Puget Sound National Estuary Program

State of Knowledge Report for the Marine Water Quality Implementation Strategy

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PUGET SOUND INSTITUTE
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2022.05.23 – Created a combined document that includes all of the review comments. Addressed comments and accepted/rejected changes. Updated tables and charts. Saved as MWQ State of Knowledge 2022.05.23 Revised Draft. Note – there remains review of SSMC tables and figures.

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2024.08.28 - Selected edits based on PSP Science Panel review comments.

2025.08.28 – Reviewed input from public review. The public review did not result in any modifications to this State of Knowledge report.

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EXECUTIVE SUMMARY

This Marine Water Quality State of Knowledge report provides a technical background to support informed decision making and recovery planning focused on the Marine Water Quality Vital Sign. The State of Knowledge Report is meant to:

- Provide a clear description of indicators and targets
- Provide sufficient technical background to allow informed development and review of strategies to meet the chosen targets
- Provide technical background and analysis of strategies
- Identify and scope Critical Analyses

The Vital Sign that is the focus of this document was originally approved by the Puget Sound Leadership Council in 2012 and provides a target related to the level of dissolved oxygen in the marine waters of the Puget Sound. It states:

> By 2020, human-related contributions of nitrogen do not result in more than 0.2 mg/L reductions in dissolved oxygen levels anywhere in Puget Sound

The Vital Sign has since been updated to include existing measures associated with anthropogenic eutrophication (e.g., dissolved oxygen, nutrient balance, and marine benthic index) while being expanded to include stressors associated with global climate change and increasing CO_2 levels in the atmosphere (e.g., marine water temperature, ocean acidification).

Since the information in this State of Knowledge document focuses largely on the impacts and outcomes of nutrient management, it is still relevant to the successful management of the revised Vital Sign.

Sources of Nitrogen to Puget Sound

There are several key points related to nutrient loading to Puget Sound.

- Nitrogen enters into Puget Sound through natural and human-related sources. Overall, natural sources make up approximately three-quarters of the annual loading, and human sources make up the remaining one-quarter of annual loading (Figure ES1 and Section 5).
- Natural nitrogen sources include ocean water, which is relatively high in nitrogen, entering Puget Sound through the Strait of Juan de Fuca. The other major natural source is surface runoff from watersheds that carries eroded soils and organic matter into Puget Sound.
- Anthropogenic nitrogen enters into the Puget Sound via two major pathways. One is through wastewater treatment systems that discharge directly into Puget Sound, or into rivers that drain into Puget Sound. The second is through surface runoff where rainwater picks up nitrogen as it runs across developed lands including urban areas, and rural areas, and particularly agricultural and livestock operations. This surface water runoff eventually enters Puget Sound, carrying the nitrogen load with it.
- Minor sources of nitrogen include atmospheric deposition and groundwater (directly to Puget Sound). Neither of these sources contributes more that 1% of total nitrogen loading to Puget Sound (Mohamedali et al., 2011b).

- Natural and anthropogenic nitrogen loadings vary geographically (between sub-basins), and temporally (across seasons and between years). For example, oceanic loads may vary by ±10% between years (Khangaonkar et al., 2021).

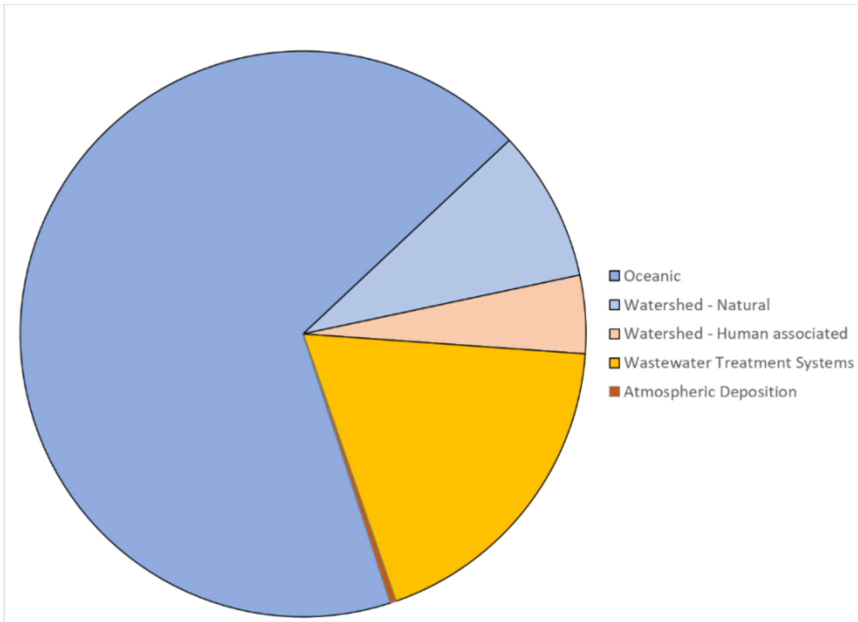


Figure ES1. Estimate of annual nitrogen loadings into Puget Sound from natural (blue) and human-associated (yellow-orange) sources. Loadings from groundwater was estimated to be 0.2 to 2.0% of total nitrogen loads (Mohamedali et al, 2011b) and is not shown. Figure based on data from Mohamedali et al, 2011b.

IMPACTS OF ANTHROPOGENIC NITROGEN TO PUGET SOUND

Changes in Dissolved Oxygen (Section 8)

Reductions in dissolved oxygen (DO) are one of the potential impacts of excess nitrogen loading to Puget Sound. Since there is no information available to understand the state of Puget Sound prior to industrialization, the Salish Sea Model is used to estimate the change in condition. Model evaluation results indicate that human-associated nitrogen inputs do affect the levels of dissolved oxygen. The impacts are mainly anticipated to occur 1) in poorly flushed embayments and terminal inlets, 2) in the bottom waters, and 3) in the late summer/early fall each year. Overall, approximately 15% of the surface area of Puget Sound is predicted to exceed the target of ΔDO<0.2 mg/L once or more over the course of a calendar year (Figure ES2). Since these effects are localized and mainly in terminal inlets and bottom waters, and over a part of any given year, less than 1% of Puget Sound water has a change in DO greater than 0.2 mg/L (measured in by volume-days with accounts for both the amount/volume of water affected and the amount of time the impact lasts). The majority of the predicted change is <0.5 mg/L. (Section 8).

The interannual variability in loading and current patterns (measured by changes in annual residence times; Ahmed et al. (2019)) results in a wide variation in anthropogenic-nutrient related impact in water quality.

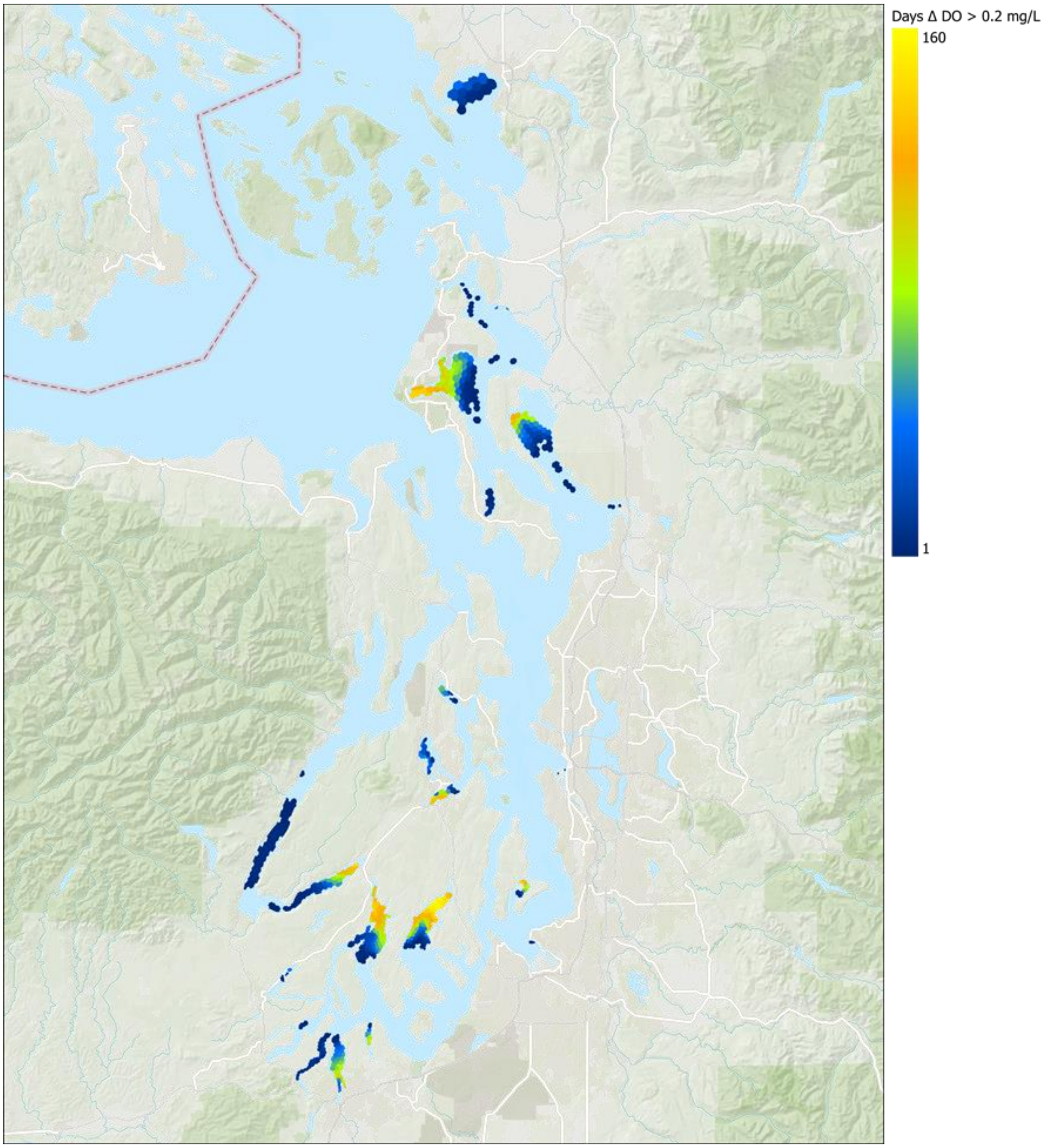


Figure ES2. Spatial extent where ΔDO>0.2mg/L over the course of the model year between Existing and Reference conditions, which is an estimate of current changes of DO associated with human activity. Color scale indicates number of days where there is an exceedance at any depth. In general, only the bottom waters are estimated to be affected.

Other potential impacts (Section 4)

There are other potential impacts related to anthropogenic nitrogen inputs including:

- change in level of primary production (Section 4.1)
- changes in phytoplankton species and abundance (Section 4.3)
- increases in harmful algal blooms (Section 4.4)
- changes in seagrass community structure and abundance (Section 4.5)
- changes in benthic community structure (Section 4.6)
- changes in pelagic species and food webs (Section 4.6)

Some of these changes have been observed in Puget Sound, for example in declines in primary production (PSEMP Marine Waters Workgroup, 2020), localized areas of seagrass losses (Christiaen et al., 2019), and declines in benthic community condition (Ecology Marine Sediment Monitoring Program). Anthropogenic nitrogen inputs have not been strongly linked to any of these changes. in the Salish Sea.

Predicted future changes associated with global climate change and population growth may exacerbate changes in water quality including reductions in DO and lower pH (i.e., ocean acidification) in the Puget Sound (Khangaonkar et al., 2019).

MANAGEMENT ACTIONS TO CONTROL NITROGEN LOADING

Three strategies have been identified to control nitrogen loading to Puget Sound. These include:

- Reducing wastewater nutrient loads
- Reducing urban stormwater and agricultural runoff nutrient loads
- Restoring natural nutrient attenuation processes

The confidence that these strategies will reduce nutrient loads varies. For example, it is clearly technically possible to upgrade the wastewater treatment systems in the Puget Sound watershed so that total effluent nitrogen is ~ 3 mg/L (current effluent N generally ranges from 20-40 mg/L). Reducing urban stormwater and agricultural runoff nutrient loads has been demonstrated (Section 10.1) through the implementation of BMPs, though the results of nutrient reduction even at the sub-basin scale is mixed (e.g., Fisher et al., 2021). Similarly, the potential effectiveness of restoration of natural nutrient attenuation has been demonstrated (Section 11), though the effectiveness even of individual projects and installations vary widely (e.g., Mendes, 2021).

Regarding the outcomes of the nutrient reduction strategies on marine water quality - work with the Salish Sea Model suggests that no single strategy, even if fully implemented would completely eliminate all areas that exceed the indicator target of ΔDO<0.2 mg/L. For example, even with complete WWTP upgrades which would greatly decrease human-associated N inputs, localized areas that exceeded the target would remain (Section 9.1). A combined and extensive nutrient reduction approach (WWTPs effluent at ~ 3 mg/L and a 65% reduction in anthropogenic nitrogen), which would require substantial and sustained investment to achieve, may result in water quality (DO) conditions that meet the MWQ Vital Signs target (Ahmed et al., 2021). An estimated US$ 2.5-5.0 billion would be required for the regional wastewater upgrades alone. (Marine Water Quality Base Program Analysis).

Uncertainties and Critical Analyses

The strategy development process included the identification of key uncertainties, those that make it difficult to implement or evaluate nutrient reduction strategies (Section 13). The key uncertainties were identified in coordination with the Marine Water Quality Interdisciplinary Team. These key uncertainties were used to developed a suite of proposed critical analysis, which provide a brief background and description of the uncertainty, and a proposed approach for addressing the uncertainty. These are intended to provide an initial scoping for follow up research and investigation; some of them are currently being addressed.

Examples include a proposed Critical Analysis on the development of a watershed nutrient model coupled to improve the quantification of nutrient sources and impacts (Section 7.2.4), and an evaluation of the DO thresholds of key Puget Sound species (Section 4.7).

1 INTRODUCTION

This Marine Water Quality State of Knowledge report provides a technical overview of the management of the Marine Water Quality Vital Sign in the Puget Sound. It is produced in connection with two additional documents titled the Implementation Strategy Narrative, which describes the results of a planning process designed to identify strategies, activities, and measures to guide the recovery of Marine Water Quality of Puget Sound; and the Base Program Analysis, which provides a programmatic and regulatory overview of the Marine Water Quality Vital Sign. These documents and the strategic plans described herein focus on reducing impacts to marine water quality associated with anthropogenic nutrients entering into the marine environment from a variety of sources and pathways.

The documents provide the following information:

- The State of Knowledge report (this document)
    - A technical description of elements related to the Marine Water Quality indicators that are necessary to understand to develop relevant strategies;
    - Evaluation of priority strategies, approaches, and activities that have been identified during the Implementation Strategy development process;
    - A review of selected technical uncertainties that are related to the management of dissolved oxygen in the Puget Sound.
- Implementation Strategy narrative
    - Description Marine Water Quality dissolved oxygen indicator;
    - The status and trends of the dissolved oxygen indicator;
    - Description of the strategy development process;
    - Identification of key barriers to achieving the recovery targets;
    - Identification of priority strategies, approaches, and approaches for achieving the recovery target.
- The Base Program Analysis
    - A review of relevant programs and regulations for the management of nutrient inputs to the Puget Sound.

1.1 PROPOSED CRITICAL ANALYSIS

This Marine Water Quality State of Knowledge report includes a suite of proposed "critical analyses," which describe focused studies meant to address part or all of a key uncertainty or question that was identified during the Implementation Strategy development process (see Section 13 for a complete description of the process). Briefly, during strategy development, the Interdisciplinary Team noted questions and uncertainties which were can impede our ability to identify activities, actions, and strategies to achieve Marine Water Quality goals and targets. Uncertainties included, for example, the effectiveness of specific approaches at nutrient management, key relationships in causal chains, the strength and direction of relationships in causal chains, biological responses to management activities, etc.

In response, and in coordination with the InterDisciplinary Team, a set of proposed critical analyses were developed to address the key uncertainties. These are listed in sections below. They are meant to provide a broad description of the questions with proposed ways to address. As there is much ongoing research in the region, the status of the applicable work is also described. The proposed critical analyses do not prescribe a specific agency or research group that should pursue the question – they are simply meant to highlight important research that could serve to improve our ability to manage Marine Water Quality in the region.

1.2 The Marine Water Quality Vital Sign

Marine Water Quality is one of 25 Vital Signs adopted by the Puget Sound Partnership to represent the health of Puget Sound. More information is available on all of the Vital Signs through the Puget Sound Partnership's Vital Signs Website.

Each Vital Sign has associated measurable indicators used to track its status. Most indicators also have quantitative targets, or goals, that are intended to represent a state of healthy recovery (Figure 1).

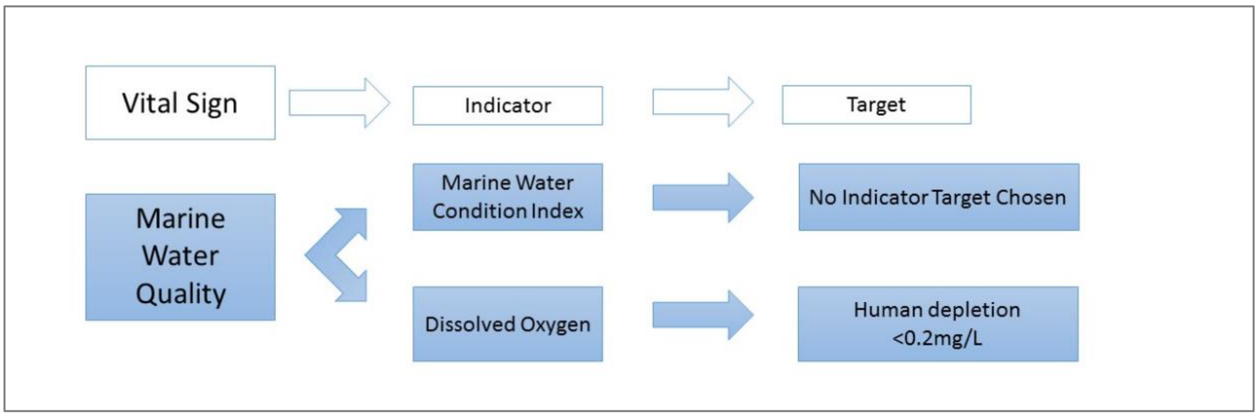


Figure 1. Overview of Marine Water Quality Vital Sign, its indicators, and targets.

1.3 Marine Water Quality Indicators

During development of the Marine Water Quality Implementation Strategy, this Vital Sign had two indicators: the Marine Water Condition Index and dissolved oxygen. Both indicators provide a measure of the impacts of eutrophication on the water quality of Puget. Eutrophication is a process where primary production is increased in a water body due to the addition of nutrients, generally nitrogen or phosphorus. Depending on the magnitude, nutrient additions can result in a variety of negative impacts, including reductions in dissolved oxygen, loss of submerged aquatic vegetation and changes in planktonic community composition (Bricker et al., 2007). So, while the primary focus on the Implementation Strategy was dissolved oxygen, the strategies and management activities that control nutrient inputs can also mitigate the negative effects of eutrophication.

1.3.1 Marine Water Condition Index

The Marine Water Condition Index (MWCI) was a Vital Sign indicator at the time of Implementation Strategy development. However, during a subsequent Vital Sign review, it was decided that it would not be included in future reporting, and would be replaced by individual measures (see Section 1.4). Details on the MWCI are described in Krembs 2012.

1.3.2 Dissolved Oxygen in Marine Waters

The second indicator under the Marine Water Quality Vital Sign is the modeled (i.e., predicted) change of dissolved oxygen in marine waters associated with human impacts.

The dissolved oxygen indicator is determined by comparing current conditions with a "reference condition" that existed prior to industrialization in the watershed; i.e., a condition without anthropogenic water quality impacts. Since there are no actual measurements of the reference condition, it is evaluated based on the modeled (i.e., predicted) levels of dissolved oxygen in a scenario without human-related nitrogen inputs. The Salish Sea Model (see section 6.1) is used for this evaluation.

1.3.2.1 Target

The Leadership Council adopted the following target on June 16, 2011, in Resolution 2011-10:

> "By 2020, human-related contributions of nitrogen do not result in more than 0.2 mg/L reductions in dissolved oxygen levels anywhere in Puget Sound"

1.3.2.2 Status of the Dissolved Oxygen Indicator

The status of the dissolved oxygen indicator is updated regularly, with updates reported on the Puget Sound Partnership Vital Sign web page for this indicator. While there are ongoing refinements to the evaluation, the most current assessment (Puget Sound Partnership, 2019) indicates:

- Human sources of nutrients have a significant impact on dissolved oxygen in Puget Sound in multiple embayments.
- The cumulative impact of all human activities causes dissolved oxygen concentrations to decrease by more than 0.2 mg/L at multiple locations throughout Puget Sound.
- In several areas throughout Puget Sound, human-related oxygen depletion persists for three months or more.

1.4 Vital Sign Indicator Revisions and Updates

The Vital Signs and indicators described above were in place during development of the Marine Water Quality Implementation Strategy. They were included here to provide context for the planning effort that resulted in the recovery strategies described in the Implementation Strategy narrative. In 2019-2020, all of the Puget Sound Vital Signs were reviewed and revised (as necessary) resulting in a somewhat modified suite of Vital Signs and indicators (McManus et al., 2020). The revised Vital Signs and indicators were accepted by the Leadership Council in 2020.

This included revisions to the Marine Water Quality indicators and the identification of three potential future indicators. The revised indicators still provide a measure of potential impacts associated with anthropogenic eutrophication (e.g., dissolved oxygen, nutrient balance, and marine benthic index) but were expanded to include stressors associated with global climate change and increasing CO_2 levels in the atmosphere (e.g., marine water temperature, ocean acidification). The new indicators are:

- Dissolved oxygen in marine waters
  Direct, field-based measurements of dissolved oxygen, at representative spatial and temporal scales for the Puget Sound ecosystem to determine where dissolved oxygen is below two biologically meaningful thresholds. Note that the threshold values were not explicitly defined.
- Marine water temperature
  Direct, field-based measurements of water temperature, at representative spatial and temporal scales for the Puget Sound ecosystem (nearshore to pelagic, annual and seasonal). This will enable an understanding of variation in positive temperature anomalies over time and space and provide insights on the habitat quality for marine organisms.
- Nutrient balance in marine waters
  Direct, field-based measurements of the ratio of silicon and phosphate to nitrogen, at representative spatial and temporal scales for the Puget Sound ecosystem. This will enable an understanding of whether Puget Sound has a nutrient balance that supports lipid-rich diatoms all year round, creating the base of the Puget Sound food web.
- Primary Production in Marine Water
  This indicator is not yet finalized. A phytoplankton and primary production indicator is under development (Jenkins et al., 2023) and expected to be finalized in 2026.
- Ocean acidification
  The ocean acidification indicator is not yet finalized. The intent is to develop a compound indicator based on (1) water carbonate chemistry measured by Omega-saturation, where: $\frac{[Ca^{2+}]\, x\, [{CO_3}^{2-}]}{[CaCO_3]} = \Omega$, and (2) ability for biological calcification in Puget Sound.
- Sediment Benthic Index
  The Sediment Benthic Index (SBI) is currently being used as a comparative measure of condition of benthic invertebrates in Puget Sound. The SBI is based on species counts, which are used to: 1) calculate nine different indices (abundance; taxa richness; evenness; species dominance; and annelid, arthropod, mollusk, echinoderm, and miscellaneous taxa abundance), 2) determine the presence, absence, and abundance of sensitive and tolerant species, 3) evaluate community structure. That information is then used to determine whether a site is "adversely affected," or "unaffected."
  The SBI will eventually be replaced by a Marine Benthic Index (Schoolmaster and Partridge, 2024).
- Sediment Chemistry Index
  The Sediment Chemistry Index (SCI) is based on a comparison of measured concentrations of 32 chemicals (metals, PAHs, phthalates, and total PCBs) with their Washington State Sediment Quality Standards (Dutch et al., 2018). The SCI is calculated by: 1) determining the mean ratio of measured concentration vs SQS of all detected chemicals (mSQSq), and 2) calculating the index by:
  $SCI = 100\, x\, \left[1 - \left(\frac{mSQSq}{1.5}\right)\right]$.
  The SCI is then used to categorize the potential chemical exposures.
- Noise in Marine Water
  This indicator is not yet finalized.

2 MARINE CIRCULATION

2.1 CIRCULATION OVERVIEW

There are several key characteristics of marine circulation in the Salish Sea. The freshwater discharged by the major rivers and streams creates a stratified, two-layer circulation system. The upper layer, which ranges from approximately 0-30 m in depth depending on location within the system, consists of the seaward flow of fresh/brackish water. The depth of the surface layer, and strength of stratification, varies over the course of the year. The deep, lower layer consists of the landward flow of saline water in from the Strait of Juan de Fuca (Ebbesmeyer and Barnes, 1980; Khangaonkar et al., 2017; Khangaonkar et al., 2011; Sutherland et al., 2011).

Another major feature affecting circulation throughout the Puget Sound, and within specific sub-basins, is the existence of predominant sills, particularly those located at the Admiralty Inlet and the Tacoma Narrows. The area between these sills defines the Puget Sound Main Basin, with the Tacoma Narrows sill separating the Main Basin from the South Sound. The sills promote vertical mixing and result in surface layer reflux, where a fraction of the outflowing surface layer flows downward, combines with the bottom layer, and returns landward. At the Admiralty sill, approximately 60% of the surface flow is re-entrained into the bottom water and returns back into the Main Basin (Ebbesmeyer and Barnes, 1980; Khangaonkar et al., 2017).

Planar views of the circulation patterns in the surface layer and the bottom layer are shown in

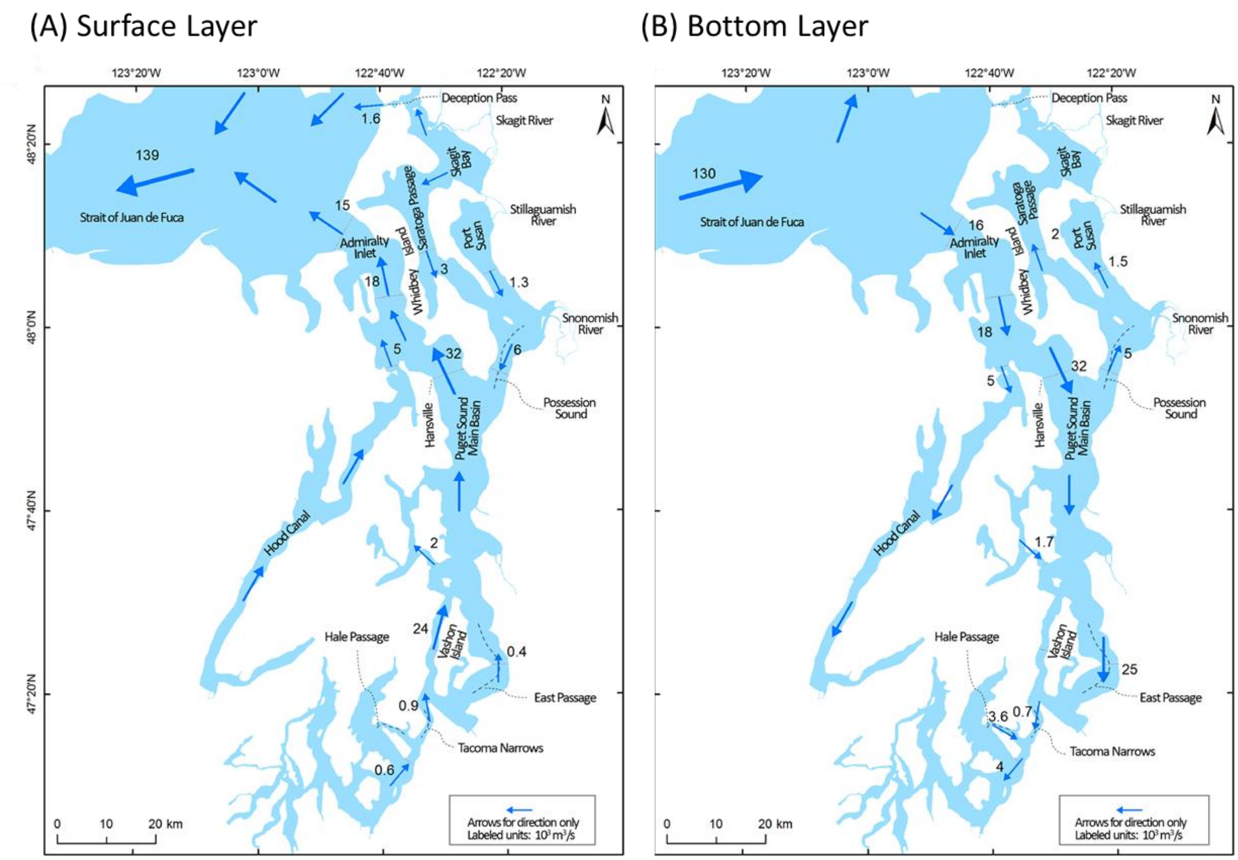


Figure 2 Figure 2A and 2B, respectively.

The depth profile of flows along a transect from the Pacific Ocean to South Puget Sound is shown in Figure 3.

A video providing an overview of flows and current patterns has been published by the Encyclopedia of Puget Sound utilizing a physical model that was constructed by John H. Lincoln and Clifford Barnes of the UW School of Oceanography. The Puget Sound Model was built in the early 1950s to better illustrate and study circulation. An illustrated description of circulation can be accessed on the video titled, “The Puget Sound Model: Tides and Currents.” It is available on YouTube.

https://www.youtube.com/watch?v=OEQP5BwVh7A

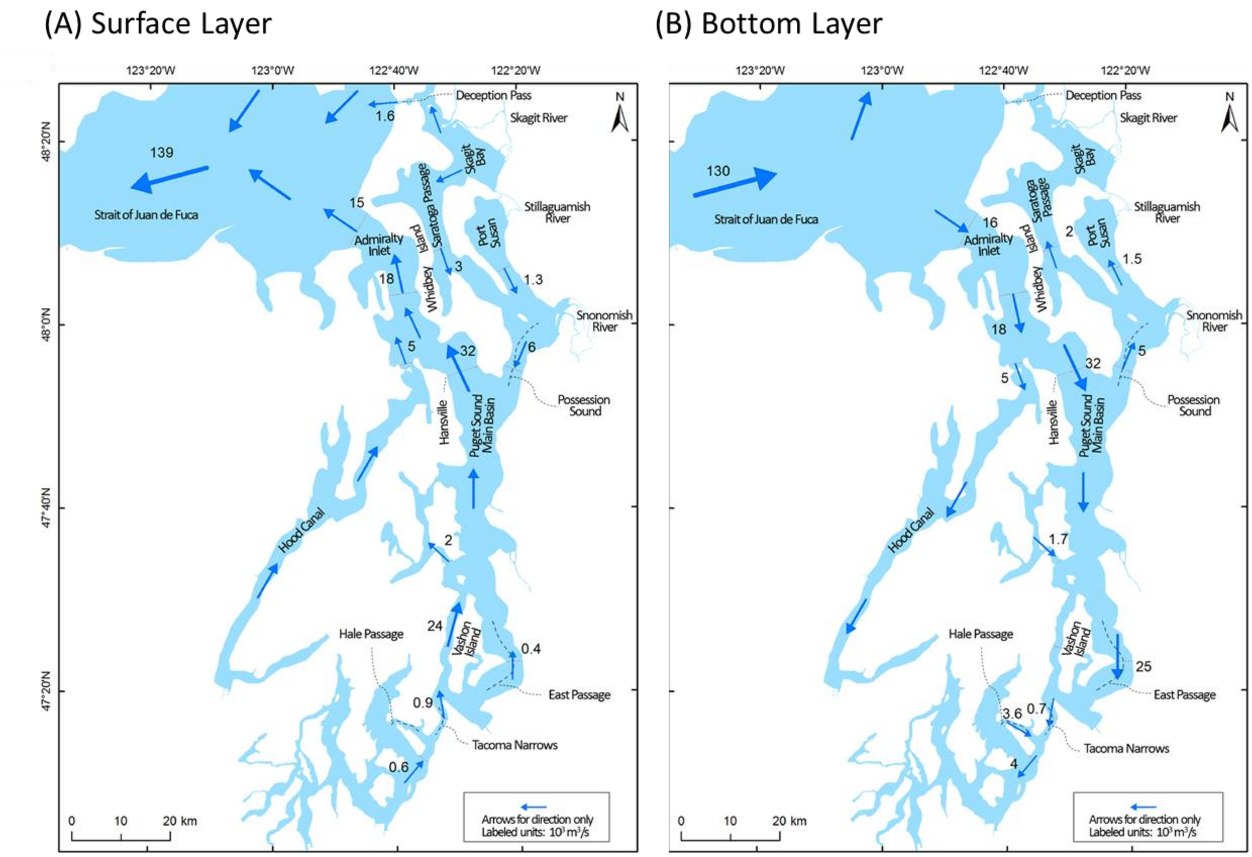


Figure 2. Tidally average flows in the surface layer (A) and the bottom layer (B) of Puget Sound region. Adapted from Khangaonkar et al. (2017)

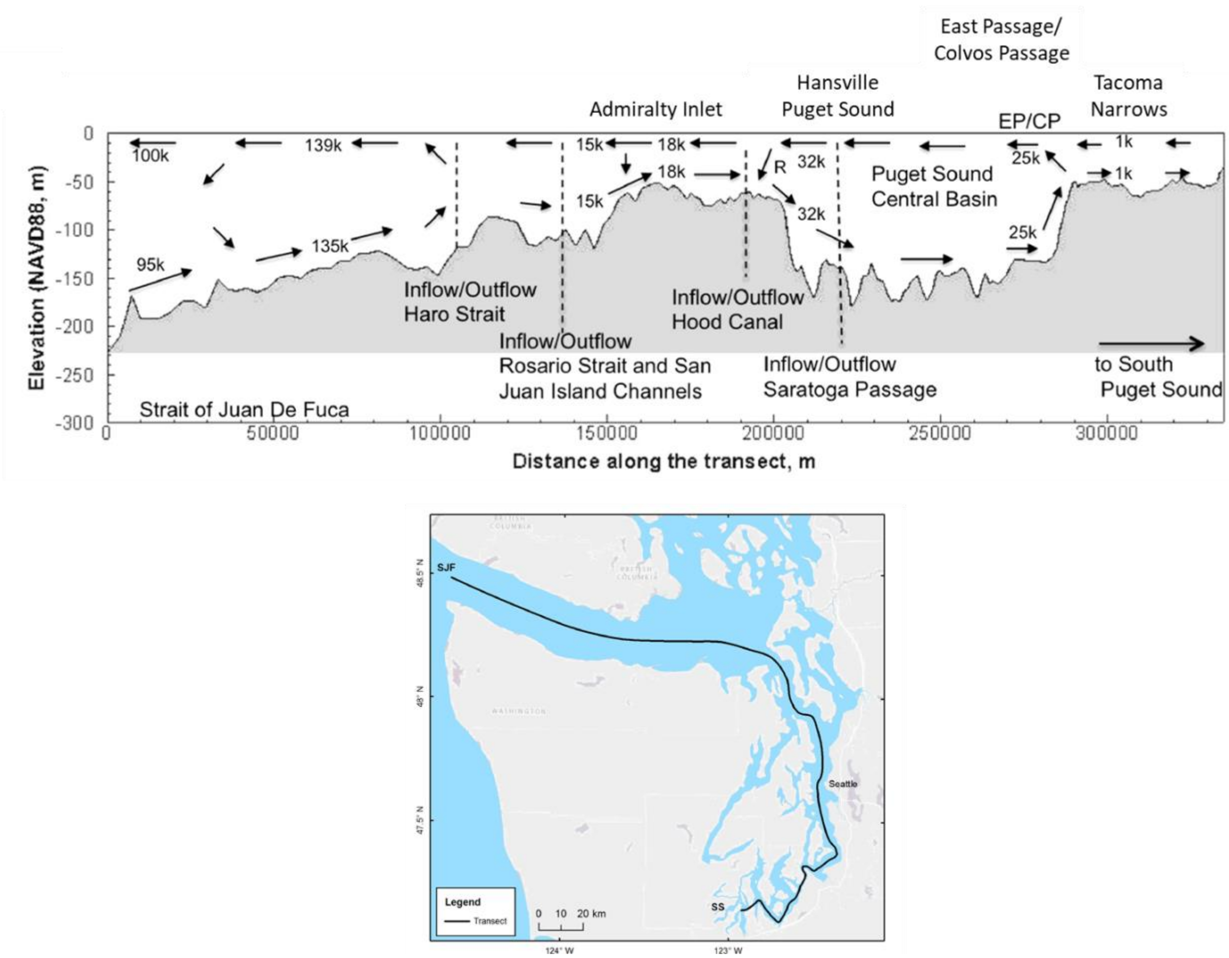


Figure 3. Representation of tidally averaged flows and exchange between the Pacific Ocean to the South Sound of Puget Sound. Arrows indicate direction only. Magnitude of flows are indicated by the numbers (10^3 m^3/s). From Khangaonkar et al. (2017)

2.2 Assimilative Capacity and Residence Times

The assimilative capacity of a water body is the amount of a contaminant that could be put into that same water body without resulting in a deleterious change of condition, or a biological impact (Krom, 1986). This concept acknowledges the fact that minor inputs, and in this particular case minor inputs of anthropogenic nutrients, may have no observable effects on a marine ecosystem. This notion is also acknowledged in the Marine Water Quality indicator target of $\Delta DO<0.2$ mg/L (where Δ = change) as it states that anthropogenic nutrient inputs from sources such as wastewater treatment systems and surface water runoff are acceptable so long as there are no predicted (or observed) changes in dissolved oxygen in Puget Sound greater than 0.2 mg/L. Therefore, the target sets the acceptable limit on inputs.

The assimilative capacity of anthropogenic nutrients in Puget Sound is not static and depends largely on the rate of flushing in the different basins and sub-basins. A system with a high rate of flushing can assimilate a larger nutrient load compared to the same system with a low rate of flushing. Flushing varies seasonally and annually. One way to measure the rate or magnitude of flushing, and thus the assimilative capacity, is with residence time.

The amount of time a parcel of water takes to leave a defined area or region is called the residence time (Monsen et al., 2002). The residence times of water in the Puget Sound varies markedly by basin and sub-basin, with additional variation due to season, weather, winds, etc. But an averaged measure of residence time in a basin is a useful in understanding potential impacts of nutrient addition on water quality. The residence time of a basin matters because it can influence water quality, and in particular the dissolved oxygen levels in the estuarine system.

The relationship between residence time and water quality can be informed by reviewing a few important processes. First, and as described above, marine water enters Puget Sound from the Pacific Ocean, through the bottom layer of Strait of Juan de Fuca, and into the greater Puget Sound at Admiralty Inlet. The quality of the bottom water, including the levels of dissolved oxygen, that enters the system generally reflects that of the Pacific Ocean at the entrance to the Strait of Juan de Fuca. Note that the levels of dissolved oxygen in the marine water at the mouth of the Strait of Juan de Fuca varies; status and trends are presented in Section 2.4.

Second, due to stratification between the surface layer and the bottom layer, mixing and oxygen transport between the two layers is limited. This means that atmospheric oxygen from the surface, or oxygen produced through primary production in the surface layer, will not necessarily diffuse downward from the surface layer into the bottom layer. Furthermore, the bottom waters may have seasonally different circulation and longer residence time than surface waters. Note that the magnitude of phytoplankton blooms varies according to nutrient availability; this is discussed more in Section 3.

Third, organic particles (phytoplankton, leaf detritus, etc.) can sink downward from the surfaces layer into the bottom layer. These organic particles are degraded through aerobic microbial processes which utilize oxygen. This biological oxygen demand is met by the dissolved oxygen in the bottom water resulting in oxygen depletion. The magnitude of the depletion depends on the flux of organic carbon entering the bottom waters and the volume of water entering the system. The volume of water can be measured by the residence time in a basin. The relationship of these three factors is summarized in Figure 4.

More water entering system → faster transport → shorter residence times
More water entering system → more oxygen → higher assimilative capacity → lower numeric depletion
Less water entering system → slower transport → longer residence times
Less water entering system → less oxygen → lower assimilative capacity → higher numeric depletion

Figure 4. Conceptual model illustrating relationships between water influx, residence times, and assimilative capacity

2.3 Residence Times – Variation

2.3.1 Spatial Variation

There is substantial variation between the residence times of the basins of the Puget Sound. Babson et al. (2006) utilized a box model to estimate the range of residence times and reported mean modeled residence times for the bottom water ranging from 37.8 days through the Main Basin to 87.7 days for the Hood Canal (modeled as northern Hood Canal and southern Hood Canal). The residence time is dependent on local and regional bathymetry and depth (i.e., sub-basin location relative to sills) and localized freshwater inputs, which suggests that basins and sub-basins will have a wide range of assimilative capacities. More recently, MacCready et al. (2021) utilized the LiveOcean model to study circulation and mixing in the Salish Sea, specifically quantifying efflux (water mixing upward into surface layer and then heading seaward) and reflux (water mixing downward into bottom layer and then heading landward). They reported that while the efflux and reflux markedly increased the average residence time of each of the basins, the relative difference in residence times between the basins was consistent with what was reported elsewhere. A summary of results is shown in Figure 5.

2.3.2 Temporal Variation

The exchange flow into Puget Sound exhibits a strong annual cycle, with the annual flow maximum generally occurring in winter, though this varies each year (MacCready et al., 2021). The exchange flow may also be influenced by the salinity of water upwelled at the Pacific coast, which would affect the density gradient at the Strait of Juan de Fuca. Babson et al. (2006) reported that the variability in salinity, driven by tidal exchange, may account for more seasonal variability than river flows for most basins except for South Sound. Additional factors that may affect flows and residence times are the seasonal changes in wind patterns (Sutherland et al., 2011), tides (Deppe et al., 2018) and temperature that, along with salinity, influence density and can lead to density gradients that affect stratification and mixing regimes. Predicted seasonal residence times for Puget Sound and major sub-basins are shown in Figure 5.

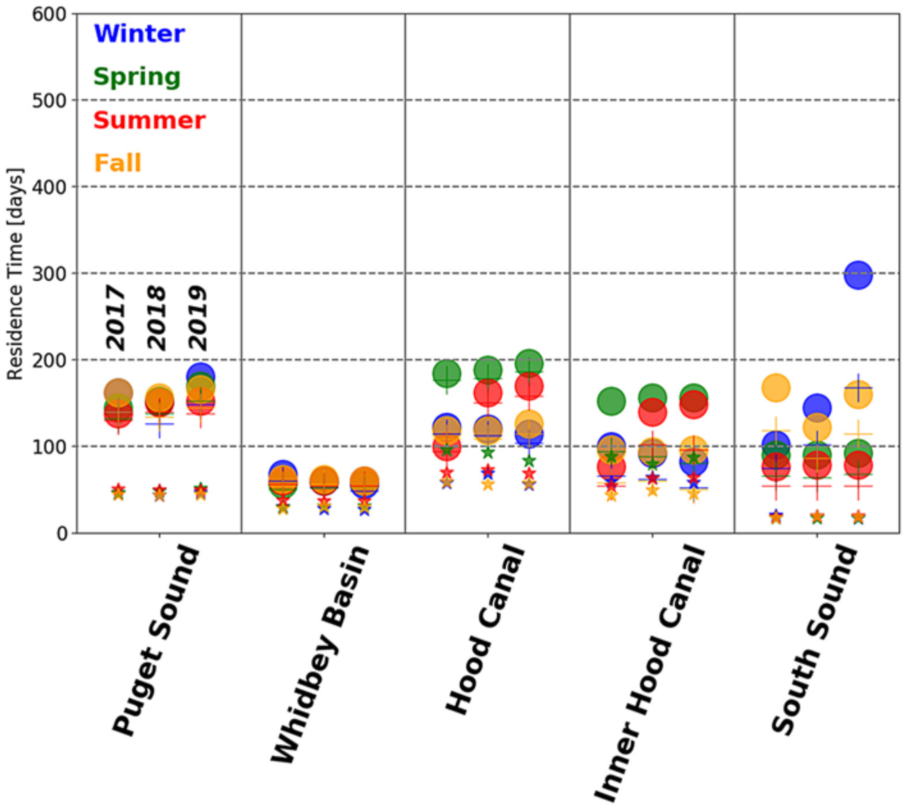


Figure 5. Modeled residence times for Puget Sound and major basins for model years 2017-2019 showing both seasonal and annual variations. Adapted from MacCready et al (2021).

