin of the current eelgrass distribution in both Zones 1 and 2.

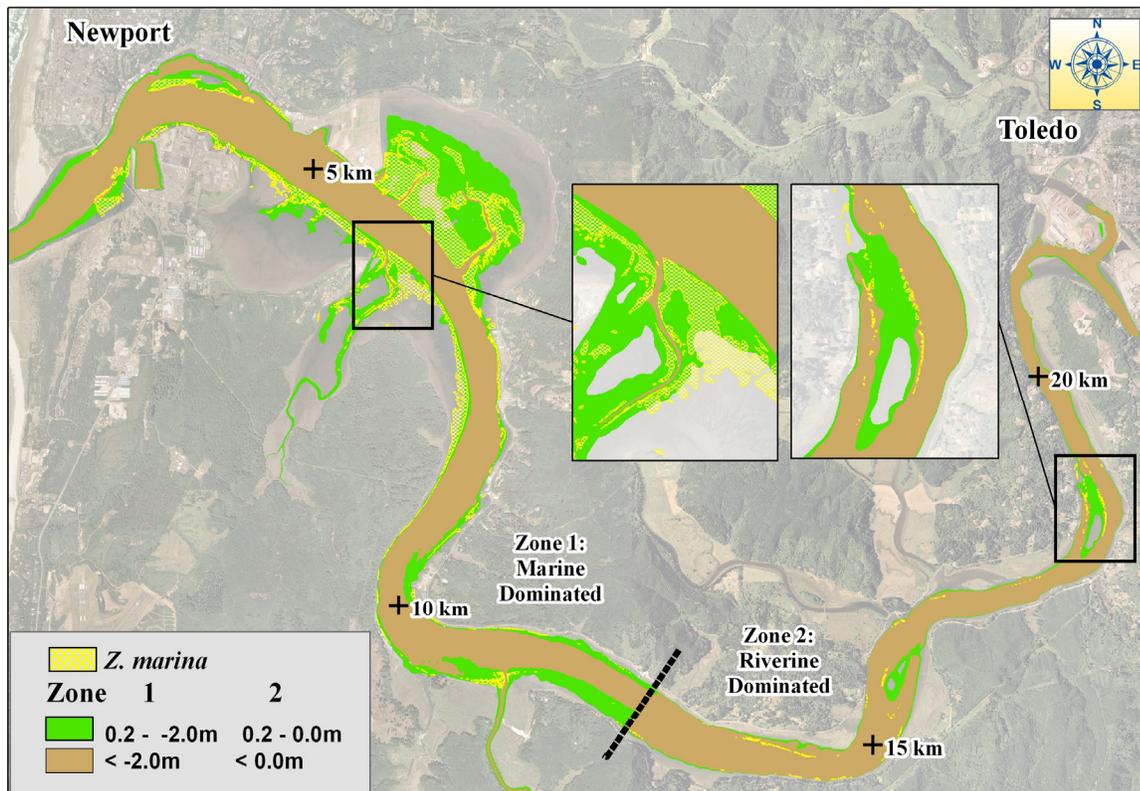


Figure 12.3 Observed eelgrass distribution (yellow cross-hatch) and simulated eelgrass (green) depth distribution based on the 75th percentile case. Brown regions are unsuitable for eelgrass survival. Inset boxes show that the 75th percentile case would eliminate much of the current eelgrass distribution in Zone 2.

12.5 Discussion

Many seagrass monitoring and assessment programs rely on presence/absence data or periodic evaluations of biomass and distribution (Pulich and White, 1997; Berry et al., 2003). However, these parameters are not very sensitive indicators of seagrass decline since they require very large sample sizes to detect modest changes (Heidelbaugh and Nelson, 1996). For example, biomass is a classic response variable; however, by the time monitoring programs can detect changes in biomass the perturbation may have caused seagrass decline. Better indicators of stress and decline are required to adequately assess seagrass condition. Non-structural carbohydrates may provide a more sensitive and integrative response variable since carbohydrate is the energy “currency” of the plant. Our assessment of the protective capacity of potential criteria was based on an evaluation of modeled carbohydrate content and modeled biomass.

Our analysis suggests that the 25th percentile and median criteria are protective of seagrass in both Zones 1 and 2 in the Yaquina Estuary (Table 12.2). Simulations using median k_d and DIN values project that eelgrass maintains its current depth distribution. The present depth distribution of eelgrass in Zone 1 (ocean dominated lower bay) is deeper than in Zone 2 (upper bay) due to greater light penetration. The 25th quartile simulations show that the seagrass permanent bed in Zone 1 might be extended from the present depth of 2 m to 3 m (below MLLW), but the depth limit in the upper bay would not change. Adoption of the 25th percentile criteria would potentially expand seagrass habitat by 38% and 41% in Zones 1 and 2, respectively, relative to the median case. The 75th percentile simulations predict a loss of 48% of habitat in Zone 2 relative to the median case. Most of the change occurs at the lower margin since the upper margin was fixed at 0.2 m above MLLW. While these changes in seagrass habitat are large in Yaquina Estuary, larger changes might be expected in shallower bays, while smaller changes might be expected in systems with steep bathymetric gradients.

Dry season median k_d values in the Yaquina Estuary (Table 12.1) are comparable to criteria that are currently being used in several other systems including Peconic Bay, NY, Long Island Sound, CT and Chesapeake Bay (Table 12.3). The criteria from the other systems are based on the requirements for restoration of *Z. marina*; therefore they may be more restrictive than those required for maintaining an existing eelgrass bed (EEA Inc., 1999). The Zone 2 median k_d values were similar to the criteria values for the restoration of seagrass to depths of 1 m below mean low water (MLW) in Chesapeake Bay (Table 12.3). The DIN criteria used by other studies are generally lower than the median values observed in the Yaquina Estuary (Table 12.1). DIN values from the 25th percentile in our study were comparable to the concentrations for Chesapeake Bay (Table 12.3). The proposed DIN criteria for Long Island Sound and Peconic Bay are much lower than observed DIN levels in the Yaquina Estuary, illustrating the importance of regional nutrient criteria. As discussed in Section 3.2, nutrient loading to Oregon estuaries is highly dynamic and naturally large as a result of coastal upwelling and alder dominated forests.

Table 12.3 Dry season median light attenuation and DIN values for Yaquina Estuary as compared to water quality management targets for other estuaries that are protective of eelgrass habitat.			
Location	k_d (m^{-1})	DIN (μM)	Citation
Yaquina Estuary, OR ¹	<0.78 (Zone 1) <1.5 (Zone 2)	14	This study
Peconic Bay, NY ²	<0.75	1.4	EEA, Inc. 1999
Long Island Sound, CT	<0.7	2.1	Holst et al., 2003
Chesapeake Bay, MD* ³	<1.5	10.7	Batiuk et al., 1992, 2000; Wazniak and Hall, 2005
Chesapeake Bay, MD* ⁴	<0.8	10.7	Batiuk et al. 1992, 2000; Wazniak and Hall, 2005
¹ Based on dry season median values; ² Based on mean summer values; *Meso and Polyhaline portions of the bay during the growth season (April-October); ³ Requirements for restoration of seagrass to 1 m MLW depth; ⁴ Requirements for restoration of seagrass to 2 m MLW depth.			

13. Conclusions and Recommendations

The Yaquina Estuary is characterized by strong seasonal variation in the magnitude of natural nutrient loading and in the dominant nutrient sources. Response variables (particularly, chlorophyll *a* and dissolved oxygen) show similar patterns of seasonal variation. During the wet season, riverine nitrogen inputs dominate, while during the dry season oceanic nitrogen sources dominate. There are also strong zonal differences in nutrient levels, response variables, and dominant nutrient sources within the Yaquina Estuary. In the lower estuary (Zone 1), water quality conditions are strongly influenced by ocean conditions, while in the upper portions of the estuary (Zone 2), watershed and point source inputs increase in importance.

The DIN and PO_4^{3-} levels in the Yaquina Estuary would represent medium levels using the criteria developed by Bricker et al. (2003) for eutrophication assessment. During the wet season, water column DIN levels within the estuary are relatively high. These high nitrogen levels are believed to be a naturally high background condition associated with the presence of red alder in the watershed. Some portion of the red alder related nitrogen inputs may be related to anthropogenic activities, since there may have been changes in red alder distribution related to logging activities in the watershed. However, we are presently unable to quantify the relative importance of natural and anthropogenic factors influencing watershed forest composition. During the dry season, PO_4^{3-} , NO_3^- , chlorophyll *a*, and dissolved oxygen levels in Zone 1 are primarily determined by ocean conditions. There is considerable interannual variability in ocean conditions that results from Pacific – scale processes, such as El Niño/La Niña and the Pacific Decadal Oscillation. The high degree of ocean-estuary coupling found for Zone 1 within the Yaquina Estuary suggests that monitoring for compliance with nutrient criteria in this region may be problematic. For example, hypoxic water and dense phytoplankton blooms at times are advected into Zone 1 from the coastal ocean during the dry season. Nutrient criteria developed for Zone 1, and any proposed monitoring process to determine compliance, would need to take into account this variability in ocean conditions. Distinguishing responses to anthropogenic nutrient inputs from those due to natural background variability for such indicators as chlorophyll *a* and dissolved oxygen may be difficult. At a minimum it may require acquisition of continuous monitoring data from multiparameter datasondes, an approach which is currently both expensive and labor intensive. WED is examining several rapid assessment approaches to allow

determination of whether conditions in Zone 1 are ocean derived, but these techniques have not yet been validated.

In addition to patterns associated with nutrients, there are strong zonal patterns in turbidity, TSS and water clarity within Yaquina Estuary, with increasing turbidity and light attenuation with distance from the mouth of the estuary. The lower depth limit of *Z. marina* habitat becomes shallower with distance from the mouth of the estuary, suggesting that light conditions may be influencing the distribution of *Z. marina* particularly within Zone 2 of the estuary. Median values of TSS are not considered at this time for inclusion in the list of potential water quality indicators. Comparison of spatial and temporal patterns of TSS with other estuaries in Oregon showed inconsistencies for reasons that are not clear at present.

Chlorophyll *a* levels within the Yaquina Estuary are typically low (median of 2-5 $\mu\text{g l}^{-1}$), and would be considered in the 'low' category when used as an indicator within the NOAA eutrophication framework (Bricker et al., 1999) and in the 'good' category using the West Coast criteria for water quality parameters from the National Coastal Condition Report (US EPA, 2004a). The present Oregon criterion (15 $\mu\text{g l}^{-1}$) is rarely exceeded except in the tidal fresh region (upper Zone 2) where the criterion is exceeded frequently during May-August.

There do not appear to have been major long-term changes in either water column nutrients or chlorophyll *a* within the Yaquina Estuary. Although the Yaquina Estuary experiences dense macroalgal blooms during the dry season (particularly in the lower estuary), we do not believe that these blooms have increased in frequency, duration or intensity, nor are they likely to be a product of cultural eutrophication. Modeling combined with determination of natural abundance, stable isotope patterns demonstrated that these macroalgal blooms are primarily fueled by oceanic nitrogen. We therefore conclude that green macroalgae biomass is not a useful indicator of cultural eutrophication in Yaquina Estuary.

Comparison of recent and historic *Z. marina* distributions suggests that there have not been any major changes in the last 30 years. The trend analyses did reveal that there was a significant increasing trend in DO levels in Zone 2 during the interval of 1960-1984. Review of watershed history suggests that current anthropogenic impacts are probably less than they were historically (particularly during 1960's-1980's).

Assessment of lower depth limits for eelgrass within the Yaquina Estuary allowed estimation of the minimum light requirements for sustaining seagrass. The mean light requirement was compared to

the estimated percentage of surface light available at various depths using the median zonal light extinction coefficients (k_d). Results indicated that the median k_d provided for persistence of seagrass at depths within the two estuarine zones that were comparable to current depth distributions, and that were consistent with results from the seagrass modeling effort.

The State of Oregon dissolved oxygen criterion (6.5 mg l^{-1}) is relatively high compared to other estuarine DO criteria (see values in U.S. EPA, 2003). The historical record of DO in Zone 2 demonstrates that this portion of the estuary may be susceptible to DO degradation. Our analyses showed that DO levels fall below of the present State of Oregon DO criterion in both Zones 1 and 2, but more frequently in Zone 1. We believe these periods of low DO in Zone 1 are related to the import of hypoxic water from the coastal ocean into the estuary; however, the causes of the low DO in Zone 2 are unknown. There are several potential causes of low DO in Zone 2 including import of hypoxic ocean water, *in situ* processes occurring within the estuary, as well as possible effects of WWTF effluent discharge. The current Oregon DO criterion should be adequately protective of estuarine resources, but is closer to the 25th percentile value rather than the median value for DO data in Zone 2. Using the present numeric Oregon DO criterion, we estimate that between 20 and 30% of measurements would not meet the criterion in Zone 2.

There are still uncertainties with respect to the development of nutrient criteria and the testing of potential criteria using the Seagrass Stress-Response Model (SRM). Uncertainties in defining the minimum light requirement of eelgrass include the effects of tidal action and temperature. The primary input used for the SRM was water clarity, which is affected by multiple factors, including turbidity that may be independent of nutrient loads. The SRM did incorporate water column DIN as a limiting nutrient, but it did not incorporate indirect nutrient effects, such as relationships between nutrient loading and water column chlorophyll *a*. The SRM does not yet include a term for epiphyte effects on light attenuation to seagrass, and thus there is still some uncertainty about precise levels of water column light required for maintenance of healthy seagrass. This uncertainty will be resolved in future versions of the SRM.