2.4 Coastal Ocean and Puget Sound Boundary Conditions

Water from the Pacific coastal shelf enters into the Puget Sound at the Strait of Juan de Fuca via Admiralty Inlet, and so the condition of the water at the ocean boundary can greatly influence water quality in Puget Sound. The Northwest Association of Networked Ocean Observing Systems (NANOOS) and the University of Washington (UW) maintain a large surface mooring (Ćháʔba) and an adjacent subsurface profiling mooring (NEMO-subsurface) to collect and record oceanographic and meteorological measurements on the Northwest Washington shelf. This system provides information on the boundary condition for Puget Sound. As reported annually in the PSEMP Marine Waters Overview reports, there are important inter- and inner-annual changes in the levels of DO in the marine waters, that could affect condition within the embayments of Puget Sound. This monitoring over time provides a measure of the variation in DO between calendar years (see the grey lines in Figure 7) and in a specific year (red and black lines in Figure 7). As shown, in 2018 the levels of DO in the deep water at Ćháʔba gradually decreased from May through mid-August, and then transitioned into a series of episodic intervals of low dissolved oxygen or hypoxia, which is commonly defined as DO<2mg/L (Diaz and Rosenberg, 1995). The DO during these periods was occasionally below 0.25 mg/L. These episodic events are associated with northward-flowing bottom water, which is related to increased winds to the north. Changes in these wind pattern led to changes in current and DO conditions at the moorings.

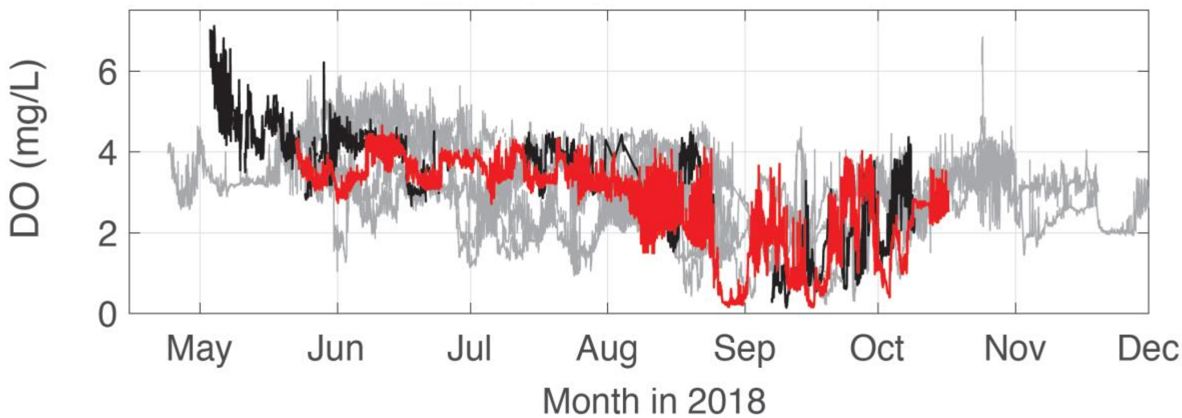


Figure 6.Dissolved oxygen (DO) measurements at Ćháʔba at 85 m depth.
Measurements for 2018 (red), 2017 (black) and 2011-2016 (gray) are shown, and describe annual trends and variation. Note wind-related episodic hypoxia events from mid-August through mid-October. Chart from Szuts et al. in 2018 PSEMP Marine Water Report.

3 NITROGEN AND CARBON CYCLING IN THE MARINE SYSTEM

3.1 NITROGEN CYCLING IN MARINE SYSTEMS

Nitrogen can occur in different forms in marine systems and can be biologically transformed depending on redox conditions, which are generally defined by the level of dissolved oxygen in the water. Predominant transformations include: fixation of nitrogen gas (N_2) to particulate organic nitrogen (PON); mineralization of PON to ammonium (NH_4^+); nitrification, the oxidation of NH_4^+ to nitrate (NO_3^-) via nitrite (NO_2^-), and; denitrification, the anaerobic reduction of NO_3^- to N_2 via nitric oxide (NO) and nitrous oxide (N_2O). See Figure 7.

The primary form of N in the marine system is Dissolved Inorganic Nitrogen (DIN) which includes nitrate, nitrite, and ammonium.

Generally speaking, nitrification occurs in the water column where there is sufficient oxygen to support these processes. Dentification occurs in or near the sediment surface where there is little dissolved oxygen available for respiration. Mineralization can occur by aerobic or anaerobic processes. Mineralization releases NH_4^+ either into the water column or pore water of sediments where it is bioavailable. Mineralization in the sediments may result in a gradient with high NH_4^+ in the sediments leading to diffusive transport to the water column.

Primary losses of N from the marine system (not including advective transport out) are via denitrification and anaerobic ammonium oxidation (anammox), where inorganic N is converted to N_2.

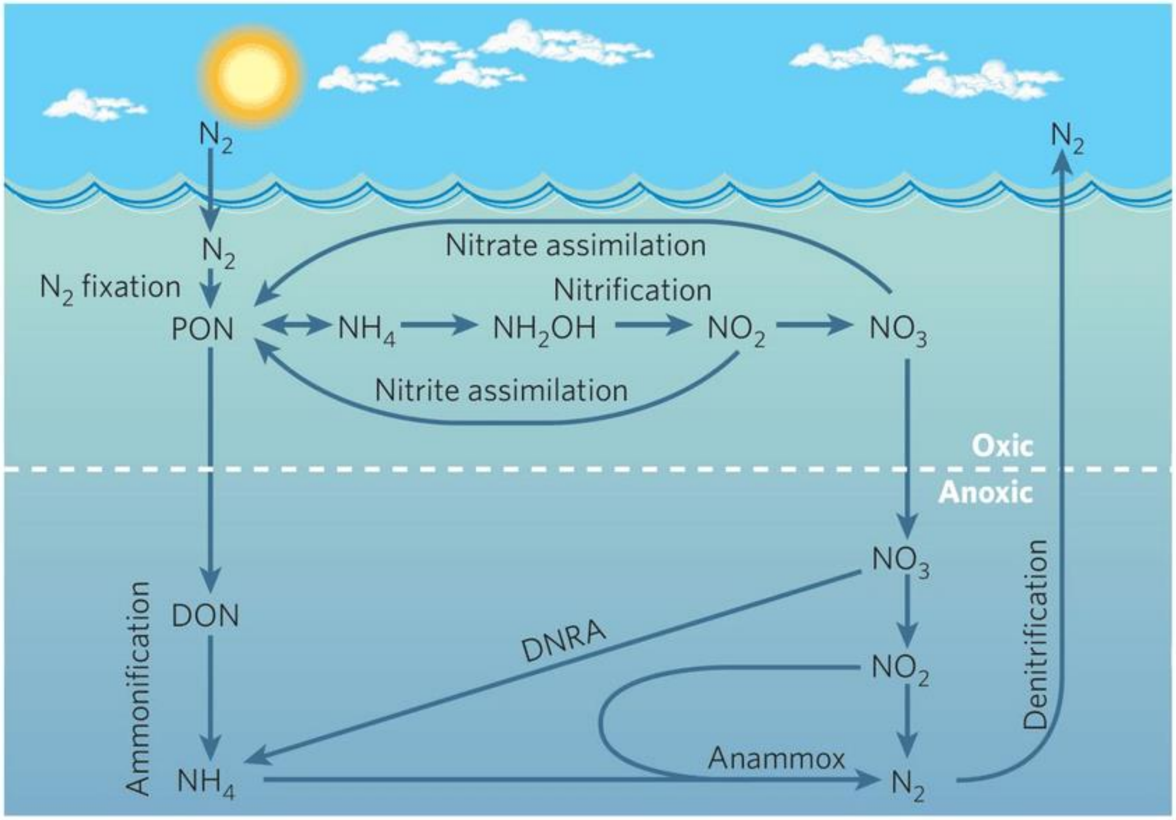


Figure 7. Marine Nitrogen Cycle.
Biological uptake of N is shown as assimilation. The oxic/anoxic threshold generally defines the redox conditions in which the shown processes are likely to occur; this threshold generally exists near to or within the marine sediments. Particulate Organic Nitrogen (PON) can physically settle from the water column to the sediments. where it can undergo mineralization. DON – dissolved organic nitrogen, DNRA - dissimilatory nitrate reduction to ammonium, "ammonification" in this context, is equivalent to mineralization.

3.2 PRIMARY PRODUCTION IS BASED ON NUTRIENT AVAILABILITY.

The process of the assimilation and conversion of inorganic carbon (carbon dioxide) and other inorganic nutrients into organic matter with the help of light by autotrophs such as phytoplankton is called primary production. A balance of elements such as carbon, nitrogen, and phosphorus is required to support the growth of phytoplankton (and, thus, to support primary production). The relative amounts of each of these has been determined based on the observed stoichiometric ratio of carbon, nitrogen, and phosphorus in marine phytoplankton and marine systems, which is approximately 106:16:1 for C:N:P (Redfield, 1934). That is, for every 106 atoms of carbon used, 16 atoms of N, and 1 atom of P are required to form biological material and support growth. And while the N:P ratio in algae and cyanobacteria varies according to localized conditions, species types, and nutrient availability (Geider and La Roche, 2002), the "Redfield Ratio" of 106:16:1 provides a benchmark to understand broad nutrient requirements for the phytoplankton growth that support marine food webs, including those of the Salish Sea. Large variations from that ratio can indicate nutrient limitations, which can limit primary production.

3.3 Primary production in the Salish Sea can be nitrogen limited

Primary production in the Salish Sea occurs in the photic zone, which is the top part of the water column that receives sufficient light to support photosynthetic activity. In the winter months, the solar radiation is low and primary production is limited by light availability. In the spring and summer, the duration and intensity of solar radiation is sufficient to support large-scale primary production to the extent that most of the available N in the photic zone is used up. In such cases, phytoplankton growth is N limited. Adding more N would then result in more phytoplankton growth and potentially eutrophication.

Newton et al. (1998) demonstrated that experimental nitrogen additions at sites in Budd Inlet resulted in increased primary production throughout the year, though there were much larger increases in the summer compared to the winter. Similarly, Newton and Van Voorhis (2002) observed a substantial increase in primary productivity in the photic zone in response to nutrient enhancements at stations in the Central Basin and Possession Sound, particularly in the summer months. The degree of response in primary productivity to nitrogen additions likely varies from year-to-year due to changes in offshore upwelling, timing and magnitude of freshwater inputs, etc.

3.4 Nitrogen transport into the photic zone can be limited by stratification

The Salish Sea is a stratified, two-layer system with a distinct upper layer ranging from 0-30 m in depth (Khangaonkar et al., 2017; Khangaonkar et al., 2011). The existence of the strong stratification can limit the vertical transport (via advection and diffusion) of N across and between the shallow layer and the deep layer. In periods of high productivity the N in the upper layer can become depleted and limiting (Khangaonkar et al., 2018). See Figure 8.

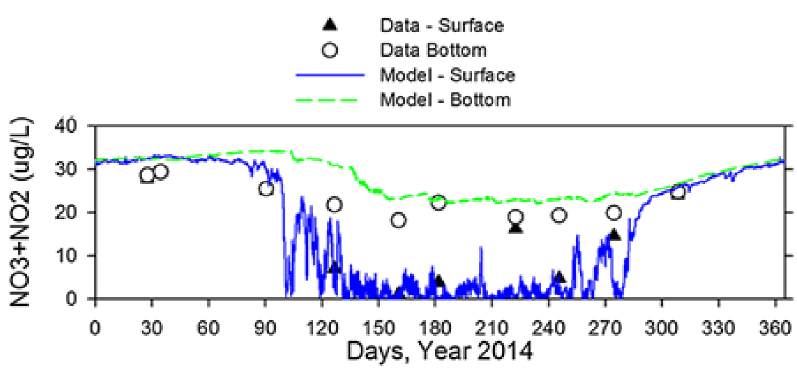


Figure 8. Modelled and measured $NO_3 + NO_2$ concentration in surface layer and bottom layer of a representative location of East Passage in Puget Sound Central Basin.

The x-axis is the Julian day of the model year, 2014. Adapted from Khangaonkar et al. (2018). Note that N in the surface layer (blue line) becomes depleted during the summer months, while the N concentration in the bottom layer is only minimally affected.

3.5 PARTICULAR ORGANIC CARBON (POC) CAN SETTLE TO SEDIMENTS AFFECTING OXYGEN DEMAND.

Particulate organic carbon (POC) can enter the Salish Sea through rivers or streams, through point source inputs, or from primary production in the photic zone. POC can physically settle out of the water column to the sediments where it contributes to oxygen demand in the bottom water and sediments, resulting in a decrease in dissolved oxygen.

4 IMPACTS OF EXCESS NUTRIENTS TO SALISH SEA – BACKGROUND AND STATUS AND TRENDS

Anthropogenic nutrient inputs, generally in excess of background levels, can change the condition of a water body. Eutrophication is the process by which a body of water becomes enriched with nutrients and/or minerals. This section describes a range of impacts from excess nutrient inputs into Salish Sea.

4.1 PRIMARY PRODUCTION

Primary production is the process of the assimilation and conversion of inorganic carbon (carbon dioxide) and other inorganic nutrients into organic matter with the help of light by autotrophs such as phytoplankton.

4.1.1 Background – Primary Production

As described above, primary production in the Puget Sound is often limited by the availability of nitrogen (Newton and Van Voorhis, 2002; Newton et al., 1998) following algal blooms which can deplete the upper water of nitrate (Eash-Loucks et al., 2016). So, in general, increases in nitrogen inputs can lead to an increase in primary productivity, mainly during the summer months.

Primary production occurs in the euphotic zone, which is the upper layer of the water column where sunlight penetrates enough to support photosynthesis. This is one of the key reasons why stratification of the water column is a critical factor for modulating primary productivity regionally. There are well defined seasonal variations in primary productivity in Puget Sound largely due to changes in incident radiation throughout the year. Variation in stratification, and spatial variation due to the shading of sediments introduced by rivers may also be important factors. There is also considerable interannual variation in primary productivity, potentially linked to changes in external forcings (Newton and Van Voorhis, 2002). As such, nitrogen inputs from different sources at different locations and timings may have different impacts on primary productivity. Harrison et al. (1983) indicated that eutrophication risks in the main basin of Puget Sound to be relatively low, but suggested that the more-poorly-flushed and often stratified bays and inlets (e.g., those with relatively long residence times) may be more susceptible to effects of localized eutrophication.

4.1.2 Status and Trends – Primary Production

The Washington State Department of Ecology performs long term monitoring at 27 stations throughout the Puget Sound. Monthly water samples are collected at each station at target depths of 0m, 10m, and 30m, as well as full depth casts which reach depths near to the bottom sediments. Long-term trends for multiple parameters are calculated and reported annually (Krembs, 2012). One parameter is the frequency of intense phytoplankton blooms, where an intense phytoplankton bloom will result on a chlorophyll A concentration of > 30 μg/L. As shown in

Figure 9, data suggests that the frequency of these intense blooms has been declining since 1999 (PSEMP Marine Waters Workgroup, 2020). Note that these changes are proxies of primary production and not actual measures of primary production. Phytoplankton biomass is often estimated by chlorophyll a concentration, which reflects primary production adding to biomass. Other loss terms such as the rates of grazing and/or sinking, and rate of advective transport (e.g., amount of phytoplankton transported out of a system through currents) may also affect chlorophyll

A concentrations. Direct measurements of primary production are scarce for Puget Sound, a gap that was identified by the Washington Academy of Sciences (Washington Academy of Sciences, 2012).

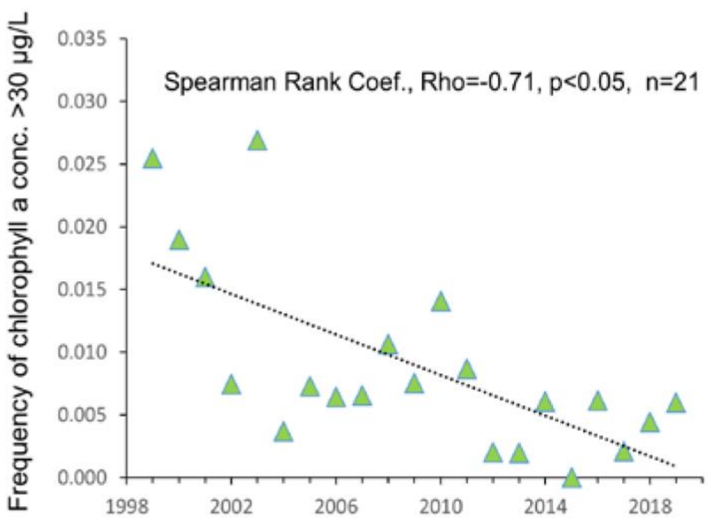


Figure 9. Phytoplankton bloom frequency of high chlorophyll events (>30 µg/L) measured from 0-30m depth. Figure from section 5.A.iii of 2019 Marine Waters Report (PSEMP Marine Waters Workgroup, 2020)

Johannessen et al. (2021) investigated the trends in primary production by examining the isotopes of organic carbon and nitrogen measured in 21 sediments cores collected in the Strait of Georgia and the Puget Sound to estimate the fluxes marine-derived and terrigenous organic matter. They reported that there were no observable trends in the flux of marine-derived organic matter for the last 100 years. This suggests, assuming that loss terms remained constant, that the overall, long-term rate of primary production has been relatively consistent over that time period.

In an earlier study, Brandenberger et al. (2011) evaluated paleo-ecological indicators in sediment cores collected in the Hood Canal and Puget Sound Main Basin. Their results suggested that there was a decrease in the proportion of marine organic matter coincident with an increase in terrestrial organic matter which, when taken into consideration with the other indicators, suggests a decrease in primary production. The authors indicated that the driver and the magnitude of this change in primary productivity was not yet clearly known.

Van Alstyne (2016) measured stable isotopes of nitrogen and oxygen in water and green macroalga (*Ulva lactuca*) in Penn Cove, a small enclosed embayment in the Whidbey basin of Puget Sound to help understand sources of nitrogen that support algae growth. The stable isotope results indicated that upwelled waters were the predominant sources of nitrogen, but that anthropogenic sources may also have contributed to algae growth.

4.2 Depletion of Dissolved Oxygen

4.2.1 Background – Depletion of Dissolved Oxygen

A primary effect of increased nutrient additions into a water body is an increase in phytoplankton biomass and consequent depletion of dissolved oxygen. As excess organic material is formed in response to nutrient additions, a significant fraction of the phytoplankton dies and settle into the deeper waters where the biomass is degraded through anaerobic microbial processes, processes which utilize the dissolved oxygen in the water column to break down organic carbon. The phytoplankton-related particulate matter can also settle rapidly on the bottom of the water body creating an ongoing sediment oxygen demand.

During oxygen depletion, oxygen demand outweighs oxygen resupply by advection from more oxygenated regions. The extent of depletion is highly dependent on the physical properties such as currents and turbulent mixing in a water body. For example, a well flushed system where high-DO water is continually entering a system may demonstrate only minimal impacts, while a more-poorly flushed system may be particularly susceptible to DO depletion. Due to the range of circulation patterns, the Puget Sound has a wide range of depletion potential (i.e., assimilation capacity) within localized bays and inlets. This can be seen by the wide range of background[1] levels of DO (see Section 8).

4.2.2 Status and Trends – Depletion of Dissolved Oxygen

Trends in dissolved oxygen are reported based on Washington State Department of Ecology's long-term monitoring program. The dissolved oxygen data are integrated from 0-50m and then compared to station-specific, monthly baseline values determined from 1999-2008 (Krembs, 2012). Differences from the 1999-2008 baseline are reported as anomalies.

The DO deficit is calculated based on a comparison of integrated measured DO undersaturation for depths >20m. The measure represents how much oxygen would be needed to achieve full saturation of a waterbody adjusted for water temperature, salinity, and pressure. Measurements are collected at a subset of 14 deeper stations. The mean monthly anomalies are shown in Figure 10,Figure 11 reflecting a pattern of higher-than-normal DO deficits since 2013. Water temperature has also been higher-than-normal during this period which could be a contributing factor since warmer water holds less oxygen. Complete data are presented in the 2019 Marine Waters report (PSEMP Marine Waters Workgroup, 2020).

[1] Background or "Reference" conditions can be defined as those that exist throughout the Salish Sea before anthropogenic impacts on water quality. Since data are not available to characterize pre-anthropogenic nutrient inputs or water quality, the Washington State Department of Ecology have explored Reference conditions through applications of the Salish Sea Model.

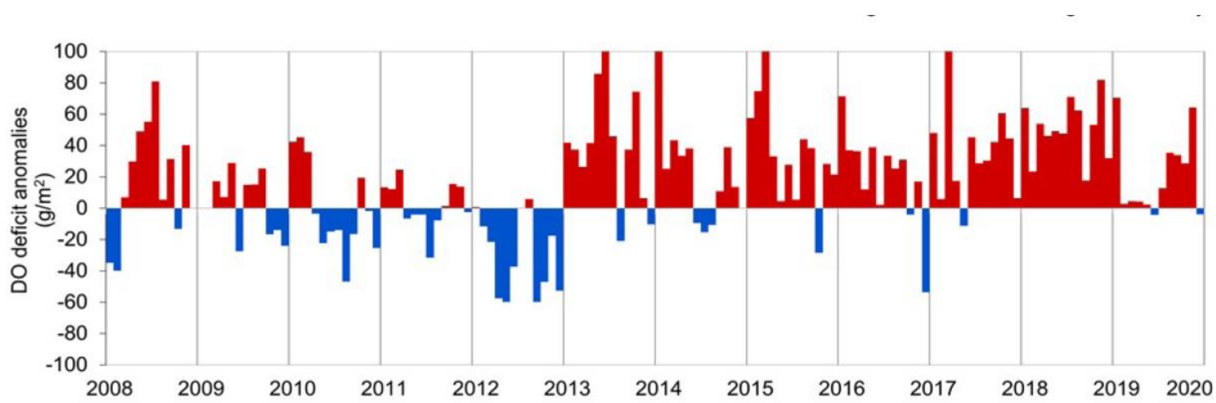


Figure 10. Mean monthly anomalies of dissolved oxygen deficit relative to 1999-2008 baseline. Data were collected at 14 stations Puget Sound wide and anomalies are determined on a month-specific, and site-specific basis. Positive values (red) describe an increased deficit (e.g., generally lower DO) compared to baseline. Negative values (blue) describe a decreased deficit (e.g., generally higher DO). Figure adopted from Section 5.A.ii of the 2019 Marine Waters Overview (PSEMP Marine Waters Workgroup, 2020).

4.3 Alterations in Phytoplankton Community Structure and Abundance

4.3.1 Background – Phytoplankton Community Structure

Phytoplankton are the base of the marine food web, and their abundance and diversity influence not only water quality but also food web structure. Just as not all zooplankton are equally valuable in supporting higher trophic levels like fish, not all phytoplankton species are equally available or valuable to zooplankton grazers. Therefore, phytoplankton community composition can have major consequences for the rest of the marine food web. Diatoms and dinoflagellates are two of the most abundant groups of microplankton in the Salish Sea; diatoms are the preferred species due to their ability to store energy in forms of lipids. The relative temporal and spatial abundances of these two groups are controlled by the availability of nitrogen, phosphorus, and silicate (Egge and Aksnes, 1992; Ittekkot et al., 2000), which indirectly has implications for the quality of food (lipids) for cold water species. Increased growth of silicate-utilizing diatoms owing to excess nutrient inputs can lead to temporal reduction in available silicate in the water column as the biologically-fixed silicate is removed via sedimentation (it is taken up and fixed by diatoms and removed as the die and sink). Other nonsilicifying species, such as dinoflagellates, can then outcompete diatoms following the spring bloom when silicate in limiting.

While seasonality naturally alters species composition, seasonal species pattern can be modified by human nutrient inputs, and alterations of the hydrological cycle of rivers.

Noctiluca scintillans is a common dinoflagellate responsible for large reddish blooms across Puget Sound at various times of the year. *Noctiluca* is a large consumer of diatoms, and may outcompete zooplankton grazers that would provide food for salmonids, and other intermediary species.

4.3.2 Status and Trends – Phytoplankton Community Structure

The silicate and dissolved inorganic nitrogen (DIN) data from the Washington State Department of Ecology long term monitoring program may provide useful background information on conditions that affect phytoplankton community structure. As described above, the Washington State Department of Ecology performs long term monitoring at 27 stations throughout the Puget Sound on a monthly basis. The resulting data are integrated from 0-50m and then compared to station-specific, monthly baseline values determined from 1999-2008 (Krembs, 2012). Differences from the 1999-2008 baseline are reported as anomalies. The median calculated anomalies for the silicate:DIN ratio, and silicate only, for all 27 stations are shown in Figure 11. Complete data are presented in section 5.A.iii of the 2019 Marine Waters report (PSEMP Marine Waters Workgroup, 2020).

The summary data suggests that there is a trend for decreasing silicate:DIN ratio as well as decreasing silicate concentration over time. The silicate:DIN ratio may reflect increases in anthropogenic nitrogen inputs or decreases in silicate concentration associated with decreased inputs. The causes of the changes are not yet known.

It is not yet clear if the changes in nutrient concentrations have altered phytoplankton community structure. Data from the 2019 Marine Waters Report (PSEMP Marine Waters Workgroup, 2020) suggest that there may be an increase in *Noctiluca scintillans* abundances since 2016 (Figure 12).

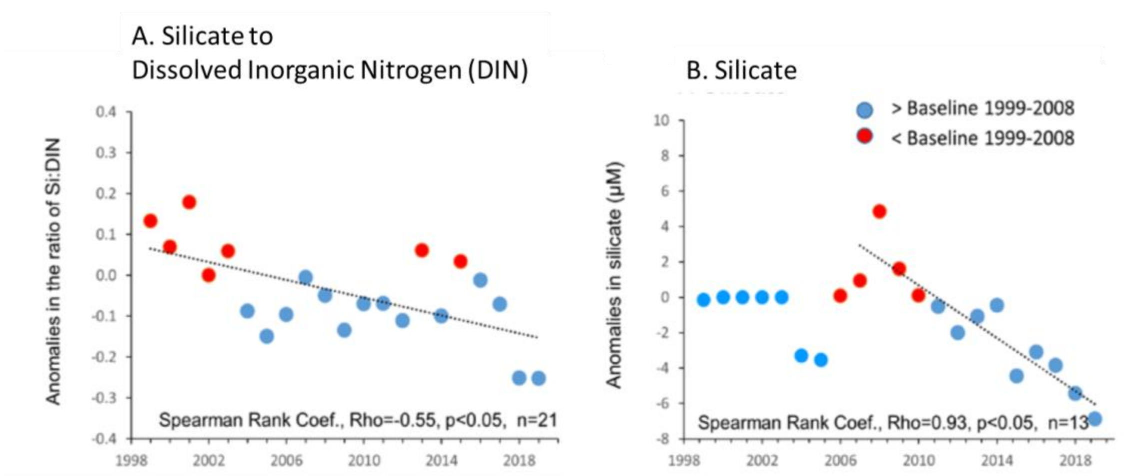


Figure 11. Changes in (A) silicate to dissolved inorganic nitrogen ratio and (B) silicate concentration in Puget Sound waters relative to a 1999-2008 baseline.

Points show median anomalies from all 27 monitoring stations. Red points are above the baseline. Blue points are below the baseline. Figures adapted from PSEMP Marine Waters Workgroup (2020)

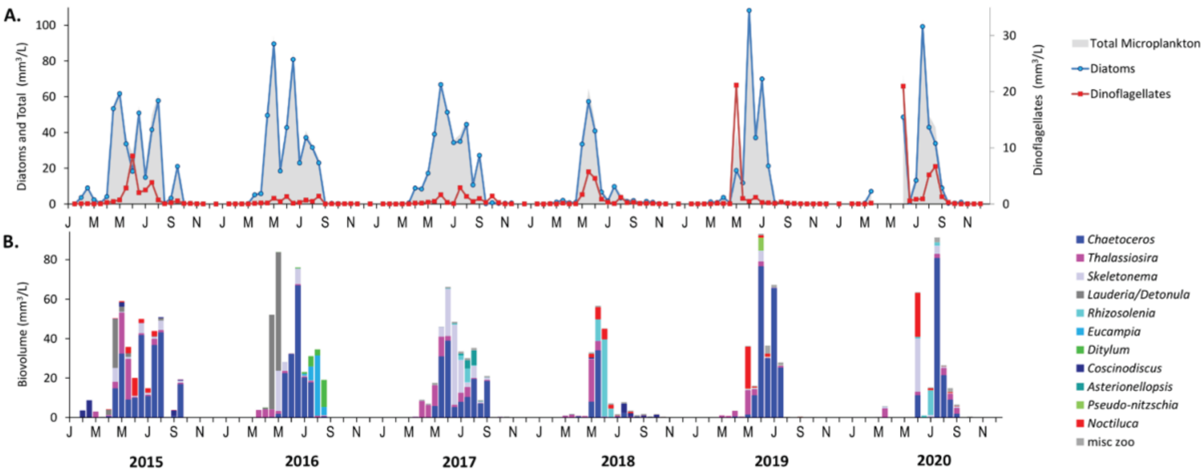


Figure 12. Total microplankton biovolume (gray area), biovolume of main phytoplankton groups, and (B) biovolumes of each year's top six taxa identified using FlowCAM between 2015 and 2020.
Plotted values are means for six Central Basin main-stem sites. Figure from PSEMP Marine Waters Workgroup (2021).

4.4 Harmful Algae Blooms

4.4.1 Background – Harmful Algae Blooms

Harmful algal blooms (HABs) result from the rapid growth of algae that can produce toxins, which can accumulate in shellfish or affect other marine organisms. Different organisms produce different biotoxins including saxitoxin (associated with paralytic shellfish poisoning; PSP), domoic acid (associated with amnesic shellfish poisoning; ASP), okadaic acid (associated with diarrhetic shellfish poisoning; DSP). These and other previously unreported HABs species, such as *Dinophysis*, and toxins such as azaspiracids have been detected recently in Puget Sound.

HABs can occur when HABs species outcompete other organisms in the estuarine ecosystem resulting in rapid growth. There are many factors that are thought to be favorable for HABs blooms such as warm surface water and air temperature, low streamflow, weak winds, and small tidal variability(Moore et al., 2009). Nutrient over-enrichment may promote toxic HAB occurrences in marine ecosystems (Paerl and Otten, 2013), though this is still is an active topic of research (Wells et al., 2020). Heisler et al. (2008) presented a set of consensus statements including that, "degraded water quality from increased nutrient pollution promotes the development and persistence of many HABs species." Additionally, nutrient forms and their proportions are important, and many HABs have physiological mechanisms that enable them to thrive in waters where nutrient loads are not in proportion to Redfield ratios (i.e., differ greatly from "natural" conditions due to anthropogenic influence; Glibert and Burkholder (2011)). Considering the complexity of the interactions that may influence HABs occurrences, a priority is to delineate the ecological space where nutrient changes make it more likely that HABs species dominate, or become a major component of the phytoplankton community (Wells et al., 2020). Research in the Salish Sea is ongoing.

4.4.2 Status and Trends – Harmful Algal Blooms

The SoundToxins program manages phytoplankton monitoring from 28 sampling stations throughout Puget Sound and reported the occurrence of *Pseudo-nitzschia*, *Alexandrium*, and *Dinophysis* blooms (all HABs species) over the course of 2020 (PSEMP Marine Waters Workgroup, 2021). Some of the occurrences exceeded the action levels which triggered warnings to WDOH.

Anderson et al. (2021) quantitatively evaluated the trends in marine HABs in the United States and reported that, nationwide, there were no trends (paralytic shellfish toxin [PSTs], neurotoxic shellfish toxin [NSTs], and diarrhetic shellfish toxins [DSTs]) or increasing trends (amnesic shellfish toxin [AST], Ciguatoxin [CTXs], *Margalefidinium*, brown tides, cyanotoxins) in frequency and/or geographic extent.

Moore et al. (2009) performed a trend analysis on the occurrence of PST in Puget Sound shellfish and did not find clear evidence that the frequency, magnitude, duration, or geographic scope increased between 1993 and 2007. They did identify several predictive factors for bloom occurrence and used that information to develop a predictive model to evaluate changes in windows of opportunity (i.e., those periods where bloom occurrence is much more likely; Moore et al. (2011)). There results suggested that, the window for opportunity for the HABs species *Alexandrium catenella* will increase by 13 days by the end of the 21st century due to the effects of climate change. In addition, toxic blooms are projected to occur two months earlier and one month later compare to current conditions.

4.5 Alterations in Macroalgae and Sea Grass Community Structure and Abundance

Cultural eutrophication (i.e., eutrophication associated with anthropogenic activities and inputs) has been implicated as a major factor in seagrass loss worldwide. Eelgrass abundance in Puget Sound has been reported to be relatively stable over the last 40 years, though a range of localized and persistent changes have been observed at smaller scales (Shelton et al., 2017). Localized declines have been documented, with meadows at the end of inlets or in protected embayments appearing particularly vulnerable (Christiaen et al., 2019). Anthropogenic nitrogen is frequently identified as a stressor to eelgrass in Puget Sound, and controlling inputs is a component of regional recovery and restoration strategies (Calloway et al., 2020; Gaeckle, 2016; Rehr et al., 2014; Short, 2014; Thom et al., 2011; Thom et al., 2014; Washington Department of Natural Resources, 2015). However, other stressors associated with human activities have also contributed to observed declines and there remains considerable uncertainty about mechanisms, scale of effects, and interactions among these stressors (Thom et al., 2011).

The most common causative mechanism linking eutrophication and loss of seagrasses is nutrient enrichment stimulating growth of phytoplankton and/or epiphytes, which reduces light availability in seagrass habitats, affecting growth and survival (Burkholder et al., 2007). Typical symptoms of excess nitrogen, including heavy epiphyte loads, entanglement in nuisance seaweeds (see below), loss at the deep meadow edge, and green water of limited clarity, have been observed at some Department of Natural Resources Puget Sound marine vegetation monitoring sites identified as declining.

Ammonia toxicity may also be a factor. vanKatwijk et al. (1997) reported that eelgrass in mesocosms exposed to ammonia at concentrations of 25 μM (450 μg/L) exhibited signs of toxicity such as a reduction in plant size and plant numbers. Factors such as light availability (Villazan et al., 2013), shoot density and pH (van der Heide et al., 2008) may also affect susceptibility to elevated ammonia concentrations. Most wastewater treatment facilities discharging into marine waters of Puget Sound discharge measurable level of ammonia in the effluent. Additionally, eutrophication can increase the ammonia flux from the sediments to the water column (Heggie et al., 1999), indicating that there a potentially direct and indirect sources of ammonia into the water column associated with anthropogenic nutrient inputs.

Due to the potential deleterious effects of ammonia on aquatic organisms, some monitoring has occurred. The highest observed ammonium level in Elliot Bay of Puget Sound was 110 μg/L (Olsen et al., 2018) suggesting that ammonia toxicity may not be a regional concern; potential for localized effects are not known.

4.5.1 Tissue nitrogen as an indicator of nutrient pollution

Measures of nutrient content in seagrass tissue, such as the ratio of carbon to nitrogen (C:N) and the relative abundance of isotopes ^{15}N and ^{14}N ($\delta^{15}N$), can be used as indicators of nutrient enrichment.

WDNR assessed the concentrations of nitrogen, carbon, metals, and contaminants in eelgrass at 15 sites throughout Puget Sound and reported a low C:N ratio, suggesting high nutrient availability (Gaeckle, 2016). Additional research is needed to determine if/how this might affect eelgrass populations throughout Puget Sound.

To detect early warning signs of eutrophication before widespread loss of eelgrass occurs, McIver et al. (2019) recommends monitoring both eelgrass tissue (tissue N, $\delta^{15}N$, and $\delta^{13}C$) and annual algae cover (epiphytic, benthic, and sediment chl-*a* as a proxy for microphytobenthos). Their analysis indicates that none of these indicators alone is sufficient to track overall risk of nutrient pollution impacts to eelgrass habitat in regions, like Puget Sound, with low or moderate susceptibility to widespread eutrophication.

4.6 Species and Food Webs

As described above, increasing nutrient inputs and eutrophication has the potential to alter the primary producer community structure and change the contributions of different sources of organic carbon to fish and invertebrates. These changes have the potential to impact local and regional food webs (Cui et al., 2021). For example, the results of a modeling exercise indicated that that nutrient-enriched sites would have greater biomass of both epiphytic algae and grazing invertebrates, though the bottom-up effects on higher trophic levels (occupied by juvenile and piscivorous fish) were not observed (York et al., 2012).

In areas outside of the Puget Sound, there have been observations of increased abundances in the abundance of jellyfish (gelatinous zooplankton) over the last 50 years, with the increases being associated with eutrophication, intensive fishing, and climate change (Purcell et al., 2007). Despite the fact that large jellyfish blooms have been observed in the Puget Sound, it is not clear if these are reflective of long-term trends; there is not sufficient or consistent monitoring data to do so (Greene, 2021). The NOAA Northwest Fisheries Science Center has conducted surface trawls in Skagit Bay, providing one of the few continuous data sets in the region. The data suggests that annual jellyfish biomass varies widely, with no obvious trends. Species specific monitoring suggests that there may be species-specific differences in response to external stressors such as climate change and eutrophication (Green and Munsch in PSEMP Marine Waters Workgroup, 2020).

The Washington State Department of Ecology sediment monitoring program has collected long term data on marine benthic invertebrate communities for over three decades. Their data suggests that, over the long term, the health of the benthic communities has declined, as described by species diversity, abundance, and type (e.g., benthic communities are less diverse and there is a higher abundance of stress tolerant species). It also appears that the most affected areas are in the terminal inlets that are characterized by low flushing rates, lower levels of dissolved oxygen, and higher inputs and accumulation of organic carbon and nitrogen. One hypothesis is that these changes are associated with changes in anthropogenic nutrient loading to the Puget Sound, though the primary driver of these changes is still an open question. Similarly, it has been hypothesized that changes in nutrient ratios leads to changes in the pelagic food web resulting in a reduction in carbon exports to the Puget Sound benthos (Krembs et al., 2014).

Overall, actual food web impacts related to anthropogenic nutrient additions in Puget Sound have not been reported. A proposed critical analysis suggests evaluating potential changes in benthic community structure is suggested, below.

4.7 Ocean Acidification

Increases in atmospheric carbon dioxide (CO_2) are well documented (e.g., https://gml.noaa.gov/ccgg/trends/) and are directly attributable to human activities. Approximately one-quarter of this human-derived CO_2 has been adsorbed by the oceans, causing the pH of the ocean to decrease (Feely et al., 2012). This process is called "ocean acidification."

While the increases in atmospheric CO_2 are probably the most important driver of ocean acidification in the region, there are other drivers that can affect the pH of Puget Sound waters including inputs of nutrients and organic carbon (Feely et al., 2012). As described above (Section 4.1), nitrogen additions to the Puget Sound can increase primary production generally in the near-surface water in the photic zone. This increased growth results in an uptake of CO_2 and an increase in pH (less acidic) in the surface waters. The algal blooms can sink into the bottom water and degrade, releasing CO_2 and decreasing pH (more acidic). Therefore, an increase in nutrient loading can lead to a localized exacerbation of ocean acidification.

Feely et al. (2012) reported that, while it was likely that anthropogenic nitrogen inputs impacted pH in Puget Sound waters though it was not possible, at that time, to differentiate the nitrogen-related changes relative to other drivers. Bednaršek et al. (2020) utilized the Salish Sea Model to assess current exposure and associated risk to potentially sensitive species, such as calcifiers, pteropods, and Dungeness crabs, due to reduced aragonite saturation state (Ω_{ar}) and pH conditions. Current conditions were compared with the pre-industrial conditions. Their results indicated an average decrease in the top 100 m of the water column of Ω_{ar} ~0.11, and decrease in pH of 0.06 since pre-industrial times. The main anthropogenic driver was increased atmospheric CO_2 uptake, while nutrient-driven eutrophication resulted in only a marginal role over spatially and temporally limited scales.

Khangaonkar et al. (2019) evaluated the projected impacts of climate change and population growth on Salish Sea water quality for historic (Y2000) and future (Y2095) scenarios. Their predictions estimated that there would be an observable decrease in pH (average ΔpH = -0.36) largely driven by a decrease in pH at the ocean boundary due to increased global atmospheric CO_2. The largest changes in pH were predicted for the continental shelf and the Strait of Juan de Fuca regions.

4.8 Proposed critical analysis – Risk to Puget Sound Species and Habitats of Low Dissolved Oxygen

Dissolved oxygen is critical for the survival and wellbeing of benthic and pelagic marine organisms. As such, spatial and/or temporal reductions in dissolved oxygen may cause harmful impacts. Threshold values describing levels that are considered protective have been identified for many species and DO concentrations below those values may impose some risk (Davis, 1975; Vaquer-Sunyer and Duarte, 2008). However, it is likely that not all species are equally susceptible to episodic low DO events or chronically low DO, due to differences in physiology, life history, and mobility, etc. (Froehlich et al., 2014; Froehlich et al., 2015). As such predicting responses and, in particular, impacts on aquatic species can be difficult (Moriarty et al., 2020; Sato et al., 2016). Several topics associated with these DO threshold values for Puget Sound species were identified as research needs and/or critical uncertainties by the Interdisciplinary Team during the Implementation Strategy development process. Topics include:

- Identifying the biological DO threshold of Puget Sound species across key life stages. This includes identifying the potential impacts on Puget Sound species and communities from low DO.
- Identifying the intersection between the geographic and temporal ranges of sensitive Puget Sound species, and those areas that are susceptible to reductions in DO associated with anthropogenic nutrient inputs.
- Identifying the uncertainty associated with the biological threshold values.

These topics were also identified as key actions in the Marine Water Quality Implementation Strategy as:

> Solicit research projects to study biological integrity with DO response (Action Code MWQ.RC5.1.1, Puget Sound Partnership (2024))
>
> Updated literature review of DO impacts on organisms of interest in priority habitats (Action Code MWQ RC5.1.2, Puget Sound Partnership (2024))

A critical analysis could be structured to incrementally address a subset of these topics.

4.8.1 Proposed approach

Outputs from existing models and data from published literature and laboratory studies can be combined to provide information on species risk and highlight geographic areas where species may be most at risk. We propose to use a risk assessment framework to conduct the following tasks:

- Identify species that may be at the highest risk for exposure to anthropogenic-related DO depletion, using a risk assessment framework. Risk assessment combines exposure (to stressors) with sensitivity (to stressors) and can be used to qualitatively determine relative vulnerability of species and habitats.
- Exposure can be evaluated using, for example, the Atlantis Ecosystem Model for Puget Sound (Morzaria-Luna et al., 2022), which provides spatially explicit distributions of Puget Sound species based on an evaluation of habitat associations. In combination with information about oxygen patterns in time and space, based for example on outputs from the Salish Sea Model that identifies the areas within the Salish Sea that are most impacted by

anthropogenic nutrient inputs (e.g., (Ahmed et al., 2019) and Section 8), we can identify the species that occur in the areas with the highest magnitude and/or duration of impact, which are at the highest potential risk (our "species of interest").

- Sensitivity can be evaluated based on published literature about, for example, lethal effects of low oxygen concentrations, or metabolic indices that combine oxygen availability with temperature to quantify oxygen demand. Sensitivity also varies by life stage, and can be quantified in terms of sublethal effects such as on growth and reproduction.
- Utilize output data from the Salish Sea Model to produce spatially explicit risk maps that identify areas of greatest species-risk, and include a description of which species are at risk in each area.

This resulting information will be of use for developing spatially-explicit mitigation and management activities that are most likely to alleviate the highest risk to the most susceptible species.

4.8.2 Status

This critical analysis has been identified by staff at the Puget Sound Institute as a focus area for the 2022-2023 fiscal year.

4.9 Proposed critical analysis – Low DO and benthic community structure

Since 1989, the Washington State Department of Ecology Marine Sediment Monitoring Program has collected data to assess the condition of sediment-dwelling invertebrate communities. These long-term data reveal declines in benthos abundance and taxa richness in parts of Puget Sound that do not appear to be correlated with concentrations of chemical contaminants in the sediment. It may be that changes in the community structure can provide information about changes in biophysical condition.

4.9.1 Proposed approach

This critical analysis will consist of 1) a focused literature review to assess the information available relating measures of marine benthic community structure to environmental conditions, and 2) explore the existing data from the Ecology Marine Sediment Monitoring Program to characterize the extent of changes over time.

4.9.2 Status

The Puget Sound Partnership has provided funding to the Puget Sound sediment monitoring team at the Washington State Department of Ecology to develop a Vital Sign indicator focusing on marine benthic community structure. That indicator should provide the tools necessary to determine status and trends of the community structure.

5 SOURCES OF NITROGEN – NITROGEN BUDGET

There are several different sources or pathways by which nitrogen enters the Puget Sound including oceanic inputs, wastewater treatment effluent, watershed runoff that is transported via rivers and streams, and subsurface transport directly into the marine waters. Mohamedali et al. (2011b) estimated that 68% of total nitrogen loading to Puget Sound enters via ocean exchange and 32% is from local sources within the greater Puget Sound watershed. The magnitude of loading from ocean accounts for advective imports (generally in the deep layers) and advective exports (generally at the surface layers) at the Puget Sound-Ocean boundary. Of the local sources, 58% is estimated to be from point sources such as WWTP effluent, 41% from nonpoint sources, entering through surface water runoff, and 1% from atmospheric deposition. The nonpoint sources are estimated to be 34% human-associated and 66% natural origin (see

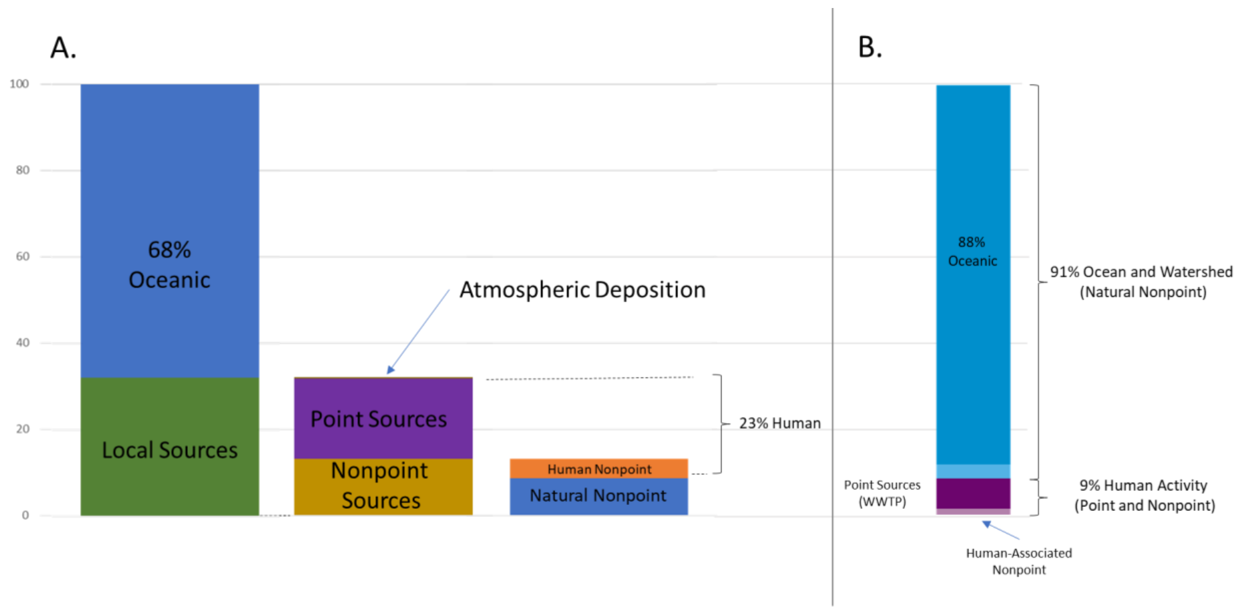


Figure 13. Two estimates of relative nitrogen loading to Puget Sound estuary by major source illustrate uncertainty and variation in loadings estimates.

It is important to note that these are estimates; there are uncertainties and differences in approaches in the loading estimates, and others have reported different relative and absolute loadings. For example, Mackas and Harrison (1997) developed estimates for the overall N budget for Puget Sound/Strait of Georgia/Strait of Juan de Fuca and reported that the oceanic inputs accounted for ~ 84%, sewage <6%, and rivers (including sewage) <9% of total N loading to Puget Sound. The University of Washington produced updated loading estimates, and reported that ~91% of loadings were from natural sources (~88% oceanic and 3% natural watershed), while 9% were associated with human activities in Washington State. (see Figure 13). Note that these loadings only considered inputs into the system. While the estimates are not precisely comparable, the numbers do provide context and magnitude of the relative loadings, and to also highlight the fact that developing a nutrient budget is a complex exercise with inherent uncertainties; the exact loadings from year-to-year are not known. The Washington State Department of Ecology are performing additional monitoring on eight major rivers entering Puget Sound to improve data quality and availability. However, estimating inputs from the ocean and other, unmonitored source, will remain with a high level of uncertainty.

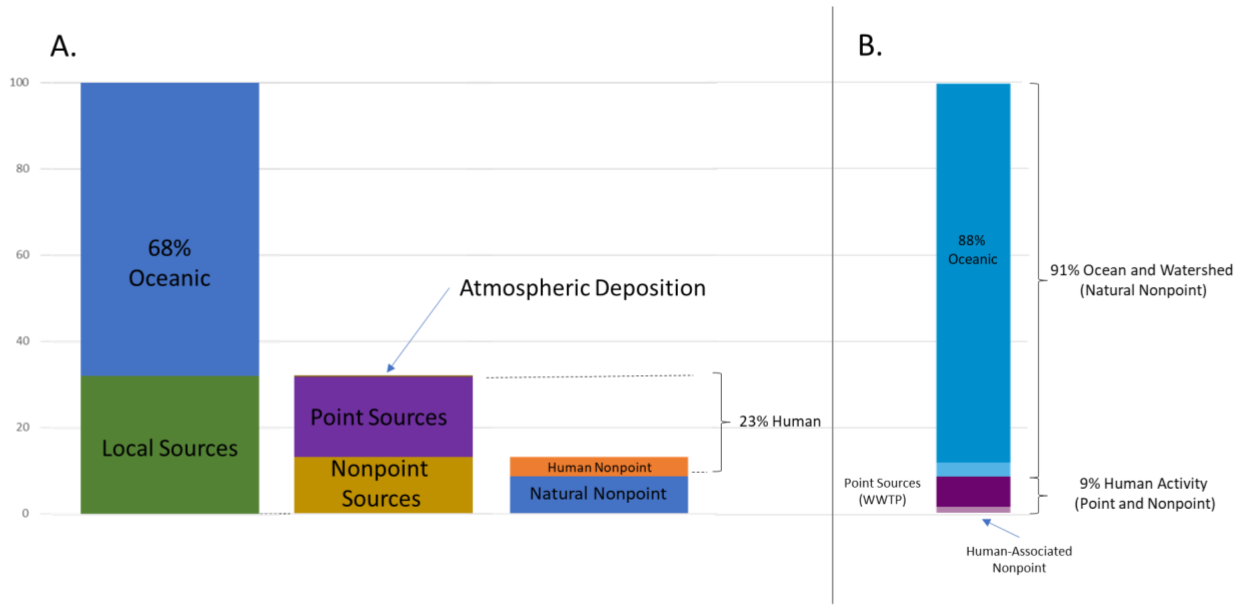


Figure 13. Two estimates of relative nitrogen loading to Puget Sound estuary by major source illustrate uncertainty and variation in loadings estimates.
Panel A), adopted from Mohamedali et al. (2011b), shows that “natural” sources (oceanic and natural watershed) account for ~77% of total loading while anthropogenic (WWTP and anthropogenic watershed) account for 23%. Panel B) adopted from Puget Sound Institute (PSI, 2023) indicate that ~91% of N loading is natural, while ~9% is anthropogenic. Note that the oceanic loadings in A) consider both N imports and N exports at the ocean boundary; loadings in B) are based strictly on imports.

The nutrient loads are not evenly distributed on a spatial (throughout the Puget Sound) or temporal (throughout the calendar year) basis. Nor are they evenly distributed vertically in the water column. Oceanic-associated nitrogen enters into Puget Sound predominantly in the deeper waters. Wastewater treatment system outfalls are located predominantly at the bottom of the Puget Sound at depths varying with the bathymetry of the region of discharge. Differences in buoyancy between the wastewater discharge from the effluent diffusors, and the surrounding marine waters, may result in upward migration of the effluent plume. Watershed loads enter predominantly via rivers and streams into the surface layer. These factors matter because phytoplankton growth, that is supported via nutrient inputs, varies by season and depth, and is generally nitrogen limited in the euphotic zone for certain periods of the year (mainly for late-spring, summer, and early-fall). An illustrative example of the timing of nutrient loads from wastewater and rivers is shown in Figure 14.

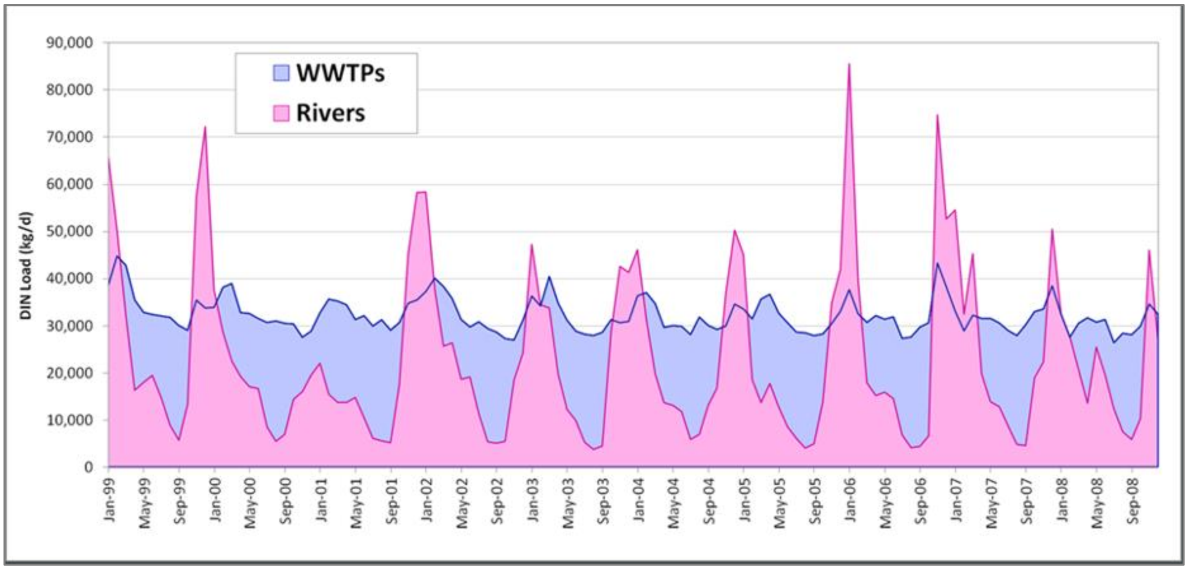


Figure 14. Seasonal Dissolved Inorganic Nitrogen (DIN) Loads to Puget Sound from rivers and wastewater treatment plant effluent (WWTP).

Loadings do not include sources into Canadian waters such as the Frasier River (Washington State Department of Ecology, 2020).

5.1 Oceanic Loading - Marine Water entering Salish Sea (Nonpoint Source)

Nitrogen inputs to Puget Sound from oceanic sources via estuarine circulation are larger than all other sources combined (see Figure 13). For example, Mackas and Harrison (1997) indicate that N imports from the ocean are ~ 84% of total loadings to Puget Sound. These are some of the only published that are based on multi-year analysis of nitrate and salinity concentration. Mohamedali et al. (2011b) estimated net oceanic input (imports in deep layers minus exports in surface layers) of approximately 68% of the total dissolved nitrogen to the Puget Sound. There is uncertainty associated with these values (see Section 5).

Upwelling from the coastal shelf carries relatively high nitrate concentration water through the Strait of Juan de Fuca and into Admiralty Inlet. Long term water quality monitoring performed by Washington State Department of Ecology has shown that there are variations in the nitrate concentration over time (see Figure 15) indicating variations in loading. For example, monitoring data suggests that 2006-2008 had higher N concentration in the oceanic waters compared to the baseline average.

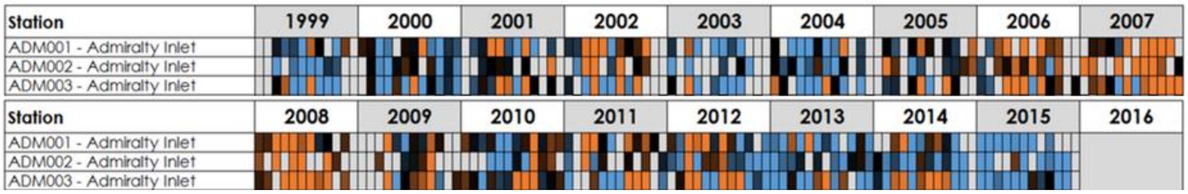


Figure 15. Variations in nitrate concentration at Admiralty Inlet monitoring stations.
Orange indicates concentrations are higher than baseline. Blue indicates concentrations are lower than baseline. Grey indicates there is no data. From Washington State Department of Ecology (2020).

Additional variation in oceanic loading occurs as a direct result of variations in exchange flows to the Salish Sea. Changes in freshwater input, either seasonally or annually, alter the freshwater-induced salinity gradients and exchange flows. For example, Khangaonkar et al. (2021) demonstrated an approximately 10% change in tidally-averaged exchange flow over a five-year model simulation resulting in roughly proportional changes in oceanic nutrient loading. This indicates that the proportion of oceanic vs anthropogenic nutrient loading varies annually and should be considered when evaluating the relative and absolute impacts of anthropogenic nutrient loading.

5.2 Watershed Inputs (Nonpoint Sources)

Rivers and streams transport natural and anthropogenic nitrogen from the upland watersheds into the Puget Sound. Mohamedali et al. (2011b) estimated that approximately 13% of total nitrogen load entered Puget Sound via rivers and streams, with approximately 34% of that (~4.5% of total) being anthropogenic of origin (Figure 13).

The magnitude of anthropogenic nitrogen varies between watersheds and is generally a function of the extent of development, type of development, and activity within the watershed. For example, the rivers that originate in undeveloped areas such as Olympic National Park and other protected forest lands on the Olympic Peninsula are considered to be relatively pristine and nitrogen loading associated with those rivers and streams is thought to be largely consistent with pre-anthropogenic (i.e., "Reference conditions"). In contrast, those rivers that run through developed, and in particular, agricultural watersheds, will have a higher proportion of anthropogenic-associated nitrogen. The Nooksack, Samish, and Stillaguamish rivers, amongst others, have a high proportion of anthropogenic-associated nitrogen loading (Mohamedali et al., 2011a). Illustrative examples of the differences in anthropogenic and pre-anthropogenic concentration and loadings are shown in Figure 16 and Table 1.

Watershed-associated nitrogen loads enter Puget Sound in the surface layer and may, depending on timing and location of inputs, be taken up by phytoplankton, advectively transported out of the Salish Sea in the surface layer, or re-entrained into the deeper waters (e.g., Figure 3). With the exceptions of large rivers such as the Frasier and the Skagit, it is expected that watershed related nutrients will not affect areas far from the river mouths, particularly in the spring and summer when uptake and phytoplankton growth rates may be high (Banas et al., 2015).

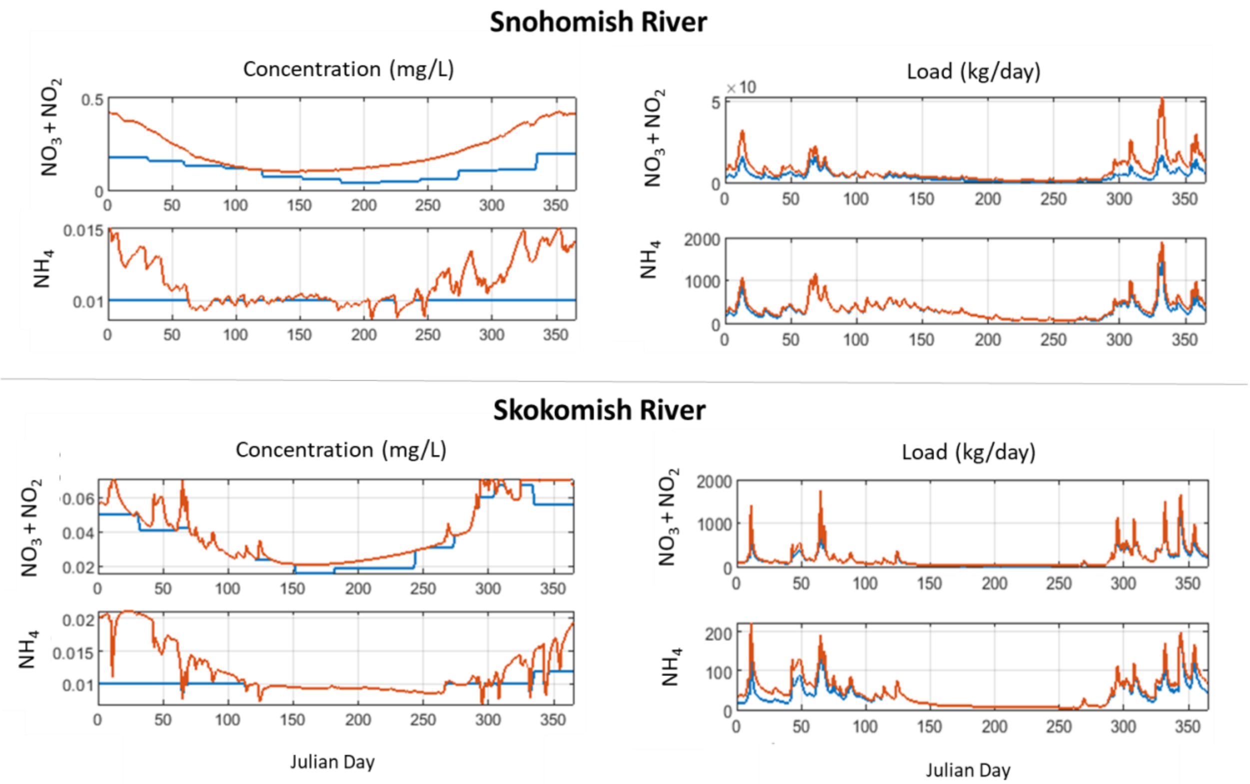


Figure 16. Estimated concentration and loading for anthropogenic (red lines) and pre-anthropogenic (blue lines) watershed conditions for the Snohomish and Skokomish rivers over the 2014 model year.
Loading reflects both concentration and flow timing. The anthropogenic conditions are used in Existing conditions scenarios; information from pre-anthropogenic conditions are used in Reference condition scenarios. The difference in the red line and the blue line is magnitude of anthropogenic impact for each watershed. Data from the Salish Sea Modeling Center (2021).

Table 1. Estimated increase in annual loadings of nitrogen and carbon due to anthropogenic development in selected watersheds.

Values are estimated by subtracting estimated or measured current loadings from estimated pre-anthropogenic (reference) loadings. 2014 model year. Data from Salish Sea Modeling Center (2021).

River	DIN [kg/yr]	NH4 [kg/yr]	DOC [kg/yr]	DON [kg/yr]	POC [kg/yr]	PON [kg/yr]
'Tahlequah'	3404	47	20669	362	949	149
'University Place'	8197	54	7301	2117	552	551
'Dabob Bay'	173	0	0	39	0	39
'Dosewallips R'	2754	0	0	589	0	589
'Duckabush R'	2870	0	0	673	0	673
'Hamma Hamma R'	4345	1756	161426	327	0	327
'Kitsap_Hood'	35296	678	0	13325	0	13325
'Lynch Cove'	17505	1599	202295	2765	101981	8782
'NW Hood'	24114	579	0	8534	0	8534
'Port Gamble'	20421	460	0	7321	0	7321
'Quilcene'	1002	0	0	219	0	219
'Skokomish R'	9356	3163	323220	681	0	681
'Tahuya'	25606	2723	290736	4202	133904	12713
'Skagit R'	169452	0	7639842	19347	2206174	19347
'Snohomish R'	1230896	14175	0	71978	0	71978
'Stillaguamish R'	394962	11256	0	45185	0	45185
'Whidbey east'	53311	428	0	2484	0	2484
'Port Townsend'	165	87	50858	95	3579	95
'Whidbey west'	60489	400	0	2410	0	2410
'Birch Bay'	66449	2124	0	1294	0	1294
'Lopez Island'	19855	138	0	731	0	731
'Nooksack R'	1133973	47212	0	29215	0	29215
'Orcas Island'	47089	314	0	1376	0	1376
'Samish_Bell south'	248673	1475	0	4472	0	4472
'San Juan Island'	35629	243	0	1166	0	1166
'Whatcom_Bell north'	187571	1047	0	2593	0	2593
'Clallam Bay'	25025	2466	488875	1368	86581	1368
'Discovery Bay'	324	136	13290	271	0	271
'Dungeness R'	1526	380	50501	1243	419	1243
'Elwha R'	19877	1929	1519825	3453	199383	3453
'North Olympic'	16025	1691	360540	1984	79088	1984
'Port Angeles'	917	249	39428	779	0	779
'Sequim Bay'	75	39	0	45	0	45
'Fraser R'	0	0	0	0	0	0

5.2.1 Watershed flow volume and timing

Watershed nutrient loadings are not uniformly distributed either spatially or temporally, throughout the Puget Sound. Nutrient loadings are roughly proportional to volumetric flows of the individual rivers and streams entering the Puget Sound, with bigger (high flow) rivers providing a higher mass loading than smaller (low volume) streams. On an annualized basis the largest loadings within the Puget Sound basin come from the Snohomish River (~5,945 kg DIN/day), the Skagit River (~4,225 kg DIN/day), the Stillaguamish River (~2,440 kg DIN/day), the Puyallup River (~2,105 kg DIN/day) and the Green River (~1,635 kg DIN/day) (Mohamedali et al., 2011b). The spatial distribution of loading from watershed sources is shown in Figure 17. Note that the loading from the Frasier River is much greater than the loading from any other river in the Salish Sea watershed (~33,140 kg DIN/day).

The flow timing of nitrogen is not consistent over the course of year as flows, and associated nitrogen loadings, vary with seasonal rainfall and snow melt patterns. A summary of load timing for Puget Sound rivers is shown in Figure 14. A summary of annual timing of the major rivers in Puget Sound, and the Frasier River, is in Figure 18.

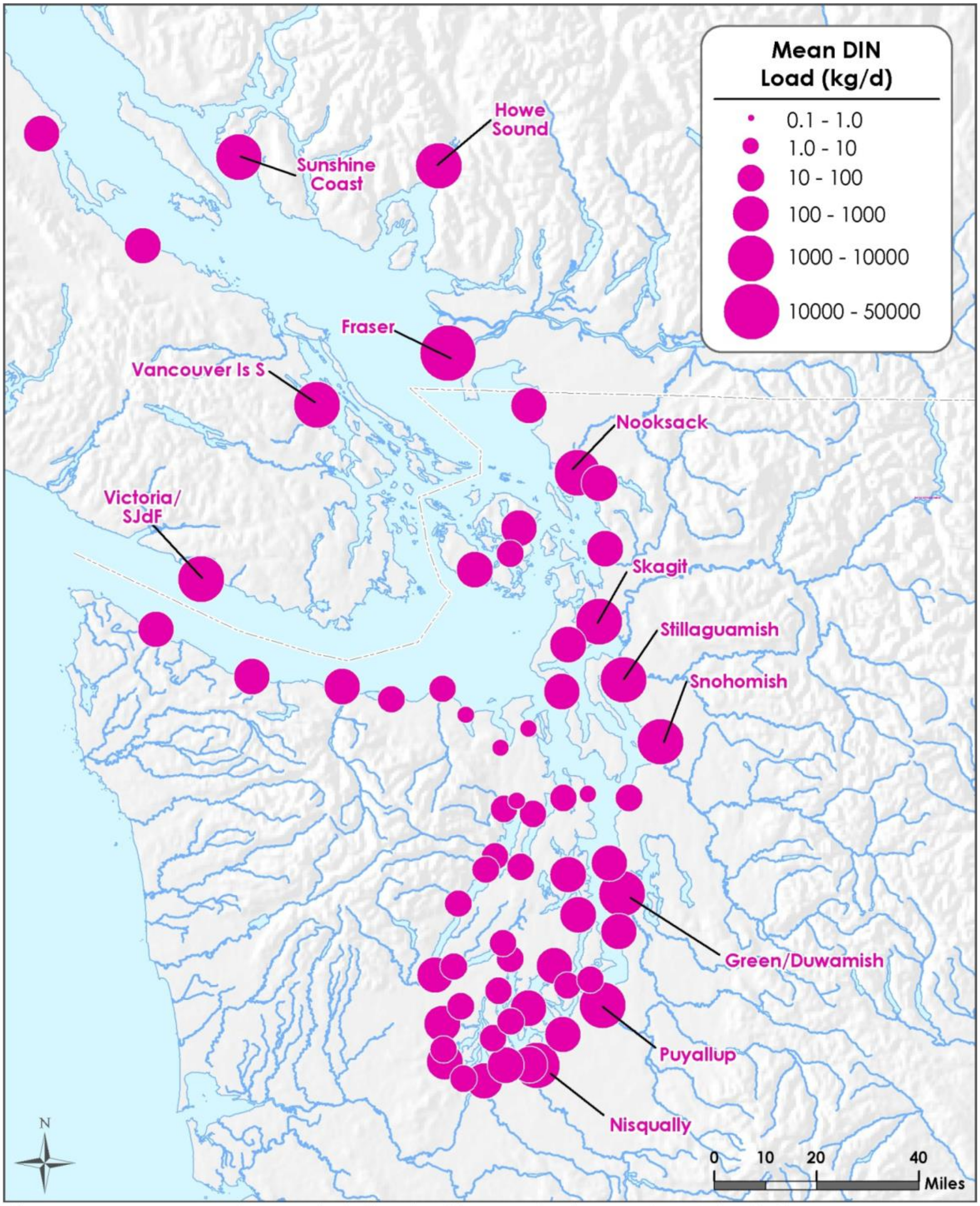


Figure 17. Estimated mean annual watershed-related dissolved inorganic nitrogen loads in the Salish Sea (1999-2008).

Loads include both nitrogen from both anthropogenic and non-anthropogenic sources. From Mohamedali et al. (2011b). Note that loads <1000 kg DIN/day are not shown.

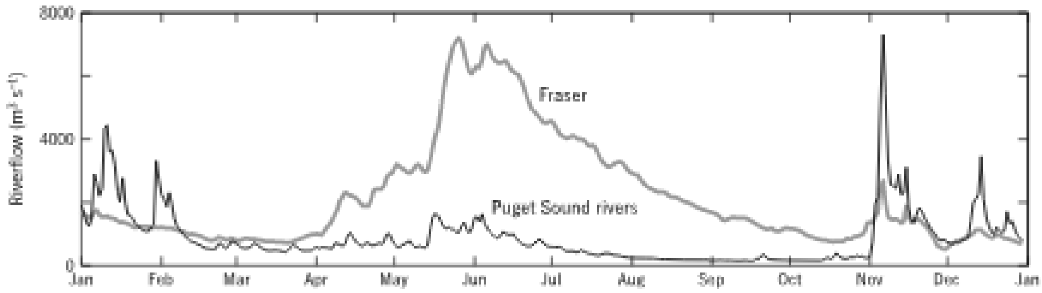


Figure 18. Annual cycle of flows for Fraser and 14 Puget Sound rivers for 2006. Banas et al. (2015)

5.3 Wastewater Treatment Plants (Point Sources)

The typical secondary wastewater treatment systems that treat the majority of the human/municipal waste in the Puget Sound basin do not actively treat nitrogen and so are a major source of nitrogen to the estuarine environment. They account for approximately 19% of overall nitrogen loading to Puget Sound, or approximately 58% of non-oceanic loadings (Figure 13; Mohamedali et al. (2011b)). Most of the flows enter through outfalls located at the bottom of the basin through outfalls that extend well offshore from the treatment facility, and so predominately enter in the deep bottom layer. Relative to the variations in timing of the watershed inputs, the loadings from the wastewater treatment systems are relatively uniform over the course of the year (Figure 14) though some combined systems collect stormwater runoff resulting in increased volumetric and mass loadings during the winter rainy season. The distribution of the loadings, shown in Figure 19, is roughly proportional to the distribution of population, with higher loads coming from the densely populated urban areas of central Puget Sound.

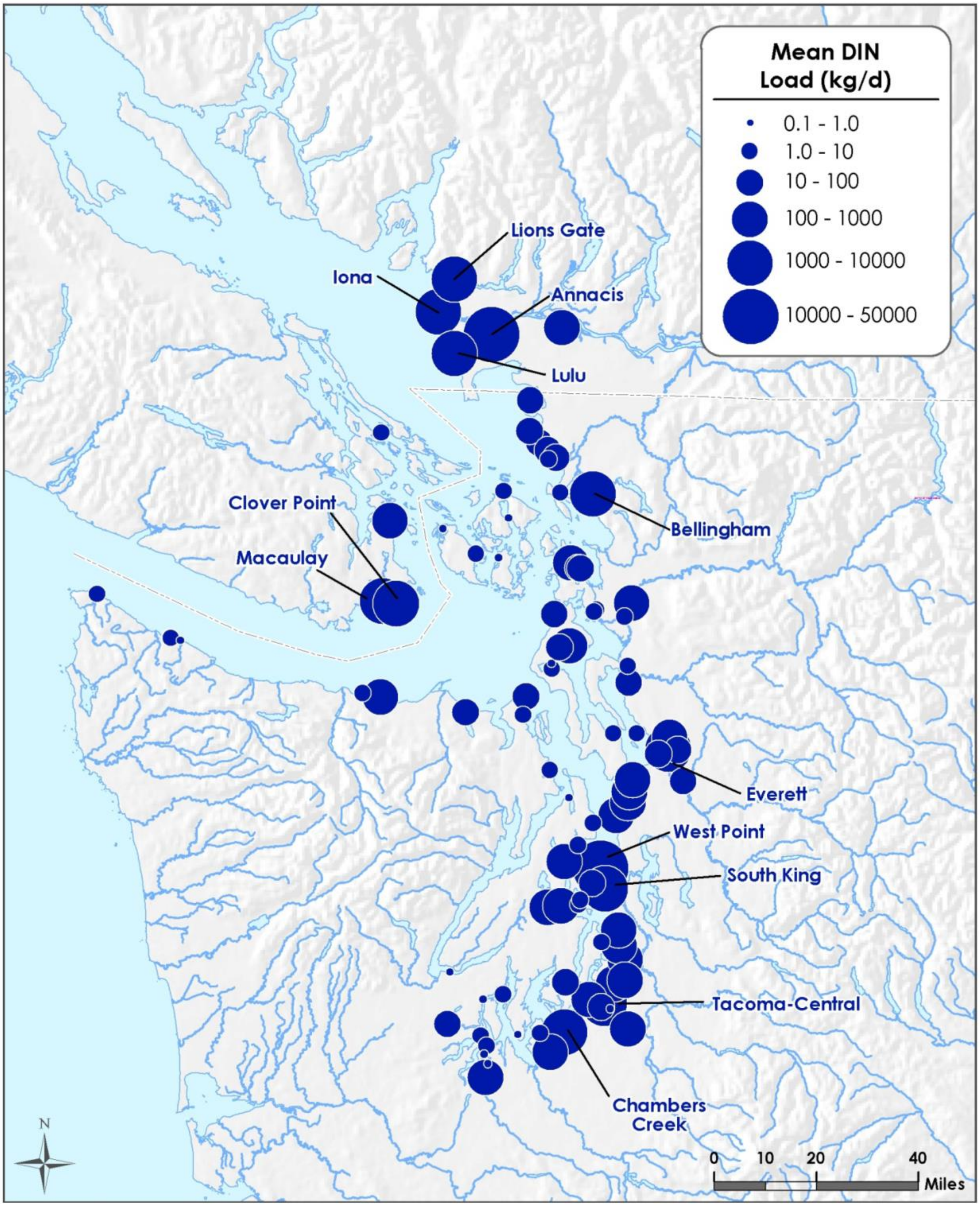


Figure 19. Mean annual dissolved inorganic nitrogen loads (DIN) associated with wastewater treatment plan effluent (point source).
From Mohamedali et al. (2011b). Note that loads <1000 kg DIN/day are not shown.

5.4 Additional Nitrogen Inputs to Salish Sea

5.4.1 Onsite Sewage (Septic) Systems

Standard onsite Sewage/Septic Systems (OSS) are not designed for nutrient removal and little nutrient reduction occurs in the septic tank. Effluent water from the septic tank is typically discharged into a drain field where pathogen deactivation, carbon oxidation, nitrification and potentially denitrification may occur. However, many poorly function septic systems, or those located near to receiving waters, may result in only minimal nitrogen removal. As such, these can be potential sources of nitrogen loading. There are tens-of-thousands of OSSs in the Puget Sound watershed, with a large fraction of them in marine protected areas; they may be a significant source of nitrogen to localized areas of Puget Sound, particularly in very poorly flushed embayments.

Lynch Cove has been identified as an area potentially impacted by nitrogen from OSSs. Cope and Roberts (2013) led a review of information related to nitrogen loading and associated impacts on dissolved oxygen in Hood Canal and Lynch Cove. They reported that approximately 20-40% of watershed loading to Lynch Cove may be attributable to OSSs. Steinberg et al. (2011) assessed nitrogen sources to Hood Canal, finding that OSS N inputs contribute ~0.5% of total N loading, possibly slightly higher to Lynch Cove. Steinberg et al. (2011) asserts that discharges from OSSs and red alders appear to be a very important N source for many streams, but a minor nutrient source for Hood Canal.

Overall, Cope and Roberts (2013) concluded that marine water upwelling delivers most of the nitrogen to the surface waters, OSSs located along the shoreline may represent the dominant human source of nitrogen, and that human impacts are highest in Lynch Cove.

5.4.2 Atmospheric Deposition

Atmospheric deposition adds nitrogen to the Puget Sound. However, only 1% of the total DIN load is estimated to be by direct atmospheric deposition to Puget Sound waters (Mohamedali et al. 2011). Atmospheric deposition does contribute additional nitrogen, but the deposition to the land mass is accounted for in the watershed loads (Section 5.2). Overall, US atmospheric loads have been stable, with general declines coupled with forest fires contributing additional seasonal loading. Atmospheric contributions to the open oceans are growing, possibly due to industrialization in Asia (Ecology 2018).

5.4.3 Groundwater

Vaccaro et al. (1998) estimated that approximately 100 to 1000 ft^3/s of direct groundwater discharge to Puget Sound compared an estimated annual average of 52,000 ft^3/s from surface waters (Czuba et al., 2011). Assuming that the groundwater DIN concentrations are equivalent to the nearby-surface waters, this would suggest that ~0.2 to 2.0% of total nitrogen loads may be associated with groundwater flows. This is generally considered to be within the measurement error of even the best flow measurements (Mohamedali et al., 2011b) and so is not incorporated into overall nitrogen loading. However, as mentioned above, groundwater affected by OSSs may have localized impacts and importance for certain terminal inlets.

6 ASSESSMENT TOOLS - SALISH SEA MODEL - EVALUATING MARINE CONDITION

The Marine Water Quality target indicates that the human-related impacts on dissolved oxygen should not be more than 0.2 mg/L anywhere in the Puget Sound. This is a comparative measure, comparing conditions before anthropogenic impacts with that currently observed. However, there are no data to describe the conditions prior to human settlement in the region. As such, a "Reference condition" has been established to describe pre-anthropogenic conditions (see Section 6.2). Anthropogenic impacts, and the results of management scenarios are determined based on the relative difference based on a comparison of two numerical model simulations. The numerical model used is the Salish Sea Model.

The development and application of the Salish Sea Model is described first. It's application for use in evaluating the nutrient management strategies described in the Marine Water Quality Implementation Strategy are presented following.

6.1 Salish Sea Model

The Salish Sea Model (SSM) is a predictive coastal-ocean model consisting of a coupled hydrodynamic and water quality model. The water-quality model is based on the Finite Volume Coastal Ocean Model (FVCOM; (Chen et al., 2003)), a 3-D hydrodynamic model that can simulate tidally and density-driven, and meteorological forcing-induced circulation in an unstructured, finite element framework. FVCOM solutions are coupled to the CE-QUAL-ICM biogeochemical model (Cerco and Cole, 1993) in the SSM FVCOM-ICM code. Both FVCOM and CE-QUAL-ICM have independently been used and published extensively (see model documentation on the University of Massachusetts and US Army Corps of Engineers websites respectively), and a recent version of the SSM FVCOM-ICM code is open source and available for download. The SSM is being used by Washington State for the Puget Sound Nutrient Source Reduction Project with the model and results available through the Department of Ecology website. In addition, versions of the model are utilized at various scales for the analysis of Salish Sea response to other water quality challenges, marine pollution, oil spill and sediment transport, climate change response, and restoration planning, among other applications – further detailed on the Salish Sea Modeling Center website.

The Pacific Northwest National Laboratory first developed the Puget Sound Dissolved Oxygen Model (PSM) in collaboration with the Environmental Protection Agency and Washington State Department of Ecology, and subsequently developed the Salish Sea Model (SSM). Each phase of the overall model development is summarized in Table 2, following the nomenclature used by the University of Washington Salish Sea Modeling Center (SSMC), and further details on specific modules and associated publications are provided in the Salish Sea Modeling Center website (https://ssmc-uw.org/). In addition to the publications listed, Ecology undertook further Quality Assurance Project Plans (QAPPs), validation, and sensitivity analysis specific to the development of the model and application in the Puget Sound Nutrient Source Reduction Project, as described in in Ahmed et al., (2019) and the summary presented during the Puget Sound Nutrient Forum, September 20, 2018). A brief summary of the model, and model development and use, is included below.

PSM 2012 (FVCOM v2.7/FVCOM-ICMv1) successfully reproduced nutrient dynamics and DO levels throughout Puget Sound, based on water exchanges with the Pacific Ocean and nutrient loads from natural and human sources within the basin (Khangaonkar et al., 2012; Kim and Khangaonkar, 2012). The model has since been updated to incorporate improved inputs, an expanded domain (improving the capability to model exchanges between Salish Sea and Pacific Ocean) and inclusion of various modules associated with more detailed and accurate representation of ecosystem function (Table 2).

Updates to the subsequent SSM included the addition of sediment diagenesis to improve sediment-water column interactions, including nutrient exchange and sediment oxygen demand and pH modules (Pelletier et al. 2017a and 2017b), refinement of freshwater loading and flows (Mohamedali et al. 2018). This version of the model (referred to as SSM 2017 -FVCOM v2.7ecy/FVCOM-ICMv2 - in Table 2) was that used by the State in Puget Sound Nutrient Source Reduction Project with an associated QAPP (McCarthy et al. 2018) and key reports and publications (e.g., Ahmed et al., 2019). However, it should be noted that the version used by the state subsequently included recalibration and harmonization of pH and DO using the parameters consistent with Khangaonkar et al. (2019) (FVCOM-ICMv2 in Table 2), and in the Bounding Scenario's update (Ahmed, 2021) the ocean boundary forcing was updated to use HYCOM for the year 2014.

Subsequent versions of the model developed by PNNL/SSMC include turbidity, zooplankton, and submerged aquatic vegetation modules (SSM 2021 - FVCOM v2.7d/FVCOM-ICMv4), and recent applications include multi-year runs examining the recent marine heat wave (Khangaonkar, et al., 2021), and refined quantification of residence and flushing times of embayments using a higher resolution bathymetric grid with approximately 100m near-shore resolution (Premathilake and Khangaonkar, 2022).

Table 2. Description of development of Salish Sea Model including additional capabilities, and associated references and publications describing the application and validation (from the Salish Sea Modeling Center).

Model Year (and FVCOM / FVCOM-ICM versions)	**Code Development Publications**	**Description, Features and Domain Extent**
PSM 2012 (v2.7/v1)	Khangaonkar et al. (ECSS 2011, Ocean Dynamics. 2012) & Kim and Khangaonkar (2012) for FVCOM-ICM_v1 specifically	Original model also referred to as the Puget Sound Model. Domain: Puget Sound and Georgia Basin
PSM 2013 (v2.7a/v1)	Khangaonkar and Wang (Applied Ocean. Res. 2013)	+ floating structure / bridge module
PSM 2014 (v2.7b/v1)	Wang and Khangaonkar et al. (JMSE 2014)	+ kelp module
PSM 2016 (v2.7c/v1)	Khangaonkar et al. (Northwest Science 2016)	+ embedded fine resolution + wetting and drying Improved: intertidal nearshore salinity and temperature
PSM/SSM 2017 (v2.7ecy/v2)	Bianucci, Long, Khangaonkar et al. (Elementa Science of the Anthropocene, 2018)	+sediment diagenesis and +pH modules (documentation in Pelletier et al. (2017a) and (2017b) respectively)
SSM 2017 (v2.7d/v2)	Khangaonkar et al. (Ocean Modelling 2017)	Domain: extended past continental shelf + Exchange flow and circulation computation
SSM 2018 (v2.7d/v2)	Khangaonkar et al. (JGR 2018)	Domain: extended to shelf break + hypoxia and net heat flux calibration
SSM 2021 (v2.7d/v3&4)	Khangaonkar et al. (Ecological Modelling 2021)	Improved: ocean boundary forcing to HYCOM, new Re-aeration formulation Recalibration for harmonization of pH and DO (v3) + turbidity, zooplankton, and submerged aquatic vegetation modules (V4)
SSM 2021 (v4.3a/v4)	In Progress.	Improved: currents and water surface elevation calibration using distributed bed friction and meteorology and FVCOM version upgrade

There are selected areas of the model domain, including intertidal and some shallow subtidal areas, where conditions are not successfully modelled and so the output is not included in reporting or evaluation. Areas with large intertidal flats are estimated to be deeper in the model compared to their actual depth due to bathymetric smoothing. As such, the bathymetry at these nearshore locations are not realistically represented (Ahmed et al. 2019) and so the intertidal and shallow subtidal zones are modeled, but results are excluded, or masked.

6.2 Salish Sea Model Application – Quantifying Change in Dissolved Oxygen using the Salish Sea Model Output.

The initial version of the Marine Water Quality Vital Sign sets the following recovery target for the Dissolved Oxygen (DO) indicator:

> *By 2020, human-related contributions of nitrogen do not result in more than 0.2 mg/L reductions in DO levels anywhere in Puget Sound* (PSP 2019).

Further detail on the PSP water quality indicators, and current proposed revisions, are included in Section 1. This target is based on a comparison of the Existing conditions (or a management scenario) against the Reference condition, which is defined as the water quality conditions of Puget Sound prior to extensive anthropogenic impacts, and is estimated by the Salish Sea Model. The approach for calculating the Existing and Reference conditions, are summarized and applied in Ahmed et al. (2019) and updated in Ahmed et al. (2021).

As described below, a significant effort is ongoing to model outcomes of management scenarios of nutrient reduction. Since it is dependent on the model, understanding what parameters are modeled, and how calculations are applied to model outputs, can help interpret scenario run results. It may provide the wider scientific community with the detail on spatial and temporal variability of DO and additional model parameters outputs relevant across other PSP recovery targets (such as N, and net primary production) that are calculated alongside DO in each model run. In the longer term, this understanding can inform the design of research and monitoring actions addressing important scientific uncertainties (see Section 13).

In developing the Marine Water Quality Implementation Strategy, the Interdisciplinary Team identified a number of scientific uncertainties specifically related to the calculation of DO concentration and change of concentration using the SSM outputs. Developing confidence intervals associated with the magnitude, extent, and duration of DO depletion were prioritized as key uncertainties, and research and monitoring actions. Accordingly, many of the questions raised by the Interdisciplinary Team can first benefit from a shared understanding of how change in DO is calculated using model results.

The purpose of this section is, therefore, two-fold. First, to provide a synthesis of how the SSM outputs from the Existing and Reference condition model runs are used to quantify changes in DO. The methodologies described in key Ecology documentation are applied (Mohamedali et al., 2011a; Mohamedali et al., 2011b; Pelletier et al., 2017a; Ahmed et al., 2019; Ahmed et al., 2021), and illustrated step by step using 2014 model output data from Ahmed et al., (2019). In the following sections four commonly raised questions are addressed to understand how the calculations are performed.

- How are Existing and Reference conditions calculated for Washington state?
- What model outputs and which model domain areas are used in the calculation?
- How are cell-layers in the water column classified?
- How are the total days and area of exceedance calculated?

Second, using the same model outputs, an approach to analysis of DO results is extended to provide further information relevant to the PSP Marine Water Quality recovery target of no less than, "….0.2 mg/L reductions in DO levels anywhere in Puget Sound" (PSP 2019). Three sections describe this method and initial analysis:

- A novel approach to calculating DO volume change against the PSP recovery target, considering proportional integration of exposure time and volume throughout the water column
- Results of volume calculations of low DO, and DO concentration throughout the water column and year (see Section 8).
- Cumulative difference in DO concentrations (see Section 8).

6.2.1 Existing and Reference conditions

Existing conditions are estimated from a combination of measured and estimated nutrient concentrations and flows from rivers and streams (for the watershed loads) and WWTPs (for the marine point sources. Washington State Department of Ecology publications describe how the existing condition model runs were set up for the years 2006, 2008, and 2014 (e.g., Ahmed et al. (2019)). They also provide some key points on the way inputs and parameterization of the Reference condition model runs are setup, which are highlighted here.

Since there are no data or information describing water quality prior to anthropogenic influences, the Reference condition must be determined through modelled simulations. In the Salish Sea these predictions are performed through an application of the Salish Sea Model. Ahmed et al. (2019) describes the development and evaluation of the Reference condition. Briefly, under the Reference condition, the major human-derived, anthropogenic, inputs of nutrients from Washington State are removed. For each year modeled, a Reference condition is created using estimated flows for that particular model year. The nutrient concentrations for all WWTP in the United States portion of the Salish Sea and for all watersheds are set at the "reference condition," which are estimated based on the measured concentrations from selected pristine watersheds in the Olympic National Park, and elsewhere. Canadian point and nonpoint sources such as the Fraser River are kept at Existing conditions, as are Washington's industrial treatment plants (Ahmed et al., 2019; Mohamedali et al., 2011a; Mohamedali et al., 2011b; Pelletier et al., 2017a). Climate, hydrology and ocean boundary forcing are also kept the same as Existing conditions for each model year.