13.1 Recommendations

Based on the analyses presented in this report, we suggest that criteria be developed for the wet and dry seasons to address the extremely strong seasonal variation in nutrient loads and sources in the Yaquina Estuary. Establishment of dry season criteria (May-October) is of first priority since during

the wet season there appears to be little utilization of nutrients within the estuary, chlorophyll *a* levels are low, and the dissolved oxygen concentrations are high. Because of the high degree of ocean influence on water quality parameters that occur in Zone 1 during the dry season and strong tidal flushing, a first priority would be the establishment of criteria for Zone 2. An additional justification for this prioritization is the potential difficulties in sustaining the data collection needed to differentiate natural from anthropogenic nutrient inputs in Zone 1. We suggest that priority for monitoring for compliance with any proposed nutrient criterion in the Yaquina Estuary be for Zone 2 in the dry season. This is the most likely region and time period where anthropogenic nutrient effects would be expressed.

EPA (2001) summarized that a “Recognized Unique Excellent Condition” estuary would have a watershed that is unimpacted with “very little human development, is distant from the influence of local population centers, adjacent land uses are undisturbed, and is outside of major atmospheric deposition of nitrogen.” With the exception of “adjacent land uses are undisturbed” the Yaquina watershed meets all of these conditions. Additionally, from a nutrient (and land use) standpoint we believe that the Yaquina watershed is undisturbed compared to many other systems in the U.S., even though extensive silviculture occurs in the watershed.

Following the recommendations in U.S. EPA (2001), median values are the suggested criteria for estuaries in “Recognized Unique Excellent Condition” (Table 13.1). The assessment of seagrass light requirements in Chapter 8 and the Seagrass Stress-Response Model demonstration in Chapter 12 indicate that the median percentiles for the water clarity criterion in both Zones would be protective of the existing *Z. marina* habitat in the Yaquina Estuary. The dry season median light attenuation values (Table 13.1) for Yaquina Estuary are comparable to water clarity criteria for the protection of *Z. marina* habitat in other estuaries, including Chesapeake Bay, Long Island Sound, and Peconic Bay. Since the existing State of Oregon dissolved oxygen criterion is based on a review of physiological requirements of biota and appears to be adequately protective of the designated uses, we would recommend that the DO criterion remain at this level.

The present Oregon chlorophyll *a* criterion is determined as a 3-month average. If the 3-month average chlorophyll *a* levels within the Yaquina Estuary were to approach the present criterion, significant trophic shifts in the estuary would be likely. Thus, the current chlorophyll *a* criterion may not prevent some impacts on designated use. A more conservative criterion would be the adoption of the medians for chlorophyll *a* within both Zones 1 and 2 (Table 13.1).

Table 13.1 Potential dry season criteria for the Yaquina Estuary based on median values for all parameters except for DO.		
Parameter (units)	Zone 1	Zone 2
DIN (μM)	14	14
Phosphate (μM)	1.3	0.6
Chlorophyll <i>a</i> ($\mu\text{g l}^{-1}$)	3	5
Water Clarity (m^{-1})	0.8	1.5
Dissolved Oxygen (mg l^{-1})	6.5	

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Appendix A: Benthic Processes in Yaquina Estuary

Benthic communities harboring actively burrowing, tube- or burrow-dwelling infaunal are often associated with elevated rates of DIN advection from sediments (e.g., Aller, 1988; Kristensen et al., 1991; Marinelli and Williams, 2003). Bioturbation and bioirrigation by infauna oxygenates sediments and mixes labile organic matter into sediments, stimulating the activity of microbial communities responsible for recycling of nutrients (Kristensen, 1988; Welsh, 2003). Benthic fauna consume organic matter from the water column (i.e., filter feeders), at the sediment surface (i.e., herbivores, surface deposit-feeders, carnivores), or below the sediment surface (i.e., sub-surface deposit feeders, carnivores). The presence of seagrasses (i.e., *Zostera* spp.), green macroalgae (i.e., *Enteromorpha* spp. and *Ulva* spp.), or microphytobenthic algae usually results in a net benthic uptake of DIN during daylight hours (Underwood and Kromkamp, 1999; Hansen et al., 2000; Sundbäck et al., 2000; Sundbäck et al., 2003). Benthic primary producers are recycled into the benthos (i.e., consumed by benthic herbivores or surface deposit feeders, buried by bioturbators, decay at the sediment surface) or are transported out of the estuary by currents.

Five studies of benthic nutrient flux have been conducted in Pacific estuaries north of San Francisco, but the reported benthic flux-chamber data in four (i.e., Dollar et al. 1991; Garber et al. 1992; Thom et al. 1994; Larned 2003) may not be appropriate for the purpose of estimating estuary-scale nutrient fluxes in Yaquina Estuary because they did not adequately take into account the presence of thalassinid burrowing shrimp. As described in Section 3.2.3 Benthic Processes, previous studies of benthic nutrient flux in PNW estuaries may have underestimated the nutrient fluxes from sediments by failing to account for the presence of burrowing shrimp. Both ghost shrimp and mud shrimp were present, or were likely to have been present, at field sites in all of the studies. Burrowing shrimp are very common, and frequently very abundant, in NE Pacific estuaries (DeWitt et al., 2004 and references therein), and they construct deep (>50 cm), branching burrows with openings 10's of cm apart. These shrimp are prodigious bioturbators, actively irrigate their burrows, and thus greatly elevate nutrient flux from sediments (Waslenchuk et al., 1982; DeWitt et al., 2004; Webb and Eyre, 2004; Papaspyrou et al., 2004).

Failure to take the presence of burrowing shrimp into account can result in water inside the chamber being exchanged with water outside of the chamber via shrimp burrows (e.g., Hughes et al. 2000), which violates the requirement that benthic chambers be closed microcosms (Hofman and de Jong, 1993, Forja and Gomez-Parra 1998). In the case of one study, benthic chamber treatments

deliberately excluded shrimp (Thom et al., 1994), and while these measurements would represent tide flat habitats that have no shrimp, such areas are only 16% of the total Yaquina tide flat area (DeWitt et al., 2004). Thus, because of these uncertainties, benthic flux measurements from those four studies (Dollar et al. 1991; Garber et al. 1992; Thom et al. 1994; Larned 2003) were not used for estimating the estuary-scale nutrient fluxes presented in Table 3.1 for the Yaquina Estuary.

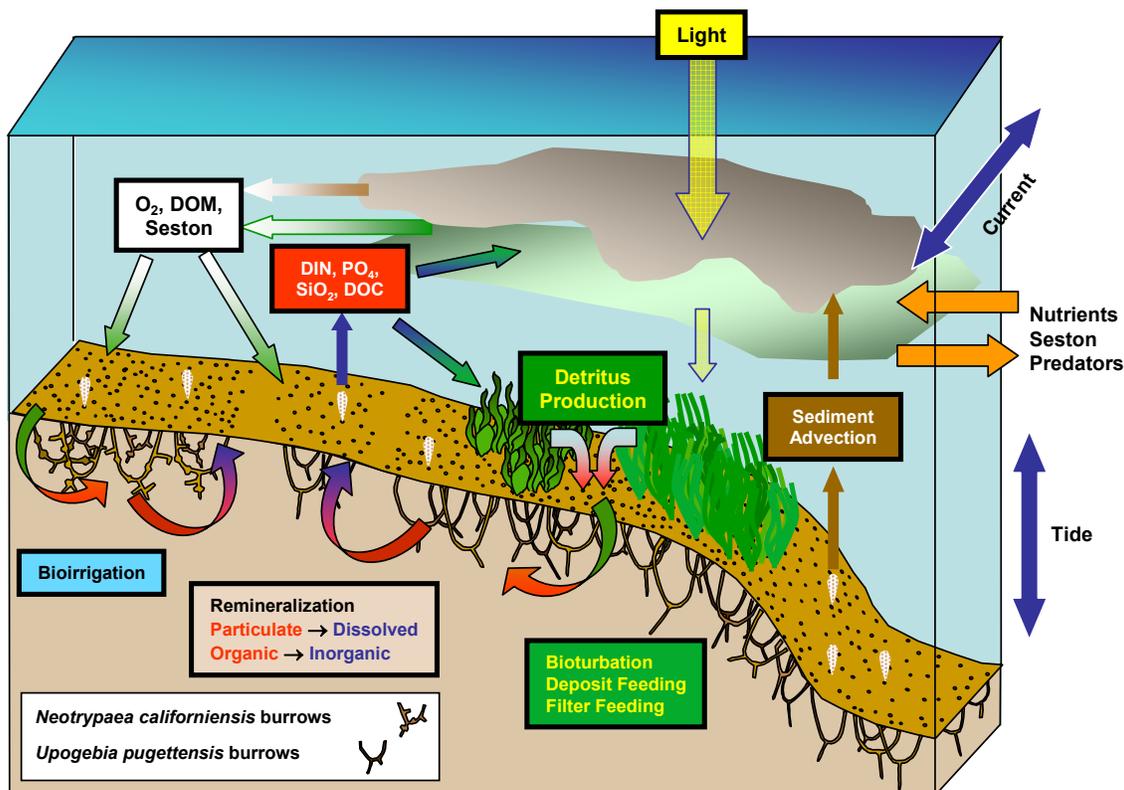


Figure A.1 Conceptual model of the dominant processes (boxes) driving carbon and nutrient flux (arrows) between the benthos and water column in Pacific Northwest estuaries.

To avoid this problem, DeWitt et al. (2004) inserted 1-m deep core barrels into sediments at their study sites, and fit benthic chambers to the tops of the core barrels to isolate water, sediments, shrimp and burrows inside the chamber from the outside world. Core barrels were inserted >10d before the chamber tops were attached so that shrimp enclosed in the barrel could construct new burrow openings within the chamber. DeWitt et al. (2004) produced estuary-scale maps of benthic-pelagic fluxes of $\text{NO}_3^- + \text{NO}_2^-$, NH_4^+ , and DIN for Yaquina Estuary (Figures A.2 and A.3) by linking estuarine-scale maps of burrowing shrimp populations to density-dependent flux measurements. DIN fluxes had great spatial variability owing to the differences in shrimp species and abundance across the

estuarine landscape. DIN efflux (release) from sediments was estimated to be much greater than DIN uptake in the presence of burrowing shrimp, but in the absence of the shrimp DeWitt et al. (2004) estimated a net uptake of DIN by sediments. Most of the DIN efflux was projected to occur in *U. pugettensis* - dominated habitat, mostly due to enhanced NH_4^+ efflux. Tide flats lacking burrowing shrimp or having low densities of *N. californiensis* were shown to have a net uptake of DIN, most of which is expected to be $\text{NO}_3^- + \text{NO}_2^-$. DIN efflux was estimated to be much greater than DIN uptake in the lower portion of the estuary, mostly due to the presence of dense populations of *U. pugettensis*. Using this shrimp species- and density-dependent nutrient flux model, the benthos in the upper (mesohaline) regions of the Yaquina Estuary was estimated to have a large net uptake of DIN because of the spatial dominance of *N. californiensis* and scarcity of *U. pugettensis*.

Much of the organic matter that is produced within, or advected into, the estuary is available for consumption by benthic herbivores, filter-feeders, or deposit feeders. The dominant benthic herbivores in Yaquina Estuary are amphipods, isopods, and nereid polychaetes. Herbivores are not abundant among benthic infauna in Yaquina Estuary, with biomass 1-10% of filter-feeders and deposit feeders (Figure A.4). Deposit feeders are abundant throughout the estuary, and dominate the upper-estuary infauna. In lower and upper reaches, the most abundant deposit-feeder by biomass is *N. californiensis*. Organic matter consumption rates have not been calculated for the infaunal deposit-feeder guild in Yaquina Estuary. Filter-feeders are more abundant in the lower than in the upper estuary, primarily because that is the distribution pattern of the dominant species of this guild, *U. pugettensis*. Griffen et al. (2004) estimated that populations of the mud shrimp in the lower Yaquina Estuary pump the entire volume of water covering the tide flats through their burrows every day. Combining per-capita grazing rates for mud shrimp (Griffen et al., 2004), patterns of mud shrimp population distribution in the estuary (Figure A.2a), bathymetry, chlorophyll *a* data and a hydrodynamic model, it has been estimated that mud shrimp populations graze approximately 60% of the phytoplankton that enters the lower estuary. This estimate is similar to measured differences in flood- and ebb-tide chlorophyll *a* concentrations, suggesting that filter feeding is an important sink for phytoplankton in the lower estuary (see Section 7.1).