6.2.2 Salish Sea Model outputs and model domain for scenario evaluation

The SSM calculates 34 parameters at each location within the model domain. The domain includes all basins of Puget Sound, the Salish Sea, and the areas of the Pacific Ocean and Columbia River that influence estuarine hydrodynamics (Figure 20). The Salish Sea Modeling Center website (https://ssmc-uw.org/) provides further details on the domain extent for the different model versions, including those used by the Washington State Department of Ecology for regulatory purposes (SSMC, 2021). The model consists of 16,012 cells representing the surface area of the domain. For each cell, there are 10 cell-layers of varying depths below, representing the entire water column from the surface to the bottom sediments (Figure 20). Altogether, there are 160,120 unique cell-layers where DO and other water quality parameters are calculated at each model time-step. All parameters such as nitrogen speciation, net primary production, and turbidity are recorded at an hourly time-step in the model run output file.

A total of 4,144 cells (41,440 cell-layers) within Washington State waters are used to calculate DO (Table 3). This cell count excludes the area close to the shoreline where SSM accuracy is limited (Ahmed, 2019). A total of 2,435 cells adjacent to the shoreline are masked and not used in the calculations.

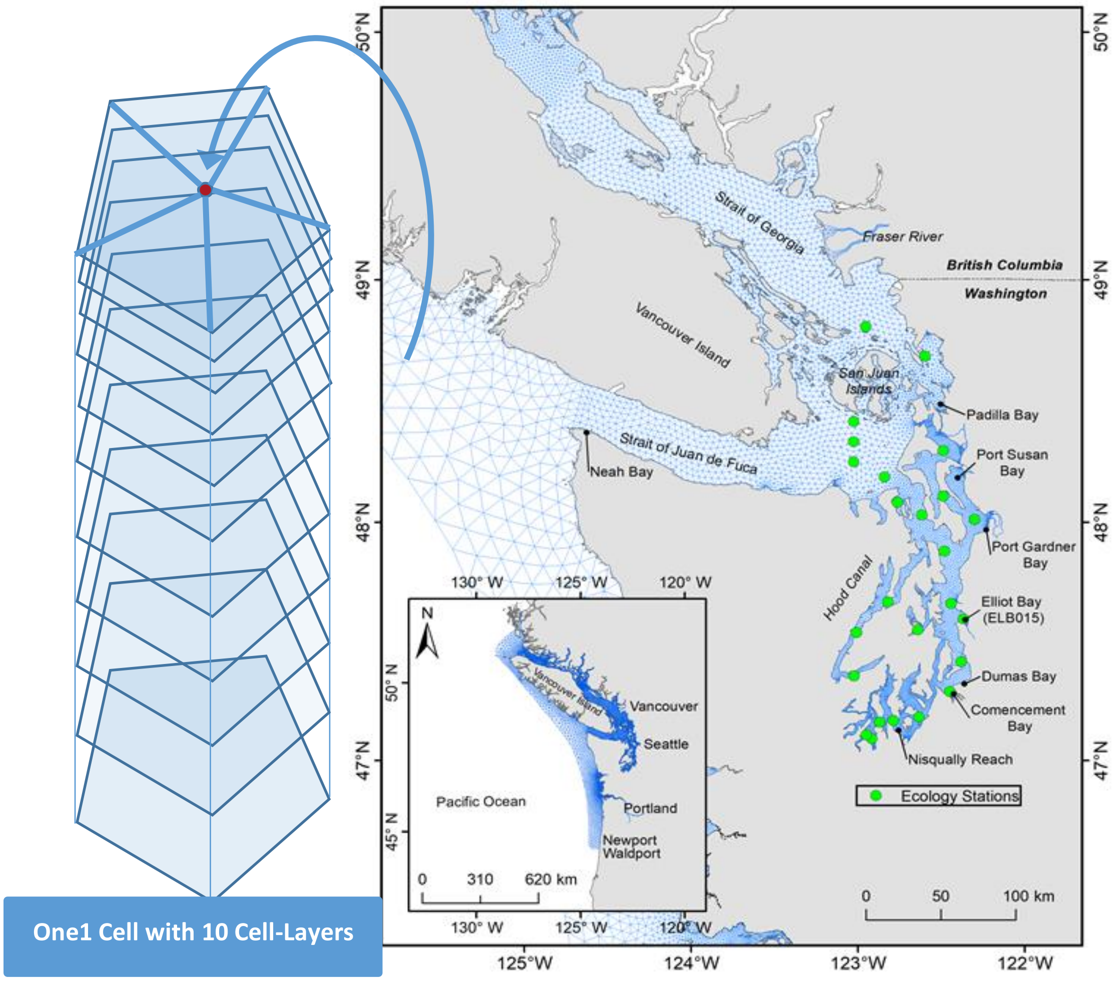


Figure 20. Salish Sea Model domain (right) and representation of a single cell for an area of the Salish Sea, and the water-column below it (left).
Each is a cell is a geometric element of the model mesh that varies in area from 185.9 km^2 on the ocean boundary to approximately 0.0024 km2 (2371 m^2) in some of the nearshore cells. This scale applies to a "medium scale" version of the grid domain that is used for scenario evaluations. Each cell has a central node with associated triangular elements described in Khangaonkar et al. (2018). Ten cell-layers of variable depth represent the water column below each cell, with cell-layer depths varying from 3.6% of total depth for the surface layers to 14.62% of total depth for the bottom layers.

Table 3. Total cells and layers for Salish Sea Model.

Cells and layers:	Count	%
Total cell number in the SSM domain	16,012	100.0%
Cells with DO standard	6,579	41.1%
Cells masked (excluded)	3,436	21.5%
Cells with DO standard and masked (subset of excluded cells)	2,435	15.2%
Cells considered in Washington State waters	4,144	25.9%
Cell-layers considered	41,440	
Days of output	360	
Cell-layer days considered in the six basins of Puget Sound	14,918,400	

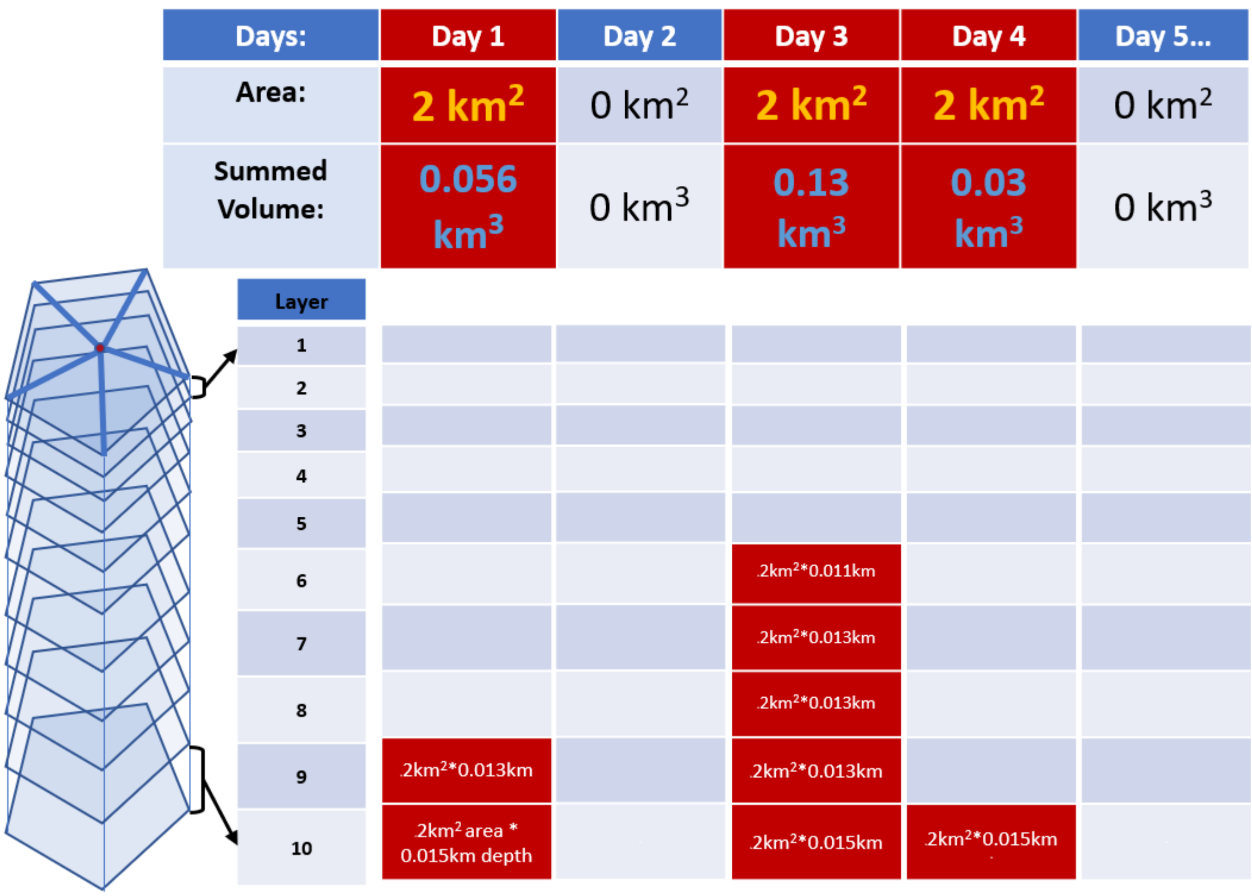


Figure 21. An illustration of different approaches to estimate exceedances of an indicator target (ΔDO>0.2 mg/L) based on area, volume, or time-volume.

In this example, each cell-layer with an exceedance of the target over 24-hour period is indicated in red. Area reporting (km^2) would not discern the differences in extent of exceedances of the different days. Volume reporting (km^3) would provide information on extent of exceedance. Time-volume reporting (km^3 days) would include important information on duration. Table adopted from Mazzilli (2022).

6.2.3 Area, Volume, and Time-Volume based reporting against indicator target.

Comparisons of Existing conditions, or a management scenario, against the Reference condition are first determined by comparing the estimated DO concentration at each cell-layer at each time point for the model domain. The approach used by Ecology in the evaluation of compliance against the Marine Water Quality standards is done on a daily basis where any exceedance in any cell-layer, at any point in the 24-hour daily period, will result in a full day of non-compliance for that cell.

To provide additional detail on the dynamics of model run outputs over time and space, which is appropriate for strategy and scenario evaluation, Puget Sound Institute has extended the approach (Mazzilli, 2022). This includes two additional steps to represent the associated volume for each day of year, and at each depth (cell-layer) in the water column (Figure 21). The first step integrates the volume of each cell-layer that exceed the target threshold (ΔDO>0.2 mg/L) to provide an estimate of the volume of water effected. The second step includes a consideration of time resulting in a time-volume integration. The resulting information, reported in volume days, provides improved detail on the actual proportion an extent of affected water under any given scenario.

Note that this time-volume integration approach is not proposed for, or used by, Washington State for regulatory purposes. However, it is designed to provide additional detail and proportional representation of the temporal and spatial dynamics of model results to support decision-making on broader Puget Sound nutrient reduction goals.

To illustrate the differences between the different reporting approaches, the extent of Washington State waters not meeting the indicator target was determined by area and time-volume. On an area-basis 5% exceeds the indicator target at least once a year. On a time-volume basis the extent not meeting the target is 0.04%. (Table 4). If just the six basins of Puget Sound and the northern bays are considered (highlighted in Figure 22), the area-based and time-volume based exceedances are calculated to be 15% and 0.13%, respectively.

Table 4. Area and time-volume reporting of extent of Washington State waters that does not meet indicator target. Total volume is reported on average annual basis.

Cell/layer/volume	**Exceeds Target**	**Totals**	**%**
Total cell area	320 km^2	6378 km^2	5%
Number of cell-layers	1993	41440	5%
Number of cell-layer days	54,275	14,918,400	0.36%
Total volume	10.7 km^3	563.5 km^3	1.9%
Total volume days	73.6 km^3 days	202860 km^3 days	0.04%

The model output and results are often presented by the major sub-basins of the Puget Sound in order to support reduction in localized impacts and improvements of management scenarios. The individual basins are show in Figure 22. Detailed information on each basin is included in

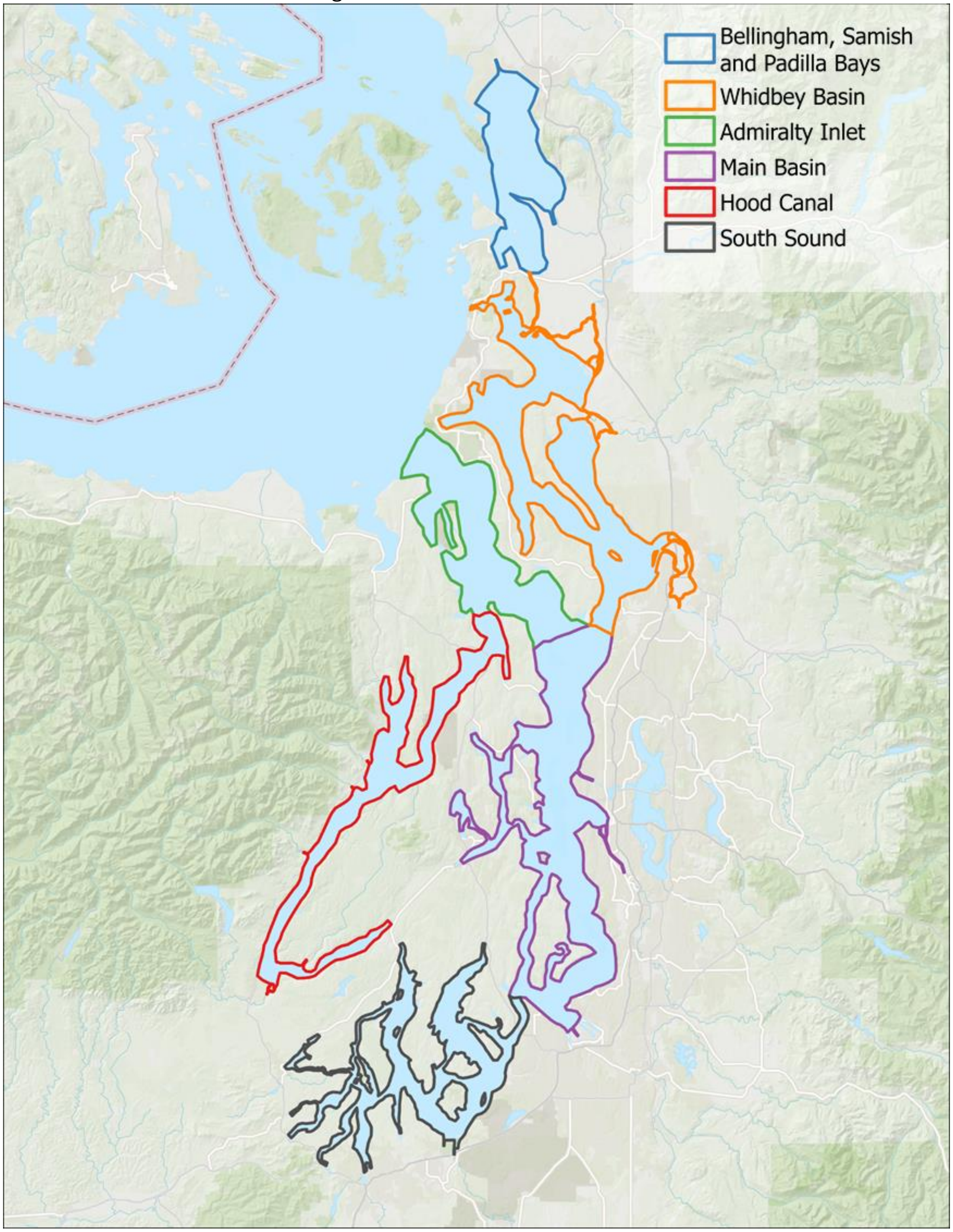


Figure 22. Major basins of Puget Sound and the northern bays (Bellingham, Samish and Padilla) used in condition and scenario evaluations with the Salish Sea Model. This excludes areas of Washington State Waters in the Strait of Juan de Fuca and Strait of Georgia.

Table 5.

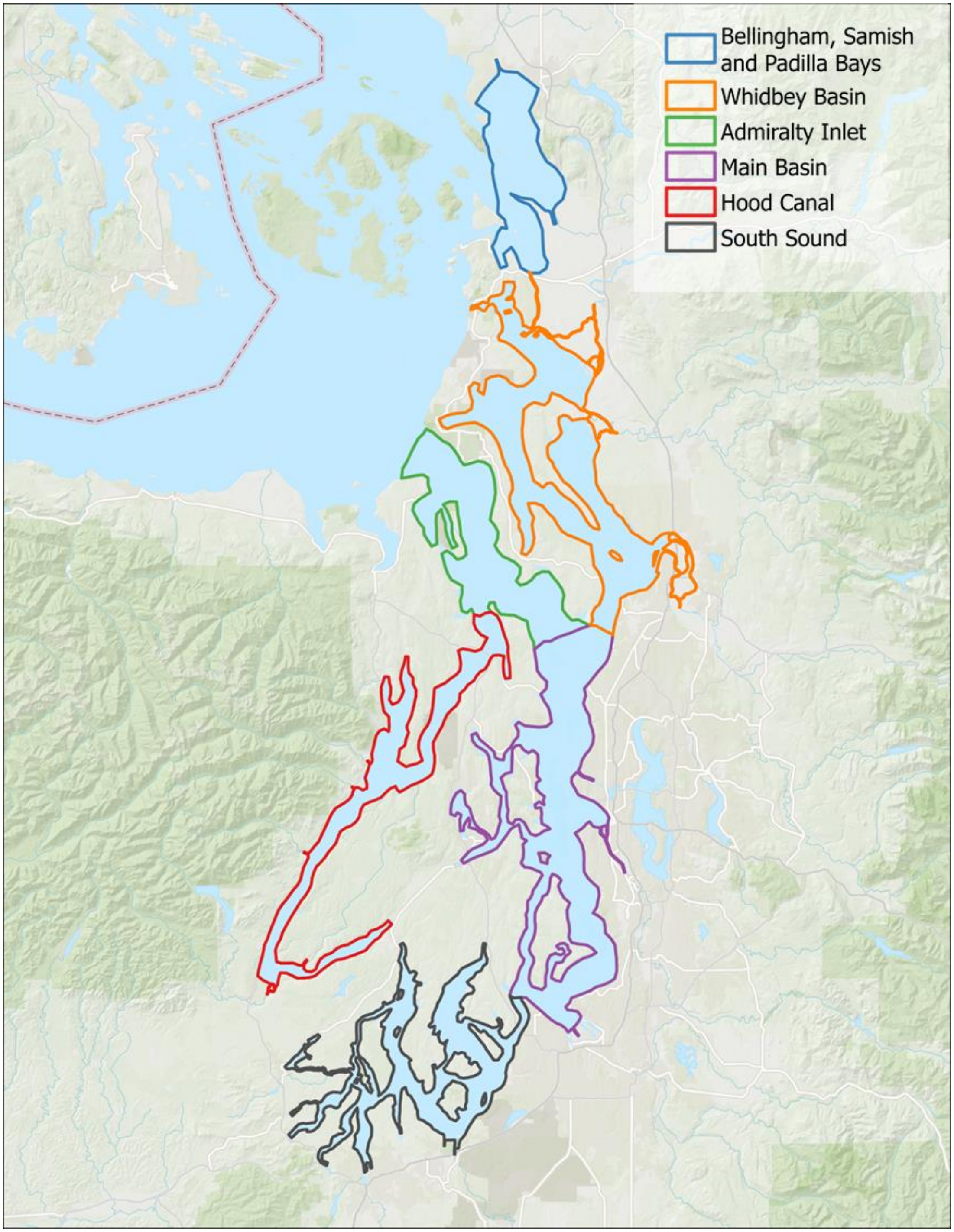


Figure 22. Major basins of Puget Sound and the northern bays (Bellingham, Samish and Padilla) used in condition and scenario evaluations with the Salish Sea Model. This excludes areas of Washington State Waters in the Strait of Juan de Fuca and Strait of Georgia.

Table 5. Model and spatial information of the major basins of Puget Sound.
Information was determined based on V2 script using Ahmed (2019) 2014 dataset. DO target is ΔDO>0.2 mg/L. Extent calculations exclude the masked area of nearshore where SSM performance is poor.

Measure	Admiralty Inlet	Bellingham, Samish, and Padilla Bays	Whidbey Basin	Main Basin	Hood Canal	South Sound	All basins
Total cell. [cells]	319	90	531	893	401	1028	3,262
Total cell-Layer. [cell-layer]	3,190	900	5,310	8930	4,010	10,280	32,620
Total cell layer days [cell-layer days]	1,148,400	324,000	1,911,600	3,214,800	1,443,600	3,700,800	11,743,200
Total volume [km3]	22	3	23	71	20	13	153
Total volume days [km3 days]	7,967	1,231	8,395	25,470	7,164	4,817	55,044

6.3 Salish Sea Model Application - Reference Conditions

6.3.1 Proposed Critical Analysis – Definition of Reference Condition

The dissolved oxygen target specifies that there should be <0.2 mg/L reduction in dissolved oxygen concentrations due to human-related activities. This is determined by comparing the current status of dissolved oxygen (i.e., Existing conditions) with a Reference condition, where the Reference condition is that which existed prior to industrialization settlement (see Section 6.2.1). Since there are no water quality measurements either from the Salish Sea or the rivers and streams from the era prior to human settlement, the Reference condition is defined by an application of the Salish Sea Model using estimated flows and inputs (Ahmed et al., 2019; Mohamedali et al., 2011a; Mohamedali et al., 2011b; Pelletier et al., 2017a). The Interdisciplinary Team, and technical review of this document, indicated that developing confidence intervals associated with the magnitude, extent, and duration of DO depletion was a key uncertainty; without such evaluation it would not be possible to determine whether anthropogenic-related impacts on DO were >0.2 mg/L. Since characterizing the DO depletion is dependent on the Reference conditions, characterizing the uncertainties associated with Reference condition is a priority.

Potential areas of uncertainty include:

- The physical forcings of the Reference condition were assumed to be the same as forcings of the Existing conditions. Forcings at that point in time would have been different due to both short- and long-terms changes in climate and oceanic currents. The uncertainties related to the changes in forcings should be estimated, and the effect on model outputs quantified.
- Nitrogen inputs of the Reference condition are based on measurements from pristine watersheds. There are numerous sources of error in those measurements, which should be quantified to understand the impact on model outputs.

6.4 Proposed Critical Analysis – Uncertainty Analysis of the Salish Sea Model

The Salish Sea Model is the primary tool that is used to understand the impacts of human-associated nutrient inputs on marine water quality of the Salish Sea. The model has been developed and extensively evaluated in order to verify and validate performance at a domain-wide scale for the Salish Sea (Bianucci et al., 2018; Khangaonkar et al., 2017; Khangaonkar et al., 2018; Khangaonkar et al., 2012; Khangaonkar et al., 2011; Kim and Khangaonkar, 2012; Pelletier et al., 2017a, b). The

Interdisciplinary Team prioritized research actions for further model evaluation and sensitivity analysis to quantify and subsequently communicate uncertainty associated with model predictions of the level/extent/duration of low DO in Puget Sound. Technical reviewers have indicated that, based on current understanding of model uncertainty, it is not currently possible to determine whether anthropogenic-related impacts on DO were > 0.2 mg/L. An uncertainty analysis would provide some measure of confidence on model prediction (Mazzilli et al., 2024).

6.4.1 Proposed approach/Status

The primary objectives of this critical analysis are to improve understanding of model-based uncertainties that have been identified during the Marine Water Quality strategy development process, develop confidence intervals in model results, and communicate these findings to the model user community. The proposed actions are consistent with the ongoing model development and evaluation by the Pacific Northwest National Laboratory, Washington State Department of Ecology, and the Salish Sea Modeling Center. Model performance, sensitivity and uncertainty analysis will be undertaken to improve the understanding of the influence of parameter- and process-specific uncertainties on model outputs.

Specific areas of uncertainty were identified during the Implementation Strategy development process (see Section 13), an independent Model Evaluation Group evaluation convened by PSI (Mazzilli et al., 2024), and review comments on the 2022 draft of this State of Knowledge report provided by the Puget Sound Partnership Science Panel. Areas of uncertainty include (adopted from Mazzilli et al. (2024)):

- There is often limited data available to inform model parameters, and so there is associated uncertainty. The parameters should be varied based on actual measurements or supported estimates, and the uncertainty calculated.
- Characterize the potential propagation of error associated with uncertainties in model parametrization, loadings, etc. on model predictions.
- Additional validation studies should be performed specifically in shallow water embayments and Hood Canal, at times of the year when phytoplankton and sediment/water processes have a high impact on oxygen reduction. Preliminary analysis suggests there is larger model error in these areas.
- Perform validation studies using sub-sets of data above/below the pycnocline to better understand model skill related to processes influencing dissolved oxygen, such as vertical mixing, stratification, phytoplankton growth, and water-sediment interactions.
- Analyze model performance for non-calibration years and across multiple years to characterize model skill beyond the three existing, single-year runs.
- Perform further sensitivity scenarios considering model years that are at opposite ends of the spectrum of interannual variability.
- Undertake model performance analysis of Sediment Oxygen Demand (SOD) in embayments. Assessment of seasonal-specific nitrogen and SOD is now possible, as well as validation of related processes/drivers using available measurements of carbon and other fluxes, and estimates of denitrification.

6.5 Proposed Critical Analysis – Strategy evaluations with Salish Sea Model (Strategy 4 – Develop Anthropogenic Nutrient Load Allocations using Modeling Tools)

Five strategies were identified that, if implemented, should result in an improved understanding of the marine condition and the effects of nutrient management activities on selected sources, and a reduction in human-associated nutrient loading to the Salish Sea. The Interdisciplinary Team indicated that a priority action is to evaluate the impacts of each of these strategies, particularly those associated with managing nutrient from selected sources (i.e., wastewater, stormwater, or agricultural runoff) in selected locations on the marine condition and the ability to achieve water quality targets. Some progress on this evaluation specific to the Implementation Strategies has been made by the Salish Sea Modeling Center and are summarized in Sections 8, 9, and 10. The Washington State Department of Ecology has an ongoing program of running scenarios as part of the nutrient reduction program of Puget Sound, and the most recent results from are summarized in Ahmed et al. (2021).

Additional model runs have been prioritized by the Interdisciplinary Team. A key aspect of this analysis is the development of anthropogenic load allocations; information that is critical in understanding the relation between sources and water quality impacts or impairments.

6.5.1 Proposed approach/Status

A suite of SSM simulations will be performed to characterize the effects of specific management scenarios such as (for example) the impacts of nutrient reductions at selected wastewater treatment systems, or the implementation of nutrient control programs, on regional water quality. The scenarios will be identified by regional stakeholders including, to the extent possible, the members of the Interdisciplinary Team. Scenarios will then be run by the University of Washington Salish Sea Modeling Center. To date, analysis has been undertaken in three studies expanding on the approach applied in Ahmed et al. (2021) to further examine the sensitivity of load reductions on water quality. Studies of the Whidbey region (Mazzilli et al., 2023) and Straits of Georgia and northern bays (Baker et al., 2023) quantified the relative sensitivity of the volume change of low dissolved oxygen within each region for different management scenarios including reductions at different wastewater treatment plants (WWTP), as well as the influence of rivers. For the Whidbey region, a theoretical scenario removing all WWTP loads reduced the regional number of volume-days where $\Delta DO>0.2 mg/L$ from 3% to 1%. In further scenario analysis reductions were generally proportional to the size of the discharge removed. However, the position of outfalls (in relation to depth and proximity to riverine influence) potentially influenced the relative reduction in DO for the larger WWTPs. (Khangaonkar and Yun, 2023) investigated the impact of depth and position of WWTP outfalls. When outfalls were moved to deeper waters, the authors found that impacts on DO were reduced only for those basins without sills. In addition, this scenario which resulted in a 22% increase in freshwater discharge to bottom waters, resulted in a 4% reduction in Puget Sound exchange flow.

6.6 Linking Salish Sea Model with Land-Use and Ecosystem Models

The Puget Sound region has a suite of cutting-edge terrestrial, estuarine, and marine ecosystem models that have on their own been useful in supporting decisions and advancing understanding of their respective systems. All models, however, have a limited domain in that they focus on specific processes and geographies. This applies equally to the models that are used in the assessment of conditions and scenarios associated with the sources, management, and impacts of anthropogenic nutrients in the Salish Sea. Individually, these models can only address a limited range of ecosystem-based management priorities and scenarios. However, a coupled environmental and human systems modeling framework has been proposed which would support the evaluation a much larger, and more holistic, ecosystem domain (McKane et al., 2017).

Linking these models would provide a platform for gaining insights into the connected systems of human use, terrestrial hydrology, estuarine-ocean circulation and biogeochemistry, and marine food webs. If implemented, the framework would (from McKane et al. (2017)):

- Synthesize existing scientific data for Puget Sound's terrestrial, estuarine and ocean environments.
- Quantify how positive or negative actions in one location propagate through the upland-floodplain-estuarine-ocean system and impact abiotic and biotic conditions in other locations.
- Link environmental models to economic and social science models to explore near and long-term outcomes of alternative planning.
- Identify tradeoffs among multiple objectives (ecological, economic, social and human health).

This work is currently going forward under a project lead by the UW Puget Sound Institute and funded by the Paul G Allen Family Foundation (https://www.pugetsoundinstitute.org/collaboration/puget-sound-integrated-modeling-framework/). This project will bring together teams representing USEPA VELMA, the UW Salish Sea Modeling Center, NOAA Atlantis, and social-ecological systems and a high-resolution land cover change models of the UW Puget Sound Institute. The project is scoped for three years. The three existing component models will be linked within a terrestrial-marine-human system framework that decision-makers can use to reliably inform recovery planning for the Salish Sea. Model linkages could address simulated exchanges of water, carbon, nutrients, contaminants, sediments and organisms across terrestrial-estuarine-ocean boundaries.

The linked set of models will be parameterized for the entire Puget Sound basin and its watersheds, delivering a tool with the ability to support regional planning and restoration decision-making under alternative climate, population, and land use futures for a broad range of regional objectives, from orcas to human wellbeing.

7 OTHER ASSESSMENT TOOLS

The Salish Sea Model is currently the only model used in the evaluation of marine water condition under the Marine Water Quality Vital Sign. However, other modeling and assessment tools are available that can help inform the impact of management activities on, for example, watershed loads and marine water quality. These are summarized below.

The Puget Sound Integrated Modeling Framework is developing a connected terrestrial-freshwater-marine-human system modeling framework to help understand how future conditions and management actions will affect the interconnected natural and human systems. The project will link a high-resolution **Error! Hyperlink reference not valid.**, VELMA, the Salish Sea Model, the Atlantis ecosystem model, and a qualitative socio-ecological model.

7.1 MARINE HYDRODYNAMIC AND WATER QUALITY MODELS

7.1.1 LiveOcean Model

The LiveOcean Model is a computer simulation used to predict ocean water properties in the NE Pacific and Salish Sea. The model provides 3-7 day forecasts of circulation and water chemistry condition, including Aragonite saturation state and pH. The model is based on the Regional Ocean Modeling System (ROMS), a free-surface, hydrostatic, primitive equation model that has been used extensively in coastal and estuarine systems. It has been utilized in the evaluation of estuarine circulation (MacCready et al., 2021; Sutherland et al., 2011), effects of local and remote forcings on exchange flow (Giddings and MacCready, 2017), an evaluation of transport pathways of Harmful Algal Blooms (Giddings et al., 2014), and nutrient transport and impacts on phytoplankton growth along the NE Pacific coastal shelf (Davis et al., 2014). While the LiveOcean model has the capability to predict dissolved oxygen condition throughout the Puget Sound (https://faculty.washington.edu/pmacc/LO/oxygen_year.html). Performance was evaluated in MacCready et al. (2021). Updates are currently underway to eventually support the evaluation of different management strategies for nutrient reduction. Watershed Models

There are currently two models that have been developed to estimate hydrodynamics and contaminant loading from individual watersheds throughout the Puget Sound watershed.

7.1.2 SPARROW Model

SPARROW (SPAtially Referenced Regressions On Watershed attributes) models predict contaminant flux, concentration, and yield in streams based on field monitoring data and a suite of watershed attributes (Schwarz et al., 2006). Applications utilize nonlinear regression to describe the non-conservative transport of contaminants from point and diffuse sources on land to rivers, and through the stream and river network. The models describe the distribution of loads and can be used to help understand the origin and transport of water-quality constituents. Smith et al. (2003) developed an application of SPARROW to predict mean annual total nitrogen and total phosphorus loading across 14 ecoregions, including the Western Forested Mountains, throughout the United States. Wise and Johnson. (2013) used SPARROW to estimate mean annual surface-water total nitrogen and total phosphorus yields and sources in catchments of the Pacific Northwest. Forestland provides the highest overall proportion of total nitrogen, mainly because it made up the largest proportion of overall catchment area. Urban sources and farm fertilizer were predicted to have the highest mean annual yield. Saad et al. (2019) utilized SPARROW to provide long term annual mean loadings of catchments throughout the United States, including several areas within the Pacific Northwest. Information and data are available at the USUS on-line portal (https://sparrow.wim.usgs.gov/sparrow-pacific-2012/).

McCarthy (2019b) compared SPARROW results with nutrient load estimates used as inputs for the Salish Sea Model for the 2002 model year. SPARROW reports results as an annual load (kg/yr) while SSM utilizes a continuous daily time series of nutrient load inputs; the comparison can be done by summing the SSM inputs over the entire year. For the entire Puget Sound basin, the SPARROW results (25.45 million kg/yr) were comparable, though slightly higher than Salish Sea Model nutrient inputs (25.43 million kg/yr). There were basin-scale differences, with the largest difference occurring in the Main basin which was mainly attributed to the way marine point source loads (i.e., WWTP effluents) were accounted for in each system.

The United States Geological Survey and the Washington State Department of Ecology are collaborating on a project to utilize SPARROW for the development of refined, seasonal loading estimates of total nitrogen and total phosphorus within Puget Sound watersheds for the period 2005-2020 (United States Geological Survey, 2022). The estimated loads will be used to improve the understanding of the influence of watershed contributions of nutrients upstream of their discharge points to marine waters. A Quality Assurance Project Plan contains further details about the technical approach, observational data, spatial and temporal source data, limitations, procedures that will be employed in this study (Figueroa-Kaminsky et al., 2022).

7.1.3 VELMA Model

The VELMA (Visualizing Ecosystem Land Management Assessments) model provides information on performance of green infrastructure to help evaluate options for controlling the fate and transport of water, nutrients, and toxics across multiple spatial and temporal scales, for different ecoregions, and present- and future-climate conditions. Abdelnour et al. (2011) describe the model development and application, focusing on an evaluation of multiple forest-harvest scenarios on catchment hydrology, demonstrating that both the extent and location of harvest matter for streamflow response. The application was expanded to evaluate the effects of harvest on carbon and nitrogen dynamics within the same modeled watershed (Abdelnour et al., 2013). VELMA accurately captured observed changes in carbon and nitrogen dynamics before and after harvest. Results suggest that exports of dissolved nutrients from the preharvest old-growth forest were generally low; and that carbon and nitrogen losses from the terrestrial system increased markedly following harvest.

VELMA has been utilized to explore changes in watershed hydrological response due to the implementation of Low Impact Development (LID; rain gardens, permeable pavement, and riparian buffers) in a watershed with diverse land use (Hoghooghi et al., 2018). A suite of scenarios covering different spatial configurations, a range of extent-of-implementation, and LID designs (e.g., different widths of riparian buffer along streams) were evaluated. Model simulation results suggest reductions in peak flows and surface runoff, and increases in evapotranspiration and subsurface flow and infiltration, with all spatial configurations of LID at the watershed scale. The results further indicated that hydrological changes from the LID implementation were modest, which suggests a potentially limited efficacy of LID practices in mixed land cover watersheds.

Barnhart et al. (2021) utilized VELMA to evaluate the hydrological impacts of green roof implementation in four urban watersheds in Seattle, WA. Results indicated that complete green roof implementation could result in 10-25% mean annual runoff reduction depending on design, which would be the upper limit of volume reductions achievable trough this approach. They also demonstrated that runoff reductions were proportionally smaller during the rainy season, as green roofs have limited storage capacity and can become saturated during large or continuing rainfall events.

Hall et al. (2018) reported on the application of the VELMA model to evaluate the impacts and outcomes of a set of forest management practices in the Nisqually Community Forest on a suite of diverse objectives including thriving salmon populations, forest products supporting local forest sector jobs, clean drinking water, carbon sequestration, recreational and cultural opportunities, and tourism. The effort requires the coupling a suite of models including VELMA (for ecohydrology), Penumbra (for stream temperatures, and EDT (fish habitat) to address the management questions.

7.1.4 National Water Model

The NOAA National Water Model (NWM) provides water flow forecasts and predictions to over 2.7 million streams and rivers throughout the United States. The core of the NWM system is the National Center for Atmospheric Research (NCAR)-supported community Weather Research and Forecasting Hydrologic model (WRF-Hydro) which utilizes forcings and data from multiple sources including: precipitation information from the Multi-Radar/Muti-Sensor System (MRMS) and Stage IV Multisensor Precipitation Estimator (MPE); forecasting information from the High Resolution Rapid Refresh (HRRR), Rapid Refresh (RAP), North American Mesoscale Nest (NAM-Nest), Global Forecasting System (GFS) and Climate Forecast System (CFS) Numerical Weather Prediction (NWP), and; the Noah-MP Land Surface Model (LSM) to simulate land surface processes. Outputs from the NWM are used to provide hindcast streamflow information for the Salish Sea region, which can be incorporated into the Salish Sea Model to provide operational feeds of river flows. The scale of the NWM does not accurately predict flows in all systems in the Puget Sound watershed and so its regional application is limited.

7.1.5 Proposed Critical Analysis – Compare loading inputs and estimates from different model sources

Watershed nutrient loads to the Salish Sea are currently determined based on a suite of measurements and extrapolated to the unmeasured watersheds based on land use characterizations (Ahmed et al., 2021). Watershed models are not currently utilized to estimate and/or refine nutrient inputs but could be useful to: 1) improve estimates for nutrient inputs for watersheds with limited data, 2) improve the understanding of relative contributions from different sources in watersheds; 3) allow the estimation of future watershed-based nutrient loads based on different growth scenarios, and 4) support the evaluation of policy and program effectiveness and source reduction strategies to reduce nutrient loads from watersheds.

The development, evaluation, and use of regional watershed nutrient models to inform Salish Sea Model has been identified as a priority action (Marine Water Quality-Result Chain4.2/Evidence Based Solution [EBS] 31; Puget Sound Partnership (2020)) by the Interdisciplinary Team during the Implementation Strategy development process. Specific recommendations were to:

- Evaluate the relative magnitude of N loads for all priority sources and evaluate source reduction strategies to adaptively manage Marine Water Quality strategies (IS-Marine Water Quality 70)

- Evaluate watershed N load with population growth and land development, and the range of uncertainties in these estimates? (IS-Marine Water Quality 49)

7.1.5.1 Proposed approach/Status

As part of Ecology's Puget Sound Nutrient Source Reduction Process, Ecology and the USGS are collaborating on the development of a new SPARROW model for the Puget Sound region. At the time of publication, Ecology was scoping a watershed nutrient strategy that will use the new Puget Sound SPARROW model and other watershed data and tools to conduct a regional watershed nutrient source assessment to understand what actions are needed to reduce watershed loads and contribute to meeting marine DO standards (Dustin Bilhimer, personal communication, 16 May 2022). The new SPARROW model will include all watersheds discharging to the Washington waters of the Salish Sea and will improve upon other SPARROW models by estimating seasonal nutrient loads rather than just annual. Ecology is coordinating with local implementation groups and other state and local agencies to update water quality, land use, and implementation activity datasets.

7.2 Atlantis Model

The Atlantis model is a marine food web and species management model that can help explore the effects of fisheries, predation, climate, pollution, salinity, etc. on marine habitat and species across trophic levels. Ecosystem dynamics are represented by spatially explicit submodels that simulate oceanographic processes, biogeochemical factors that drive primary production, and food web relations. The model provides information of marine species biomass and population dynamics for major functional groups such as bacteria, eelgrass, Pacific herring, Chinook salmon, and Orcas. The model is modular and thus supports the inclusion of socioeconomic drivers in addition to the biological and environmental drivers and hydrodynamic forcing (Audzijonyte et al., 2019). Atlantis has been used in over 30 systems throughout the world, including the Strait of Georgia and Puget Sound.

The Atlantis Model for Puget Sound (Morzaria-Luna et al., 2022) is linked to the Regional Ocean Modeling System model for Puget Sound to force temperature and salinity fluxes. The food web model includes 73 functional groups, including salmon (21 groups), demersal fish (9), pelagic fish (1), forage fish (3), elasmobranchs (4), seabirds (3), mammals (7), zooplankton (4), primary producers (4), invertebrates (10), bacteria (2), and detritus groups (3). Historical abundance and catch data were used to establish biomass, catch, and effort trends in Puget Sound. The model was tested for its ability to represent historical fishing pressure from 2011 to 2017; results suggest that it can reasonably approximate historical spatial distribution and abundance for most functional groups and fisheries in the region. The Atlantis Model for Puget Sound is suitable for applications such as evaluating ecosystem indicators, assessing the effects of climate change, and identifying trade-offs associated with alternative management scenarios.

8 EXISTING CONDITIONS

The Salish Sea Model has been used to estimate the water quality impacts of human-related nutrient inputs to the Salish Sea. These results have been extensively reported by the Washington State Department of Ecology in terms of impairments (Ahmed et al., 2019), where an impairment is defined as a point the falls outside of water quality standards.

The Salish Sea Modeling Center performed a similar evaluation to describe changes in dissolved oxygen between Existing and Reference conditions based on current measured or estimated nitrogen loadings from WWTPs (marine point sources) and rivers and streams (watersheds). A selection of the results are presented in this section.

Note that for the SSMC runs, initialization files for the year 2014 were used following Ahmed et al., (2019) accessed from the Ecology website, and masking and exclusion of near shore cells was applied, and sub-basin analysis followed the same delineation. Methods for data processing are described in Section 6 and in Mazzilli et al., (2022).

8.1 Temporal Variation

The temporal distribution of occurrences of ΔDO>0.2 mg/L is not uniform throughout the year. The greatest differences between Existing conditions and Reference condition are observed in late-summer or early-fall. Ahmed (2019) modeled the seasonal volume of hypoxic (DO<2 mg/L) and low DO (2mg/L to 3 mg/L) water for Existing and Reference conditions (Figure 23). Seasonal hypoxia occurred in July-December, with a peak hypoxic volume in September with a maximum of 2.97 km^3 for Existing conditions in 2006 (0.2% of the entire domain volume). The difference in hypoxic volume between Existing and Reference conditions was calculated to be 28% for the 2006 model year. This is represented by the difference between the blue and orange lines in Figure 23, indicating the duration and timing of low DO events.

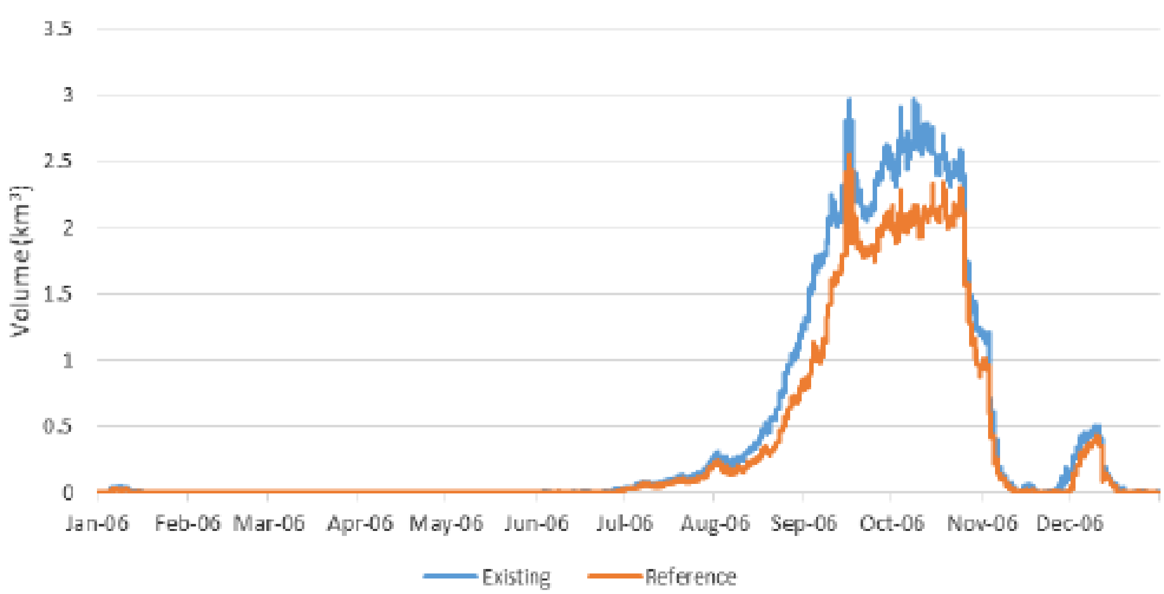


Figure 23. Estimation of hypoxic (DO<2 mg/L) volume for Existing and Reference scenarios for the2006 model year (Figure 38, Ahmed 2019).

The Salish Sea Modeling Center performed a similar analysis for a single cell-layer. To do so, the daily minimum DO concentrations, and difference in minimum DO concentrations between Existing and Reference conditions, was estimated for the 2014 model year. One cell-layer was selected as an illustrative worst-case example of anthropogenic impacts as it is located in the bottom layer (deep waters are more affected by eutrophication than shallow layers) and an embayment (embayments are more affected by eutrophication than open waters). Results are shown in Figure 24 and Figure 25. The figures show a representative annual pattern of dissolved oxygen in the bottom layer of an enclosed embayment demonstrating absolute concentration over the course of the year, and also the magnitude of difference that is likely attributable to anthropogenic nitrogen inputs into the Puget Sound.

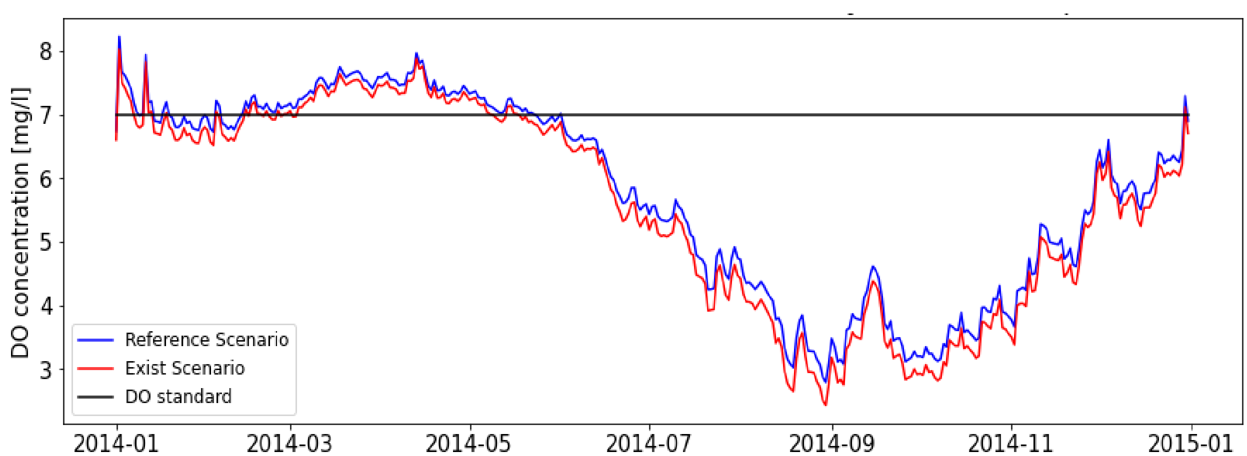


Figure 24. Daily minimum DO concentration of the Existing and Reference scenarios at one cell-layer in one location for the year 2014.

The x-axis presents model year and month (YYYY-MM). The water quality standard for this location is shown for reference. The data is from the bottom layer in a cell-layer in Quarter master harbor which had a depth of 15m (cell node id: 13549, 10th cell-layer).

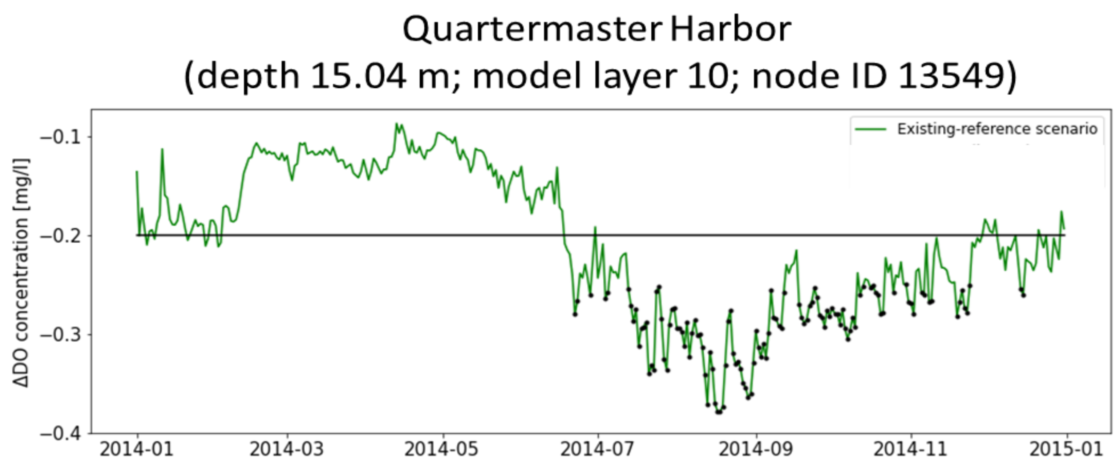


Figure 25. The difference in daily minimum DO concentration between the Existing and Reference scenarios at one cell-layer in one location for the year 2014. The Marine Water Quality indicator target is shown (black line).

These results also provide insight on the maximum potential improvement in seasonal low DO concentration and persistence at this location. As shown in Figure 25, the maximum improvement for this location would be on the order of 0.3-0.4 mg/L in the late-summer and fall. This comparison illustrates outcomes if all anthropogenic nitrogen loading to Puget Sound were stopped or diverted. As it is not technologically feasible to completely eliminate nutrient from wastewater treatment system effluent, there are no scenarios which propose a complete elimination of anthropogenic-related nutrient loadings to Puget Sound. However, these results are illustrative of potential water quality improvements in the most sensitive areas of Puget Sound; as the dataset is drawn from the bottom layer in one of the shallow embayment and terminal inlets most impacted by anthropogenic loadings.

8.2 Spatial Variation

Differences in basin specific results highlight how the geometry, hydrodynamics and connectivity of each sub-system may influence its capacity to respond to changes in nutrient loading.

For example, based on estimates of extent of each of the major sub-basins that did not meet indicator target of ΔDO<0.2 mg/L (Table 6), the South Sound has the highest exceedance by volume days totaling 43.12 km^3 days, likely driven by the relatively high proportion of shallow waters and terminal inlets and most landward location at the end of Puget sound, while still being proximate to major N input sources. This represents an exceedance of 0.90% of the total for South Sound basin volume of 4,817 km^3. Conversely, the Main Basin and Admiralty Inlet have less than 0.1% volume days not meeting indicator targets, with 1.85 and 0 km^3 days respectively. These results are likely influenced by the relatively deeper waters in the Main basin, the connectivity of Admiralty Inlet to the open ocean, and the prevailing circulation patterns throughout the sound. Further application of this analysis to specific shallow water embayments, and times of the year, would provide more detailed information on how propose scenarios can improve identified impacted zones of Puget Sound.

Table 6. Modelled extent of six major sub-basins in Puget Sound where ΔDO>0.2mg/L between a given scenario e.g., Existing, versus Reference conditions over the 2014 model year.
The scenarios are (row 1-3): Existing – nitrogen loading from WWTPs and watershed is at current estimates; WWTP-3 mg/L – total nitrogen concentration of all WWTP = 3 mg/L and watershed loads at existing conditions; River management – nitrogen concentration from all rivers/streams set at half-way between Existing and Reference conditions, and WWTP loads at Existing conditions. All values in km^3 days. Total km^3 days for each sub-basin shown for comparisons (row 4).

Scenario	Bellingham, Samish, and Padilla Bays	Whidbey Basin	Admiralty Inlet	Main Basin	Hood Canal	South Sound
Existing	0.77	24.40	0.00	1.85	3.42	43.12
WWTP-3mg/L	0.18	0.00	0.00	0.03	0.38	1.18
River management	0.00	5.41	0.00	1.20	1.21	18.95
Total sub-basin volume-days	1,231	8,395	7,967	25,470	7,164	4,817

Khangaonkar et al. (2018) modeled areas of hypoxia (DO <2 mg/L) in Puget Sound based on Existing conditions for the 2014 model year. The majority of hypoxic volume was within the Hood Canal, as well as sub basins and landward ends of inlets with longer residence times, based on the extent of bottom-layer hypoxic volume (Figure 26).

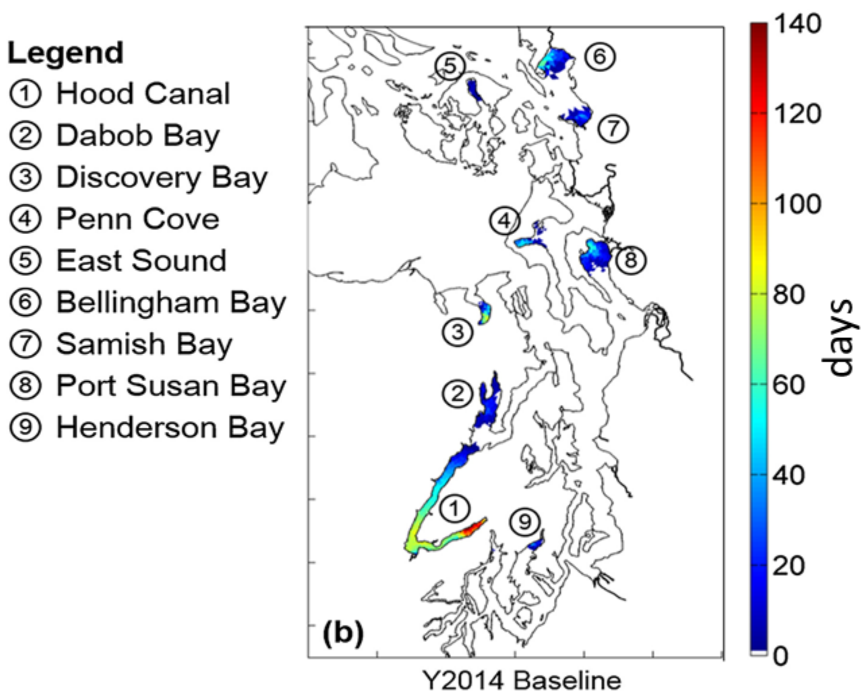


Figure 26. Duration (days) and extent of hypoxic bottom-layer waters of Puget Sound (DO<2mg/l) for an Existing conditions scenario based on 2014 model year.
Bottom layer is the lower 15% of water column. Reproduced from Khangaonkar et al. (2018).

Additional characterization of temporal and spatial distributions of the difference between Existing conditions and Reference conditions are shown for the Puget Sound (Figure 27) and for the six major basins of Puget Sound (Figures 28-32). The calculated differences in volume, shown in volume-days, are shown in Table 6.

The results suggest the depth distribution of areas with a ΔDO>0.2 mg/L is not uniform. The greatest volume in each of the sub-basin occur in the bottom layer.

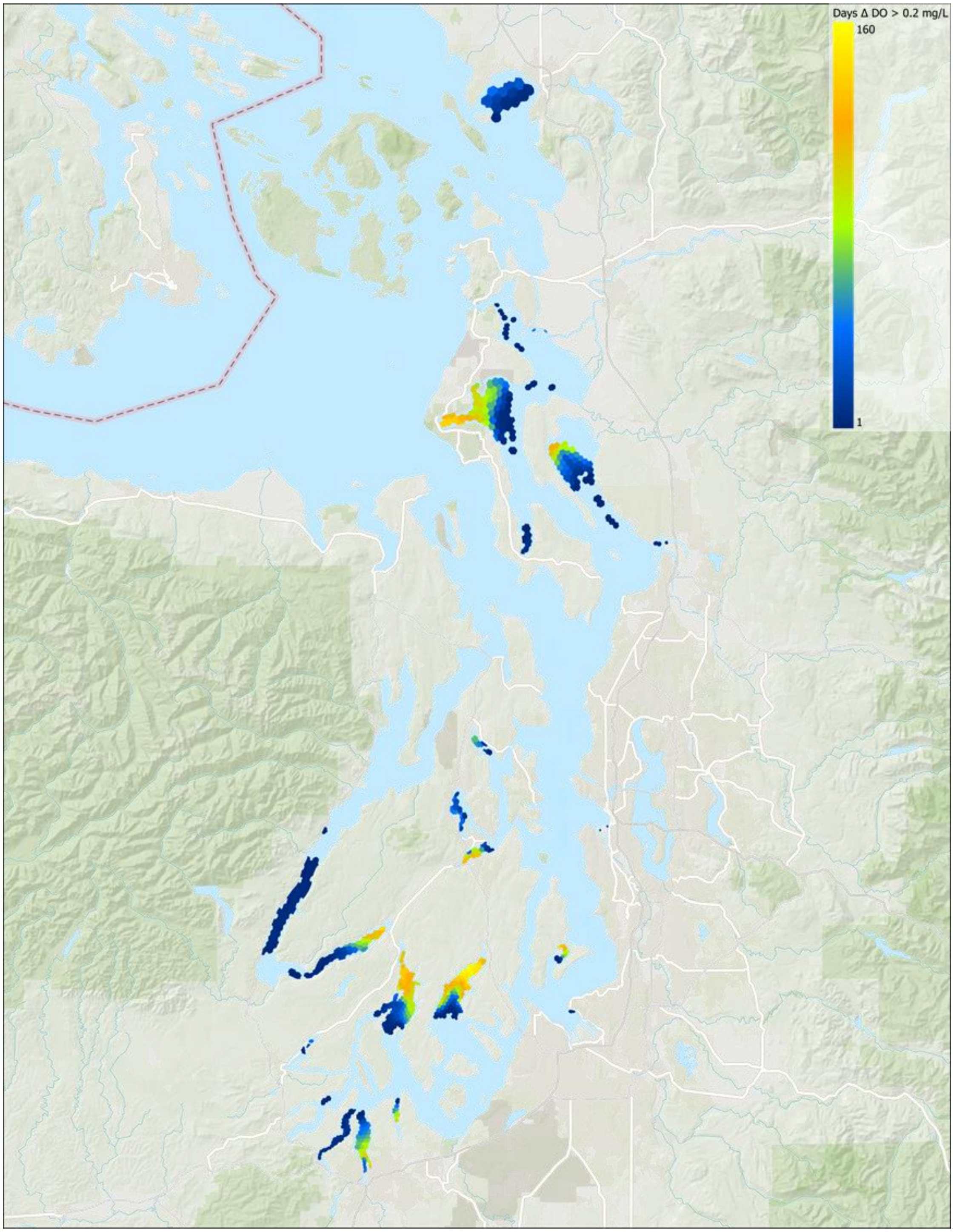
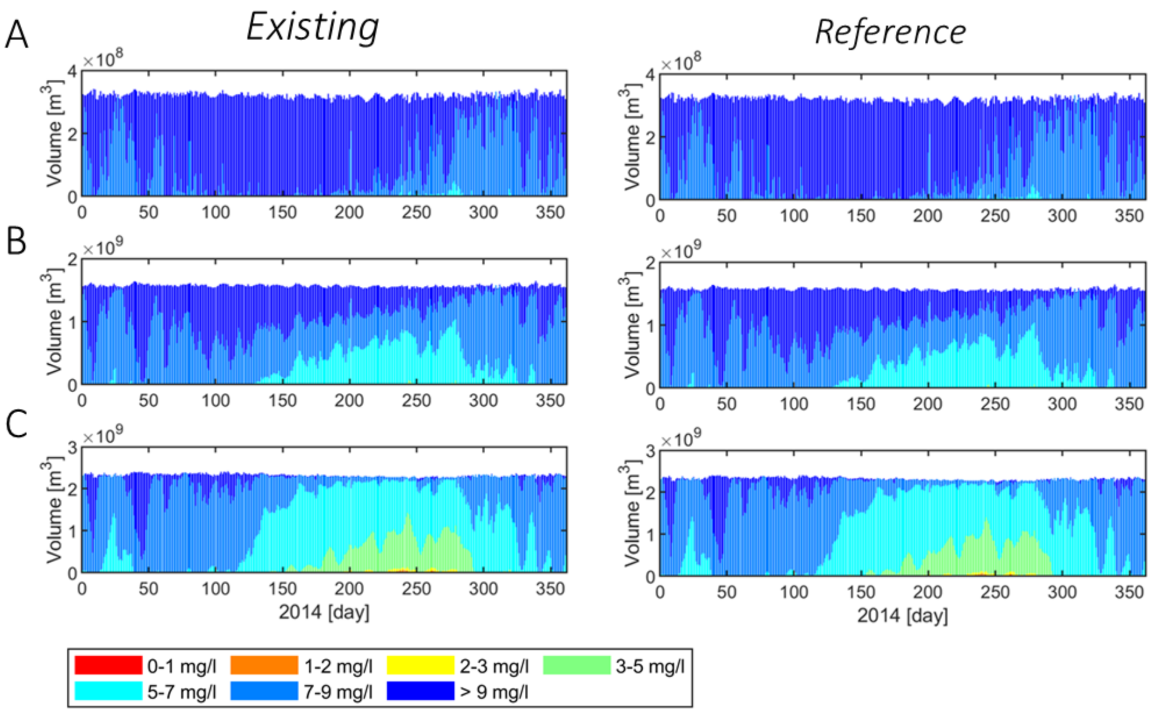


Figure 27. Summary of areas in Puget Sound where ***a comparison of Existing vs Reference conditions indicates that ΔDO>0.2 mg/L.***
Results based on two runs from the Salish Sea Model. These areas are those that are predicted not to meet PSP indicator target. Color indicates number of days that ΔDO>0.2 mg/L for 2014 model year.

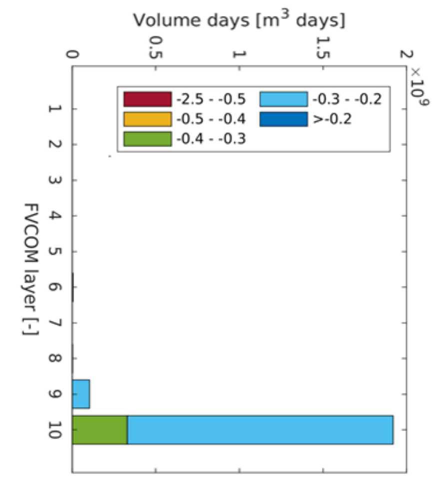
Bellingham, Samish, and Padilla Bays

Modeled Condition
(mg/L dissolved oxygen)



Model Layers: A. 1-2, B. 3-6, and C. 7-10

Annualized Difference in Condition by Layer
(m^3 days)



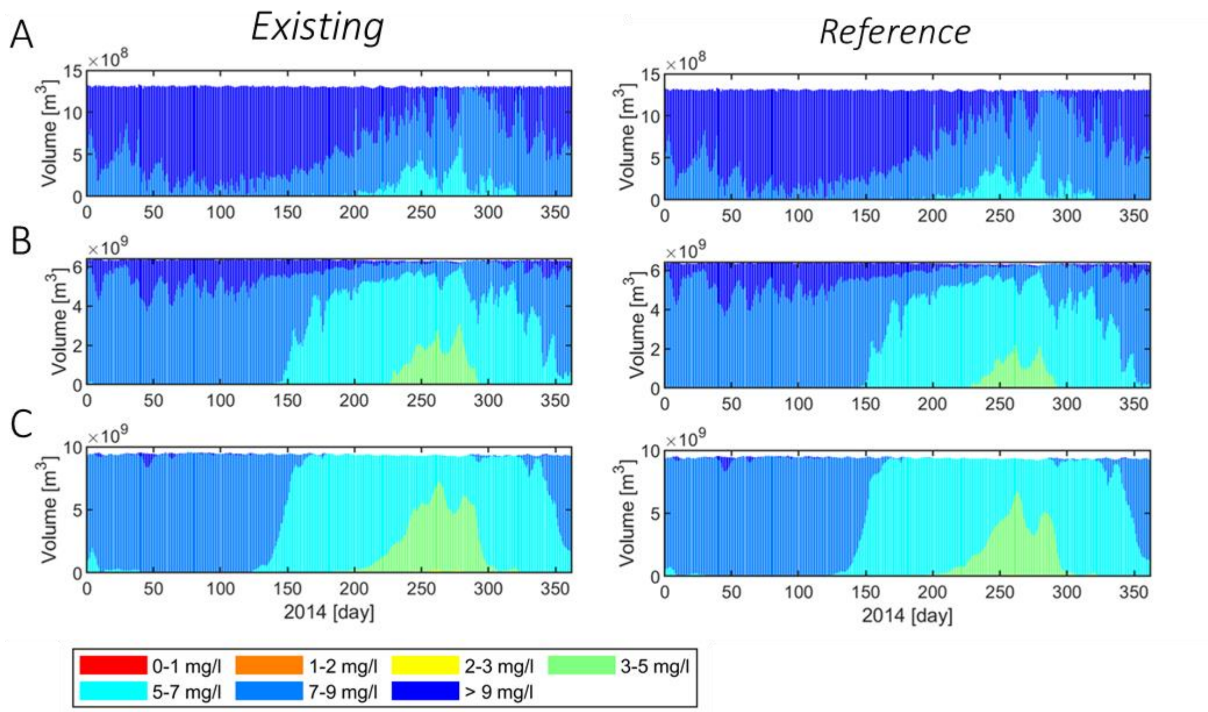
Note: Total volume-days for Strait of Georgia and Northern Bays is 1.32×10^{12} m^3 days

Figure 28. Modeled dissolved oxygen concentration of Bellingham, Samish, and Padilla Bays comparing Existing versus Reference conditions scenarios presented over different model layers.

Top model layers (1-2) are in A, middles model layers (3-6) are in B, and bottom model layers (7-10) are in C. Note volume scale varies between model layer sets. The figure to the right presents the annualized difference in modeled condition is presented in total volume-days (m^3 days). Volume-days are calculated by summing the volume of water with ΔDO>0.2 mg/L in a given basin for each day of the model run. Estimates based on 2014 model year.

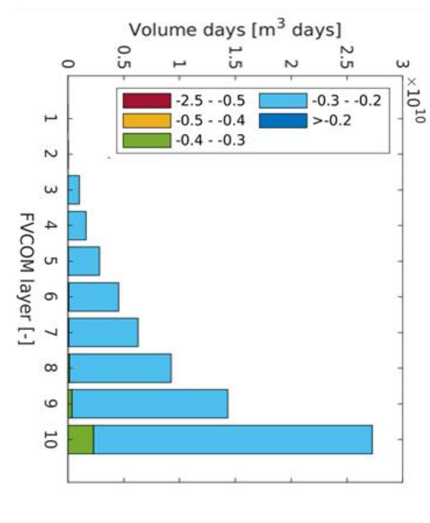
Whidbey Basin

Modeled Condition
(mg/L dissolved oxygen)



Model Layers: A. 1-2, B. 3-6, and C. 7-10

Annualized Difference in Condition by Layer
(m^3 days)



Note: Total volume-days for Whidbey Basin is 8.65x10^{12} m^3 days

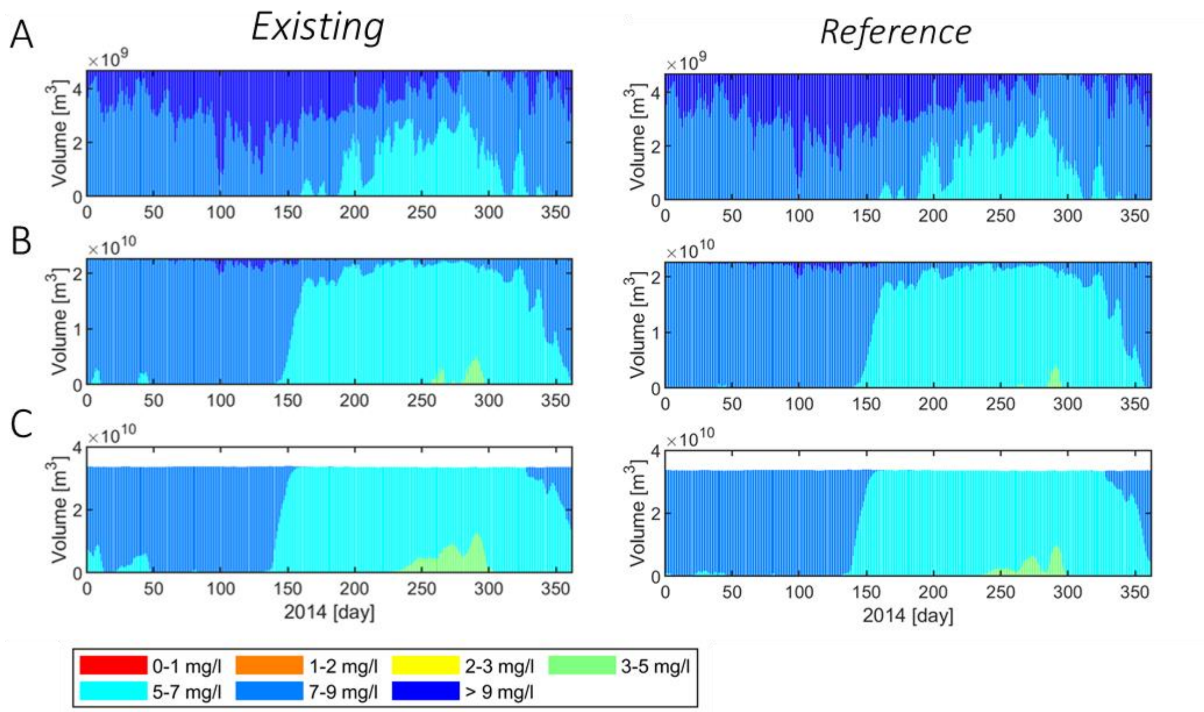
Figure 29. Modeled dissolved oxygen concentration of Whidbey basin comparing Existing versus Reference conditions scenarios presented over different model layers.

Top model layers (1-2) are in A, middles model layers (3-6) are in B, and bottom model layers (7-10) are in C. Note volume scale varies between model layer sets. The figure to the right presents the annualized difference in modeled condition is presented in total volume-days (m^3 days). Volume-days are calculated by summing the volume of water with ΔDO>0.2 mg/L in a given basin for each day of the model run. Estimates based on 2014 model year.

Main Basin

Modeled Condition

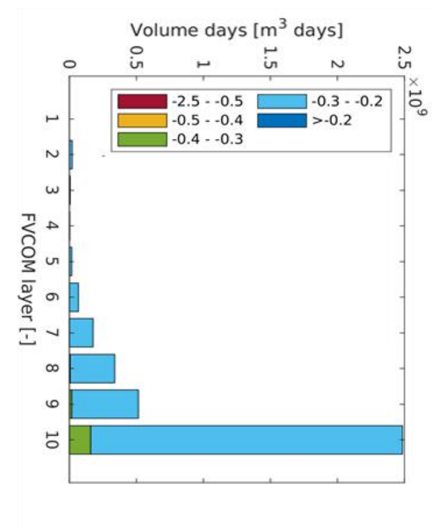
(mg/L dissolved oxygen)



Model Layers: A. 1-2, B. 3-6, and C. 7-10

Annualized Difference in Condition by Layer

(m^3 days)



Note: Total volume-days for Main Basin is 2.52×10^{13} m^3 days

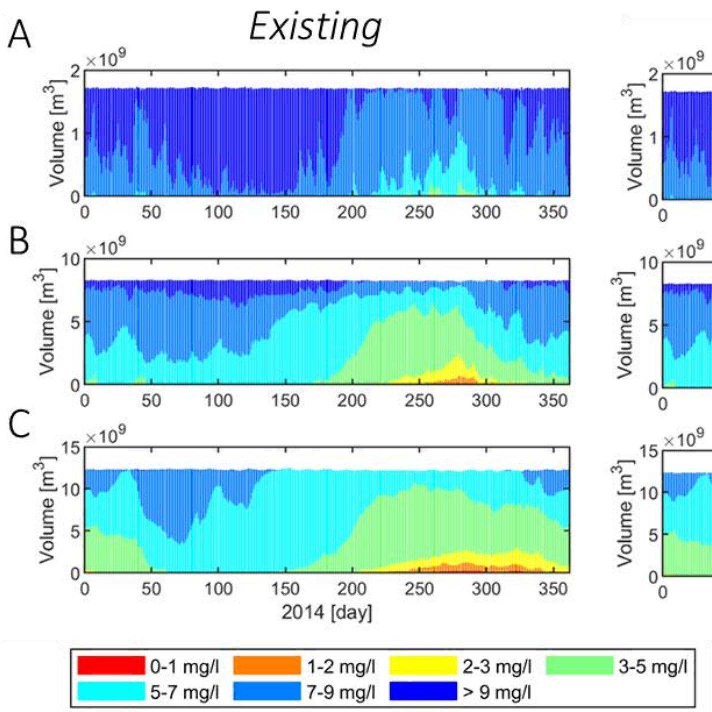
Figure 30. Modeled dissolved oxygen concentration of Main Basin comparing Existing versus Reference conditions scenarios presented over different model layers.

Top model layers (1-2) are in A, middles model layers (3-6) are in B, and bottom model layers (7-10) are in C. Note volume scale varies between model layer sets. The figure to the right presents the annualized difference in modeled condition is presented in total volume-days (m^3 days). Volume-days are calculated by summing the volume of water with ΔDO>0.2 mg/L in a given basin for each day of the model run. Estimates based on 2014 model year.

Hood Canal

Modeled Condition

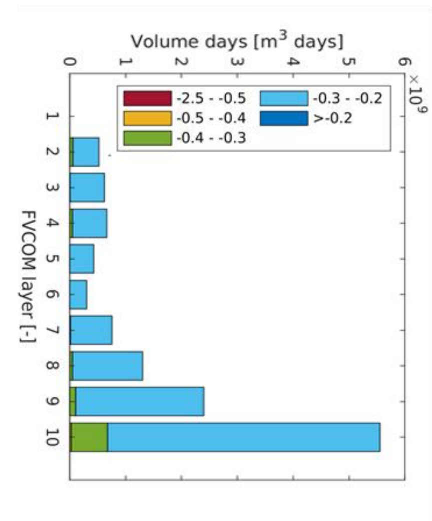
(mg/L dissolved oxygen)



Model Layers: A. 1-2, B. 3-6, and C. 7-10

Annualized Difference in Condition by Layer

(m^3 days)



Note: Total volume-days for Hood Canal is 7.32×10^{12} m^3 days

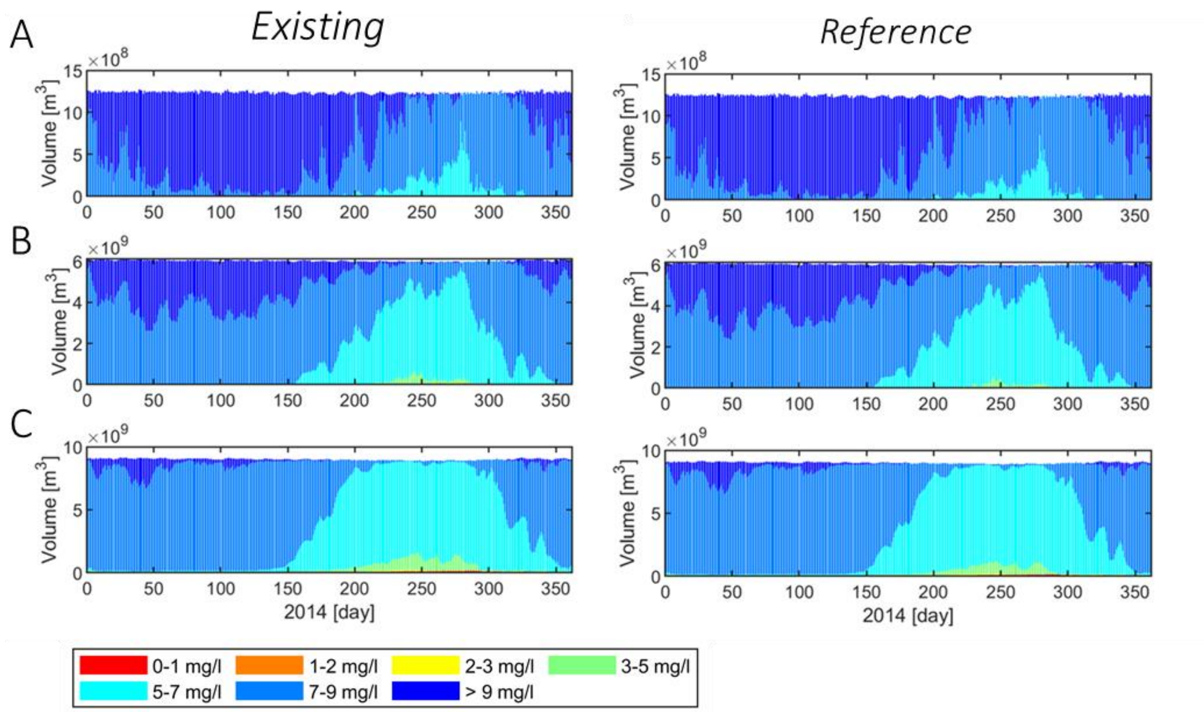
Figure 31. Modeled dissolved oxygen concentration of Hood Canal comparing Existing versus Reference conditions scenarios presented over different model layers.

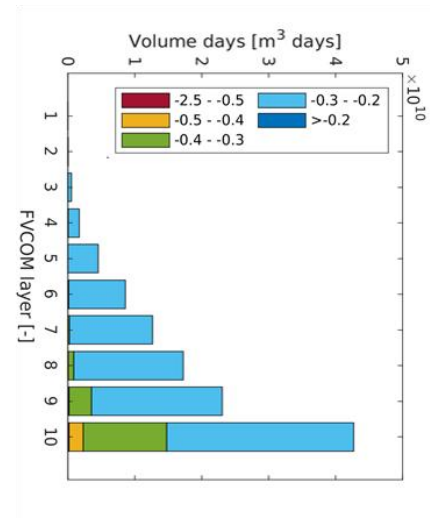
Top model layers (1-2) are in A, middles model layers (3-6) are in B, and bottom model layers (7-10) are in C. Note volume scale varies between model layer sets. The figure to the right presents the annualized difference in modeled condition is presented in total volume-days (m^3 days). Volume-days are calculated by summing the volume of water with ΔDO>0.2 mg/L in a given basin for each day of the model run. Estimates based on 2014 model year.

South Sound

Modeled Condition
(mg/L dissolved oxygen)

Annualized Difference in Condition by Layer
(m^3 days)





Note: Total volume-days for South Sound is 5.17×10^{12} m^3 days

Model Layers: A. 1-2, B. 3-6, and C. 7-10

Figure 32. Modeled dissolved oxygen concentration of South Sound comparing Existing versus Reference conditions scenarios presented over different model layers.

Top model layers (1-2) are in A, middles model layers (3-6) are in B, and bottom model layers (7-10) are in C. Note volume scale varies between model layer sets. The figure to the right presents the annualized difference in modeled condition is presented in total volume-days (m^3 days). Volume-days are calculated by summing the volume of water with ΔDO>0.2 mg/L in a given basin for each day of the model run. Estimates based on 2014 model year.

8.3 MAGNITUDE OF CHANGE OF DO CONCENTRATIONS

The percent of area not meeting Marine Water Quality indicator targets (row 1), and the breakdown of distribution and magnitude of those exceedances (rows 2-6), is shown for each sub-basin in Table 7. The results show several important features in modeled DO response of the differences between Existing versus Reference conditions. In particular, that the majority of the water volume in most basins has dissolved oxygen values ranging from 0.25 to 0.47 mg/L less than the reference condition (i.e. those in row 3, less than the 75th percentile). However, the maximum ΔDO is often considerably more and varies widely by sub-basin. Again, this ΔDO value represents the maximum predicted improvement in DO in the basins of Puget Sound if all anthropogenic nitrogen loads are eliminated. The extent of volume-days affected is also shown (row 1), providing an estimation of extent of improvement possible.

Table 7. Percent of total volume days exceeding indicator target of ΔDO<0.2 mg/L for each basin for Existing conditions. The distribution of the DO difference between Existing and Reference scenarios is also shown in mg/L. SoG - Strait of Georgia.

	Hood Canal	**South Sound**	**Admiralty Inlet**	**SoG and Northern Bays**	**Whidbey Basin**	**Main Basin**
Percent of volume days exceeding indicator target [%]	0.048%	0.895%	0.000%	0.063%	0.291%	0.007%
ΔDO max	-0.44	-2.15	0	-0.39	-0.43	-1.4
ΔDO 75th percentile	-0.3	-0.38	0	-0.32	-0.29	-0.47
ΔDO 50th percentile	-0.28	-0.31	0	-0.29	-0.27	-0.33
ΔDO 25th percentile	-0.26	-0.27	0	-0.27	-0.26	-0.28
ΔDO min	-0.25	-0.25	0	-0.25	-0.25	-0.25

9 Nutrient Management - Wastewater Treatment Plants (Strategy 1 – Reduce Wastewater Nutrient Loads)

9.1 Strategy Outcome Evaluation

Strategy 1 from the Marine Water Quality Implementation Strategy identifies a reduction in wastewater-associated nutrient loads as a key part of the overall nutrient reduction program for Puget Sound. Initial loading assessments indicate that loadings from WWTPs contribute ~50-60% of total anthropogenic loads to the Puget Sound (see Figure 14 and Mohamedali et al. (2011b))through more than 100 individual dischargers in the United States. Currently there are only a few WWTP systems in the region specifically designed to treat nitrogen in the waste stream. There are two general approaches to nitrogen reduction: Biological Nutrient Removal (BNR), which achieves effluent DIN concentrations of 8-10 mg/L and enhanced nutrient removal (ENR), which achieves effluent DIN concentrations of 2-3 mg/L. Opportunities under this strategy are to reconfigure and/or reconstruct the existing facilities to incorporate nitrogen reduction processes. Ahmed et al. (2019) evaluated the potential of this strategy by modeling a set of WWTP nutrient reduction scenarios. They focused on evaluating a scenario where BNR was implemented at all, or a subset of facilities resulting in a discharge water quality of 8 mg/L dissolved inorganic nitrogen (DIN) and 8 mg/L carbonaceous biological oxygen demand (CBOD). The modeling results demonstrated that these scenarios would result in a general, though uneven, increase in minimum DO in many locations throughout Puget Sound and would reduce, though not eliminate the areas of non-compliance.

The estimated maximum-technically-feasible nitrogen reduction wastewater treatment system processes can result effluent DIN concentrations as low as ~ 3 mg/L (e.g., US EPA (2021)), which defines the maximum extent of impact from the complete implementation of this strategy. In order to understand the potential effect on levels of dissolved oxygen in Puget Sound, the Salish Sea Modeling Center performed a model run where the effluent DIN concentration of all WWTP facilities (marine point sources) was set at 3 mg/L for the entire year (notes on model version and run are in Section 6.1 and footnote 3). Results indicate that there is a significant reduction in the occurrence of $\Delta DO>0.2$ mg/L in the management scenario compared to Existing conditions. However, the model results suggest that there would remain areas within each sub-basin where $\Delta DO>0.2$ mg/L compared to the Reference condition.

The implementation of this strategy is not, in itself, predicted to eliminate anthropogenic-related changes in dissolved oxygen greater than 0.2 mg/L.

Several additional scenarios were modelled to illustrate the changes in marine condition from different WWTP (point source) management scenarios. One of the scenarios, titled WWTP-1.5x, provides an illustration of potential impacts of population growth only, with no other changes to wastewater management or treatment facilities. In such a case nutrient loading would increase in proportion to population. Results of scenarios are summarized in Table 8.

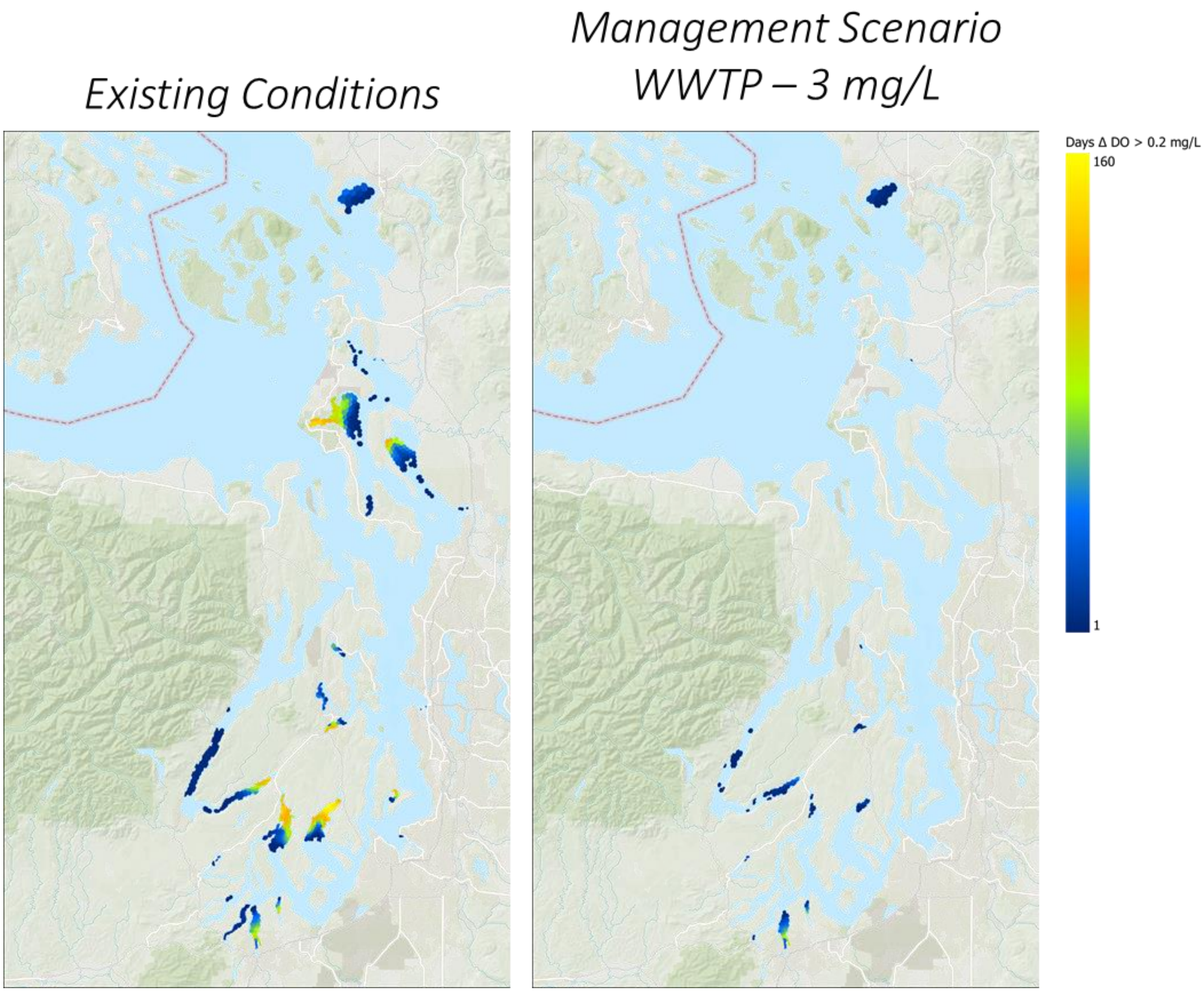


Figure 33. Summary of changes in water quality in Puget Sound between Existing conditions and a management scenario where all WWTP effluent total nitrogen concentration is at 3 mg/L.
Right shows spatial distribution of areas that would not meet Marine Water Quality indicator target of ΔDO<0.2 mg/L compared to Reference condition. Left shows spatial distribution after full implementation of management scenario. Color indicates number of days where ΔDO>0.2 mg/L for 2014 model year.