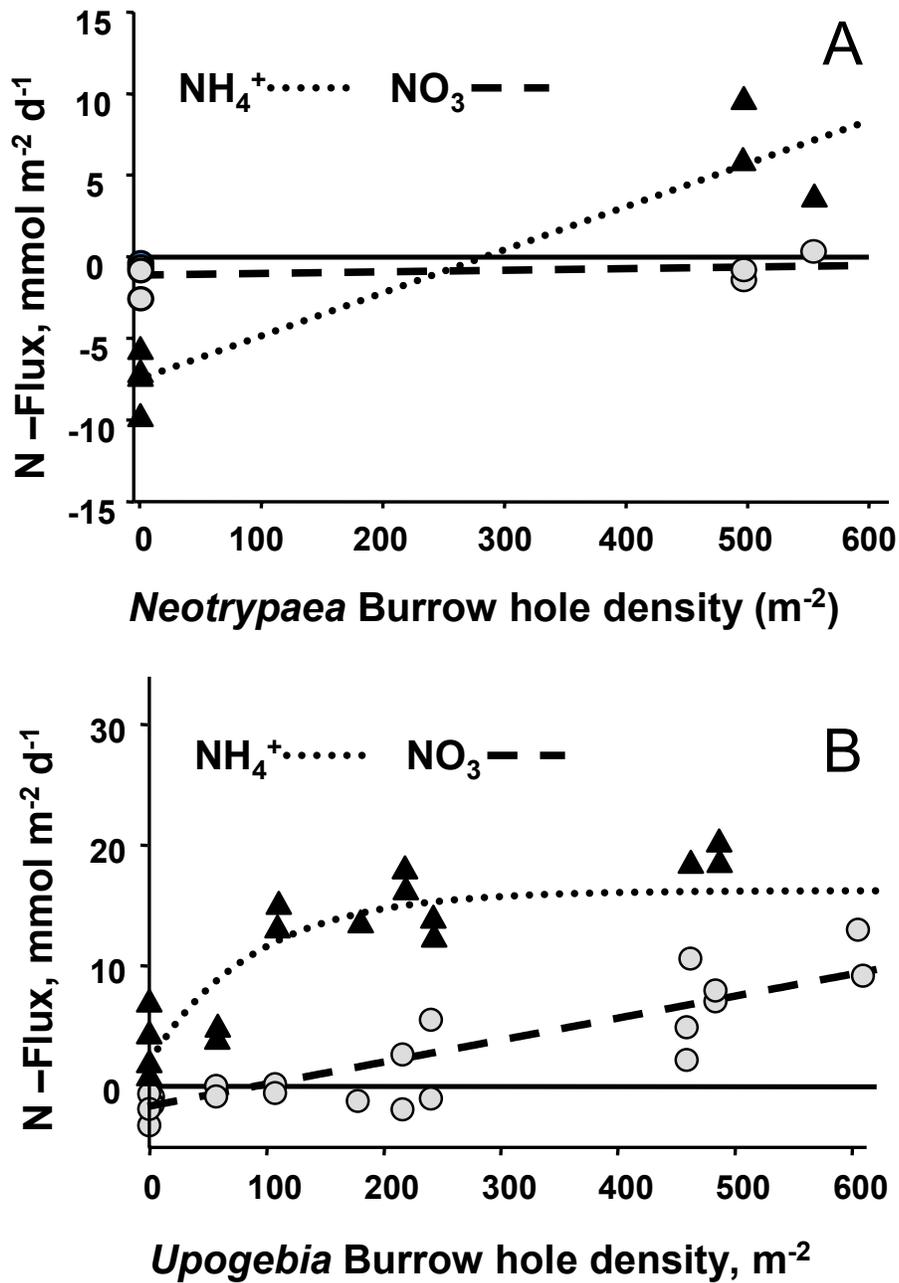


Figure A.2 Density-dependent flux of NH₄⁺ (black triangles and dotted line) and NO₃⁻ (gray circles and solid line) associated with *Neotrypaea californiensis* (a) and *Upogebia pugettensis* (b). Positive values = efflux from sediments, negative values = uptake by sediments. Dashed and dotted curves represent best-fit regression models for flux = $f(\text{shrimp burrow density})$. Solid black line represents 0 nitrogen flux.

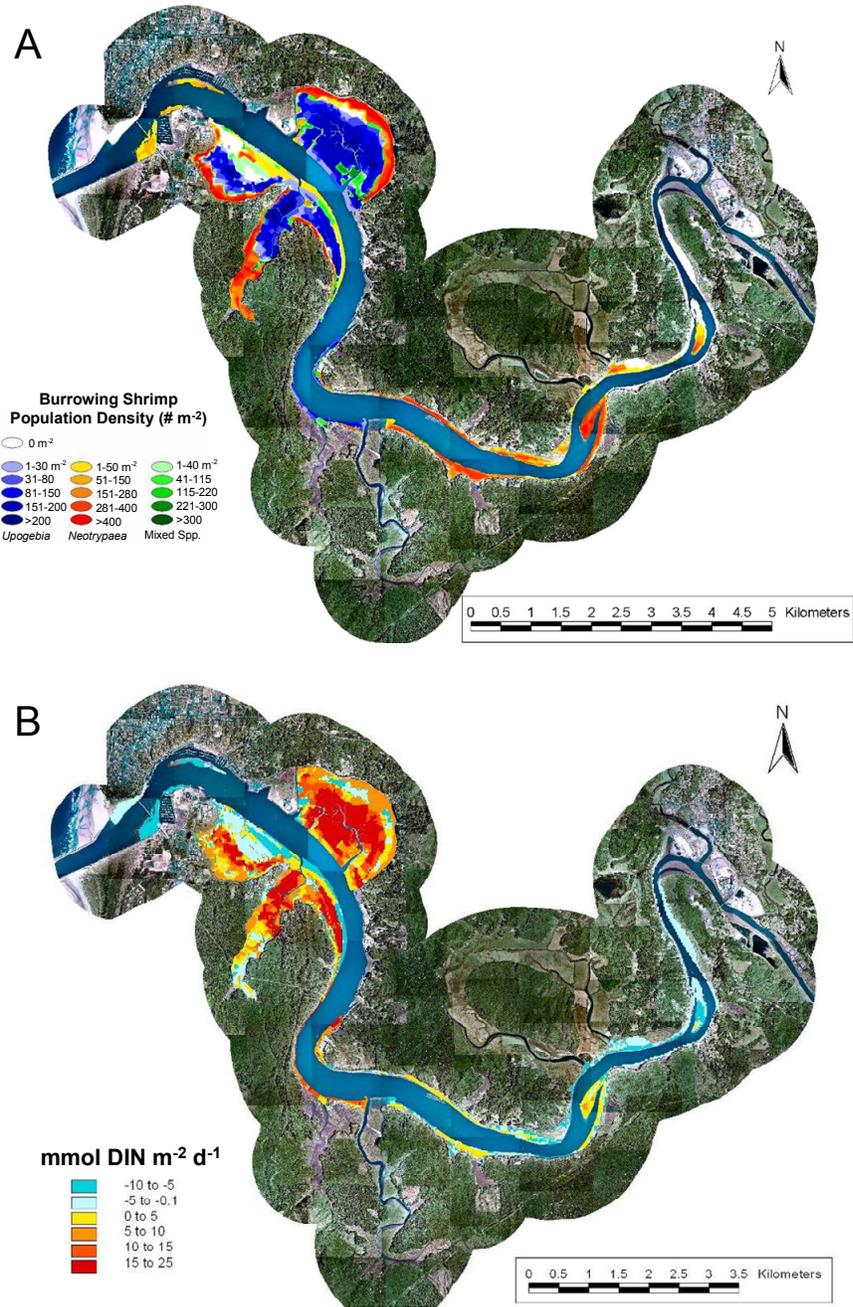


Figure A.3 a) Distribution and abundance of two species of burrowing shrimp (red = *Neotrypaea californiensis*, blue = *Upogebia pugettensis*, green = mixed species; darker color = higher density). b) Distribution and magnitude of benthic-pelagic flux of DIN in Yaquina Estuary (positive values = efflux from sediments, negative values = uptake by sediments).

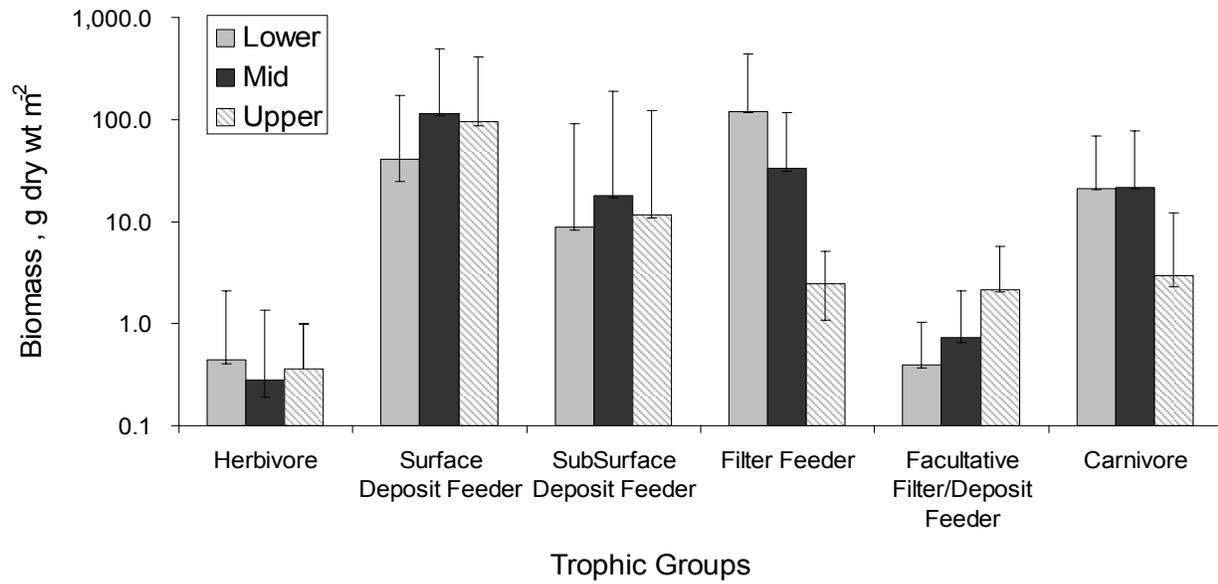


Figure A.4 Mean biomass distribution among trophic guilds in the lower, mid, and upper reaches of Yaquina Estuary. The error bars indicate minimum and maximum observed biomass values.

Appendix B: Description of Methods and Quality Assurance Procedures

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, 2001). Measurements Data Quality Objectives (MQOs) 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 data sets, 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 MQOs and were established from considerations of instrument manufacturer’s specifications, scientific experience, and/or historical data. A measure of systematic error (measured vs. true or expected): accuracy and the random error (precision) is presented. Accuracy is a measure of how close measured values are to true values. In this appendix, accuracy is calculated using the following equation:

$$Accuracy(\%) = (1.0 - (\sum (|V_t - V_n|) / n / V_t)) * 100$$

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:

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

where SD is the standard deviation, and \bar{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
Sea-Bird CTD and YSI Multiparameter Sondes					
Conductivity Sea-Bird CTD	mS cm ⁻¹	0-100	± 0.5% of reading	0.01 mS cm ⁻¹	SOP FSP.03, SeaCat manual
Salinity Sea-Bird CTD	psu	0 – 35 psu	± 0.5% of reading or 0.1 psu	0.01 psu	Calculated from conductivity and temperature
Temperature Sea-Bird CTD	°C	0 - 25 °C	± 0.15°C	0.01°C	SOP FSP.03, SeaCat manual
Turbidity Sea-Bird CTD	ntu	0-125 ntu	± 2% of reading or 2 ntu	0.1 ntu	SOP FSP.03, SeaCat manual
Depth Sea-Bird CTD	meters	0-15 m	± 0.018 m	0.001 m	SOP FSP.03, SeaCat manual
Conductivity YSI Sonde	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 YSI Sonde	psu	0 – 35 psu	± 1 % or 0.1 psu	0.1 psu	Calculated from conductivity and temperature
Temperature YSI Sonde	°C	0 to 25 °C	± 0.15°C	0.01°C	SOP IOP.09, YSI manual
Dissolved Oxygen YSI Sonde	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
Depth on YSI Sonde	meters	0-15 m	± 0.018 m	0.001 m	SOP IOP.09, YSI manual

Table B.1 Measurement Quality Objectives for data collected by Western Ecology Division.					
Parameter	units	Expected range	Accuracy	Precision	SOP(s) or other
Water Column Nutrients					
Dissolved NO ₃ ⁻	μM	0 - 100	±5%	5%	MSIAL UCSB, 2005
Dissolved NO ₂ ⁻	μM	0 - 1	±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
Total Nitrogen	μM	0-100	±5%	±5%	WRS 34A.3, 2005
Total Phosphorous	μM	0-3	±5%	±5%	WRS 34A.3, 2005
Total Suspended Solids	mg l ⁻¹	1 - 150	15%	15%	WRS 14B.2
Water Column chlorophyll <i>a</i>	μg l ⁻¹	0-200	±15%	±15%	MES SOP06.rev0
<i>Zostera marina</i> Data					
Biomass	gdw m ⁻²	0 -300 gdw m ⁻²	0.1 g ⁻¹ sample	0.1 g ⁻¹ sample	QAPP02.01
Shoot Density	shoots m ⁻²	1-15,000 m ⁻²	± 2 m ⁻²	1 shoot per m ² quad	QAPP02.01
Growth rate of leaf blades	gdw m ⁻² d ⁻¹	0 to 5 gdw m ⁻² d ⁻¹	95 %	95 %	QAPP02.01
Tissue CHN	g C/g tissue as %	0.1 - 50 %	± 10% of standard	CV ≤ 10%	QAPP02.01

Table B.1 Measurement Quality Objectives for data collected by Western Ecology Division.					
Parameter	units	Expected range	Accuracy	Precision	SOP(s) or other
Tissue P	g P/g tissue	0.1 - 10 %	± 10% of standard	CV ≤ 10%	QAPP02.01
Shoot Length	mm	5 - 1200 mm	± 2 mm	± 1 mm	QAPP 04.01
Shoot Width	mm	1 - 7 mm	± 1 mm	± 1 mm	QAPP 04.01
Shoot Dry wt.	mg	0-1500 mg	± 1 %	± 0.1 mg	QAPP 04.01
<i>Z. marina</i> lower limit depth	cm	0-10 MLLW	≤60 cm	n/a	
Epiphyte Biomass dry wt.	mg	0-1500 mg	± 1 %	± 0.1 mg	QAPP 04.01
%Plant Cover (Visual)	% area	0-100%	10%	10%	QAPP 98.01
Macroalgae					
% cover of algae, or Bare Substrate (visual estimate)	percent m ⁻²	0-100%	± 15%	0.70	QAPP 2000.01
Biomass	gdw m ⁻²	0 -300 gdw m ⁻²	0.1 g ⁻¹ sample	0.1g ⁻¹ sample	QAPP02.01
Macroalgae nitrogen isotope ratio (δ ¹⁵ N)	‰	-2 to +25	0.5 ‰	0.5 ‰	QAPP02.01; QAPP 01.02
Burrowing Shrimp Parameters					