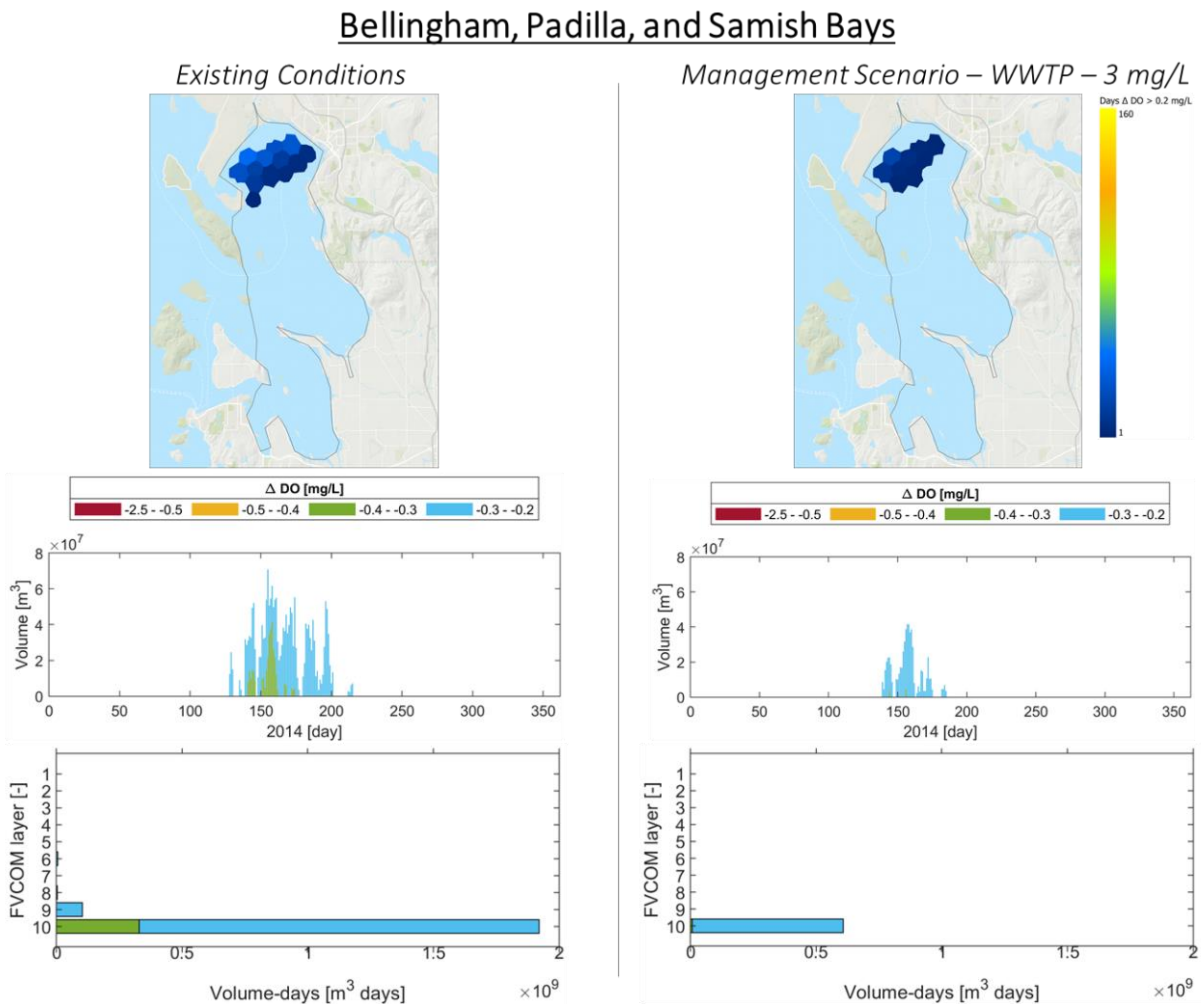


Figure 34. Changes in water quality in Bellingham, Padilla, and Samish Bays between Existing conditions and a management scenario where all WWTP effluent total nitrogen concentration is at 3 mg/L.
Top set shown spatial distribution of areas that would not meet Marine Water Quality indicator target of ΔDO<0.2 mg/L compared to Reference condition. Middle set describes the magnitude of change of DO between scenario and Reference condition, and bottom set describes depth distribution of change of DO between the scenario and Reference condition.

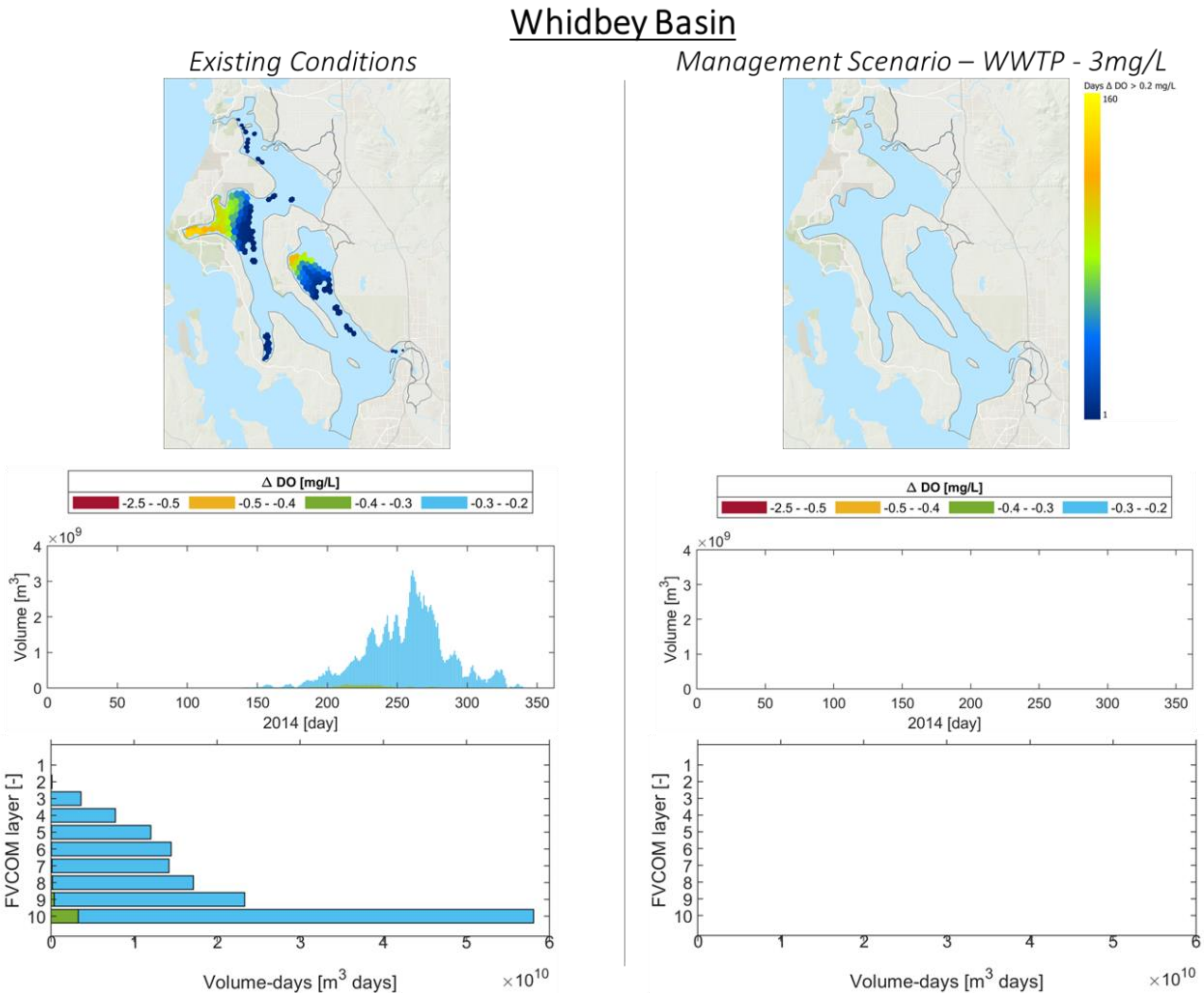


Figure 35. Changes in water quality in Whidbey Basin between Existing conditions and a management scenario where all WWTP effluent total nitrogen concentration is at 3 mg/L.

Top set shown spatial distribution of areas that would not meet Marine Water Quality indicator target of ΔDO<0.2 mg/L compared to Reference condition. Middle set describes the magnitude of change of DO between scenario and Reference condition, and bottom set describes depth distribution of change of DO between the scenario and Reference condition.

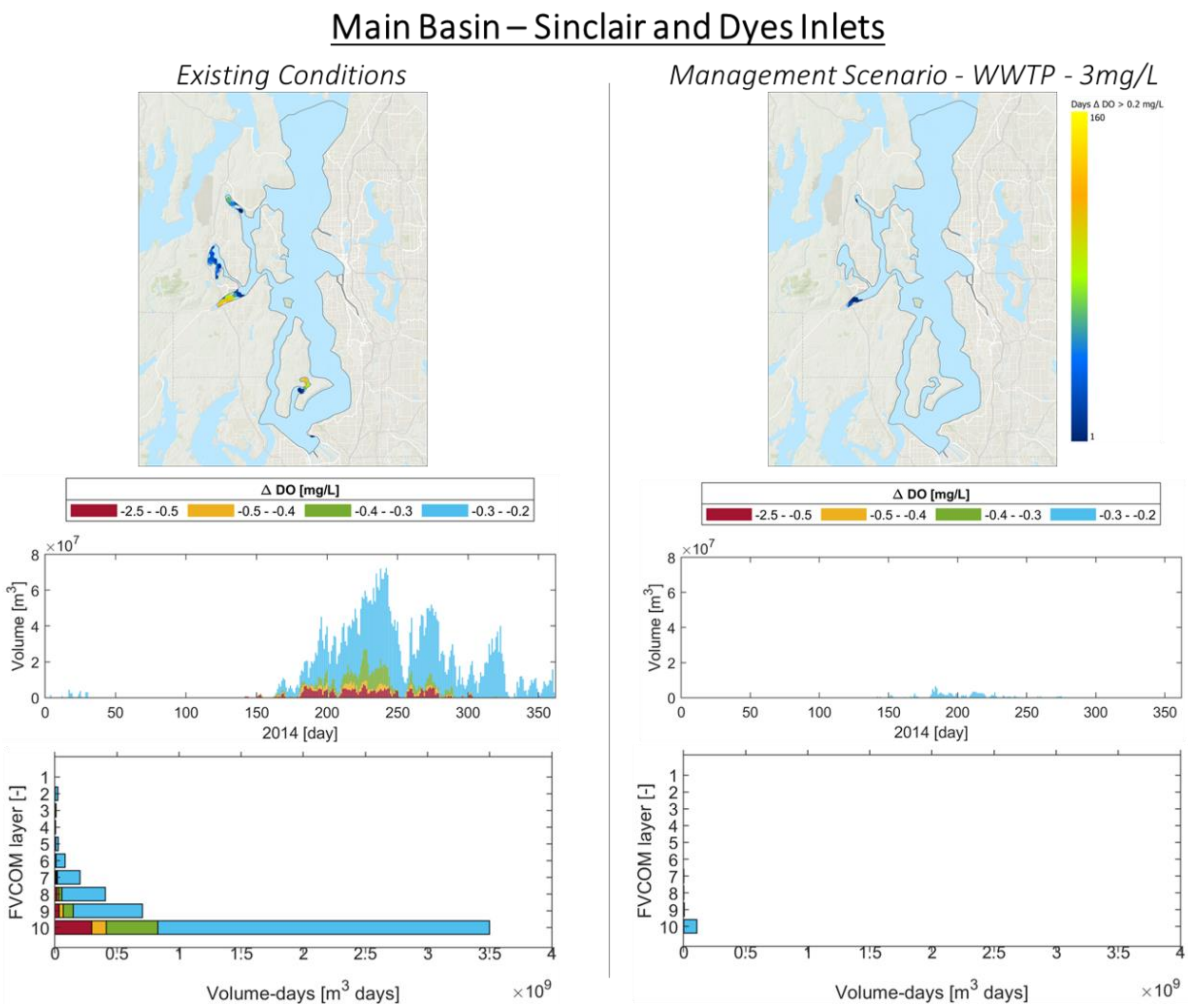


Figure 36. Changes in water quality in the Main Bain between Existing conditions and a management scenario where all WWTP effluent total nitrogen concentration is at 3 mg/L.

Top set shown spatial distribution of areas that would not meet Marine Water Quality indicator target of ΔDO<0.2 mg/L compared to Reference condition. Middle set describes the magnitude of change of DO between scenario and Reference condition, and bottom set describes depth distribution of change of DO between the scenario and Reference condition.

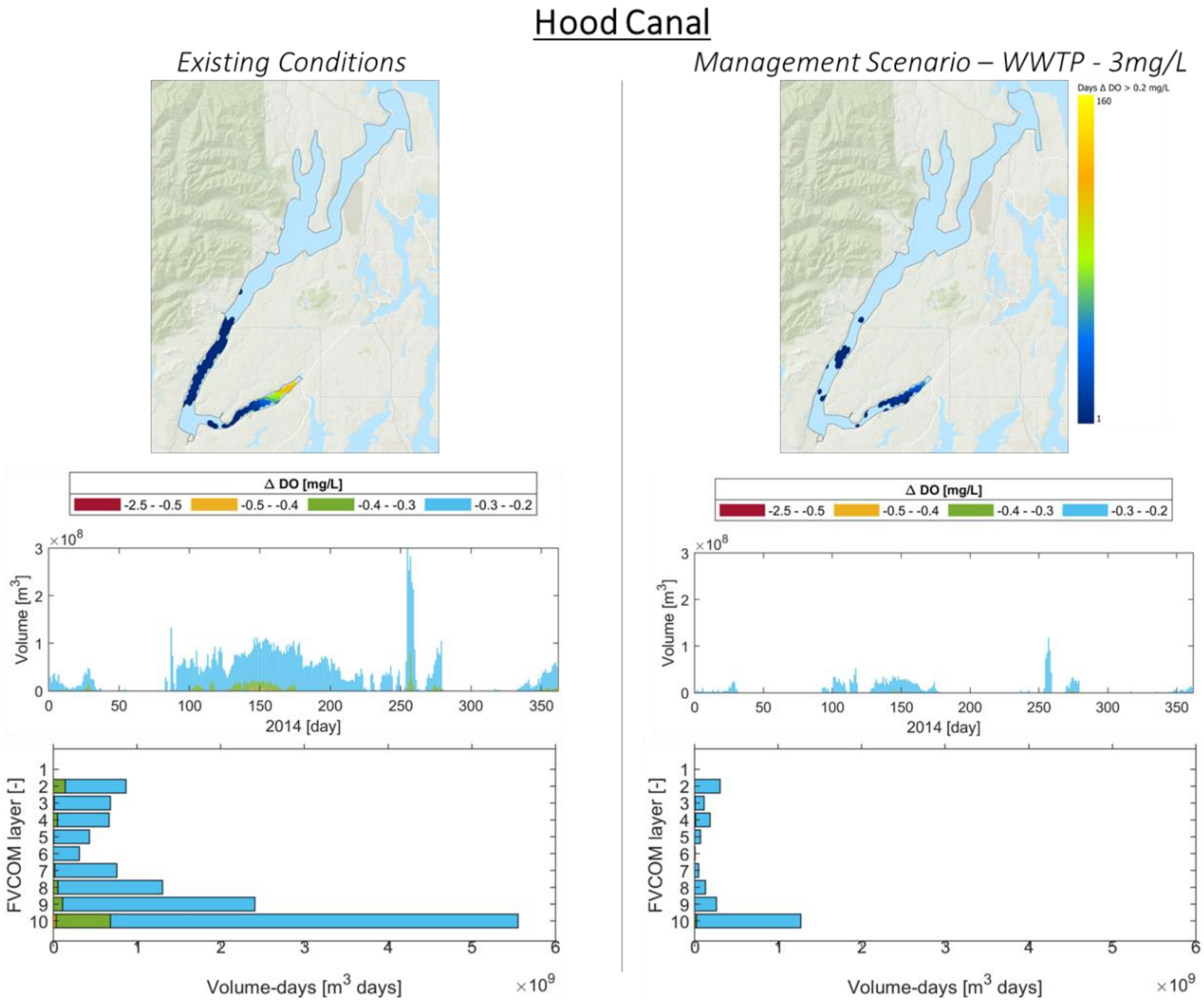


Figure 37. Changes in water quality in Hood Canal between Existing conditions and a management scenario where all WWTP effluent total nitrogen concentration is at 3 mg/L.

Top set shown spatial distribution of areas that would not meet Marine Water Quality indicator target of ΔDO<0.2 mg/L compared to Reference condition. Middle set describes the magnitude of change of DO between scenario and Reference condition, and bottom set describes depth distribution of change of DO between the scenario and Reference condition.

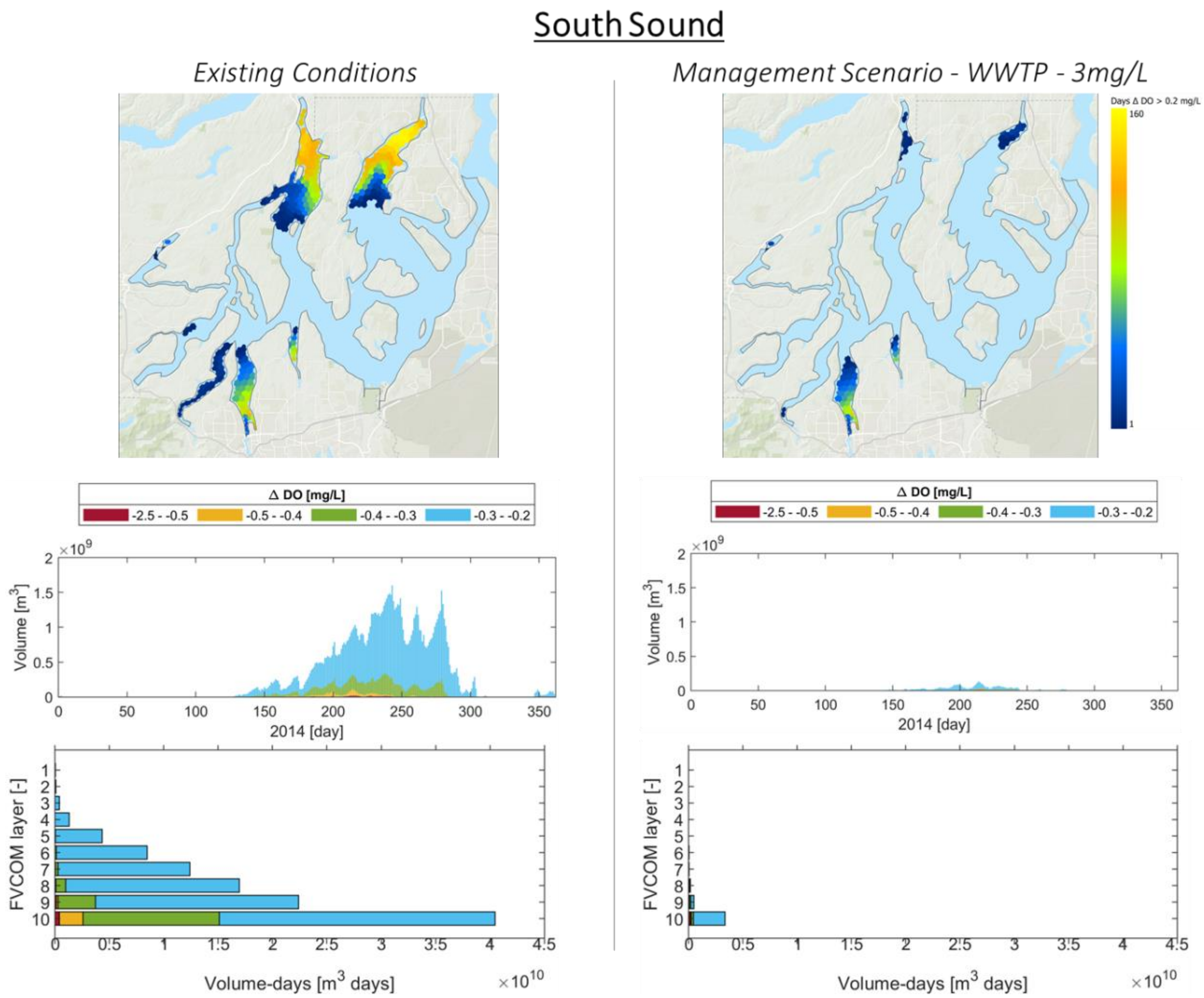


Figure 38. Changes in water quality in South Sound between Existing conditions and a management scenario where all WWTP effluent total nitrogen concentration is at 3 mg/L.

Top set shown spatial distribution of areas that would not meet Marine Water Quality indicator target of ΔDO<0.2 mg/L compared to Reference condition. Middle set describes the magnitude of change of DO between scenario and Reference condition, and bottom set describes depth distribution of change of DO between the scenario and Reference condition.

Table 8. Modeled extent of change between selected WWTP-associated management scenarios versus the Reference condition.

Values reflect extent of sub-basins where ΔDO>0.2 mg/L over the course of the 2014 model year. All values are in km^3 days. Existing conditions scenarios and total sub-basin volume-days are included for reference.

Scenario	Scenario description	Bellingham, Padilla, and Samish Bays	Whidbey Basin	Admiralty Inlet	Main Basin	Hood Canal	South Sound
Existing	All nitrogen loads from WWTPs and watersheds at Existing conditions.	0.77	24.40	0.00	1.85	3.42	43.12
WWTP-0x	No nitrogen loads from WWTPs. All nitrogen loads from watersheds at Existing conditions. Note that this scenario is illustrative only; it is not possible to completely eliminate WWTP-associated nitrogen loads.	0.17	0.00	0.00	0.01	0.31	0.92
WWTP-0.5x	Nitrogen concentration from WWTPs set at 0.5x Existing conditions. All nitrogen loads from watersheds at Existing conditions.	0.41	0.31	0.00	0.60	1.28	9.14
WWTP-1.5x	Nitrogen concentration from WWTPs set at 1.5x Existing conditions. All nitrogen loads from watersheds at Existing conditions.	1.40	199.86	0.00	6.06	8.83	110.76
WWTP-3mg/L	Total nitrogen concentration from all WWTPs set at 3 mg/L. All nitrogen loads from watersheds at Existing conditions.	0.18	0.00	0.00	0.03	0.38	1.18
Sub-basin volume-days (km^3 days)		1231	8395	7967	25470	7164	4817

9.2 Wastewater Treatment Plant Nutrient Reduction Approaches

9.2.1 Biological Nutrient Removal (BNR)

The conventional wastewater treatment systems that discharge to Puget Sound do not, in general, effectively remove nitrogen or phosphorus at part of their treatment processes. Some nitrogen reduction is achieved by supporting the assimilative growth of bacteria during the treatment process, which are subsequently collected as biosolids. These biosolids are not part of the wastewater treatment system effluent and are often used as fertilizer for farming and forestry. Land-applied biosolids have the potential to re-enter into aquatic systems through surface runoff. Conventional wastewater treatment plants remove approximately 10-30% of total nitrogen from the influent (Tchobanoglous et al., 1991) The reported median effluent DIN concentration of conventional wastewater treatment plants in the region is 24 mg/L (Roberts et al. 2014).

Biological Nutrient Removal (BNR) processes can reliably reduce nitrogen and phosphorus effluent concentrations, though many facilities would require a complete system modification or upgrade to incorporate them. Biological nitrogen removal takes place in a two-step process. In the first step, ammonia nitrogen that is released as part of the waste treatment and degradation processes is converted to nitrate. This often requires the addition of oxygen to the waste stream to support the process. In the second step, the nitrate is converted to nitrogen gas through denitrification. This step must take place in an anaerobic environment (one without oxygen) which is often achieved through the addition of external organic carbon to the waste stream. A schematic diagram showing the fundamental processes of BNR is included in Figure 39. There are many different physical configurations of BNR treatment plants. BNR processes often have a design effluent DIN concentration of ~ 8 mg/L (Roberts et al. 2014).

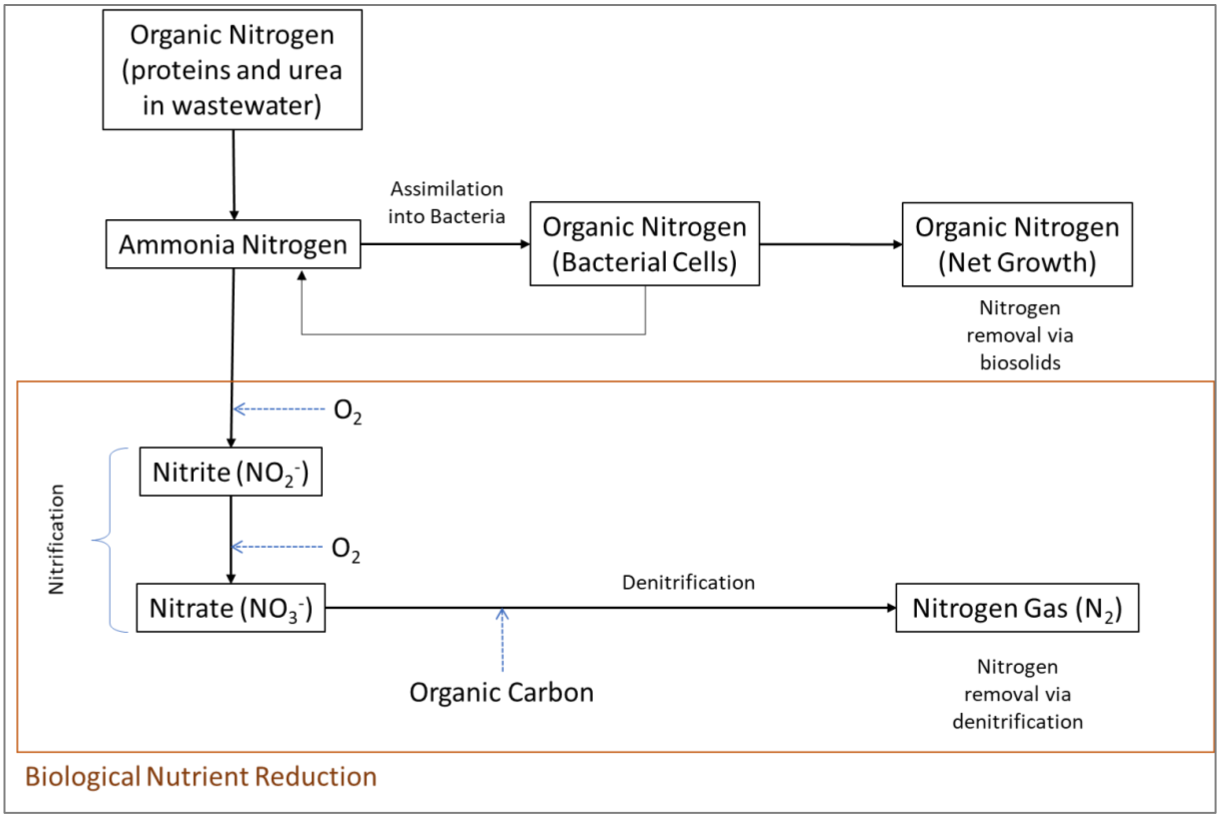


Figure 39. Schematic diagram of nitrogen transformations during the Biological Nutrient Reduction process.
Nitrogen removal is achieved through a two-step transformation process: first ammonia to nitrate (nitrification) and then nitrate to nitrogen gas (denitrification). Adopted from Tchobanoglous et al. (1991).

9.2.2 Enhanced Nutrient Removal

Lower effluent nitrogen concentrations can be achieved through more controlled and complex system processes. For example, the EPA utilized the proprietary WWTP design program CaptedWorks (Hydromantis, Inc) to evaluate systems configurations that would result in an effluent concentration of ~ 3 mg/L (US EPA, 2021). One configuration was based on a 5-stage Bardenpho with denitrification filter and a second was based on a 4-stage Bardenpho with Membrane Bioreactor. Although there is a large diversity among processes and configurations, a survey of installed systems reported that most are either Bardenpho type or UCT type processes (Pagilla et al., 2006; Sattayatewa et al., 2009; Urgun-Demirtas et al., 2008). There are a range of technology gaps and challenges with the design and operation of these systems (Pagilla et al., 2006) in addition to increased cost of design and construction, and increased costs, energy and chemical inputs for operation.

The Budd Inlet wastewater treatment plant is currently the only regional facility that utilizes enhanced nutrient removal. It has a four-stage Bardenpho processes with alternating anoxic and aerobic basins to facilitate nitrification/denitrification (Washington State Department of Ecology, 2018). The enhanced removal processes operate only during the summer months in accordance with permit requirements. The facility reported a mean and maximum total inorganic nitrogen (TIN) concentration of 2.13 mg/L and 4.1 mg/L, respectively, for the operating period of October 2011 through September 2016. A fraction of the treated, secondary effluent is directed to produce class A reclaimed water, further reducing the nitrogen loading to Puget Sound from the facility by diverting flows away from Puget Sound.

9.2.3 Modifications to Existing WWTPs to Improve Nutrient Removal

Enhanced nutrient removal generally requires major plant upgrades, including new/additional infrastructure for specific processes, or the construction of a completely new facility. Depending on the facility, there may be some operational or low-cost modifications that can be made which improve effluent quality. The EPA evaluated a set of case studies and their results suggest that there are often opportunities for low-cost nutrient removal optimization (US EPA, 2015). Many of the case studies incorporated modifications to create or optimize anoxic zones that support denitrification for enhanced biological nutrient removal. Examples included:

- Aeration modifications: changes to physical aeration equipment, controls, operation, and function of equipment and aerated areas. They include installing energy efficient blowers, variable frequency drives, improved diffusers, airflow meters, airflow control valves, and on/off cycling; all for dissolved oxygen, ammonia, or oxidation reduction potential control.
- Process modifications: adjustments to process control characteristics, including solids retention time and recycle/return rate. Physical process improvements might include adding return activated sludge pumps for internal recycling; adding online monitoring equipment for process control and optimization; or providing new screens or grit removal equipment at the headworks to improve the performance of the treatment process.
- Configuration modifications: changes to channels; manipulating gates; or modifying or adding piping, such as adding internal recycle lines or step-feed provisions; and are frequently employed to create or enhance environments for denitrification (e.g., by returning nitrate rich mixed liquor back to an anoxic zone).
- Chemical modifications: addition of, or changes to supplemental alkalinity and organic carbon feed to support biological nitrogen removal.
- Discharge modifications are made at the end of the treatment system to further reduce nutrients prior to delivery to receiving surface waters. They generally use natural systems and might include soil-based treatment systems or wetland assimilation discharge.

The case studies focused on activated sludge facilities; examples low-cost modification for lagoon or trickling filter systems were not reported.

9.2.4 Proposed critical analysis: Life cycle assessment of enhanced nitrogen removal from wastewater treatment plants

The Marine Water Quality Implementation Strategy presents explicit recommendations to incorporate advanced nutrient removal processes into wastewater treatment system upgrades to improve water quality. While there may be measurable benefits of implementing nutrient reduction approaches, there are also costs and impacts associated with increased use of energy, chemicals, and other resources that such processes require. Changes in treatment processes may also alter emissions of greenhouse gasses, such as carbon dioxide (CO_2) associated with increased energy demand, and nitrous oxide (N_2O) which can be generated as part of denitrification (Kosonen et al., 2016). Increases or decreases in greenhouse gas emissions are also an important part of the Life cycle assessment (LCA) process. LCA can support the holistic evaluation of costs and benefits and provide more complete information needed for decision making.

9.2.4.1 Proposed approach

There are several studies that focus on performing LCAs for wastewater nutrient removal (Corominas et al., 2013a; Corominas et al., 2013b; Lam et al., 2020; Rahman et al., 2016; Rodriguez-Garcia et al., 2014; Zang et al., 2015). We will begin by performing a detailed literature review providing examples and approaches. A case study may be performed to demonstrate the approach based on a regionally-appropriate example.

9.2.5 Proposed Critical Analysis: Evaluating change in contaminant removal associated with the enhanced nitrogen removal technologies in wastewater treatment plants.

The Toxics in Fish/Aquatic Life Implementation Strategy identifies wastewater treatment plant effluent as an important pathway for anthropogenic contaminants to enter into Puget Sound. This evaluation was largely based on the existing wastewater treatment infrastructure. Some compounds specifically associated with WWTP effluent, including Contaminants of Emerging Concern (CECs), have been identified as priority compounds based on their potential to cause biological harm (James and Sofield, 2021). Based on the recommendations in the Marine Water Quality Implementation Strategy it is likely that some wastewater treatment systems will be upgraded to incorporate nutrient removal technologies. These modifications may affect the fate and transport of anthropogenic contaminants, which would potentially change contaminant loadings to Puget Sound. Since managing toxic contaminants and their effects on marine biota and the predators that consume them is a stated goal for the region (see Toxics in Aquatic Life Vital Sign), the impact of potential changes in contaminant loading associated with WWTP nutrient removal upgrades may affect the overall approaches and plans to toxics reduction in the region (Washington Department of Ecology et al., 2021). This information may support a more efficient allocation of resources and/or improve the understanding of potential for multiple benefits from WWTP technology investments.

The Chesapeake Bay Program recently reported on an effort to understand contaminant removal (primarily PCBs) with wastewater treatment plant nutrient control upgrades by surveying information on published databases and in the literature (Tetra Tech, 2019). Their results supported several broad points, such as:

- Most efforts to reduce PCB in wastewater treatment plants focused primarily on source control.
- Since PCBs are hydrophobic and sorb to organic particles, upgrades that reduce total suspended solids (TSS) in the effluent will also reduce total PCB concentration in the effluent stream. PCBs will remain bound to the solids so the proper handling and disposal of solids is an important aspect of contaminant control.
- The degradation of PCBs through wastewater treatment systems improves with longer solids retention time.
- A primary degradation mechanism is the reductive dichlorination of PCBs, which occurs under anaerobic conditions. Therefore, Biological Nutrient Reduction, which utilize anaerobic conditions as part of the treatment process, may also improve PCB dichlorination.

While the data suggests that nutrient upgrades will likely reduce PCB concentration in the effluent, it is difficult to predict the extent of improvement. Additionally, there is limited data available for changes in treatment effectiveness for other chemicals of concern.

9.3 Onsite Septic Systems Nutrient Reduction Technologies

Traditional onsite septic systems are not designed to remove nitrogen; organic nitrogen is converted to ammonia in the septic tank, which may then be oxidized to nitrate in the drain field. Further removal is dependent on subsurface conditions.

Septic systems can be modified to improve nitrogen removal by facilitating the nitrification/denitrification process. While approaches vary, this is typically achieved by first, directing the effluent through a porous media with aerobic conditions to support nitrification, followed by addition of, or contact with a source of organic carbon which produces an anoxic environment for denitrification. Various examples include a wood-based filter (Robertson et al., 2005), a vegetated recirculating gravel filter (Wei, 2013), a recirculating gravel filter followed by a vegetated denitrifying woodchip bed (Grinnell, 2013), and an enhanced recirculating gravel filter. Pilot studies of these technologies demonstrated a total nitrogen removal of ~70-90%. All of these designs demonstrated consistent and effective nitrogen removal, though performance varied due to temperature, and all systems required more maintenance, and cost more than traditional systems.

The Hood Canal OSS nitrogen reduction project installed two nitrogen-reducing OSSs on nearshore properties and sampled monthly for two years to evaluate their performance (McCarthy, 2019a). Nitrogen removal ranged from ~45% at the beginning of operations to ~85% at the end of the monitoring period. One of the two systems was decommissioned shortly after the study due to maintenance and design issues.

The Washington State Department of Health has recently published design standards for the installation and operation of the recirculating gravel filer systems (Washinton State Department of Health, 2021).

9.4 Other Nutrient Management and Nutrient Recovery Approaches

It has been argued that wastewater should be considered as a resource from which water, nutrients/fertilizers, and energy can be recovered (Guest et al., 2009). This approach would provide more opportunities to achieve sustainability and satisfy a broader range of environmental and social goals, including reducing the impacts of effluent discharge to the environment. It should be noted, however, that the most sustainable solutions may not necessarily maximize any single measure (i.e., reduction or recovery of nutrients from wastewater effluent) but will likely focus on broader goals. Additionally, a primary challenge of incorporating sustainability and recovery considerations into decision-making is not necessarily the availability of technologies for resource recovery, but rather the lack of planning and design methodologies amongst resource management agencies to identify and pursue sustainable solutions (Guest et al., 2009).

A full evaluation of the regional implementation of sustainable treatment and use of wastewater is outside the scope of this State of Knowledge; a brief presentation of technical considerations of some resource recovery options is described below.

9.4.1 Water Reclamation

Water reclamation has the potential to reduce nutrient loading to receiving waters by reducing the overall discharge volumes. Reclaimed water is not used as a source of drinking water in the State of Washington but can be used for irrigation, to replenish groundwater, or as source of water for toilets in commercial or industrial buildings. Reclaimed water is handled through a different infrastructure than traditional municipal water supplies, in specially marked purple pipes.

All reclaimed water must be treated and disinfected. It is certified based on level of treatment and number of total coliforms present, ranging from Class A Reclaimed water to Class D Reclaimed Water. The potential uses of the reclaimed water vary by class. For example, only Class A reclaimed water can be used for spray irrigation on food crops, while Class D (or better) can be applied to non-food related forestry lands (Washington State Department of Ecology and Washington State Department of Health, 1997).

There are currently fourteen facilities in the region that produce and use reclaim water mainly for irrigation and groundwater recharge. These include the LOTT Budd Inlet facility, the King County Brightwater plant, the Port Gamble Resource Recovery facility, and the Belfair Wastewater and Water Reclamation facility. The total design capacity for reclaimed water is ~ 34 MGD, though actual production and use is less.

9.4.2 Nutrient Recovery

Nitrogen fertilizer is a valuable resource and the production of synthetic nitrogen fertilizer is an energy intensive process. Smith (2002) estimated that ~ 1% of global energy consumption was associated with supporting the production of synthetic fertilizers. Phosphorus is a non-renewable resource. Both nitrogen and phosphorus are abundant in human waste streams and so methods for recovering them continue to be evaluated. Perera et al. (2019) provided a review of technologies for recovering nutrients from wastewater. They identified and described several technologies that had the potential to capture nitrogen, though all had challenges that made scale-up and practical application difficult. A principal challenge is that ammonia salts are highly soluble. The formation of struvite precipitates (a phosphate mineral; $NH_4MgPO_4{\cdot}6H_2O$) is one possibility though the approach often requires the addition of Mg^{2+} and PO_4^{3-} to the waste stream to achieve N recovery. Jia et al. (2017) evaluated recovery scenarios and concluded that an optimized approach was feasible, resulting in over 96% NH^{4+} recovery, sustainable, and economically viable. To our knowledge, only a few full-scale recovery systems have been constructed. Examples include the CleanWater Services facilities in the Tualatin River watershed, which performs phosphorus recovery to meet effluent limits for discharge into the Tualatin River, and the Hampton Road Sanitation District which provides some nutrient recovery for water discharged to the Chesapeake Bay.

Algae harvesting, where wastewater is used to support the production of algal biomass in dedicated ponds, has drawn considerable attention and shows promise, particularly when the algae is used as an animal feed in livestock production (Madeira et al., 2017). Production costs, however, remain high. Others have considered the use of algae harvesting to support biofuel production and as a direct fertilizer for crops.

9.4.3 Source separation

Source separation refers to the practice of handling and collecting different waste streams differently in order to maximize their collection and reuse. Urine, for example, contains 81% of the nitrogen and 50% of phosphorus in domestic wastewater (Larsen et al., 2009) and so urine separation may be a viable option to end-of-pipe wastewater treatment. A key concept to this is that treating, and recovering nutrients, from concentrated solutions is more efficient than efforts from combined wastewater (see Section 9.4.2). Such approaches would require investment in infrastructure for separate piping systems and/or onsite treatment technologies. Source separation has the potential to offer ecological benefits compared to conventional, centralized treatment, depending on the system configuration and the nutrient recovery intentions; full nutrient recovery (both P and N) is the least cost- and energy-efficient due to the substantial chemical inputs required for N recovery (Ishii and Boyer, 2015; Remy and Jekel, 2008). And while there are some case studies where urine-based fertilizers were collected and used there remain barriers to acceptance to their use in industrialized nations (Lienert and Larsen, 2010). Primarily is that the actual facilities do not yet meet the functional standards of current systems.

10 Nitrogen Management - Approaches to Reduce Watershed Loads (Strategy 2 – Reduce Urban Stormwater and Agricultural Runoff Nutrient Loads)

Strategy 2 from the Marine Water Quality Implementation Strategy identifies a reduction in watershed-associated nutrient loads as a key part of the overall nutrient reduction program for Puget Sound. The strategy aims to reduce or eliminate anthropogenic nitrogen loading and pathways in stormwater runoff; the strategy specifically identified the following land use categories:

- Municipal stormwater within jurisdictions included in the NPDES Municipal Stormwater Phase I and Phase II Western Washington General NPDES Permits
- Construction and industrial stormwater
- Urbanized/rural stormwater runoff from areas not covered under an NPDES permit
- Commercial and non-commercial agricultural land use nonpoint loads

Stormwater runoff from impervious surfaces and/or agricultural areas can transport nutrients to surface waters, which contributes to the pool of dissolved inorganic nitrogen in the marine environment. This can influence the timing and magnitude of algae and dinoflagellate blooms.

In the urban environment sources may include fertilizers that are improperly stored or over-applied, particulate nitrogen and carbon from plant debris or soil erosion, improper food waste disposal, and pet waste. In addition, illicit connections between the stormwater conveyance system and sewer pipes may provide more continuous inputs into receiving waters.

Agricultural practices, including large-scale commercial production and small-scale non-commercial operations, can lead to the transport of excess nitrogen in runoff. For example, the presence of tile drains on agricultural lands to facilitate drainage can lead to nutrient-laden leachate discharged to surface waterbodies. Other examples include poor manure management, inappropriately timed or located nutrient application, and over fertilization. The lack of functioning riparian areas increases the potential for nutrient discharges to surface waters through reduced or impaired natural nitrogen attenuation function that a healthy riparian area may provide.

Initial loading assessments indicate that loadings from watershed runoff contributes ~40% of total local loads to the Puget Sound; about 34% of that is associated with anthropogenic activities (see Figure 13 and Mohamedali et al. (2011b)). That means that approximately 5% of total annual Puget Sound nitrogen load (or ~20% of total anthropogenic load) is associated with anthropogenic watershed sources.

There are several general approaches that have been identified to reduce nitrogen loading from surface water runoff. They include:

- Implement agricultural BMPs for crop and livestock producers
- Support nitrogen reduction for non-commercial agriculture operations
- Support illicit discharge detection and elimination (IDDE) programs
- Support development of nitrogen reduction technologies and BMPs for urban stormwater management.

These approaches will be discussed in further detail, below.

10.1 Strategy Outcome Evaluation

10.1.1 Background

Managing diffuse nutrient pollution has been identified as a "wicked" problem in that, despite years of research, improvements in technology, and investments, anthropogenic-nutrient related eutrophication continues to be a major problem in the United States and elsewhere (Lintern et al., 2020; Patterson et al., 2013). Several specific challenges to management have been identified including: ineffective implementation of best management practices, time lags between implementation and water quality improvements, the existence of legacy nutrients pools, and the overall complexity of diffuse pollution sources and watershed conditions. Social and economic issues, such as lack of motivation of private land owners and cost of construction and maintenance, also can impact the successful mitigation on diffuse nutrient runoff. Due to these challenges, managing diffuse nutrient pollution is challenging and outcomes are mixed. Selected case studies are presented below.

10.1.2 Watershed Nutrient Reduction – Selected case studies

This strategy focuses on reducing nitrogen loading to Puget Sound through increased and improved management of agriculture, residential, and urban runoff. Reducing watershed pollution through improved management of agricultural sources is a challenge for coastal communities around the world (Kroon et al., 2016).

Lintern et al. (2020) reviewed field and modeling studies to understand the rate of success of nutrient management, and to highlight the key factors that may affect the lack of improvement of nutrient levels in receiving waters. They reported that 60% of all studies reported improvements in water quality following the implementation of best management practices. However, most of the studies that showed improvements were based on modeled, rather than measured results. Of the field studies, approximately one-third showed water quality improvements, while the remainder either showed no improvement, or mixed results. This suggests that 1) performance of field best management practices is more variable than predicted, and 2) that variability is not well captured in existing models.

Some specific case studies of watershed-scale nutrient management are presented below.

Green et al. (2021) performed case study on three watersheds in the U.S. to understand if were possible to associate the implementation of nonpoint source nutrient management projects and programs, with changes in estuarine habitat and water quality. They reported that in two of three watersheds, the BMP installation was sufficient to significantly reduce nutrient concentrations and harmful algal blooms, a common symptom of eutrophication, within ~20 years. In contrast, BMP implementation in a third coastal watershed, was not sufficient to improve water quality most likely due to the fact that the management activities focused on surface-water inputs, while the majority of the nutrient loading was associated with groundwater and atmospheric deposition. This suggests that focused and sustained management activities can result in improving water quality in coastal systems.

There has been a wide range of efforts to monitor and reduce the inputs of N and P in the Chesapeake Bay watershed over the last 10-15 years including the implementation of agricultural and urban BMPs (Hively et al., 2018). Moyer and Blomquist (2019) reported that widespread improvements in water quality have not yet been observed. Murphy et al. (2022) utilized generalized additive models to link nutrient loads from watersheds to water quality trends in the estuary on a station-by-station basis. While there were few long term observed trends in riverine loading (loading from two rivers decreased, while one increased, with six no change), an analysis of flow-normalized trends indicated decreasing flow-normalized trends at six stations, and increasing flow-normalized TN trends at three stations. Since decreasing flow-normalized trends included the three largest rivers, the overall flow-normalized riverine load to Chesapeake Bay is decreasing. These improvements were attributed to decades of management effort throughout the region. An overview of the Chesapeake Bay watershed plans and recent case studies on BMPs and nutrient reduction will be included in the Marine Water Quality Base Program Analysis.

Fisher et al. (2021) performed a focused evaluation of nutrient loading to the Choptank estuary, an estuary within the Chesapeake Bay watershed, to characterize changes in loading from atmospheric deposition, point sources (e.g., WWTP effluent), and watershed sources (e.g., agriculture). Water quality improvements in the estuary were observed and associated with reductions in atmospheric deposition and point sources. The export of nitrogen from diffuse watershed sources increased over the study period over the entire watershed, with variable changes (increase, decrease, no change) across the sub-watersheds that were monitored. The results suggested that the efforts to improve nutrient exports from agricultural lands were not largely successful. Several possible explanations were provided: there is a lack of incentives, and some costs, for individual farmers to adopt BMPs; there may be insufficient numbers of BMPs installed, or they could be poorly located; there is a reservoir of legacy nutrients that provide a long term source, and; there may be a time lag between BMP implementation and improvements in downstream water quality (Fisher et al., 2021).