Table B.1 Measurement Quality Objectives for data collected by Western Ecology Division.					
Parameter	units	Expected range	Accuracy	Precision	SOP(s) or other
Burrowing Shrimp Hole Density	holes m ⁻²	0-1500 m ⁻²	± 10%	0.70	QAPP 2000.01
Burrowing Shrimp Density	individuals m ⁻²	0-600 m ⁻²	± 10%	0.70	QAPP 2000.01
Benthic Invertebrate Abundance	individuals m ⁻²	0-5000 m ⁻²	± 10%	± 10%	QAPP 2000.01
Burrow Surface Area	cm ²	5-1600 cm ²	± 5 cm ²	± 1 cm ²	QAPP 2000.01
Burrow Volume	cm ³	1-800 cm ³	± 5cm ³	± 1 cm ³	QAPP 2000.01
General Lab					
Mass	mg	0.5-150	± 5%	± 5%	SOP IOP.01
Physical features					
Estuary bathymetry	ft	-40 to +10 MLLW	± 0.5	NA	survey contract D.Young PI
Ground position CMT GPS system, HP- GPS-L4	UTM: meters a) Easting b) Northing	414480 - 428210 4933695-4942656	± 0.5-2.5 m	± 1.5 m	CMT GPS system, HP- GPS-L4 Manual

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
SBE-19 package:					
Conductivity	SeaBird factory calibration	Solution of known conductivity	If drift is suspected	MQO	Factory return
Temperature	SeaBird factory calibration	Place in water bath with NIST traceable thermometer	If drift is suspected	MQO	Factory return
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
PAR sensors	LiCor factory calibration- every 2 years	Solar noon clear sky exposure	Bi-annually	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 $\pm 25\%$ of initial	Clean and re-test
YSI Multiparameter Sondes (6000 and 6600 models)	Factory calibrated for depth; calibrations performed to manufacturer's specifications	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		All parameters checked against factory standards			Clean and re-calibrate or return to factory
CMT-DGPS	Self-calibrates with satellites (speed of light as the calibrant)- 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 – every 2 years	Solar noon clear sky exposure	Bi-annually	MQO	Return to factory
Balance (5-place)	Factory representative adjusts on annual basis		Before each session	Within than class tolerance	Contact balance maintenance personnel
Drying oven		Check reading of thermometer	Before each use	MQO	
Fluorometer Turner 10-AU	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 $\pm 1\%$ from true value	Re-calibrate and re-run samples
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
Mass Spectrometer		Peak centering	Before each analysis	Center for each of the masses being measured	Adjust magnetic field and source high voltage; service if necessary
Mass Spectrometer		Nitrogen isotope ratio of working organic standard	Once every 10 samples	$\pm 0.2\%$ of long-term measured value	Repack combustion and reduction columns, reanalyze; 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

Water Quality CTD Profiles and Grab Samples

Profiles of water quality parameters were measured using the Seabird SBE 19 CTD with data logging capability. Variables measured by the CTD and other associated instrumentation are depth, temperature, conductivity, turbidity, photosynthetically active radiation (PAR), and *in situ* fluorescence, and calculated variables include salinity and density. During 2006 cruises, dissolved oxygen was measured at discrete depths. Discrete water samples were collected at representative depths (surface, mid-depth, and bottom) for analysis of chlorophyll *a*, total suspended solids, and dissolved inorganic nutrients. Since this report includes data collected for various projects and principal investigators, the number and location of discrete samples varied with sampling interval.

For each water quality profile, the CTD was lowered to the bottom, and discrete water samples were collected during the upcast at bottom, mid-depth and surface depths. 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. During 2006 cruises, dissolved oxygen was collected using a YSI 6600 Sonde attached to the CTD cage. 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 , $\text{NO}_3^- + \text{NO}_2^-$, NH_4^+ , and PO_4^{3-}), total nitrogen and phosphorous (2006 cruises only), 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 within several hours of collection.

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.

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 laboratory. Typically, the samples were filtered within 1-2 hours of sample collection. For TSS analysis, 1-liter of unfiltered seawater was collected at relative depths 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. Standard Operating Procedure for Preparation and Analysis of Estuarine Water Samples for Determination of Chlorophyll-*a* content. The samples were stored in the freezer until analyses. Chlorophyll *a* was extracted by sonicating the filters in 90% acetone and quantified using a fluorometer (10 AU Fluorometer, Turner Designs, Inc, Sunnyvale, CA). 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 $\pm 1\%$, the instrument was recalibrated 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 fluorometric chlorophyll *a* standards supplied by the manufacturer (Turner Designs) consisted of a high concentration ($181 \mu\text{g l}^{-1}$) and a low concentration ($18.2 \mu\text{g l}^{-1}$). 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* presented in this report. The accuracy and precision for the chlorophyll *a* data reported in this study are estimated to be 98.7%.

Nutrient Data

Nutrient analysis for nitrate+nitrite, 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 analysis 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 are analyzed periodically to audit performance. Deionized water blanks and sea water blank (low-nutrient natural sea water, aged to allow nutrient values to drop to near-zero levels) are also run. An independently-prepared “control” solution containing an intermediate concentration of each of the nutrients is also prepared. 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 is used to adjust the measurement timing parameters to compensate for the refractive index effect. Instrument calibration is checked at the beginning of a sample-batch run, at the end of the run, and periodically during the run. Each calibration sample is analyzed in duplicate, and the resulting data is used to establish calibration curves for each nutrient species. If the mean of the two replicates of any standard differs from the known concentration of that standard by more than ten percent or more than one-half the concentration of the lowest standard, whichever is greater, the calibration for that species is considered invalid and the calibration run is repeated. (See MSIAL Quality Assurance Manual 2005 for more details)

Table B.3 Precision and accuracy (expressed as %) of phosphate, silicate, nitrate+nitrite, ammonium, and nitrite Data.					
Measure	PO ₄ ³⁻	Si(OH) ₄	NO ₃ ⁻ +NO ₂ ⁻	NH ₄ ⁺	NO ₂ ⁻
Precision, %	97.6 (n=265)	98.3 (n=258)	98.2 (n=266)	98.0 (n=265)	94.6 (n=7)
Accuracy, %	98.2 (n=82)	98.2 (n=253)	96.5 (n=195)	96.8 (n=140)	97.9 (n=4)

Total Phosphorous and Total Nitrogen

Total nitrogen and total phosphorous analysis was performed by the Willamette Research Station (WRS), an EPA (WED) research facility in Corvallis Oregon. A Lachat Quikchem[®] 8000 Two-Channel FIA was used for analysis. WRS adheres to strict EPA QA/QC procedures and the following procedures are detailed in the standard operating procedure document WRS 34A.3 (2005).

A second source check standard (SSCS) (NIST traceable) was included in each automated analyses run. Instrument calibration and stability was validated after every tenth samples. A blank (Reagent water) was run after each SSCS to ensure negligible carryover. Calibration verification was monitored throughout the run by checking SSCS recovery. If measurement exceeded $\pm 10\%$ of the theoretical value, the instrument was recalibrated. A control check standard (QCCS) is a bulk sample digested at least once each digest batch set to show inter-run consistency. The bulk sample is collected as needed from a local stream or river and prepared in the same manner as samples. An analytical duplicate was run as a separate analysis (digestion and analysis) no less than once every 10 samples. Inter-run consistency and column performance was monitored with the QCCS (bulk sample) that is analyzed once each analytical run.

Three of each of the TN and TP quality control check standards (nicotinic acid and ascorbic acid) were digested with each batch to verify TN and TP recovery. Three reagent water blanks were digested to determine the nitrogen and orthophosphate blank present in the mixed persulfate digestion reagent. Digested standards and blanks were used to monitor digestion efficiency and background contribution of the persulfate. A method detection limit is the minimum concentration of an analyte that can be measured and reported with 99% confidence.

A MDL was established for each analysis based on at least seven repeated measurements (within a run) of a low level standard.

Table B.4 Precision, accuracy, and recovery for total nitrogen and phosphorous analyses.		
Measure	Total Nitrogen	Total Phosphorous
Precision, %	98.7 (n=39)	96.7 (n= 36)
Accuracy, %	101.8	97.7
Digestion Recovery, %	96.3	98.5

Handheld YSI Meters

The handheld multiparameter Yellow Springs Instruments (YSI) meters were checked prior to use with manufacturers (YSI) conductivity standard. To check the dissolved oxygen reading the DO probe was placed in a 100% saturated environment for several minutes. If the percent saturation value was several points away from 100% the electrodes were cleaned, a new KCL solution was reapplied and the membrane was replaced. Another percent saturation reading is taken after the DO probe has been serviced. All QA/QC checks and calibrations recorded in a database (Microsoft[®] Office Access 2003). The frequency of checks and calibrations was intermittent and depends on how often the units are being used.

YSI Multiparameter Sonde

YSI 6600 Multiparameter 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 dissolved oxygen (DO) sensor was calibrated for dissolved oxygen 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 can not be calibrated but its performance was checked. The data sondes 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 is defined as the accuracy of the probe in a standard solution. Post deployment accuracy is defined as the accuracy of the probes after they are retrieved from the field and are tested against the known standard solutions.

	Temperature	Salinity	Conductivity	Turbidity	DO
Calibration Accuracy Pre-deployment	99.0	98.2	100.0	100.0	99.6
Post Deployment Accuracy (includes the effect of biofouling)	98.7	96.5	97.8	97.1	94.4

Burrowing Shrimp Densities

Shrimp densities were assigned classes (0 holes, 1-10 holes, 11-50 holes, 51-100 holes, and 101-175) by direct counts of burrow holes within a 0.25 m² quadrat. These were designated as classes 0-4, respectively. Shrimp burrow identity and density-class, date, time, and geographic location data were collected along a survey track using a dynamic line setting recording the data onto a March II GPS-data logger (Corvallis Microtechnology Inc., Corvallis, OR) at 1-s intervals. As a change in species or density was observed, the line feature was ended, and a new dynamic line was started with the new species and density attribute. Burrow identity and density classification was verified approximately every 30 min during the survey by qualitatively sampling burrows using a bait pump and counting burrow hole densities using a 0.25 m² quadrat. The same procedure was also used whenever the survey team was uncertain of burrow attributes. Each 'QA' quadrant was also photographed with a digital camera mounted on a PVC frame to provide a consistent reference. As a check on burrow hole counting, the burrow holes were later re-counted from the digital pictures.

To quantify the relationships between burrow density class, burrow opening density, and shrimp density, spatially coincident “baseline” samples were collected at 90 random sites throughout Yaquina estuary. Shrimp density was determined by hydraulically excavating a megainfaunal core barrel (40 cm diameter x 100 cm depth) using a suction-dredge, and washing the core barrel contents through a 3 mm mesh to retain burrowing shrimp and other large infauna.

Stable Isotope Data

Macroalgae samples were collected monthly during 2003 and 2004 from five locations along the salinity gradient ranging from polyhaline to oligohaline conditions. Algal material was collected from hard substrates to eliminate contamination from any additional nutrient sources other than the water column. Five replicates of healthy macroalgae were collected from each sampling site. Samplers wore sterile lab gloves while collecting to prevent contamination. Each algae sample was washed thoroughly in RO 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 samples for $\delta^{15}\text{N}$ according to SOP CL-6 (1999). Nitrogen isotope ratios were measured on macroalgae samples combusted in 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.

Table B.6 Precision and Accuracy of $\delta^{15}\text{N}$ data.	
	$\delta^{15}\text{N}$
Precision, %	99.1 (n=23)
Accuracy, %	96.6 (n=63)

Aerial Mapping of Seagrass and Macroalgae

The remote sensing procedure used in this study to map the intertidal distribution of eelgrass and benthic green macroalgae (conducted from 1997 – present) utilizes aerial photography with false-color near-infrared (color infrared, CIR) film. This allows an aerial survey to be conducted during daylight low tide (typical tide level about -0.5 m MLLW) when the majority of the eelgrass habitat in the Yaquina Estuary is exposed. CIR film has been found to provide substantially better spectral resolution of exposed intertidal vegetation than has true color film (Young et al., 1999). The mapping method is able to detect inundated and submerged *Z. marina* to a depth of about 1 m below water level at the time of the aerial photograph. To map perennial eelgrass habitat, the surveys are conducted in late spring or early summer before the summer bloom of benthic green macroalgae that can interfere with the classification of eelgrass habitat. Mid-summer surveys are used to map macroalgal distributions upslope of the eelgrass meadows. Photoscales utilized range from about 1:6,000 to 1:20,000. The aerial photographs are digitally scanned and georectified while correcting for terrain and camera distortions to produce 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 $0.72 \text{ m} \pm 0.27 \text{ m}$ (95% CI; Clinton et al., in review). The digital orthophotos are classified into eelgrass and bare substrate habitats, defined as $\geq 10\%$ cover or $< 10\%$ cover, respectively. On-the-ground resolution of 0.25 m is 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., in review). 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.

Another part of the ground survey employs 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 review). Results were obtained in spring 2004 from 51 randomly positioned stations within intertidal eelgrass 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 of 97%, with a Kappa Index value of 0.9447 ± 0.0024 , indicating 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 eelgrass meadow margins (Young et al., in review).

*Depth Distribution of *Z. marina**

The amount of *Z. marina* at a specific tidal height (Figures 11.2 and 11.3) was determined by overlaying the *Z. marina* maps with the results of an extensive bathymetric depth survey conducted by the U.S. Army Corps of Engineers in 2002. The bathymetric model and the seagrass classification are both ArcInfo format grids. The bathymetric depth data was interpolated onto a grid using TopoGrid in ArcInfo. A 2.0 m floating point bathymetric model grid was integerized and resampled to 0.25 cell size to match the binary seagrass classification grid. The bathymetric and seagrass grids were then overlaid using the map algebra function COMBINE, which produces a grid value attribute table with counts of cells for each unique combination of cell values from each grid.

The aerial mapping method utilized is capable of classifying some submerged seagrass beyond the depth at which near-infrared radiation is absorbed by water; however, some of the data returned by the COMBINE function is undoubtedly a result of spatial misregistration between the aerial photo classification and the bathymetric model. A few outliers were trimmed from both tails of the distribution curve. It is also quite possible that a physical survey could return some higher percentages of overall seagrass distributions below -1 m MLLW. The result of this analysis is presented as distribution curves of total area of intertidal eelgrass for ocean and river dominated estuarine areas.

Macroalgal Biomass and Cover

Benthic green macroalgae coverage and biomass measurements were conducted as part of several studies in Yaquina Estuary between 1998 and 2004 (Table B.7). Most of the measurements were made during the dry season (May – October) in the marine-dominated sector (Zone 1). Although both non-random and random sampling designs were employed in different survey efforts, all the macroalgal data within Yaquina Estuary (WED unpublished data; Kentula and DeWitt, 2004) were combined for analysis. To obtain a regional perspective, the data from the Classification Study (Section 4.2) also were summarized for comparison (Lee et al., 2006).

Table B.7. Sources of ground survey macroalgae data used in this study. All data were collected by WED. Random and non-random sampling designs are denoted by R and NR, respectively.				
Period	Zone	No. of Samples		Sampling
		Zone 1	Zone 2	
1998	1	69	-	NR
1998-99	1	65		R
1999-2003	1	4159	-	R
2001-2002	1	2094	-	NR
2002-2004	1	140	-	NR
2004	1 & 2	47	53	R

To examine seasonal variability in benthic green macroalgae cover and biomass, six band transects were established within Zone 1 of Yaquina Estuary during 1999. These sites (I –VI, respectively) were situated 3.9, 5.0, 6.3, 7.8, 8.6, and 10.6 km from the ocean end of the jetties at the mouth of the estuary (Figure 2.1). The transects were 30 m wide, and extended 100 m upslope from the MLLW tide line perpendicular to the channel. Along each transect, sampling stations were regularly spaced at 10 m intervals. At Sites II and III, the bathymetric slope was lower than for other sites, and in August 1999 the transects were extended to about 350 m from MLLW, with the additional sample stations situated at ~ 40m intervals. Stations along the transects were generated with a random number generator prior to field sampling. For each station, three randomly-selected distances between 1 and 30 m perpendicular to the sampling transect were sampled. Between June and December 1999, most of the sites were sampled every other week during daylight low tides, while less frequent sampling of these sites continued through May 2000.

At each plot, a percent cover value within a 0.25 m² quadrant was visually estimated for SAV and macroalgae. For much of the data, a frequency-of-occurrence value was also measured by recording which class of cover occurs directly beneath each of 25 point intercepts formed by two orthogonal sets of string intercepts. The purpose of such frequency-of-occurrence measurements was to provide a quality check on the estimated percent cover values. Pilot studies comparing the linear regression of the estimated percent cover values against the measured percentage frequency of occurrence values have yielded r² values of 0.91 - 0.97. For this current study only the green macroalgae data was used for analysis.

Samples of the alga taxa was collected from a 0.05 m² area of the 0.25 m² quadrant at two of the three replicates sites along the station line. After cleaning, the alga was identified by qualified individuals. Alga keys and identification aides were available for cross-referencing. The alga biomass was determined by drying the alga at 80°C until dry. A sample was considered dry when there was no further weight change due to loss of water after additional oven drying. Returning a subset of sample to the oven after recording the initial dry weight was also a measure of repeatability. The final dry weighs for the 0.05 m² sample was converted into gdw m⁻².

An alternative method was used to determine macroalgae biomass in the 2002 field season. A volumetric biomass estimate was collected according to the graduated cylinder method as described in Robbins and Boese 2002. The macroalgae was collected from a 0.25 m² quadrant and placed into the 2000 ml cylinder. The algae were pressed with a plunger to remove excess water before recording the algal volume in ml. This quick field method was determined to be an accurate surrogate for biomass dry weight determination with a linear relationship yielding a r² value of 0.78 to 0.88 for macroalgae species (Robbins and Boese 2002).

***Zostera Marina* Lower Depth Limit**

The lower depth margin of *Z. 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 by 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 waters surface and could easily be approached by foot. Personnel walked from shore along the transect to the deepest *Z. marina* patch, recorded a GPS location, measured the depth with a lead line and collected other water quality parameters.

Tidal corrections for Lower Depth Limits of Z. marina

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 of *Z. marina* was measured, suggests that conditions were relatively calm during the sampling and not collected during extreme tides. In addition, 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 amplitude and phase lags of tides are available for four locations (South Beach, Yaquina, Winant, and Toledo) in the Yaquina Estuary (<http://co-ops.nos.noaa.gov/tides05/tab2wc1b.html#132>). 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 maximum distance along the river any given depth station from a tide station was 4.5 km. The largest source of error in the tidal corrections results 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, we compared the differences in tidal corrections between the two stations that are located upstream and down stream of the observations. 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.

Epiphyte Methods

Epiphytes growing on *Z. marina* leaves were collected within the Yaquina Estuary from 2000 through 2004. Data were collected at six stations distributed between 3.5 and 17 km upriver from the mouth of the Yaquina Estuary. Leaves from collected plants were subdivided into outer (older) and inner (younger) leaves. Epiphytes were scraped from these leaf groups with dry weights (24-36 hours at 60-70 °C) of the removed material determined for each individual plant. The effect of epiphyte cover in terms of reducing light (PAR) availability to eelgrass 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.

Station Location

The geolocation 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 1-5 m.

Distance Upriver Calculations

The distance from the mouth of the Yaquina to each station was calculated using GIS mapping software Arcview. After a center line shape file extending the length of the river was converted into a route file, the orthogonal distance from each station to the nearest point along that route file was calculated. The ArcToolBox linear referencing tool, `LocateFeaturesAlongRoutes`, was used to calculate the distances in meters for each station from the mouth of the estuary to the nearest point along the centerline route feature.

Landscape Data

The Yaquina landscape analysis was done as part of a larger Pacific west coast estuary survey. Watershed boundaries subtending the Yaquina estuary basin were primarily determined from the Sixth Field hydrologic unit code (HUC) 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.—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. Boundary lines and water bodies were plotted and reviewed for accuracy of coding and fidelity to the original sources. The Yaquina watershed delineated in this project captures the entire drainage area (EDA). By delineating the entire watershed, the watershed area is 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://spo.nos.noaa.gov/projects/cads/description.html#caf>).

Land Cover Sources

The estuary watershed was 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 Yaquina watershed boundary. The 1992 NLCD data contains 21 classes of land cover (see <http://erg.usgs.gov/isb/pubs/factsheets/fs10800.pdf>). 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 use 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. The 2001 NOAA data are based on 22 land use 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/classification.htm> for a crosswalk of the two schemes).

Land Use 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 use 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 a watershed. Accuracy of the NLCD data of the MRLC zone that contains the Yaquina watershed is estimated to be 86.1%. These data sets 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, 2004b). The MRLC 2001 impervious surface layer, which represents an estimate of developed impervious surface per pixel by percent imperviousness, was clipped and summarized for the 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 use 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) DEMS (digital elevation models). 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.

Population Density

Human population estimates from the 1990 and 2000 censuses (<http://www.census.gov/>) were generated for the drainage unit. Area weighted estimates of total population by census block were summed for each drainage and population density (individuals/sq. kilometer) was calculated from the total drainage population estimate.

Other Data Sources

Historical sources (data collected in the 1960, 70 and 80's and presented in technical reports, thesis dissertations or journal publications) were carefully reviewed for information on quality procedures implemented in their data collection. Methodological approaches for measuring DO, chlorophyll-*a*, and nutrients were reviewed for analytical method utilized. Data obtained from the Oregon Department of Environmental Quality database were previously QAed and only data with grades of A and A+ were used as part of this report. Often historical data collected before the use of GPS systems were given as locations on map, common station names, or as kilometers upriver. From these descriptions, the northing and easting UTM locations were estimated using Topozone or Google Earth. Locations given in Latitude and Longitude were converted into UTM units using a batch converter located at <http://www.uwgb.edu/dutchs/UsefulData/HowUseExcel.HTM> or in Topozone. Units were converted to metric units (i.e. feet to meters, Fahrenheit to Celsius, etc.). Most of the data was hand entered from tables to electronic format in Excel. Data was entered electronically by one person and checked for errors by another independent person. Data points presented only in graphical format were digitized into electronic format and translated into tables. Errors and any resulting changes were documented and traced to the source. All data were entered into an Access database and data were reviewed to ensure that there were no duplicate entries.

Models

Stress-Response Model

All field and laboratory data used in the development of the Stress-Response model (SRM) were collected in accordance with WED SOP's and QAPP's. (Tables B.1 and B.2). A complete list of these parameters is found in Kaldy and Eldridge (2006) Volume II-Tables 1, 2, 3, 4, and 5. All the measurements of physical and biological quantities used in our modeling are subject to uncertainties. Further these measurements were often combined to produce new derived quantities, each of which has its own uncertainties. We calculated these uncertainties using means and standard deviations of the data and error propagation algorithms from http://teacher.nsrj.rochester.edu/phy_labs/AppendixB/AppendixB.html.

Once the SRM was calibrated to the biomass data (see Kaldy and Eldridge 2006, Figure F.1), we conducted a series of tests to examine the models sensitivity to parameters. A model that is overly sensitive to parameters is considered unstable. Kaldy and Eldridge (2006) Volume II-Table 8 provide a complete sensitivity analysis of the model used herein and a discussion of the sensitive results is presented on page 55 of Kaldy and Eldridge (2006).

Another aspect of the model quality assurance is the development of validation experiments. We developed the model using local seagrass and environmental data, but plan to use the model to address regional or national level questions. *Z. marina* physiology and genetic alleles (minor variations of the same gene) diverge significantly in different regions of the continental United States. The regional differences in *Z. marina* require that we run our validation experiments at multiple scales. At the local scale we have developed plant level-tracer experiments to evaluate the allocation of carbon within a plant (Kaldy and Eldridge 2006). At the regional scale we are planning running validation experiment in Puget Sound, Washington during 2007. At the national scale we have conducted *Z. marina* mesocosm experiment in Narragansett, Rhode Island in collaboration with AED and the University of Rhode Island. These data will be used to validate or recalibrate the SRM for regional or national level implementation of the SRM. The combination of the data uncertainty analysis, the model calibration and sensitivity analyses, and the local, regional, and national scale SRM validations constitute our QC/QA program.

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 SOP's and QAPP's. (Tables B.1 and B.2). 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 $\mu = 0.1 \text{ d}^{-1}$. 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_O 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 ($\delta_W = 15\text{-}22\text{‰}$) 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 ($\delta_R=2\text{‰}$, $\delta_W=20\text{‰}$, and $\delta_O=8.4\text{‰}$) 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).

List of Quality Assurance Project Plans (QAPPs) Used in This Study

- QAPP98.04. Evaluation of the Susceptibility of Eelgrass Beds in Oregon Estuaries to Changes in Watershed Uses. R. J. Ozretich, EPA, 1998.
- 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.
- Marine Science Institute Analytical Laboratory University of California, Santa Barbara. Quality Assurance Manual-Draft. 2005.

List of 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.
- MES EP01.rev 0. Draft. Standard Operating Procedure for Collecting and Processing *Zostera marina* and Associated Epiphytes for Light Attenuation Measurements. Dynamac Corporation. 2004.
- MES SOP09.rev 0. Standard Operating Procedure for Preparing Water Samples for Nutrient Analysis. Dynamac Corporation. 2003.
- 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.
- WED SOP06.rev 0. Standard Operating Procedure for Preparation and Analysis of Estuarine Water Samples for Determination of Chlorophyll-*a* Content. Dynamac Corporation. 2004.
- 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.

WRS 34A.3. Standard Operating Procedure for the Digestion and Analysis of Fresh Water Samples for Total Nitrogen and Total Phosphorus. 2005.

Clinton, P.J., Young, D.R., and Specht, D.T. In Review. 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. *SOP-NHEERL/WED/PCEB/PJC/06-01-000 09/15/06*, U.S. Environmental Protection Agency, Pacific Coastal Ecology Branch, Newport, OR.

Appendix C: Classification of Oregon Estuaries

Of the estuaries that were sampled as part of the Classification and NCA data sets, the number of classes of estuaries (or types) depends upon the scale of the classification system as well as the classification system utilized (see Table C.1). Of the seven estuaries sampled by WED in 2004-2005 which form the Classification data set, five were also included in a NOAA classification scheme, with 4 estuaries classified as “river dominated with straits and terminal bay”, and 1 estuary classified as “coastal embayment – v-shaped and semienclosed.” Lee et al. (2006) classified all seven of the estuaries sampled in the Classification effort as “drowned river valley” and Bottom et al. (1979) classified 6 of them as “partially mixed” and the 7th (Coos) as “well mixed.” For the Oregon estuaries sampled as part of the NCA effort, the NOAA classification would define eight as “river dominated with straits and terminal bay”; one as “river dominated with salt wedge” (Columbia), one as “coastal embayment” (Coos), and one as “lagoon” (Netarts).

Quinn et al. (1991) classified West Coast estuaries based on their relative susceptibility to nutrient pollution, defined as an estuary’s capability to concentrate dissolved and particulate pollutants. In their 1991 study, Quinn et al. classified the estuaries by dissolved concentration potential (DCP), which is the ability of the estuary to concentrate dissolved substances, and particle retention efficiency (PRE), which is a measure of the ability to retain suspended particulates within the estuary. In this classification system, 8 Oregon estuaries (including Alsea, Coos, Nehalem, Netarts, and Siletz, Siuslaw, Tillamook, and Yaquina) classified as having “high” DCP and “low” PRE. Umpqua Estuary classified as “medium” (near border of high) DCP and “low” PRE, while the Columbia River Estuary had “low” DCP and “low” PRE. All of the Oregon estuaries classified as being in the medium category of nitrogen concentrations (estimated using the loadings and DCP).

Burgess et al. (2004), Engle et al. (2007), and Bricker et al. (in prep.) classified estuaries of the United States using statistical cluster analysis of physical and hydrologic variables to determine the response of estuaries to nutrient loading. The Burgess et al. (2004) classification included physical and hydrologic parameters including estuarine area, estuary drainage area, area of mixing, seawater and tidal fresh portions of the estuary, tide, riverflow, estuary volume, tidal prism volume, salinity, depth, DCP and PRE. The primary variables contributing to the separation of 11 clusters (or classes of estuaries) in the Burgess et al. (2004) classification were

size of the estuarine drainage area, estuary area and volume, riverflow, depth and salinity. In a more recent estuarine classification, Engle et al. (2007) updated the Burgess et al. (2004) classification scheme to incorporate average air and water temperature and surface and bottom water temperature and found that the estuaries clustered into 9 classes. Bricker et al. (in prep.) also used a cluster analysis to classify the same estuaries and they found that the estuaries were best clustered by estuary depth, tide, ratio of freshwater input to estuary area, temperature, and mouth openness, resulting in a classification with 10 classes of estuaries.