The Mississippi river watershed, which exports significant quantities of nutrients to the Gulf of Mexico, is another example of a watershed that has received significant attention focused on nitrogen management, yet has not been a marked reduction in exports. Van Meter et al. (2018) explored the effects of legacy N in intensively farmed watersheds, on N export and the ability to achieve water quality goals in the Gulf of Mexico. They demonstrated that, even if agricultural N use became 100% efficient (where nutrient additions were completely utilized by crops), it would take ~30 years to meet target N loads within the Mississippi River basin. Further, the authors noted that, although that large reductions in N loading may be possible, these reductions would require changes in land management as well as a fundamental alteration of the agroecosystem.

Successful reduction programs have been reported on. Bunnell-Young et al. (2017) investigated the changes in groundwater over time following the conversion of active farm land to conservation. They noted the exponential decline in groundwater nitrate concentrations over the 16 years following the farm conversion. Nitrate in the shallowest water declined from ~ 11 mg/L to 0.5 mg/L within 3-5 years.

Schilling and Spooner (2006) monitored stream nitrate concentrations in pared ~5000-ha watersheds to observe changes associated with the conversation of row crop agriculture to native prairie and savanna. In one of the watersheds, approximately 25.4% of row crop area was converted to grass land; nitrate concentrations decreased by 1.2 mg/L at the watershed outlet over. In the second watershed, row crop area increased by 9.2% and nitrate concentration increased by 1.9 mg/L at the watershed outlet over the study period. Water quality changes in stream nitrate concentration were more pronounced in smaller basins within these watersheds. Additionally, simple groundwater modeling suggested that the lag time for observing changes was on the order of a decade. Study results highlight the challenges of detecting land use management-associated changes in water quality even in the small watersheds that were the focus of this study. Detecting water quality improvements in larger watersheds will likely require a dedicated long-term monitoring effort on the order of several decades.

10.1.3 Outcome Evaluation – SSM

The Salish Sea Modeling Center evaluated the potential of this strategy (managing watershed nutrient loads) to affect estuarine dissolved oxygen by modeling a set of nutrient reduction scenarios (notes on SSM model version are in Section 6.1 and footnote 3). One of these model runs included a management scenario where the nitrogen concentration in all watersheds was reduced to a level that was half-way between the Existing conditions and the Reference conditions.

$$N_{t,x} = \frac{\left(N_{existing} + N_{reference}\right)}{2}$$

This scenario was selected as the potential result of a successful nitrogen management program in any given watershed; reducing nitrogen exports to full pre-anthropogenic/reference conditions is not likely achievable in any developed watershed.

Figure 40 - Figure 45 show the difference in extent of sub-basins meeting the indicator target (ΔDO<2 mg/L) for the nutrient management scenario.

The modeling results are shown in Table 9. They demonstrate that these scenarios would result in a general, though uneven, increase in minimum observed DO in many locations throughout Puget Sound and would reduce, though not eliminate the areas where ΔDO> 0.2 mg/L, as specified in the recovery target.

The implementation of this strategy is not, in itself, predicted to eliminate anthropogenic-related changes in dissolved oxygen greater than 0.2 mg/L.

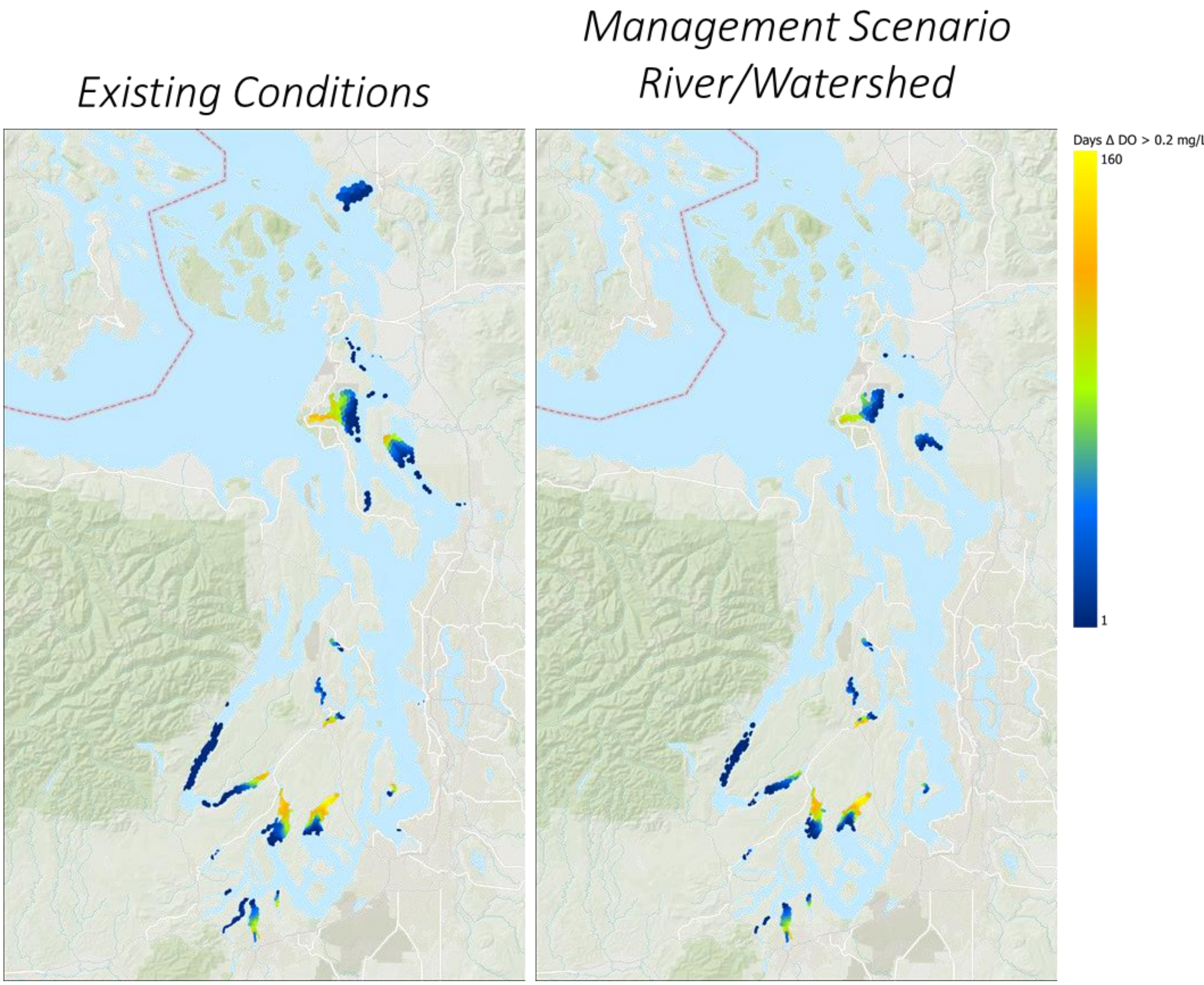


Figure 40. Changes in water quality between Existing conditions and a River/Watershed management scenario for the Puget Sound basin.

Under this scenario, all watershed N concentrations are 50% between Existing and Reference conditions. Right shows spatial distribution of areas that would not meet Marine Water Quality indicator target of ΔDO<0.2 mg/L compared to Reference condition. Left shows spatial distribution after full implementation of management scenario. Color indicates number of days where ΔDO>0.2 mg/L for 2014 model year.

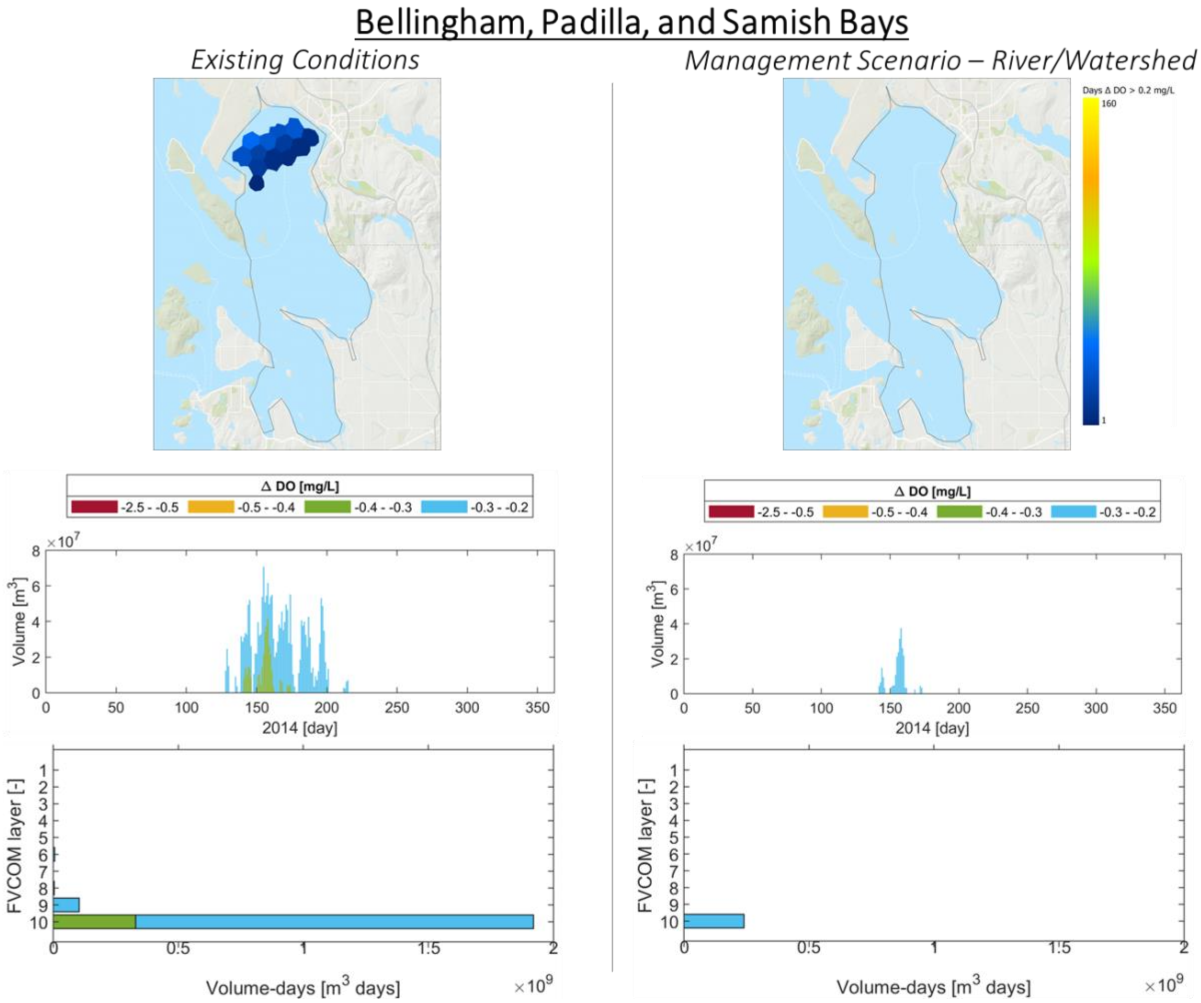


Figure 41. Changes in water quality between Existing conditions and a River/Watershed management scenario for Bellingham/Padilla, and Samish Bays.

Under this scenario, all watershed N concentrations are 50% between Existing and Reference conditions. Top set show spatial distribution of areas that would not meet Marine Water Quality indicator target of ΔDO<0.2 mg/L compared to Reference condition for at least one day. Middle set describes the magnitude of change of DO between scenario and Reference condition, and bottom set describes depth distribution of change of DO between the scenario and Reference condition.

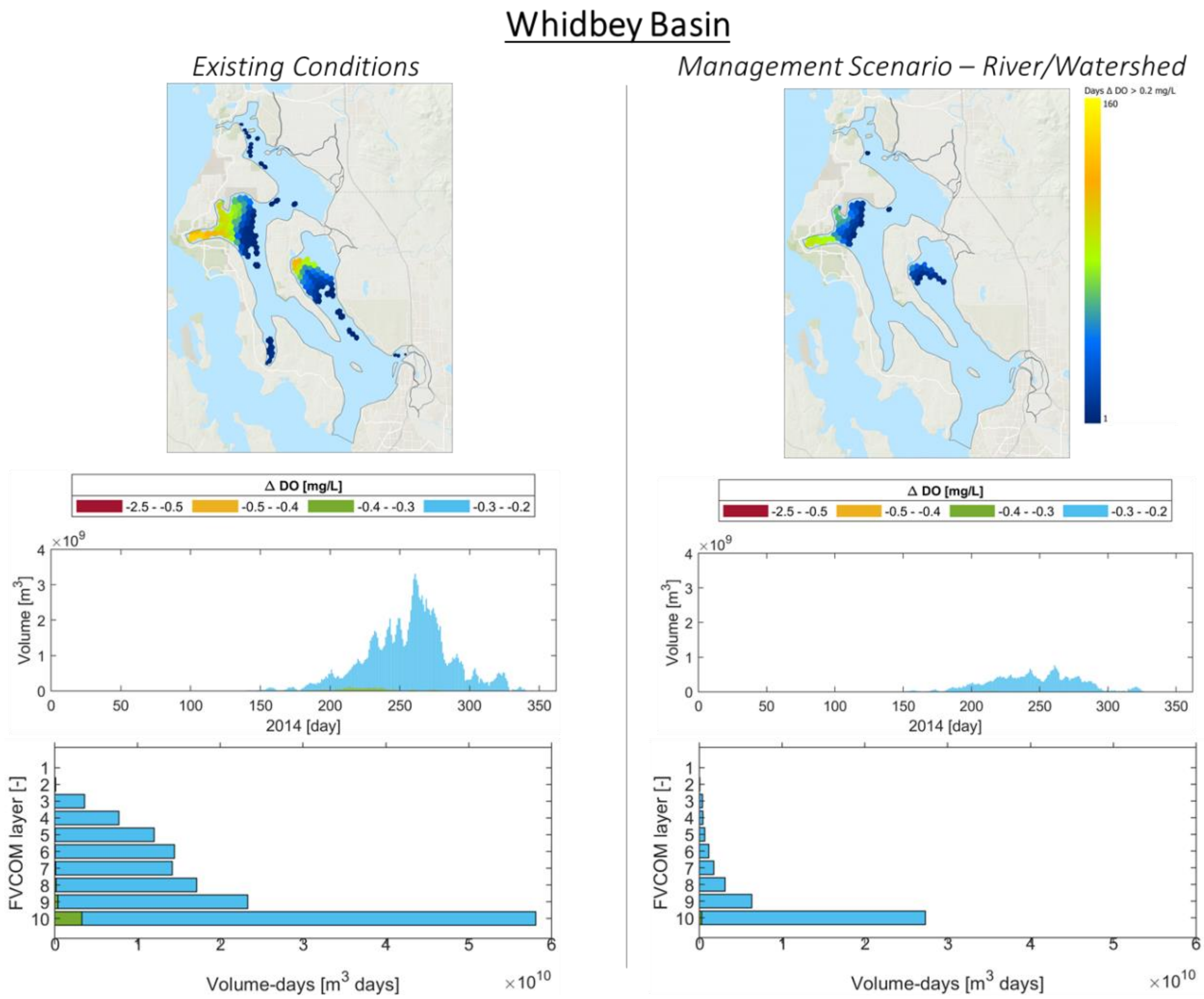


Figure 42. Changes in water quality between Existing conditions and a River/Watershed management scenario for Whidbey Basin.

Under this scenario, all watershed N concentrations are 50% between Existing and Reference conditions. Top set show spatial distribution of areas that would not meet Marine Water Quality indicator target of ΔDO<0.2 mg/L compared to Reference condition for at least one day. Middle set describes the magnitude of change of DO between scenario and Reference condition, and bottom set describes depth distribution of change of DO between the scenario and Reference condition.

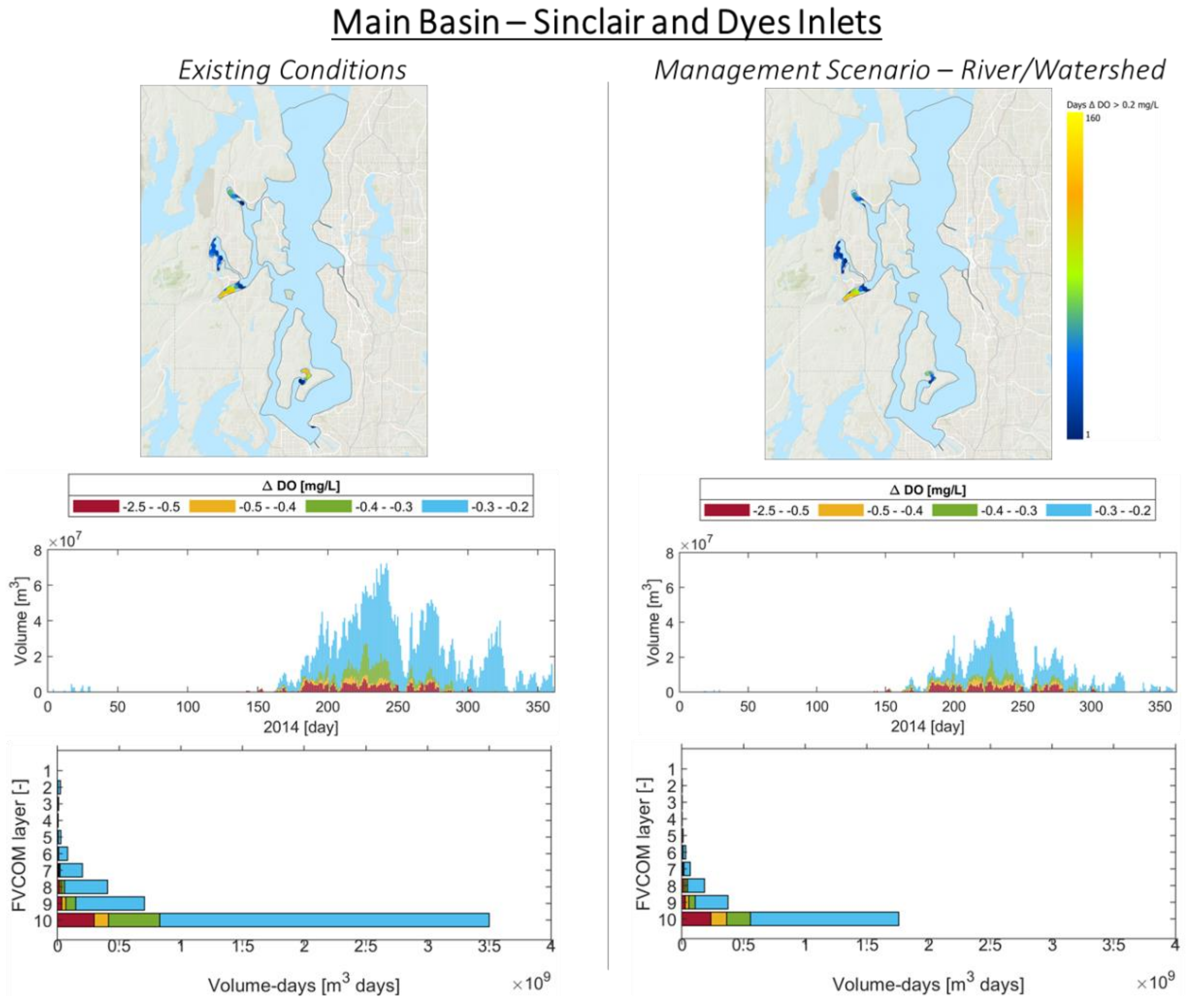


Figure 43. Changes in water quality between Existing conditions and a River/Watershed management scenario for the Main Basin.

Under this scenario, all watershed N concentrations are 50% between Existing and Reference conditions. Top set show spatial distribution of areas that would not meet Marine Water Quality indicator target of ΔDO<0.2 mg/L compared to Reference condition for at least one day. Middle set describes the magnitude of change of DO between scenario and Reference condition, and bottom set describes depth distribution of change of DO between the scenario and Reference condition.

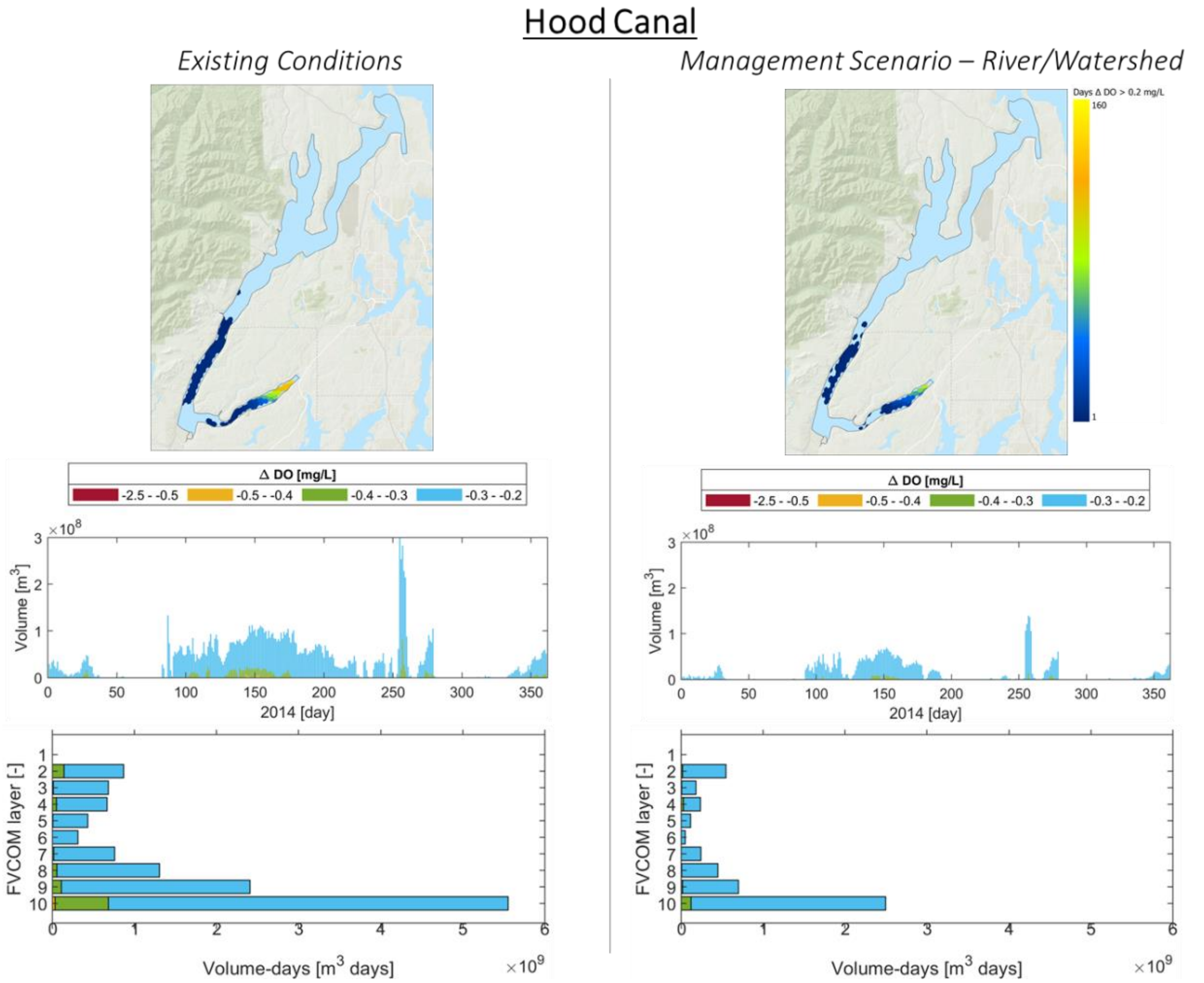


Figure 44. Changes in water quality between Existing conditions and a River/Watershed management scenario for Hood Canal.

Under this scenario, all watershed N concentrations are 50% between Existing and Reference conditions. Top set show spatial distribution of areas that would not meet Marine Water Quality indicator target of ΔDO<0.2 mg/L compared to Reference condition for at least one day. Middle set describes the magnitude of change of DO between scenario and Reference condition, and bottom set describes depth distribution of change of DO between the scenario and Reference condition.

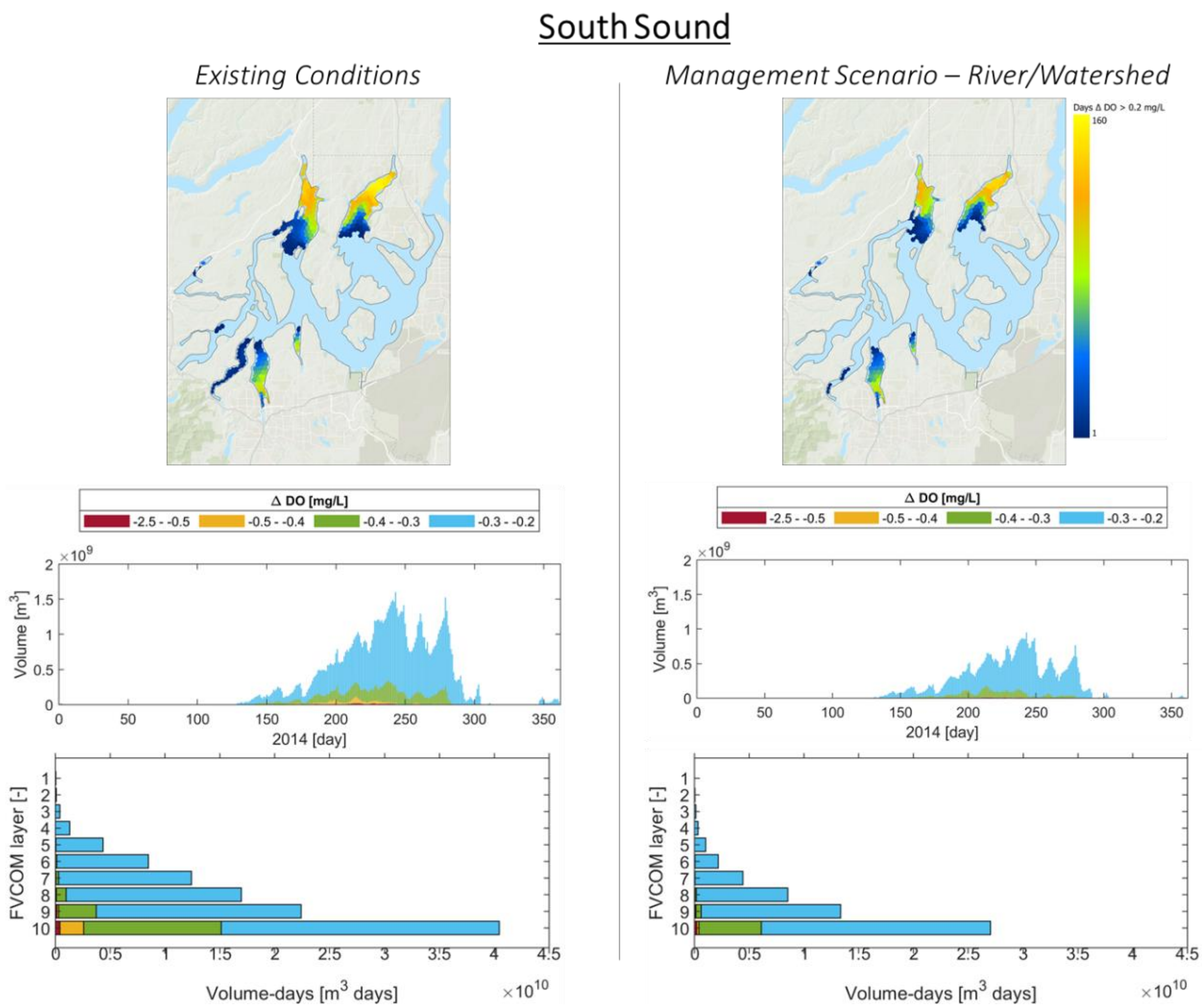


Figure 45. Changes in water quality between Existing conditions and a River/Watershed management scenario for South Sound.

Under this scenario, all watershed N concentrations are 50% between Existing and Reference conditions. Top set show spatial distribution of areas that would not meet Marine Water Quality indicator target of ΔDO<0.2 mg/L compared to Reference condition for at least one day. Middle set describes the magnitude of change of DO between scenario and Reference condition, and bottom set describes depth distribution of change of DO between the scenario and Reference condition.

Table 9. Modeled extent of change between selected watershed-associated management scenarios versus the Reference condition.
Values reflect extent of sub-basins where ΔDO>0.2 mg/L over the course of the 2014 model year. All values are in km^3 days. Existing conditions scenarios and total sub-basin volume-days are included for reference.

Scenario	Scenario Description	Bellingham, Padilla, and Samish Bays	Whidbey Basin	Admiralty Inlet	Main Basin	Hood Canal	South Sound
Exist	All nitrogen loads from WWTPs and watersheds at estimated Existing conditions.	0.77	24.40	0.00	1.85	3.42	43.12
Watershed-0x	No nitrogen loads from watersheds. All nitrogen loads from WWTPs at Existing conditions. Note that this scenario is illustrative only; it is not possible to completely eliminate watershed-associated nitrogen loads.	0.00	0.00	0.00	0.57	0.04	0.31
Watershed-0.5x	Nitrogen concentration from watersheds set at 0.5x Existing conditions. All nitrogen loads from WWTPs at Existing conditions.	0.00	0.00	0.00	0.84	0.17	6.02
Watershed-1.5x	Nitrogen concentration from watersheds set at 1.5x Existing conditions. All nitrogen loads from WWTPs at Existing conditions.	3.98	266.59	0.00	4.94	23.59	111.98
Watershed-Ref	All nitrogen loads from watersheds at estimated Reference conditions. All nitrogen loads from WWTPs at Existing conditions.	0.00	0.32	0.00	0.86	0.23	4.90
River/Watershed Management	All nitrogen concentrations from watersheds at half-way between Existing and Reference conditions. This represents a 50% reduction in anthropogenic loading from watersheds. All nitrogen loads from WWTPs at Existing conditions.	0.00	5.41	0.00	1.20	1.21	18.95
Sub-basin volume-days		1231	8395	7967	25470	7164	4817

10.2 Agriculture and Livestock Management

Agricultural operations (both livestock and crop production) are associated with an increase in nitrogen runoff and loading into nearby receiving waters. Reduction of nitrate leaching can be achieved through different approaches such as changes in farm management and the installation of treatment systems. Examples of best management practices include (adopted from Kroll and Oakland (2019) and Dzurella et al. (2012)):

- Modify crop rotation and utilize cover crops to reduce nitrogen leaching potential.
- Optimize rate, timing, and placement of fertilizer, animal manure, and organic amendment application to reduce losses.
- Avoid fertilizer material and manure spills during transport, storage and application.
- Plant edge-of-field or streamside (riparian) buffers with trees or native grasses.
- Exclude livestock from streams.
- Stabilize stream banks.

Kroll and Oakland (2019) provided a recent review on the effectiveness of agricultural BMPs on reducing nitrogen, phosphorus, and sediment loadings from agricultural areas. The reported a wide range of effectiveness, which they attributed to factors such as synergistic effects of BMPs implemented within the same sites, upstream conditions, the influence of weather and other regional factors such as soil conditions, poor BMP placement, implementation and management, lack of maintenance, and poor experimental design which would preclude the collection of quality monitoring data. A summary of effectiveness is provided in Table 10.

Table 10. Estimated nitrogen reduction efficiency associated with different BMP types. From Kroll and Oakland (2019) and references therein.

BMP category	Estimated efficiency	Factors
Cover crops	3-45%	Geology, soils, cover crop type, timing and method of cover crop planting
Livestock exclusion	0-65%	Off-stream water, fencing, buffers, crossings
Riparian buffer	0-65%	Soils, vegetation type
Bank protection	35-65%	

The Water Research Foundation has compiled information on the effectiveness of agricultural BMPs at reducing nutrient and sediment loads (Leisenring et al., 2016; WRF, 2020); the database is available at: https://bmpdatabase.org/agricultural-bmp-database.

They acknowledge the complexity of designing and managing BMPs across agricultural areas, indicating that "one-size-fits-all" solution is not realistic for agricultural water quality issues. They identified a series of challenges in assessing BMP performance, similar to those described above. These include:

- Site-specific conditions such as soil, slope, groundwater depth, and climate will impact the efficiency of a BMP.
- Some practices, such as the management of nutrient use and application, are very complex. Factors that may affect the effectiveness of nutrient management include differences in nutrient source, application rate, timing and method of application, and cropping history.
- Many study sites include multiple BMPs with overlapping or related effects.
- There may be a lag-time between BMP implementation and observed effect.

Despite these challenges they did note several conclusions based on the information included in the database. These were:

- Nutrient management practices show reductions in surface runoff phosphorus and subsurface nitrate loads.
- No-till and conservation tillage practices show reductions in surface runoff sediment loads and subsurface nitrate concentrations compared to conventional tillage.
- Cover crops show reductions in subsurface nitrate loads.

Melland et al. (2018) reviewed the results of studies that measured impacts of agricultural mitigation activities in medium (1–100 km^2) watersheds. The mitigation measures included improved landscape engineering, improved crop management, and reductions in farming intensity. Improvements in water quality were reported in 17 of 25 studies, though it took from 4 to 20 years to detect such improvements. In most catchments where water quality improvements were measured, combinations of practices, rather than single practices, had been implemented. These addressed more than one nutrient source or pathway. Additionally, positive effects were often associated with a reduction in agricultural land use intensity, rather than with a change in practice. Finally, Melland et al. (2018) highlighted the notion that improved water quality may not always lead to meeting water quality standards.

The Washington State University Center for Sustaining Agriculture and Natural Resources has ongoing research, and education and outreach programs focusing on the implementation of anaerobic digestions and small-scale biogas production (https://csanr.wsu.edu/). Additional information on this program, programs from the Conservation Districts, and the Washington State Department of Ecology Voluntary Clean Water Guidance for Agriculture programs are presented in the Marine Water Quality Base Program Analysis.

10.3 Other Approaches

10.3.1 Shellfish Aquaculture and Shellfish Mitigation

Shellfish aquaculture has been identified as a potential source of diffuse (watershed) nitrogen to Puget Sound. Quantifying the actual and relative loads from shellfish-aquaculture activities has been identified as a key uncertainty that should be addressed.

Shellfish bed restoration is another proposed method for nutrient management particularly in eutrophic estuaries. Nitrogen in phytoplankton that is taken up by shellfish can be assimilated into shellfish tissue, buried in the sediments, or returned to the atmosphere via denitrification (Kellogg et al., 2014). Of those three processes, only denitrification will result in nitrogen removal from the system; removal via burial and assimilation is dependent on several factors. The review suggested that restored oyster reefs could result in increased nitrogen removal, though rates were not consistently enhanced. And, generally, oysters can have effects on water quality that vary by orders of magnitude among sites, seasons, and growing condition.

Shellfish aquaculture can be used as a mitigation tool to reduce the potential impacts of eutrophication when shellfish are removed from the local marine environment through harvest. Nutrient removal is a co-benefit of traditional commercial shellfish farms which are designed and operated to maximize the production of high-quality shellfish for consumption. Rose et al. (2015) reviewed published model estimate of nitrogen removal by shellfish farms and reported a range of 105-1356 lbs N ac^{-1} yr^{-1} (average of 520 lbs N ac^{-1} yr^{-1}). These results were site specific and local factors such as species, temperature, food/phytoplankton concentration and supply will likely affect uptake rates. They do serve to illustrate the potential uptake by farmed and harvested shellfish.

There have been some efforts to develop and optimize “mitigation mussel cultivation,” (also called “nutrient bioextraction”) which are shellfish operations specifically meant for nutrient removal (Nielsen et al., 2016; Petersen et al., 2014; Petersen et al., 2019; Taylor et al., 2019). Nutrient uptake can be optimized and the labor time and costs can be reduced for mitigation mussel production compared to consumption mussel production (Petersen et al., 2014).

Petersen et al. (2014) evaluated the nutrient removal potential and costs associated with a full-scale mussel mitigation farm in a eutrophic estuary in Denmark. After two harvest cycles, they reported that mussels can be an efficient tool for mitigation of nutrients in the coastal environment, with an average removal of 0.6–0.9 t N ha^{-1} yr^{-1} (535-803 lbs N ac^{-1} yr^{-1}). This area normalized nutrient removal rate for mitigation cultivation is higher than the average estimated from theoretical studies using mussels designated for human consumption (Møhlenberg, 2007). Costs for this approach were reported as 14.3 **€** kg^{-1} N, which is comparable to the higher-range costs of land-based mitigation measures.

Taylor et al. (2019) evaluated the mitigation performance of additional full-scale mussel farms with different density configurations of conventional setups and potential harvest times, and also with different cultivation technologies. The optimized configurations demonstrated a removal potential of 0.6–2.0 t N ha^{-1} (per year; 535-1784 lbs N ac^{-1} yr^{-1}). These removal rates were based on the condition of the model estuaries and removal rates would likely be lower for systems with lower rates of primary production due to potential food limitations. Additional considerations include the accumulation fecal and pseudofecal matter, and organic enrichment of sediments directly within and around the farm, and the use of harvested mitigation mussels. Many mussels will be appropriately sized for human consumption market, other uses may need to be identified.

In addition to the nutrient reduction benefits, Petersen et al. (2016) reported on several of the potential co-benefits including: improved water clarity, increased biodiversity and habitat around the farm structures, provisioning of food for humans and livestock, fertilizer and shell material from shells, and culture services such as recreation, science, and education.

A regional pilot study was performed to test nutrient bioextraction using native blue mussels at a site in Budd Inlet (Pacific Shellfish Institute, 2017). Mussels were grown and harvested for compost over a course of two seasons and water quality and extracted nutrient mass were recorded. During the trial period, ~ 50 lbs of nitrogen was removed from the system and the additional potential benefits of water clarification, biodeposition, and habitat creation were also noted.

10.3.2 Anaerobic Digestion for Manure Management

Anaerobic digestion has been used on some dairies and other livestock operations to treat manure. It has the potential to be a source of renewable energy and reduce greenhouse gas emissions, while effectively controlling pathogen and odors. Anaerobic digestion does not alter the nutrient content and so careful management of the digestor waste is needed to avoid nutrient over-application (Camarillo et al., 2013).

10.4 Urban Stormwater Management Effectiveness

The Marine Water Quality Implementation Strategy identifies mitigating and managing stormwater-associated nutrient loads across the full spectrum of urbanization as a high priority. Monitoring across different land uses indicates that, nitrate concentration in streams in residential and commercial basins is much lower than in agriculture basins (Figure 46). However, there are selected residential basins that can have elevated nitrate. Additionally, runoff from commercial land use areas may be higher in Total Kjeldahl Nitrogen (TKN; organic N + ammonia) compares to low density or high density residential land use areas (Figure 47). These results suggest that, while nutrient runoff from built (non-agricultural) areas may be less significant than from agricultural areas, it still may be elevated compared to natural conditions. (see also McCarthy (2019b)).

The use of BMPs and Low Impact Development techniques has been suggested for nutrient control and management from developed watersheds. Successful implementation remains a challenge. Lintern et al. (2020) reviewed the effectiveness of best management practices for controlling diffuse nutrient pollution and reported that 40% of studies in mixed use watersheds, and 44% in urban watersheds reported little or no evidence of water quality improvements. A review of the data included in the International Stormwater BMP Database suggested that removal effectiveness is variable (Clary et al., 2020).

This variability is related to the complexities of the nitrogen cycle, nitrogen transformations during treatment, and the different properties of the different forms (e.g., ammonia vs nitrate vs organic nitrogen). As such, it is important to recognize that removal of one form of nitrogen may result in an increase in another form later in the cycle. For example, BMPs with permanent pools such as retention ponds and wetlands may reduce nitrate concentrations but may be ineffective (or increase) organic nitrogen. These systems may sequester nitrate in sediments and vegetation during the growing season and then release nitrogenous particulates after vegetation die-off. Conversely, biofilters and media filters can capture nitrogenous particulates, but the conditions are not conducive for denitrification or sequestration. Combinations of treatment BMPs, such as a permanent wet pool followed by a vegetated swale or media filter, may lead to permanent nitrogen reduction. Harvesting of vegetation and removal of captured sediment may also be key maintenance practices for reliable removal of nitrogen (Clary et al., 2020).

Driscoll et al. (2015) utilized information reported in the literature and the International Stormwater Best Management Practice database to describe the contaminant-removal by various types of green infrastructure. They reported a wide range of efficiencies with bioretention removing an average of 57% of total nitrogen mass, accounting for changes in concentration and volumetric reductions in of flow. Green roofs exported nitrogen, likely due to the use of fertilizers in their installation and maintenance. Summary of results are shown in Figure 48.

Note that there are currently no treatment technologies approved through the Washington State Technology Assessment Protocol – Ecology (TAPE) program. Such approvals would allow nitrogen reduction BMPs to be included in Washington State stormwater management manuals, and used for mitigation purposes under the NPDES Municipal Stormwater Permit program.

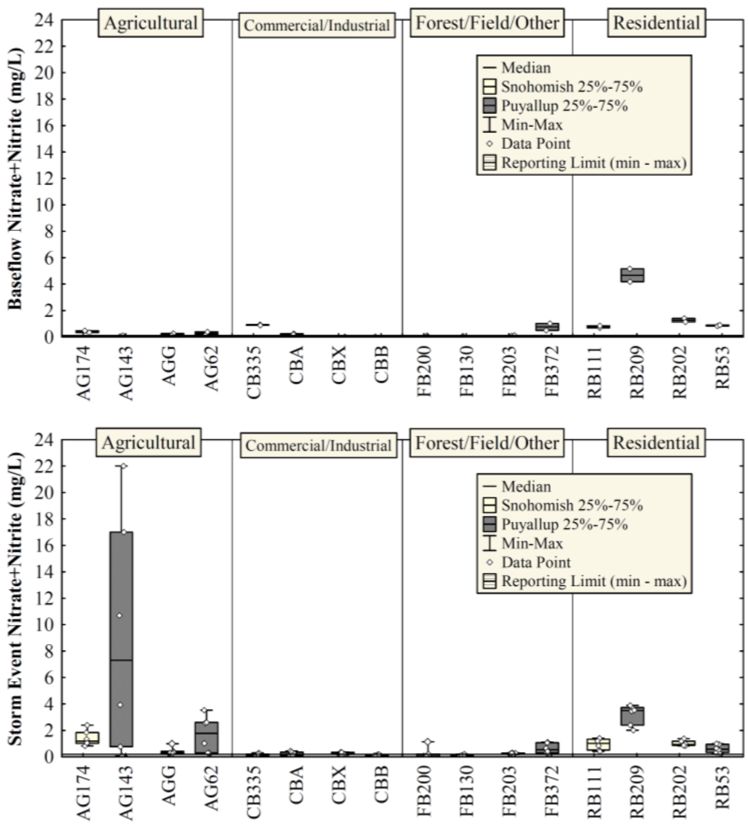


Figure 46. Nitrate/nitrite concentrations in streams in basins with predominantly agricultural, commercial/industrial, forest, or residential land use.

Measured concentration from base flow (top pane) and storm events (bottom pane) are shown. From (Herrera, 2011)

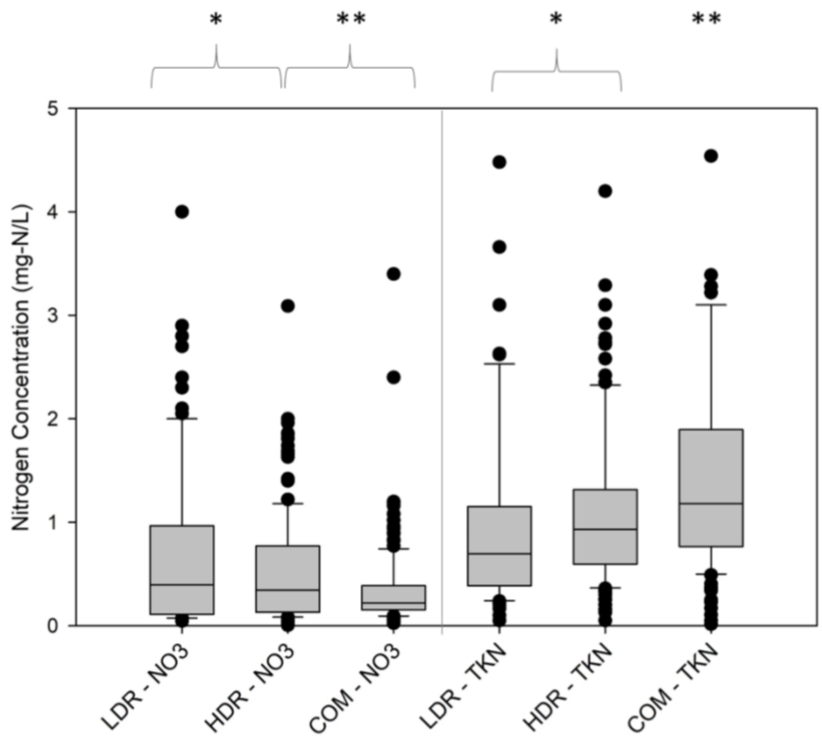


Figure 47. Nitrate/nitrite (NO3) and Total Kjeldahl Nitrogen (TKN) concentrations from basins with different predominant land use types.