Using the Engle et al. (2007) classification, the estuaries sampled in the Classification data set fall into two classes (Alsea and Umpqua in one; and Coos, Tillamook, and Yaquina in a second class). In the Bricker et al. classification, the estuaries sampled in the Classification data set fall into two classes; however, Alsea, Tillamook, Umpqua and Yaquina are in one class and Coos is in another. The Oregon estuaries sampled in the NCA data set fall into 3 classes in the Engle et al. (2007) classification with 8 of the 11 classified as within one class, while in the Bricker et al. classification these same estuaries fell into 2 classes with 9 of the 11 being in the same class.

Table C.1. Classifications of 14 Oregon estuaries sampled in WED Classification Study and NCA data sets (data set denoted as C and N, respectively in first column). Lee et al. (2006) Classification based on Oregon Coastal Atlas (<http://www.coastalatlantlas.net>) and Bottom et al. (1979).

Estuary	NOAA		Geomorphology Lee et al (2006)	Stratification Bottom et al. (1979)	Burgess et al. (2004)	Engle et al. (2007)	Bricker et al. (in prep.)	Quinn et.al (1991)			
	Class	Type						Dissolved Concentration Potential	Particle Retention Efficiency	Predicted Concentrations	
										N	P
Alesea (C,N)	River Dominated	Straits w/ Term. Bay	Tidal dominated drowned river	Partially Mixed	9	1	4	H	L	M	M
Columbia (N)	River Dominated	Salt Wedge	River dominated drowned river	NA	1	5	4	L	L	M	M
Coos (C,N)	Coastal Embayment	V-shaped & semi-encl.	Tidal dominated drowned river	Well Mixed	6	8	1	H	L	M	M
Nehalem (N)	River Dominated	Straits w/ Term. Bay	Tidal dominated drowned river	Partially Mixed	9	8	4	H	L	M	L
Nestucca (C,N)	NA	NA	Tidal dominated drowned river	Partially Mixed	NA	NA	NA	NA	NA	NA	NA
Netarts (N)	Lagoons	Limited FW Inflow	Bar built	Well Mixed	11	8	1	H	L	M	L
Rogue (N)	River Dominated	Straits w/ Term. Bay	River dominated drowned river	Partially Mixed	9	8	4	M	L	M	M
Salmon R. (C,N)	NA	NA	Bar built or drowned river	Partially Mixed	NA	NA	NA	NA	NA	NA	NA
Siletz (N)	River Dominated	Straits w/ Term. Bay	Tidal dominated drowned river	Partially Mixed	9	8	4	H	L	M	M
Siuslaw (N)	River Dominated	Straits w/ Term. Bay	Tidal dominated drowned river	Partially Mixed	9	8	4	H	L	M	L
Tillamook (C,N)	River Dominated	Straits w/ Term. Bay	Tidal dominated drowned river	Partially Mixed	9	8	4	H	L	M	L
Umpqua (C,N)	River Dominated	Straits w/ Term. Bay	River dominated drowned river	Partially Mixed	9	1	4	M	L	M	M
Yachats (N)	NA	NA	Tidally Restricted coastal Creek	NA	NA	NA	NA	NA	NA	NA	NA
Yaquina (C,N)	River Dominated	Straits w/ Term. Bay	Tidal dominated drowned river	Partially Mixed	6	8	4	H	L	M	M

Legend: NA denotes classification not available; H, M, and L represent high, medium, and low classes (respectively).

Appendix D: Survey of Effects of Macroalgae on Biota

Publications concerning the ecological effects of macroalgae were reviewed, and reported threshold values of percent cover and/or biomass for effects on infauna, epifauna, fishes, and shorebirds are summarized in Table D.2

Table D.2. Summary of literature regarding the effects of macroalgae (biomass or percent cover) on estuarine infauna, epifauna, fishes, and shorebirds.

Taxa	Type	Location	T (° C)	Sediment H ₂ S or Low Eh	Effect*	Macroalgal Density	Citation
Shorebirds	S	England (S. coast)		Y	↓ abundance	Cover: 75% in 20% of intertidal	Tubbs (1977)
Shorebirds	S	England (S. coast)		Y	Neg. correl.: areas densest algal mats vs. abundance	Cover: >25% in 40% of intertidal	Tubbs & Tubbs (1980)
Infauna Epi. fauna Shorebirds	S	England (S. coast)		Y	↓ infauna ↑ epibenth. fauna ↓ shorebirds	Cover: 42 %	Nicholls et al. (1981)
Infauna Shorebirds	S	England (S. coast)			Little effect on zoobenthos or shorebirds	~ 300 gdw m ⁻²	Soulsby et al. (1982)
Shorebirds	S	England (S. coast)			↓ abundance (refutes Soulsby et al., 1982)	Cover: >25% in 40% of intertidal	Tubbs & Tubbs (1983)
Infauna/ Epi. fauna	F	Ireland (S.W. coast)		Y	↓ infauna ↑ epi. fauna		Thrush (1986)
Infauna	F	Scotland (E. coast)		Y	↓ amphipods ↑ polych. & bivalves	60 gdw m ⁻²	Hull (1987)
Infauna	F	Sweden (S. coast)	~20		↓ larval settlement	143 gdw m ⁻²	Olafsson (1988)
Infauna	S	Scotland (E. coast)			Altered infaunal composition	440 gdw m ⁻²	Raffaelli et al. (1989)
Infauna	F	Central California		Y	↓ bivalves ↓ phoronids	800 gdw m ⁻² Cover: 100 %	Everett (1991)
Infauna	F	Scotland (E. coast)		Y	↓ amphipods by ~90 %	200 gdw m ⁻²	Raffaelli et al. (1991)
Infauna	S	North Baltic Sea	~16		↓ bivalves by 73 %	832 gdw m ⁻²	Bonsdorff (1992)
Epibenthic fauna	S	Sweden (W. coast)	14 - 20		↓ epi. fauna by > 50 %	Epiphyte cover on eelgrass: 80%	Isaksson & Pihl (1992)
Juvenile flatfish	S	Sweden (W. coast)	4 - 20		↓ abundance	Cover: 50%	Pihl & van der Veer (1992)
Infauna	F	Central California			↓ bivalves by ~ 70 %	220 gdw m ⁻² Cover: 100 %	Everett (1994)
Decapods Pred. Fish	F	Sweden (W. coast)	10-20		↑ decapods ↓ foraging	Cover: 35 % Cover: 75 %	Isaksson et al. (1994)
Fishes	S	Sweden (W. coast)	14-20		↓ fish biomass	7 – 225 gdw m ⁻²	Pihl et al. (1994)
Juvenile Flatfish	L S	Sweden (W. coast)	9–12		↓ settling ↓ abundance	Cover: 80 % 179 gdw m ⁻²	Wennhage & Pihl (1994)
Epibenthic fauna Juvenile Flatfish	S	Sweden (W. coast)	14-20		↑ epi. fauna ↓ epi. fauna ↓ fish sp. & foraging	Cover: 30-50% Cover: 90% Cover: 30-40%	Pihl et al. (1995)
Shorebirds	S	Portugal			No effect on feeding	30-60 gdw m ⁻²	Múrias et al. (1996)

Zoobenthos	F	North Baltic Sea	11-16	Y	↓ abundance & biomass by 87-94 %	440 gdw m ⁻²	Norkko & Bonsdorff (1996a)
Gastropod & bivalve Mollusks	F	North Baltic Sea	11-16	Y	↓ abundance & biomass	440 gdw m ⁻²	Norkko & Bonsdorff (1996b)
Infaunal bivalve & shrimp	L	North Baltic Sea	6 - 20	Y	↓ survival by 80% ↓ survival by 83%	200 gdw m ⁻² 440 gdw m ⁻²	Norkko & Bonsdorff (1996c)
Zoobenthos	S	North Baltic Sea		Y	Alters community	300 gdw m ⁻²	Bonsdorff et al. (1997)
Zoobenthos	F	Scotland (E. coast)		Y	↓ abundance of invert. prey of fishes/birds	≤ 600 gdw m ⁻²	Raffaelli et al. (1998)
Infauna	L	North Baltic Sea	20		↓ juv. bivalve moll.	Cover: 50 %	Norkko (1998)
Infauna	S	Maine, U.S.A.		Y	Shifted community structure to detritivores	50 – 200 gdw m ⁻²	Thiel & Watling (1998)
Infauna	F	Scotland (E. coast)		Y	Altered infauna	200 gdw m ⁻²	Bolam et al. (2000)
Infauna	F	Portugal			↑ infaunal detritivores	158 gdw m ⁻²	Lopes et al. (2000)
Zoobenthos	S L	North Baltic Sea	19	Y	Altered structure. ↓ bivalve & amphipod	181 gdw m ⁻²	Norkko et al. (2000)
Amphipods	S	Portugal		Y	Population zeroed by algae crash	413 gdw m ^{-2**}	Pardal et al. (2000)
Shorebird Black-tailed Godwit	S	Ireland (S. coast)			↓ abundance	Cover: 40-70%	Lewis & Kelly (2001)
Infauna	F	Sweden (W. coast)	17-18	Y	↓ suspension feeding bivalves by ~90 %	300 gdw m ⁻²	Österling & Pihl (2001)
Epibenthic Mudsail	S	Portugal (W. coast)			↑ abundance	250 gdw m ^{-2**}	Cardoso et al. (2002)
Copepods	F	New York, U.S.A.		Y	↓ copepods by ~85%	100 gdw m ⁻²	Franz & Friedman (2002)
Infauna Pred. birds	F	Ireland (S. coast)			↓ amphipod abundance; Black-headed gulls avoid algal cover	206 – 277 gdw m ⁻²	Lewis et al. (2003)
Zoobenthos	F	Portugal (W. coast)		Y	↑ abundance mudsnail, polych. worm	600 gdw m ⁻²	Cardoso et al. (2004)
Infauna	F	Australia (E. coast)		Y	↓ polychaete, molluscan abundance	450 gdw m ⁻²	Cummins et al. (2004)
Infauna	S	England (S. coast)		Y	↓ infaunal abundance ~70% ↓ infaunal biomass 70%	Cover: 90 %	Jones & Pinn (2006)
Zoobenthos	S	NE Baltic Sea			Minor alterations to community structure	347 gdw m ⁻² Cover ~25%	Lauringson & Kotta (2006)
Infauna	F	Netherlands			No major changes in infauna	80 gdw m ⁻² buried in sediment	Rossi (2006)
Type of Study: S is survey, F is field manipulation, L is lab experiment; Effects: ↓ decreased; ↑ increased; * For surveys, negative correlation suggesting a possible effect; ** Ash free dry weight							

Appendix E: Stressor-Response Model Calibration, Input Data, and Results

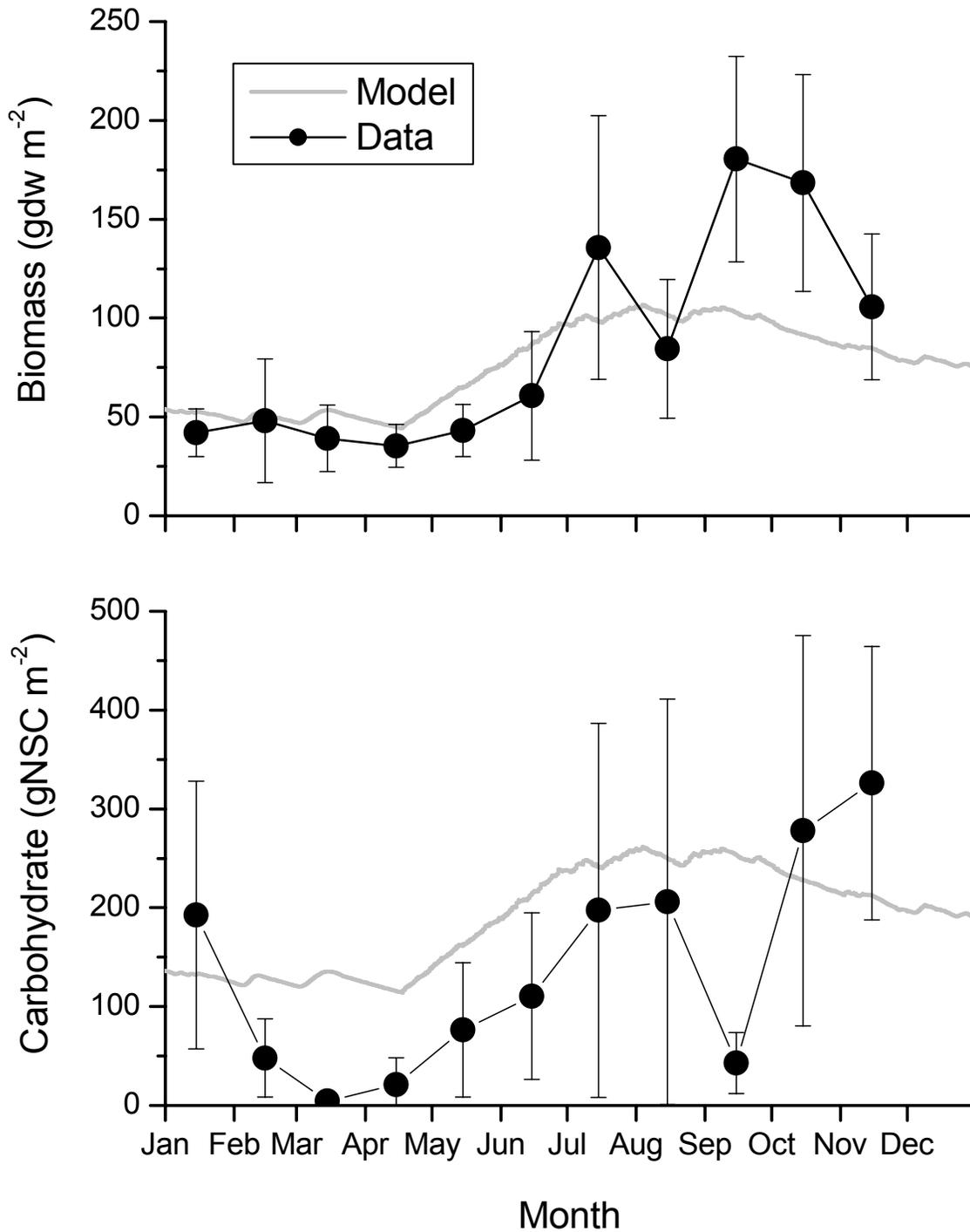


Figure E.1 Stressor-Response Model Calibration for biomass and carbohydrate during 2002 in *Zostera marina*. Error bars represent standard errors.