LDR - low density residential. HDR- high density residential. COM - commercial. Data collected during storm events under the NPDES Phase I Municipal Stormwater Permit programs.

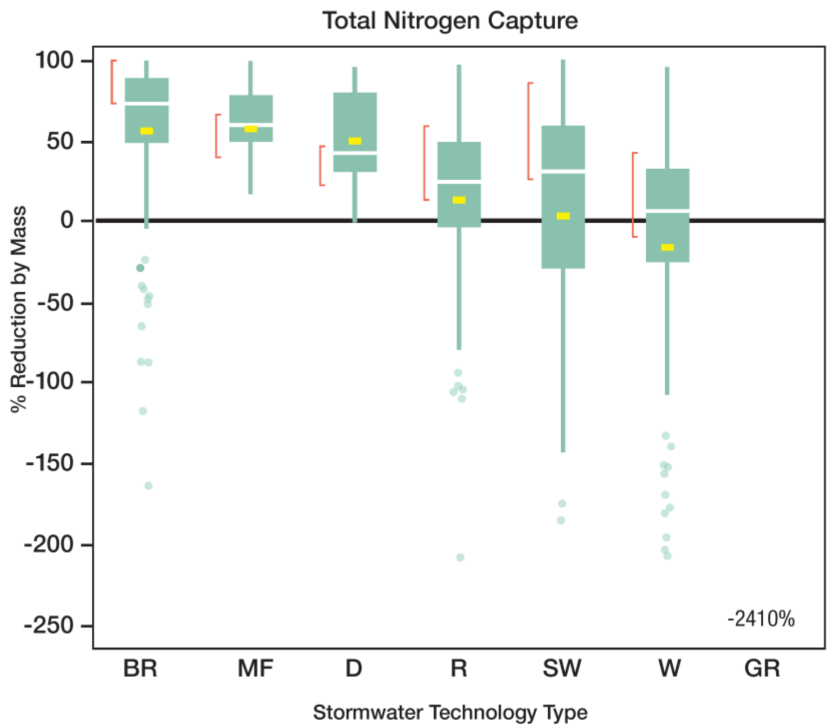


Figure 48. Summary of total nitrogen removal effectiveness on a mass basis, by technology type. Negative values indicate an export of total nitrogen from the system. The values shown are the average of performance by event, and may consist of more than one observation. Plot for green roofs is not shown due to scale. From Driscoll et al. (2015).

11 NUTRIENT MANAGEMENT - NATURAL NUTRIENT ATTENUATION PROCESSES (STRATEGY 3 – RESTORE NATURAL NUTRIENT ATTENUATION PROCESSES)

The Marine Water Quality Implementation Strategy identified the natural processes that lead to nutrient attenuation as a topic that should be addressed. The argument is that there are numerous physical, chemical, biological, and watershed factors that affect the transformation and uptake of nitrogen and phosphorus, and these factors are reduced or impaired by human activities resulting in lower attenuation and higher overall nutrient loads. Human land use activities that degrade natural nutrient attenuation processes include: degraded riparian areas, disconnected floodplains, loss of instream complexity, and loss of wetland areas.

Human land use activities that may impact attenuation processes include agriculture (ditching, leveling, installation of tile drains, and channel straightening) and other land development (loss of riparian zones and wetlands, construction of roads and other impervious areas, and construction of dikes and levees). It has been estimated that, as of 1988, about 20 to 40. percent of wetlands in Washington State, have been lost during the past two centuries, a continuing wetland loss of 700 to 2,000 acres per year. Some urbanized areas of the Puget Sound area have lost from 70 to 100 percent of wetland area. In addition, most of the State's remaining wetlands have been significantly degraded (Fretwell et al., 1996). Mitigating these landscape impacts may lead to improved nutrient removal and decreased loading into the Puget Sound.

Sheibley et al. (2016) performed an analysis of nutrient attenuation in the Puget Sound identifying key physical, chemical, and biological factors related to nutrient attenuation; estimated the nutrient attenuation for 17 major rivers draining into Puget Sound using two different models (RivR-N and v_f), and; highlighted potential management approaches for increasing nutrient attenuation. Their literature review highlighted the finding that, in general, attenuation increases with the amount of time spent within a given river reach. Additionally, most nitrogen cycling takes place within streambed sediments. As such attenuation may be improved through restoration of large, shallow floodplains (increase travel time and increase relative bed sediment area), improved channel diversity, and improved hyporheic connectivity.

Projects that might restore natural attenuation processes include: increasing instream channel complexity through additions of large woody debris and constructions of pools and riffles; restore off-channel (riparian, wetland, and estuaries) habitat, floodplain reconnection, and wetland restoration and reconnection.

The co-benefits of such restoration activities were also highlighted, including: improved fish habitat, reduction of instream temperature, improved flows, better salmon spawning and rearing habitat, increased intra-gravel and hyporheic flows, and reduced sediment loads.

11.1 Strategy Outcome Evaluation

Although the restoration of natural attenuation processes has been recognized as a potentially valuable approach for nutrient management, there has not been a watershed scale implementation demonstrating their successful application. There is information on the functionality of individual elements and some planning and/or modeling has been done to understand scope and scale of intervention required.

11.1.1 Wetlands

The use of wetlands has been identified as an approach to mitigate agricultural-related nutrient pollution. For example, Mitsch and Day (2006), suggested that the use of farm runoff wetlands (situated between farms and adjacent streams and rivers) and river diversion wetlands (diversion of river water into adjacent constructed and restored wetlands along main river channels and deltas) to augment agricultural best management practices in the Mississippi–Ohio–Missouri river basin. Based on their evaluations, neither approach would be sufficient to meet water quality goals on their own. They estimated that, in addition to management practices such as changing cropping systems, reducing fertilizer application, controlled drainage, and managing manure spreading and timing, approximately 2 million ha of wetland (<1% of total watershed area) area would need to be restored or constructed. Prior to undertaking such a large, watershed scale restoration effort, the authors argued that a comprehensive research effort was needed to address key uncertainties such as timing of floodwater input, methods for retention, fate of nitrogen, and investigation of factors, such as temperature, soils, microbiology, etc., on loading-uptake relationships.

Verhoeven et al. (2006) estimated that a relatively large proportion of a watershed (~2%-7%) would need to be devoted to wetlands to achieve water quality goals in intensive agricultural areas in Europe. This was based on observed nutrient loading capacities; they indicated that loadings beyond 25 kg N ha^{-1} yr^{-1} would results in a change of species composition of wetland vegetation, while loading beyond a critical value of 1000 kg N ha^{-1} yr^{-1} would result in substantial nutrient leaching and export into downstream receiving waters. Both numbers were presented as guidelines and would certainly vary based on local conditions. In addition, the authors noted the potential for the emission of the greenhouse gas nitrous oxide from nitrogen processing in wetlands.

Cheng et al. (2020) evaluated the potential for improving nutrient removal through the targeted restoration of wetlands. They used the National Wetland Inventory to estimate current N removal on a national scale and compared nitrogen removal base on a random scenario, a targeted scenario where restoration was focused on areas with highest nutrient loads, and a scenario that completely avoided active agricultural areas. They reported that the targeted scenario achieved approximately 4.3 times greater than random placement, and 40 times greater than that with no loss of agricultural land, thus quantitatively demonstrating the value of the targeted approach. Such targeting would demonstrably improve return on investment. This suggests that current opportunistic wetland restoration approaches are limited by a disconnect between N hotspots and wetlands.

Regarding the nutrient uptake by individual wetlands, Cheng and Basu (2017) performed a literature review and modeling study to understand nutrient processing in wetlands, the mechanisms, and the relationship between nutrient processing and wetland size on a landscape scale. They reported a mean removal of total nitrogen by all wetlands (both natural and constructed) to be ~ 50%, regardless of wetland size. This suggested that there was an inverse relationship between rates and size, a conclusion that was supported by modeling. This highlighted the potential value of small wetlands, particularly those situated near nutrient hotspots (Mitsch and Day, 2006) in mitigating nutrient pollution from watersheds. Indeed, Cheng and Basu (2017) suggested that, on a landscape scale, smaller wetlands (<300m^2) account for ~50% of total nitrogen removal, meaning they play a disproportionally large role in landscape nutrient processing.

Van Meter and Basu (2015) evaluated wetland loss patterns in the Midwest United States and found that there has been a preferential loss of smaller wetlands, and a reduced likelihood that smaller wetlands were located in upland areas relative to historic conditions. The change patterns could lead to the disproportionate loss of biogeochemical processing across a watershed. Their results suggest that restoration efforts should focus on understanding and recreating the historic size distribution and spatial organization of wetlands, rather than focusing primarily on increasing overall wetland area.

There are many factors that affect the nutrient-removal performance of constructed and natural wetlands including temperature, hydraulic residence time, vegetation type, wetland bathymetry, water depth (and variation of depth), and sediment type (Lee et al., 2009). A recent review reported a wide range of nutrient removal efficiencies (-10% to ~100%) and nitrogen removal rates (0.5 to 1000 g m^{-2} yr^{-1}) for surface flow wetlands and identified many of the same variable as reported elsewhere (Mendes, 2021). These studies clearly highlight the need for careful design and monitoring to understand the effectiveness of presumed attenuation processes.

11.1.2 Riparian Restoration

Riparian buffers, which are vegetated areas adjacent to streams and rivers, may reduce nitrogen loads into surface waters. Riparian-related mitigation mechanisms include plant uptake, microbial immobilization, soil storage, and denitrification. Denitrification is the only of these processes to remove nitrogen from the system; plant uptake may also lead to nitrogen removal if they are subsequently harvested. The effectiveness of buffers for nutrient attenuation varies widely. Mayer et al. (2007) performed a meta-analysis of the published literature to better understand the relationship between nitrogen removal in riparian buffers and buffer width, hydrological flow path (surface vs subsurface), and vegetation type (forest, forested wetland, wetland, herbaceous, herbaceous/forest mix). The overall mean removal was 67%, ranging from -258% (i.e., significant export of nitrogen) at one site, to 100% at several. Consistent with expectations, buffer width was positively associated with nitrogen removal with large buffers (>50m) removing significantly more nitrogen than smaller buffers (<25m).

Nitrogen removal effectiveness differed based on flow pattern, with subsurface removal being greater than surface removal. Furthermore, subsurface removal of nitrogen was apparently not related to buffer width, though surface removal of nitrogen was.

King et al. (2016) reported on a long-term study evaluating nitrate removal in groundwater passing through buffers with different widths (8 and 15 m) and with varying vegetation types (trees, switchgrass, fescue, native, and a control). Results indicated that the wide buffers were approximately 2.5x more effective than the narrow buffers, and that vegetation type did not make any significant difference. The presence of buffers did not affect nitrate concentrations in the deeper groundwater (2.1-3.5 m depth).

Valkama et al. (2019) performed a weighted meta-analysis of buffer zone impacts on nitrate and total nitrogen in surface water and groundwater. Overall, nitrate concentration was reduced by an average of 33% (17-48%) in the surface water runoff, and 70% (62-78%) in groundwater compared to control areas without buffer zones. Total N was reduced by an average of 57% (43-68%) in surface runoff. They reported that groundwater quality was more consistently improved by the presence of buffers compared to surface waters, which seemed to be affected by factors such as buffer age, where N removal decreased with age and vegetation type, and removal was more sensitive to buffer width compared to removal from groundwater. And consistent with other studies, the authors did highlight that fact that nitrogen removal through buffers is highly variable and so performance should be monitored, and not simply inferred.

Lyu et al. (2021) performed another meta-analysis focusing on the effectiveness of N removal through riparian zones, and the process that contributed to the removals. For surface runoff, they estimated increasing and highly variable removal of nitrogen from surface waters through approximately 10 m; there was no increase in removal at widths greater than 10m achieving an average of ~ 79%. For groundwater, they reported an increasing removal rate through increasing widths to ~15m, after which there was little significant increase. The average reported removal was 75%. Overall, the authors concluded that despite the high variability, riparian buffers can effectively reduce nitrogen in surface and groundwater.

Lind et al. (2019) expanded their analysis to include benefits of riparian areas towards other ecosystem functions such as provision of habitat and supporting biodiversity in a concept they described as Ecologically Functional Riparian Zones. They focused their review on publications that quantified services provided by riparian zones and developed recommendations for minimum buffer widths. They found that drainage size matters for nutrient and sediment removal, but that a 11 m buffer zone would act as a nutrient filter. However, a 24 m buffer was needed to maintain a high floral diversity, while a 144 m buffer would preserve bird diversity. Buffer width required for other services, such as shading (~ 21 m) and amphibian habitat (~ 53 m) were also estimated. This work highlighted the potential for multiple benefits from riparian restoration noting that, while there is generally no optimal buffer width to provide ecosystem services, the decision making can be informed by the literature in light if the specific goals and objectives of each riparian zone.

Note that there are different buffer width requirements applied to different recovery projects and land uses in western Washington, and requirements are not always well aligned leading to a loss of multi-benefit opportunities. This is discussed further in the Marine Water Quality Base Program Analysis.

11.1.3 Channel Complexity

Agricultural and urban development have resulted in channelization and straightening, ditching, and diking that directly impacts flow regime and channel structure. Additionally, watershed development affects stream hydrology and geomorphology of stream and river channels. Overall these changes decrease habitat diversity and can alter stream bio-geochemical processes. Tuttle et al. (2014) measured denitrification rates in streambed sediments seasonally to characterize the physicochemical drivers of nitrogen transformations in restored urban streams. Mean denitrification rates were highly variable though the importance of channel complexity, water depth and temperature were notable. Denitrification was dependent on streambed heterogeneity. The authors suggest that additional complexity provides more opportunities for habitat conditions for microbial biofilm formation, which may create a more active denitrifying environment in the sediments near geomorphic structures. It has also been reported that structurally stable areas accumulate organic matter and therefore function as denitrification hot spots (Harrison et al., 2012).

In addition, complex channels can increase the hyporheic connection and exchange. Hyporheic flow increases the proportion of stream water contact with the reactive surfaces of sediment grains and periphyton, and as a result can increase denitrification (Harvey et al., 2013).

12 FUTURE CONDITIONS

The condition of the marine water in Puget Sound is affected by many different, often interrelated factors. Many of these are expected to change in the future which will likely impact water quality and the effectiveness of management actions. For example, the population in the Puget Sound watershed is predicted to continue to increase. The Puget Sound Regional Council (PSRC) estimated that 1.8 million more people will reside in central Puget Sound by 2050 (Puget Sound Regional Council, 2018)) This will result in more development in the region and the potential for increased nitrogen loadings. Ecology has done some work estimating the potential nutrient loading increases from both WWTPs and watersheds based on future growth scenarios (see below) but has yet to incorporate potential changes in watershed condition associated with climate change.

Additionally, there is ample evidence that climate change will alter the magnitude and timing of flows of the rivers entering the Salish Sea, may alter marine circulation patterns, and/or alter species composition in the open oceans and the Puget Sound. Roberts et al. (2014) prepared a conceptual diagram relating the potential influence and potential uncertainties related to future and current conditions. For example, changes in future ocean condition may have very large impacts on the condition of water in Puget Sound, but the remains a large degree of uncertainty about what those conditions might be. Future oceanic conditions are both highly influential and highly uncertain (Roberts et al., 2014).

The results of selected potential future scenarios are presented below.

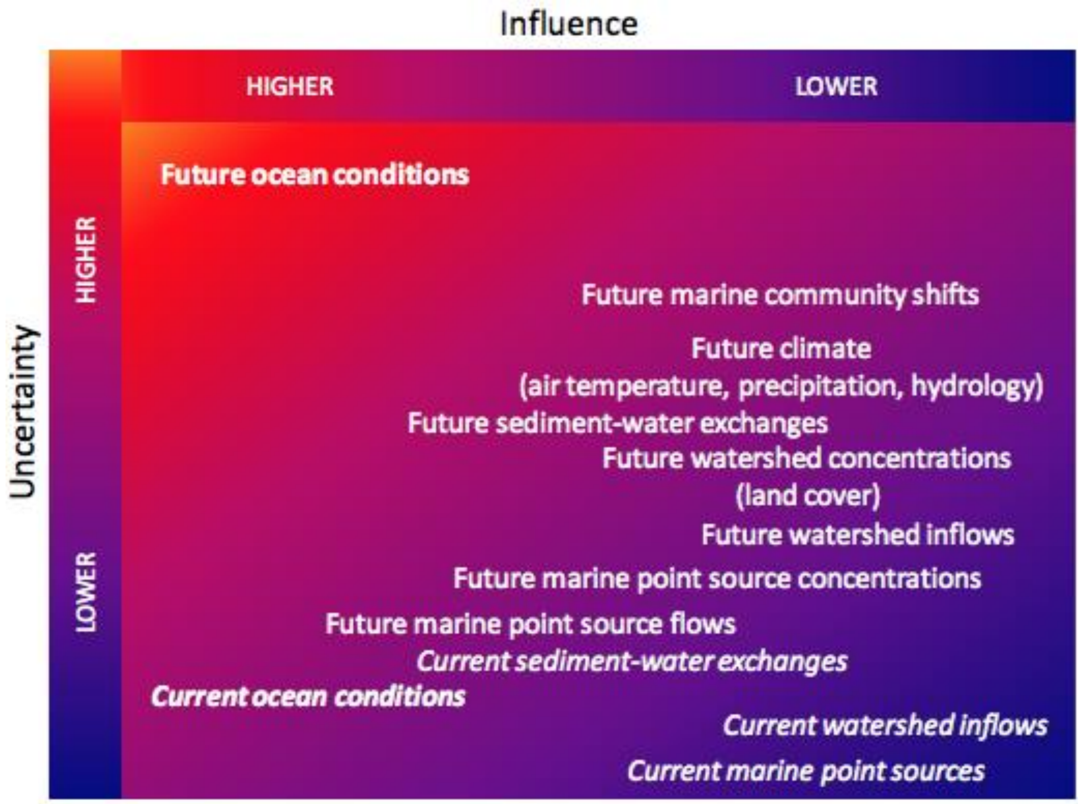


Figure 49. Conceptual diagram of the relative influence and uncertainty of selected current and future scenarios. From Roberts et al. (2014).

12.1 Future Wastewater Treatment Plant Loads (Point Source) - Increased Populations

Roberts et al. (2014) estimated future nutrient loads based on projected future populations in the Puget Sound watershed. Future loads were estimated by updating flows through the regional WWTPs based on population increases, while keeping the effluent N concentration the same. This method assumes that there would be no changes in nutrient removal technologies in any of the WWTPs. Based on this approach, WWTP-associated nitrogen loads would roughly double from ~ 33,000 kg/d in 2006 to ~ 55,000 to 65,000 kg/d in 2070. It should be noted that any WWTP upgrades that resulted in decreased N concentration in the effluent would offset the projected increases due to population growth. Completely upgrading all of the WWTPs to BNR (with effluent concentration of ~ 8 mg/L) would result in an overall reduction in WWTP-associated N loading to ~ 15,000 to 18,000 kg/d, which is well below current levels.

12.2 Future Watershed Loads – changes due to development

Roberts et al. (2014) estimated the future changes in watershed-associated nitrogen loadings by first, utilizing land use change predictions from the Oregon State University Futures Project, and then estimating the nitrogen loading based on a Land Use Index which describes the relationship between land use and average annual DIN concentration in associated basins. The “status quo” future scenario was used, which described predicted condition without major policy changes. DIN concentrations increase relative to current levels for all watersheds, with the greatest increases where forested land is converted to developed land. There is minimal predicted change for several watersheds located in the Olympic Peninsula and San Juan Islands due to low anticipated population growth/development, or due to the protected status of the watersheds.

Relative to the 2006 baseline load of ~ 29,000 kg/d, Roberts et al. (2014) reported that predicted watershed inflow DIN loads would increase to approximately 41,000 kg/d in 2070. This represented an increase of human-associated watershed loads of 14% in 2040 and 51% by 2070. It does not include changes from Canadian watersheds and in particular the Frasier River; alterations in the magnitude and timing of flows in rivers and streams within the Salish Sea watershed were also not included.

12.3 Future Marine Conditions and Climate Change

Khangaonkar et al. (2019) utilized the Salish Sea Model with outputs from the National Center for Atmospheric Research climate model Community Earth System Model to investigate potential changes in ocean condition, and how those changes might affect water quality in Puget Sound comparing current (year 2000) with future conditions (year 2095). In addition to changes in the ocean boundary condition, changes in monthly flows of the major rivers due to climate change were also incorporated into the analysis. Future nutrient loads were as described above, and in Roberts et al. (2014). The modeled output suggested that there would be marked changes at the ocean boundary that translated to changed conditions in Puget Sound. Highlights of findings include:

- oceanic exchange flows would not be significantly affected;
- there will be warming throughout the Salish Sea with a mean increase of ~ 1.5 °C;
- the maximum area of deep-water hypoxia (DO < 2 mg/L) will increase to cover ~ 16% of the Salish Sea, and a majority of Hood Canal.
- the mean annual DO will decrease by ~ 0.77 mg/L (see Figure 50);
- there may be shifts in algal species composition, with conditions being more favorable for dinoflagellates and much less favorable for diatoms. Bloom timing is also expected to change.

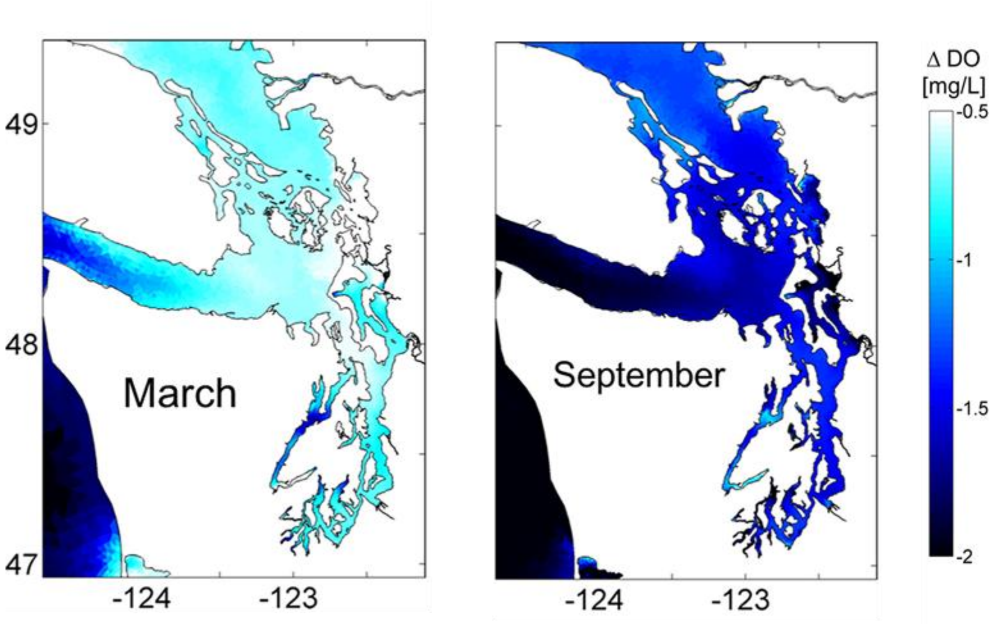


Figure 50. Change in predicted dissolved oxygen concentration between historical (Y2000) and future (Y2095) scenarios, for bottom 15% of water column.
Predicted change is for shown for March and September. Predicted future change at Ocean boundary is also shown.

Note that some fraction of these predicted changes can be attributed to local increases in human-associated nutrient loading, with the remaining fraction attributed to global climate change. The authors did not quantify the relative impacts of either stressor in these future scenarios. Roberts et al. (2014), however, suggested that, assuming that the long-term declining trends in North Pacific Ocean DO concentrations continue, Salish Sea DO would decline far more due to changes in oceanic conditions than from human nutrient loads. Future ocean conditions are highly uncertain.

13 DEVELOPING A RESEARCH AGENDA IN SUPPORT OF THE MARINE WATER QUALITY IMPLEMENTATION STRATEGY

The Marine Water Quality Implementation Strategy acknowledges that there remain many uncertainties associated with the causes, impacts, and management of marine water quality in the Salish Sea. As such, one of the strategies specifically identifies advancing marine water monitoring and research programs as a priority, and the majority of other strategies have specific research actions identified.

To support this, a focused process was undertaken to identify critical uncertainties and research and monitoring priorities, where "critical" refers to those questions and gaps in knowledge that are likely to impact our ability to either understand the causes of degraded marine water quality, or to implement an effective solution. This was done during the Implementation Strategy development process, and the results were reviewed by members of the Interdisciplinary Team. Some research and monitoring activities were translated into proposed Critical Analyses as described throughout this document, and identified as actions in the Regional Marine Water Quality Recovery Strategies (Chapter 3) of the Implementation Strategy narrative.

Developing and addressing a research agenda is an ongoing part of adaptive management, and continues after the finalization of this Implementation Strategy. The steps that were taken during Implementation Strategy development are described in Sections 13.1 and 13.2; the ongoing process is described in Section 13.3.

13.1 Developing a Research Agenda During the Implementation Strategy Development

During the development of the Marine Water Quality Implementation Strategy, the Interdisciplinary Team were charged with identifying key uncertainties, and research and monitoring priorities. PSI captured those uncertainties and questions in a matrix. There were three subsequent forums and workshops focused on refining uncertainties and research and monitoring priorities. They were:

- Interdisciplinary Team 4 - Discussion of general agreements and disagreements
- Interdisciplinary Team 5 a/b - Identifying what uncertainties require further investigation
- Puget Sound Partnership's Marine Water Scientists Meeting (July 23, 2020)

13.1.1 Interdisciplinary Team Meeting 4 – Initial Discussions

During Interdisciplinary Team meeting 4, members shared key certainties and uncertainties related to the Marine Water Quality Implementation Strategy. There was general agreement on a set of points that were generally factual (i.e., certainties), and a set of points where additional research was needed. These are listed in Table 11.

Overall, a total of 77 uncertainties were identified.

Table 11. Statements from Interdisciplinary Team members that were generally agreed upon as certainties (to be true) or uncertainties (key questions) by participants.
These were reviewed separately by Ecology to identify technical-related and policy-related uncertainties (see Appendix 4 of the Implementation Strategy Narrative)

Certainties	Uncertainties
• Clear patterns of impairment of beneficial uses of Puget Sound based on low DO levels • Increased incidence of algal blooms • Salish Sea Model connects N to DO • Rivers contribute nutrients to Puget Sound • Agricultural systems (both animal and crop agriculture) contribute nutrients to Puget Sound • Human population around Puget Sound is increasing • Non-human derived nutrients are the dominant source of nutrients • There are areas of Puget Sound with low DO • Productivity can be a good thing • More jellyfish in water	• What is the appropriate scale and type of approach to address impairment? • Primary causes for algal blooms • DO patterns • Lack of understanding of relative importance of different sources and pathways, and whether controlling them will make an impact to Puget Sound • How to use model results to regulate an allocation when the variability of model results is larger than the increment of the effect? • Quantifying the amount of control agriculture has over its contributions to nutrient exports. What are the major drivers of these contributions? Tillage? Amount of nutrients? Timing? • Relative effect of organic C and N on DO; nature and relationship between nutrients and DO • The Marine Water Quality indicator target is highly conservative (suggesting that it allows for only minimal change), highly sensitive, and is not measurable with observations. • If implemented, will strategies improve water quality in Puget Sound? • Is the vital sign target of DO impact less than 0.2 mg/L a relevant measure? • Is marine life in Puget Sound impacted by low DO?

13.1.2 Puget Sound Partnership's Marine Water Scientists Meeting (July 23, 2020)

The purpose of the meeting was to bring together regional scientists to discuss disagreements on uncertainties and identify potential research. Prior to the meeting, a prompt was sent to all participants. They were asked to reflect on, and submit written responses to the prompt so that organizers could identify broad areas of agreement and disagreement, and focus the discussions. The prompt read:

> "What is known, with what level of confidence, about the contributions of nitrogen and carbon inputs derived from regional human activity to changes in biogeochemical cycles, and in particular, dissolved oxygen reductions in bottom layers and ocean acidification in Puget Sound marine waters?"

Submissions were sent from several Interdisciplinary Team members, Ecology's modeling team and marine scientists, King County, PSI, and the Washington Environmental Council. The general concerns were narrowed to questions about Ecology's use of the Salish Sea Model and quantifying the impact of nutrient loading on ambient dissolved oxygen. Questions on how findings would be used for Ecology's regulatory and policy decisions were deferred for Ecology to address in their Puget Sound Nutrient Source Reduction project.

The workshop generated suggestions and ideas for improved communication and potential activities that the Partnership and the Strategic Initiative Leads could pursue to further explore modeling uncertainties, with help from the newly formed Salish Sea Modeling Center at University of Washington, Tacoma.

13.1.3 Interdisciplinary Team Meetings 5 a/b – Identifying what uncertainties require further investigation

The goal of Interdisciplinary Team meeting 5a and b, was to develop a research agenda focusing first on the consensus uncertainties; i.e., those that needed to be resolved before decisions related to nutrient reduction actions could be made, and which required further research. As such, Interdisciplinary Team Meeting 5a focused on those uncertainties where consensus could be reached. Interdisciplinary Team meeting 5b focused on identifying alternative pathways for the scientific community to identify a path forward.

In Interdisciplinary Team Meeting 5a, participants noted key gaps and barriers in the Strategies and Results Chains via interactive Mural boards. Consensus was reached on 28 uncertainties.

In Interdisciplinary Team Meeting 5b, those 28 uncertainties were reviewed and prioritized, and potential actions were identified to resolve them. Eight were deprioritized as they could be readily answered or regrouped under one of the other prioritized uncertainties. Results of the activity are available on Mural, and have been incorporated in the implementation strategy actions, and results chains.

The discussion at 5b also focused on the process by which future consensus could be reached for the 41 remaining uncertainties. Based on that, a plan was developed to engage the wider community of experts to build that consensus, particularly on key modeling uncertainties, beyond the Marine Water Quality Implementation Strategy development period, and to undertake further analysis to build confidence in the model's application. Subsequently, a series of workshops and a Model

Evaluation Group was hosted by PSI through the "Science of Puget Sound Water Quality" project, funded by King County.

13.1.4 Marine Water Quality Research Agenda based on Implementation Strategy Development

A draft research agenda was developed, focusing on the 28 consensus uncertainties (see Section 13.1.3). These uncertainties are described in more detail in Section 13.2, as they apply to specific recovery strategies. Some of them were also used to develop proposed Critical Analyses, that are included in this document. The purpose of those proposed Critical Analyses is to provide a brief presentation on the background and status so that researchers can begin to develop approaches to address them.

The 28 consensus uncertainties were broadly grouped as:

- Watershed source load quantification (n= 10)
    - Watershed sources and relative loading
    - Program targets
    - Model development and uncertainty
- Strategies (n=11)
    - Program targets and solution effectiveness (dependent on load calculations)
    - Optimization across multiple benefits
- DO thresholds (n=7)
    - Biological focus on Puget Sound specific species and life stages
    - DO thresholds -What information is available and what more is needed?

As described above (Section 13.1.3) an additional 41 uncertainties required further refinement. Ecology has reviewed the uncertainties and provided comments related to their relevance and applicability, and responses are included in an appendix to the Marine Water Quality Implementation Strategy. These 41 uncertainties are broadly grouped as:

- Model performance and interpretation
- Biogeochemistry
- Interpretation and application of science across a number of topics including DO, N, and marine loadings.

The entire list of the uncertainties is included in Appendix F of the Implementation Strategy narrative. All of the uncertainties have been incorporated into the Grand Uncertainty Matrix (see Section 13.3) to track relevant research, summarize new science, and support adaptive management.

Lastly, there was clear consensus on the prioritization and advancement of monitoring needs across most areas where uncertainties were identified (Strategy 5).

13.2 Draft Uncertainties and Research Priorities

The draft uncertainties align with three main strategies of the Implementation Strategy. They are discussed briefly in that context, below.

13.2.1 Strategy 2: Reduce Stormwater and Agricultural Runoff Nutrient Loads

Several uncertainties were specifically related to Strategy 2 (Reduce Stormwater and Agricultural Runoff Nutrient Loads). They can be described in three areas:

- Watershed sources and loading, including: relative magnitude of N loading from sources and sub-basins; climate change and population growth; erosion and flooding; on-site septic systems (OSS); groundwater; and fish hatcheries and shellfish aquaculture. These focused on gaps quantifying the (relative) magnitude of sources of anthropogenic nitrogen to Puget Sound, and within and between sub-basins.
- The effectiveness of policies and programs to deliver nutrient reductions, and particularly those related to OSSs, stormwater, and agricultural Best Management Practices.
- Watershed model development, including characterizing model performance, and the expansion of available models, or development/coupling of new models.

Overall, a broad area of research centered on the development of a watershed nutrient loading model that could be coupled with the SSM to evaluate anthropogenic load allocations. This could help address uncertainties related to policies and programs associated with managing stormwater and agricultural runoff. Many of these uncertainties align with the recommendations from the PSP Science Work Plan for 2020-2024 (Puget Sound Partnership, 2020), specifically those highlighted under EBS number 31 focusing on the development of biogeochemical modeling and nutrient load allocation.

13.2.2 Strategy 4: Develop Anthropogenic Nutrient Load Allocations Using Salish Sea Model and Regional Watershed Modeling Tools

Additional work is needed to address uncertainties related to the use and communication of the SSM, particularly related to load allocations. In response, a collaborative process was established by the Interdisciplinary Team for PSI, the Salish Sea Modeling Center (SSMC) and Ecology to engage the broader scientific community in further model performance, sensitivity and uncertainty analysis. The aim was to improve the understanding of the effects of parameter-and process-specific uncertainties on model outputs, and to communicate confidence in model results.

There were several components, including:

- Following published methodologies to further explore parametrization specification errors (Bowen & Hieronymus, 2020; NRC, 2001) recognizing that improved understanding of uncertainty is possible, but completely quantifying all uncertainty is not.
- Include sensitivity analysis of existing condition, (for example, using simplified Monte Carlo), and considering co-variance between parameters.

These priorities are consistent with the PSP Science Work Plan for 2020-2024 (Puget Sound Partnership, 2020), and the earlier PSP Science Meeting. Specifically, the following priorities were identified as the basis of a research agenda in nutrient load allocation modeling: access to data, code and documentation; input data and model refinement; further model runs addressing uncertainty, and communication among scientists.

13.2.3 Strategy 5: Advance Marine Waters Monitoring & Research Programs

Strategy 5 specified the need for additional monitoring and research.

Seven uncertainties were related to DO thresholds and requirements for biota in Puget Sound, and the science on biological and ecosystem-wide requirements for DO. Proposed research was in two parts. The first part would be, a literature review focused on Puget Sound-specific species, and key life stages, in the context of larger food web impacts, particularly if such a review included comparative studies globally in similar systems, the temporal and spatial extent of species range, and marine water condition. The second part would focus on the science around DO thresholds, and include:

- Meta-analysis of DO criteria, and associated uncertainty,
- Analysis of the data collection and research needs to further refine DO thresholds, and
- Research on the relationship of other eutrophication-related indicators in addition to DO, considering how they relate to anthropogenic nutrient loads.

13.3 Adaptive management of the Implementation-Strategy Related Research Agenda

The primary point of developing a research agenda is to identify and address key uncertainties, those that make it difficult to plan and/or implement effective recovery activities, or reach our recovery goals. Therefore, by addressing the uncertainties through focused research, we would learn and should then be better at ecosystem management and recovery. The broad research enterprise would be part of an adaptive management system.

Several tools have been developed to support the adaptive management of research and monitoring in Puget Sound, primarily the Grand Uncertainties Matrix (GUM). The GUM is centralized record of uncertainties, and associated research- and monitoring-related information for each of the Implementation Strategies (including Marine Water Quality) that have been developed under the Puget Sound National Estuary Program. It includes:

- A compiled list of uncertainties and information their sources (e.g., Interdisciplinary Team, planning workshops, academic publications, etc.);
- The associated Implementation Strategy and strategies;
- An indication of their priority status and a description of the prioritization exercise; and,
- A description of research that addresses elements of the uncertainty.

The GUM is a living document and will be updated regularly by staff at PSI. These updates will be performed based on an identified need such as planning, decision making, or implementation of a recovery activity, that could be informed by updated research, or focus on priority uncertainties. These updates may be in the form of research briefs, presentations, manuscripts, or annotated bibliographies.

The GUM should be considered the most up-to-date information on Implementation Strategy related research and monitoring. It will at first reflect the information provided herein, and then supersede it as more information becomes available.

14 CONCLUSION

This Marine Water Quality State of Knowledge report provides a technical background to support informed decision making and recovery planning focused on the Marine Water Quality Vital Sign. The State of Knowledge Report is meant to:

- Provide a clear description of indicators and targets
- Provide sufficient technical background to allow informed development and review of strategies to meet the chosen targets
- Provide technical background and analysis of strategies
- Identify and scope Critical Analyses

The Vital Sign that is the focus of this document was originally approved by the Puget Sound Leadership Council in 2012 and provides a target related to the level of dissolved oxygen in the marine waters of the Puget Sound. It states:

> By 2020, human-related contributions of nitrogen do not result in more than 0.2 mg/L reductions in dissolved oxygen levels anywhere in Puget Sound

The Vital Sign has since been updated to include measures associated with anthropogenic eutrophication (e.g., dissolved oxygen, nutrient balance, and marine benthic index) while being expanded to include stressors associated with global climate change and increasing CO_2 levels in the atmosphere (e.g., marine water temperature, ocean acidification).

Since the information in this State of Knowledge document focuses largely on the impacts and outcomes of nutrient management, it is still relevant to the successful management of the revised Vital Sign.

14.1 Sources of Nitrogen to Puget Sound

There are several key points related to nutrient loading to Puget Sound.

- Nitrogen enters into Puget Sound through natural and human-related sources. Overall, natural sources make up approximately three-quarters of the annual loading, and human sources make up the remaining one-quarter of annual loading (Section 5).
- Natural nitrogen sources include ocean water, which is relatively high in nitrogen, entering Puget Sound through the Strait of Juan de Fuca. The other major natural source is surface runoff from watersheds that carries eroded soils and organic matter into streams and rivers entering Puget Sound.
- Anthropogenic nitrogen enters into the Puget Sound via two major pathways. One is through wastewater treatment systems that discharge directly into Puget Sound, or into rivers that drain into Puget Sound. The second is through surface runoff where rainwater picks up nitrogen as it runs across developed lands including urban areas, and rural areas, and particularly agricultural and livestock operations. This surface water runoff enters streams and rivers and eventually reaches Puget Sound, carrying the nitrogen load with it.
- Minor sources of nitrogen include atmospheric deposition and groundwater. Neither of these sources contributes more than 1% of total nitrogen loading to Puget Sound.

- Natural and anthropogenic nitrogen loadings vary geographically (between sub-basins), and temporally (across seasons and between years). For example, oceanic loads may vary by ±10% between years (Khangaonkar et al., 2021).

14.2 Impacts of Anthropogenic Nitrogen to Puget Sound

Changes in Dissolved Oxygen (Section 8)

Reductions in dissolved oxygen (DO) are one of the potential impacts of excess nitrogen loading to Puget Sound. Since there is no information available to understand the state of Puget Sound prior to industrialization, the Salish Sea Model is used to estimate the change in condition. Model evaluation results indicate that human-associated nitrogen inputs do affect the levels of dissolved oxygen. The impacts are mainly anticipated to occur 1) in poorly flushed embayments and terminal inlets, 2) in the bottom waters, and 3) in the late summer/early fall each year. Overall, 15% of the surface area of Puget Sound is predicted to exceed the target of $\Delta DO<0.2$ mg/L once or more over the course of a calendar year (Figure ES2). Since these effects are localized and mainly in terminal inlets and deep waters, and only at certain times over any given year, this means that less than 1% of Puget Sound water has a change in DO greater than 0.2 mg/L (measured in by volume-days with accounts for both the amount/volume of water affected and the amount of time the impact lasts). The majority of the predicted change is <0.5 mg/L.

The interannual variability in loading and current patterns (measured by changes in annual residence times; Ahmed et al. (2019)) results in a wide variation in anthropogenic-nutrient related impact in water quality.

Other potential impacts (Section 4)

There are other potential impacts related to anthropogenic nitrogen inputs including:

- change in level of primary production (Section 4.1)
- changes in phytoplankton species and abundance (Section 4.3)
- increases in harmful algal blooms (Section 4.4)
- changes in seagrass community structure and abundance (Section 4.5)
- changes in benthic community structure (Section 4.6)
- changes in pelagic species and food webs (Section 4.6)

Some of these changes have been observed in Puget Sound, for example in declines in primary production (PSEMP Marine Waters Workgroup, 2020), localized areas of seagrass losses (Christiaen et al., 2019), and declines in benthic community condition (Ecology Marine Sediment Monitoring Program). Anthropogenic nitrogen inputs have not been clearly linked to any of these changes.

14.3 Management Actions to Control Nitrogen Loading

Three strategies have been identified to control nitrogen loading to Puget Sound. These include:

- Reducing wastewater nutrient loads
- Reducing urban stormwater and agricultural runoff nutrient loads
- Restoring natural nutrient attenuation processes

The confidence that these strategies will reduce nutrient loads varies. For example, it is clearly technically possible to upgrade the wastewater treatment systems in the Puget Sound watershed so that total effluent nitrogen is ~ 3 mg/L (current effluent N generally ranges from 20-40 mg/L). Reducing urban stormwater and agricultural runoff nutrient loads has been demonstrated on a small scale (Section 10.1) through the implementation of BMPs, though the results of nutrient reduction even at the sub-basin scale is mixed (e.g., Fisher et al., 2021). Similarly, the potential effectiveness of restoration of natural nutrient attenuation has been demonstrated (Section 11), though the effectiveness even of individual projects and installations vary widely (e.g., Mendes, 2021).

Regarding the outcomes of the nutrient reduction strategies on marine water quality - work with the Salish Sea Model suggests that no single strategy, even if fully implemented would completely eliminate all areas that exceed the indicator target of ΔDO<0.2 mg/L. For example, even with complete WWTP upgrades which would greatly decrease human-associated N inputs, localized areas that exceeded the target would remain (Section 9.1). A combined and extensive nutrient reduction approach (WWTPs effluent at ~ 3 mg/L and a 65% reduction in anthropogenic nitrogen), which would require substantial and sustained investment to achieve, may result in pre-anthropogenic water quality (DO) conditions (Ahmed et al., 2021). An estimated US$ 2.5-5.0 billion would be required for the regional wastewater upgrades alone.

14.4 Uncertainties and Critical Analyses

The strategy development process included the identification of key uncertainties, those that make it difficult to implement or evaluate nutrient reduction strategies (Section 13). The key uncertainties were identified in coordination with the Marine Water Quality Interdisciplinary Team. These key uncertainties were used to develop a suite of proposed critical analysis, which provides a brief background and description of the uncertainty, and a proposed approach for addressing the uncertainty. These are intended to provide an initial scoping for follow up research and investigation; some of them are currently being addressed.

Examples include a proposed Critical Analysis on the development of a watershed nutrient model coupled to improve the quantification of nutrient sources and impacts (Section 7.2.4), and an evaluation of the DO thresholds of key Puget Sound species (Section 4.8).

ACRONYMS

Acronym	Definition
ABC	American Biogas Council
AKART	All Known, Available, and Reasonable methods of Treatment
BMPs	Best Management Practices
BNR	Biological Nutrient Removal
CW	Constructed Wetland
DIN	Dissolved Inorganic Nitrogen
DNR	Washington State Department of Natural Resources
ENR	Enhanced Nutrient Removal
DO	Dissolved oxygen
EPA	US Environmental Protection Agency
ERGF	Enhanced Recirculating Gravel Filter
HAB	Harmful Algae Blooms
IS	Implementation strategy
LA	Load Allocations, under NPDES permit
LOTT	Lacey, Olympia, Tumwater, Thurston Clean Water Alliance
MWCI	Marine Water Condition Index
MWQ	Marine Water Quality
NANOOS	Northwest Association of Networked Ocean Observing Systems
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
NRCS	Natural Resources Conservation Service
OA	Ocean Acidification
OPG	Olympic Property Group
ORCA	Oceanic Remote Chemical Analyzer
OS	Open Standards (for the Practice of Conservation Science)
OSS	Onsite Sewage Systems/Septic
PIC	Pollution Identification and Correction Programs
PNNL	Pacific Northwest National Laboratory
PSNSRP	Puget Sound Nutrient Source Reduction Project
PST	Paralytic Shellfish Toxin
SPARROW	Spatially Related Regressions on Watershed attributes Model
SSM	Salish Sea Model
SST	Sea Surface Temperatures
STI	Straight to Implementation projects
TMDL	Total Maximum Daily Load
USACE	US Army Corp of Engineers
USDA	US Department of Agriculture
USFWS	US Fish and Wildlife Service
USGS	United States Geological Survey

VDWB	Vegetated Denitrifying Woodchip Bed
VGRF	Vegetated Recirculating Gravel Filter
VS	Vital Sign
WAC	Washington Administrative Code
WLA	Waste load Allocations, under NPDES permit
WQS	Water Quality Standards
WWTP	wastewater treatment plants

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