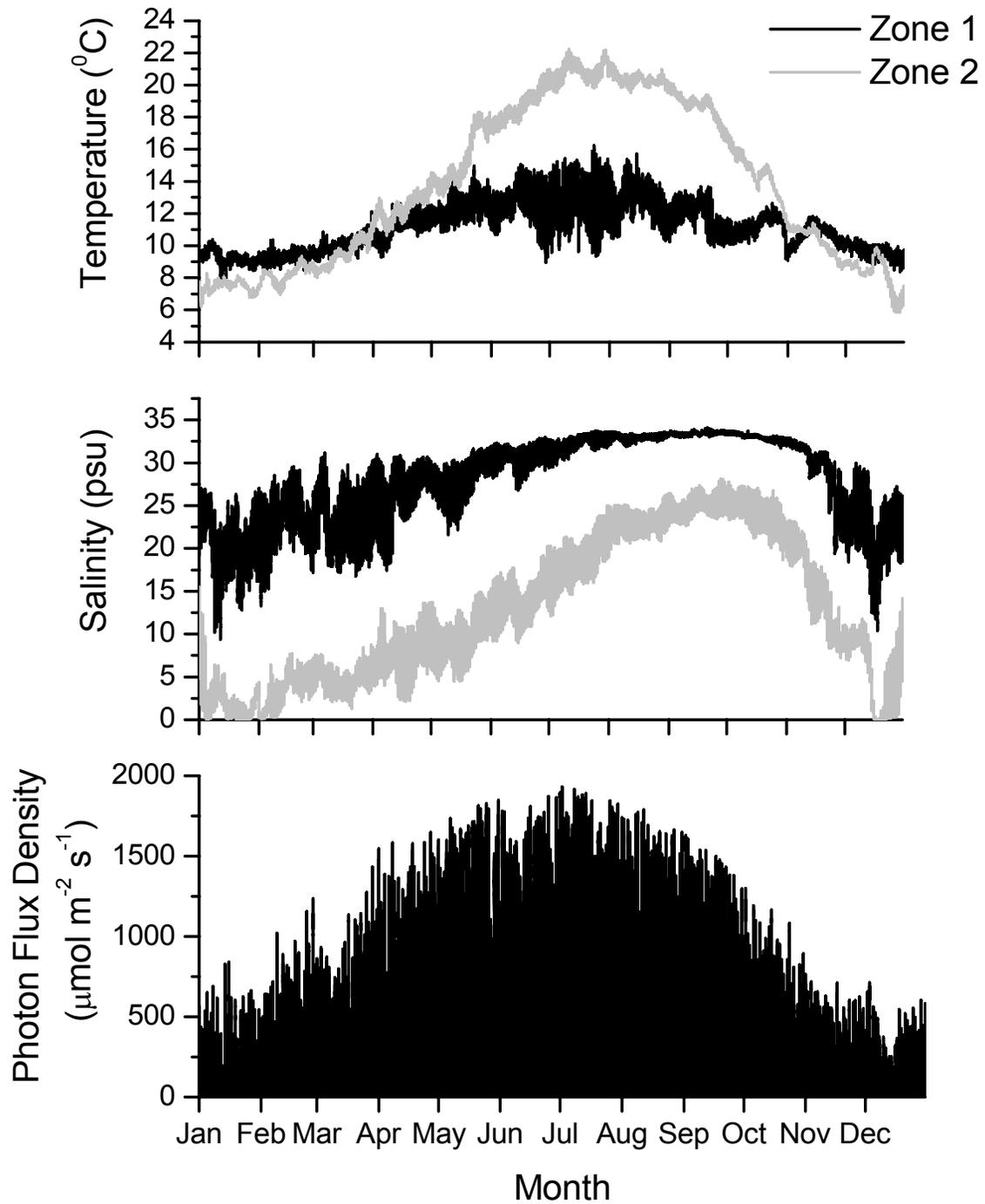


Figure E.2 Environmental input data (temperature, salinity and photon flux density) used in the SRM simulations for Zone 1 (lower estuary) and Zone 2 (upper estuary). Data presented are a composite created by averaging data from 1999-2003.

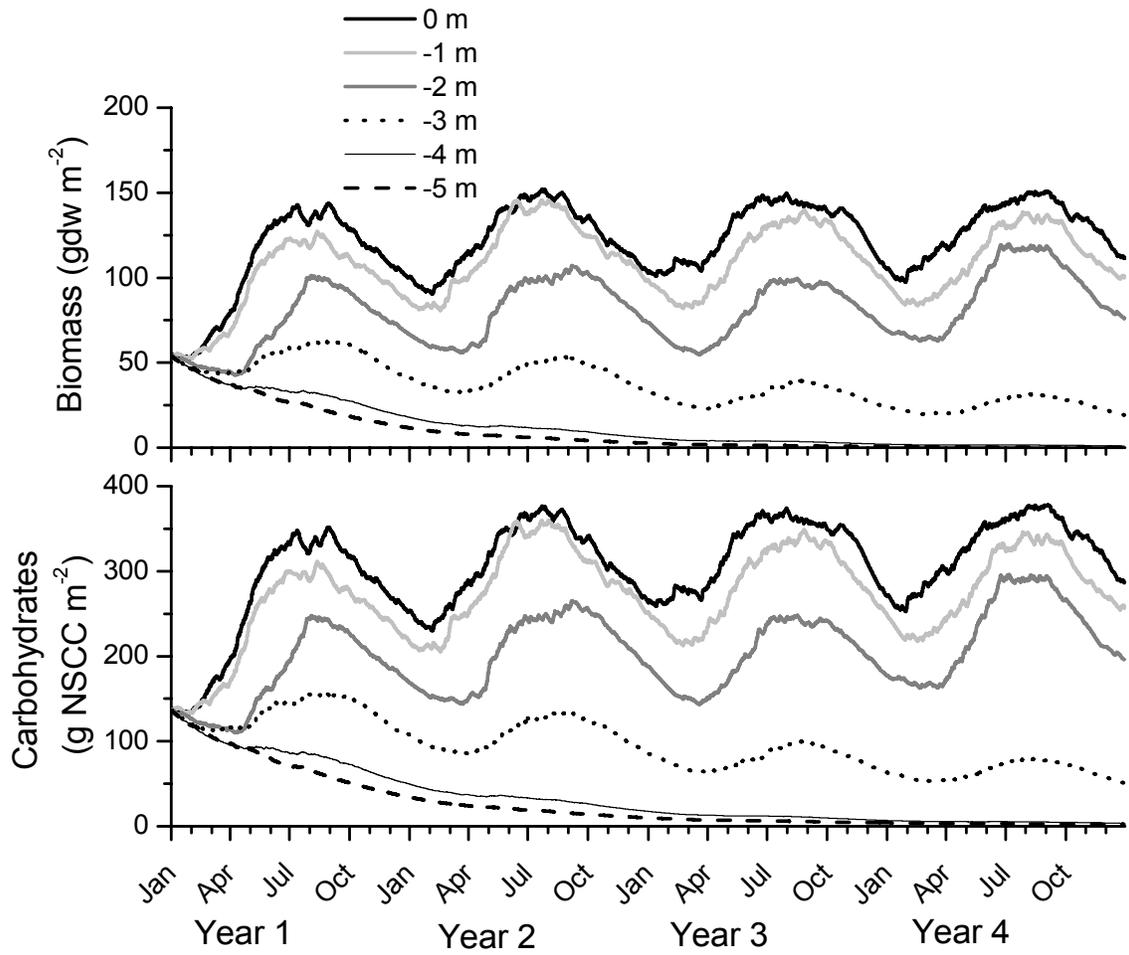


Figure E.3 Case 1, Zone 1; simulations using median light attenuation and DIN values. *Zostera marina* biomass and carbohydrate trajectories for depths from 0 to 2 m below MLLW are stable indicating that median values are protective in Zone 1. At depths greater than 2 m below MLLW, the trajectories indicate that the median values are not protective.

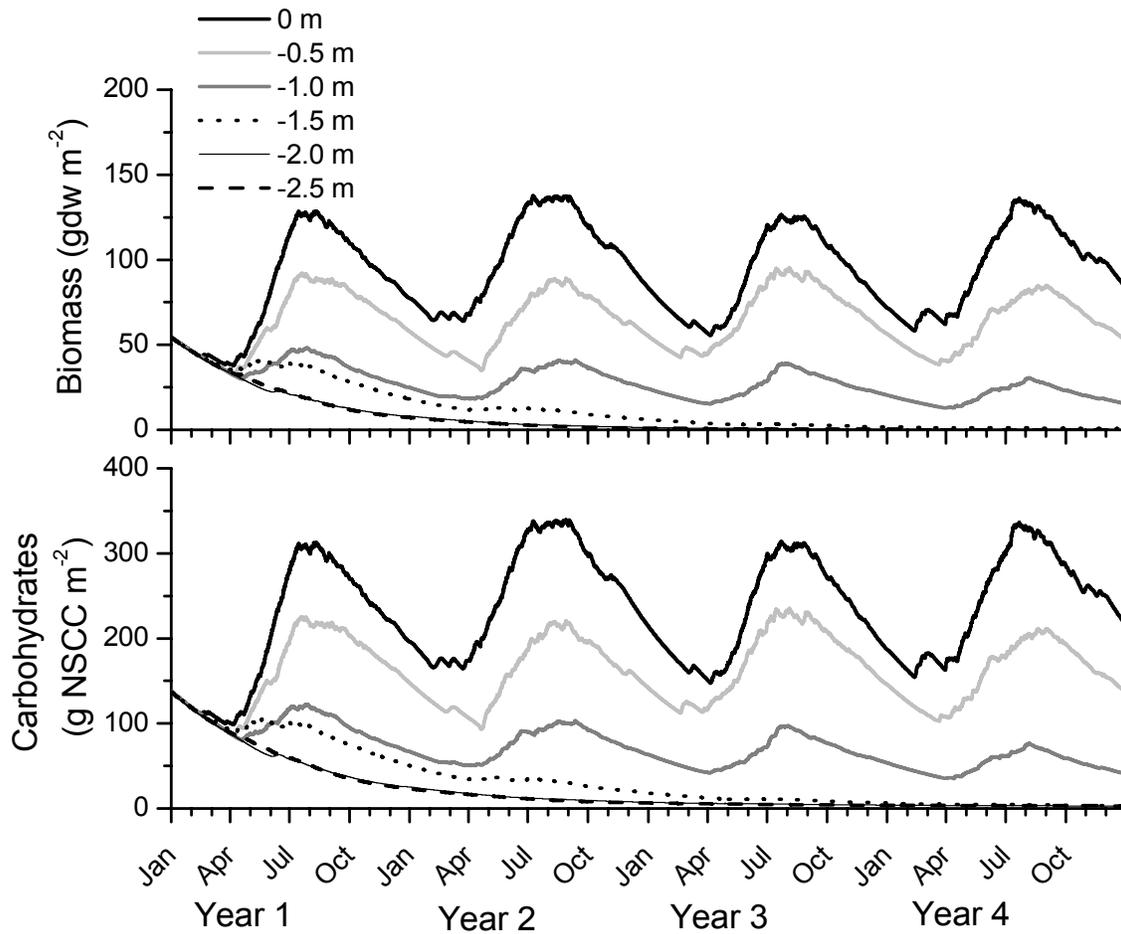


Figure E.4 Case 1, Zone 2; simulations using median light attenuation and DIN values. Stable *Zostera marina* biomass and carbohydrate trajectories indicate that the median values are protective in Zone 2 for depths between 0 and 0.5 m below MLLW. Simulations at the 1 m depth contour appear to be more closely associated with the deeper depths (1.5 to 2.5 m below MLLW) suggesting that this is a break-point that would be susceptible to decline from minor perturbation. At depths below 1 m MLLW, the trajectories indicate that the median values are not protective.

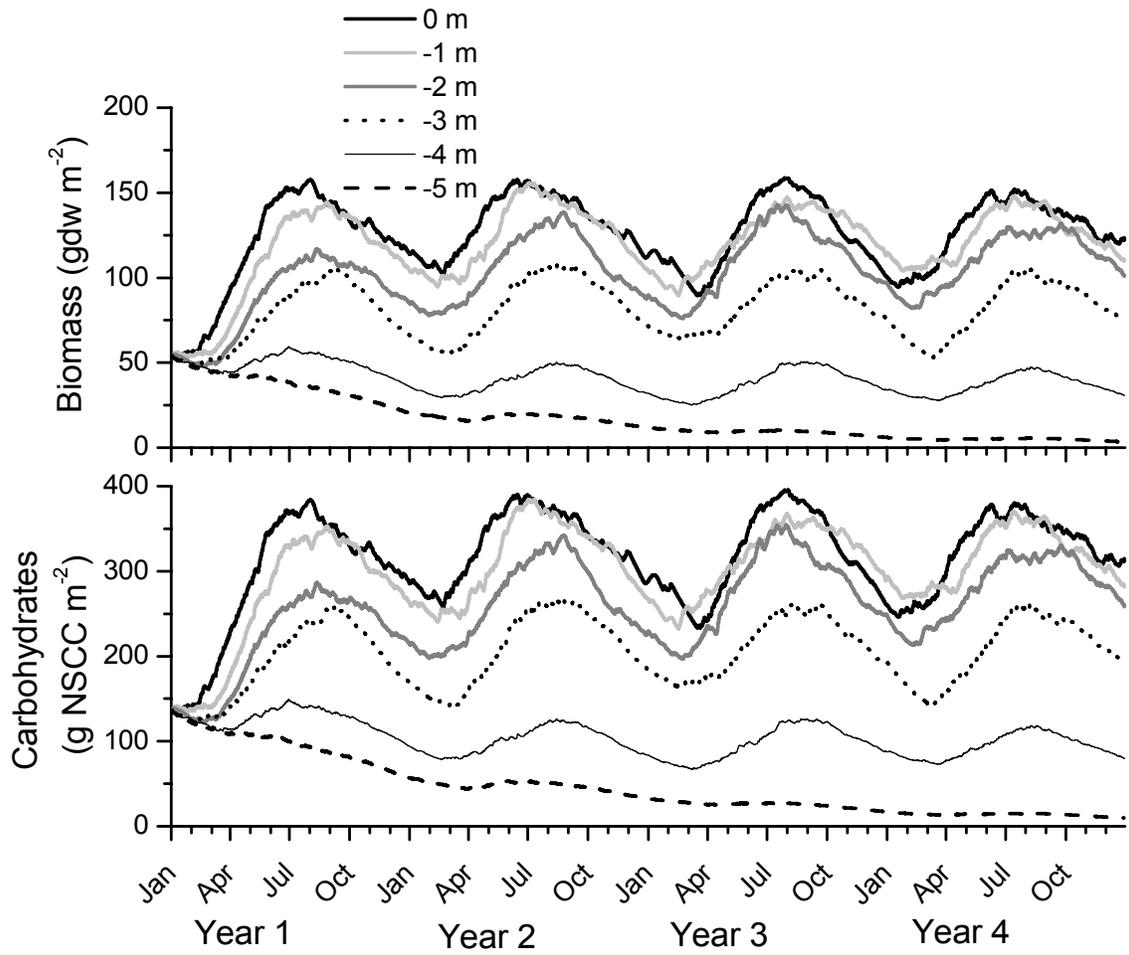


Figure E.5 Case 2, Zone 1; simulations using 25th percentile light attenuation and DIN values. Stable *Zostera marina* biomass and carbohydrate trajectories indicate that the 25th percentile values are protective in Zone 1 for depths between 0 and 3 m below MLLW. Simulations for depths below 3 m (MLLW) exhibited trajectories which indicate that the 25th percentile values were not protective.

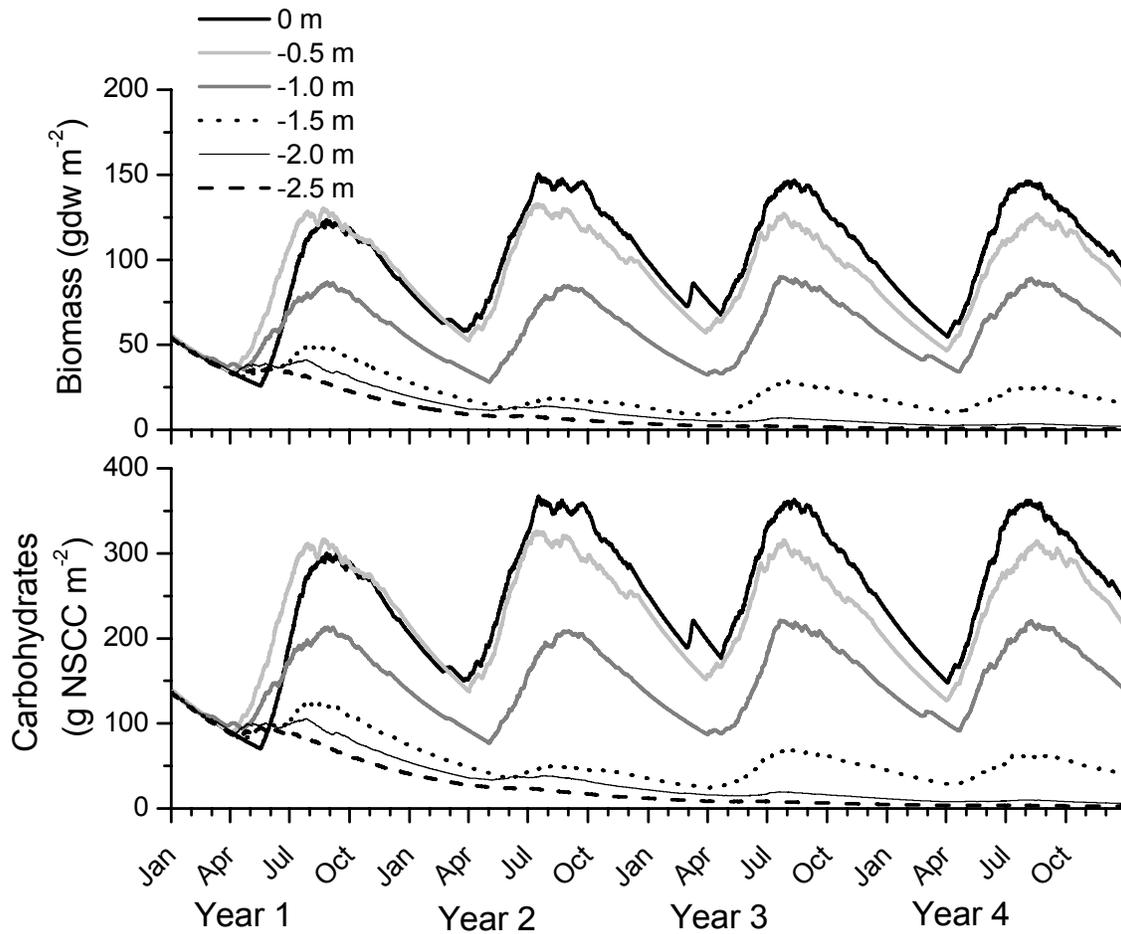


Figure E.6 Case 2, Zone 2; simulations using 25th percentile light attenuation and DIN values. Stable *Zostera marina* biomass and carbohydrate trajectories indicate that the 25th percentile values are protective in Zone 2 for depths between 0 and 1 m below MLLW. Simulations for depths below 1 m (MLLW) exhibited trajectories which indicate that the 25th percentile values were not protective.

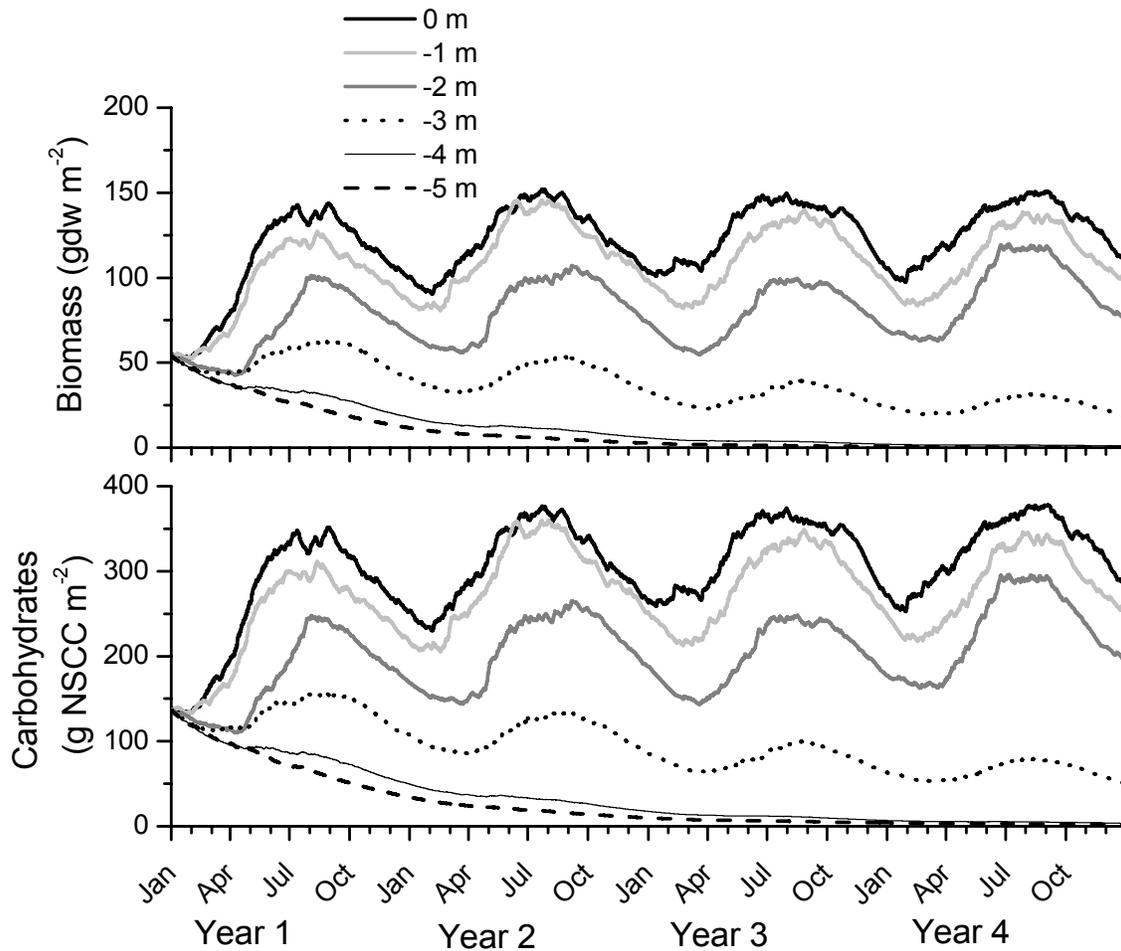


Figure E.7 Case 3, Zone 1; simulations using 75th percentile light attenuation and DIN values. Stable *Zostera marina* biomass and carbohydrate trajectories indicate that the 75th percentile values are protective in Zone 1 for depths between 0 and 2 m below MLLW. Simulations for depths below 2 m (MLLW) exhibited downward trajectories which indicate that the 75th percentile values were not protective.

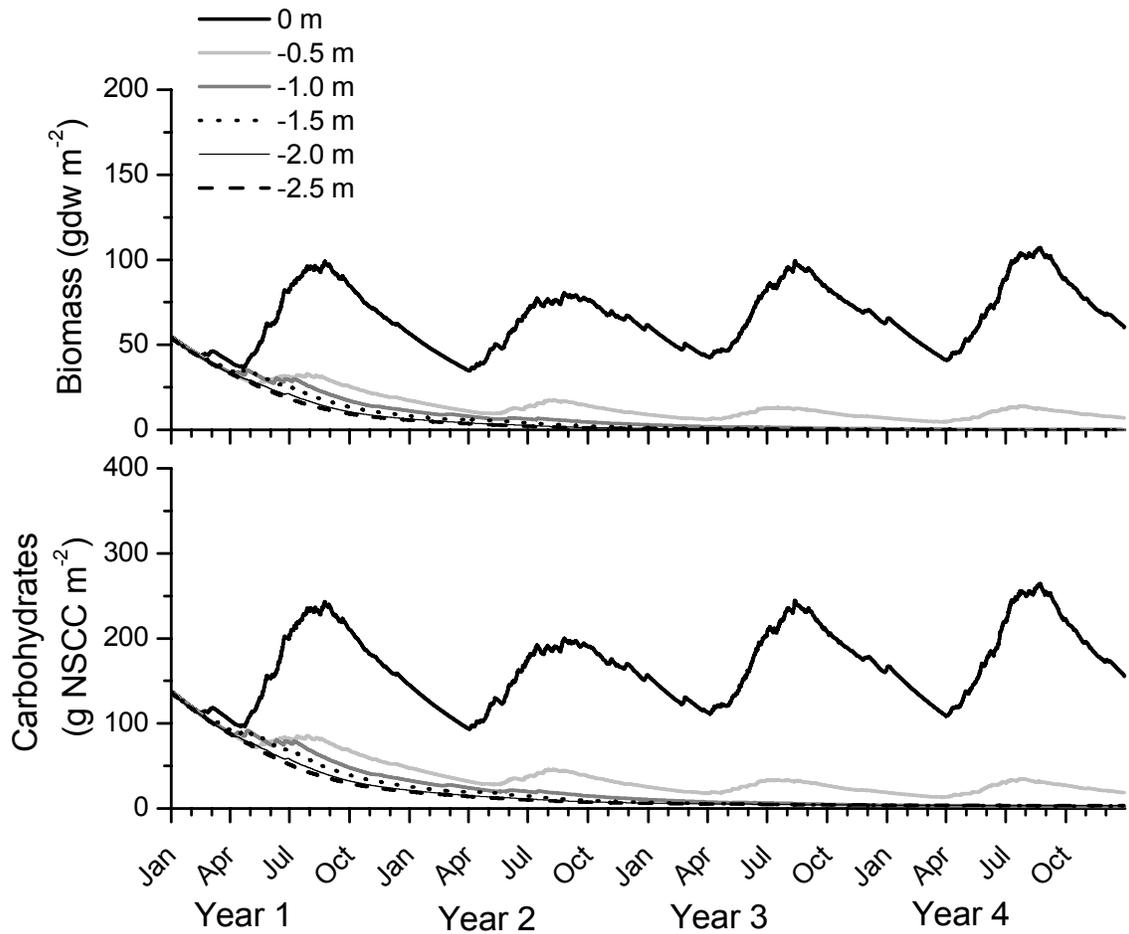


Figure E.8 Case 3, Zone 2; simulations using 75th percentile light attenuation and DIN values. *Zostera marina* biomass and carbohydrate trajectories for depths greater than 0 m MLLW exhibited downward trajectories which indicate that the 75th percentile values were not protective at these depths in Zone 2.