NATURAL RESOURCES

Bull Kelp Monitoring in South Puget Sound in 2017 and 2018

June 10, 2019

WASHINGTON STATE DEPT OF
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Cover photo: Bull kelp bed near Point Defiance State Park. Credit: Aaron Barna

The Nearshore Habitat Program is funded by the Washington State Department of Natural Resources as part of the agency's work as steward of public lands to ensure environmental protection (www.dnr.wa.gov/programs-and-services/aquatics). It is a component of the Puget Sound Ecosystem Monitoring Program (PSEMP) (www.psp.wa.gov/evaluating-PSEMP.php).

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Executive Summary

The Washington Department of Natural Resources (DNR) is the state steward of 2.6 million acres of state-owned aquatic land. DNR manages aquatic lands for the benefit of current and future citizens of Washington State. As part of this responsibility, DNR's Nearshore Habitat Program monitors the health of nearshore vegetation and other indicators of habitat health along Puget Sound's shorelines.

Kelp refers to large brown seaweeds in the order Laminariales. More than 20 species of kelp occur in Washington State. In greater Puget Sound (the portion of the Salish Sea within Washington State), bull kelp (*Nereocystis luetkeana*) is the primary kelp species that forms a floating canopy. Kelp provides critical habitat to a wide range of species and other ecosystem functions, including primary production and cycling of nitrogen. Kelp also responds to a variety of environmental conditions.

There is concern that bull kelp is declining within the Salish Sea, yet data are limited.

During 2017 and 2018, we collected data on bull kelp status and condition in South Puget Sound, the southernmost basin of the Salish Sea. We compared our observations with earlier surveys, where available. We also assessed environmental factors that are known to influence bull kelp performance and condition. In a preliminary effort to compare beds in good and poor condition in different regions of the Salish Sea, we compared bed characteristics and conditions at two sites in South Puget Sound (SPS) to a bull kelp bed in the eastern Strait of Juan de Fuca.

Findings:

<u>In 2017, bull kelp had limited distribution in South Puget Sound (SPS). Major declines occurred at all four beds that were monitored between 2013 and 2018.</u>

- In 2017, bull kelp beds occurred along 2% of the SPS shoreline. The vast majority of shorelines with bull kelp (87%, 9.5 km) were located along the Tacoma Narrows and its approaches at Day Island and Fox Island. The shoreline with the next largest proportion of bull kelp was located at Squaxin Island (11%, 1.2 km), the southernmost historically recorded bed within Puget Sound.
- Intensive monitoring at Squaxin Island, Brisco Point, Devils Head, and Fox Island showed that bull kelp bed area significantly declined at all four locations in 2017 and 2018, compared to 2013. Two of the four beds were absent in 2018. No beds were intensively monitored in the Tacoma Narrows between 2013 and 2018, an area that appeared stable over this time period.
- The most detailed recent data record is from Squaxin Island. The smallest bed area was recorded in 2017, 18% of the 2013 areal extent. Maximum bed depth

also decreased significantly. While both bed area and maximum bed depth increased slightly in 2018 relative to 2017, both measures remained a fraction of 2013 values.

Bull kelp was more abundant in the Tacoma Narrows vicinity and in healthier condition than other locations in SPS. We compared Salmon Beach (in the Tacoma Narrows) to Squaxin Island (the innermost bull kelp site in SPS), using a series of metrics of stressor intensity and kelp morphology (detailed below). Better condition in the Tacoma Narrows vicinity may be related, in part, to strong currents and intense tidal mixing.

- The length, weight and number of bull kelp blades was significantly greater at Salmon Beach than at Squaxin Island. Additionally, at Squaxin Island almost all bull kelp individuals lacked blades by September, whereas robust blades persisted at Salmon Beach. Bull kelp blades are important because they are the location for both photosynthesis and reproduction.
- Mean maximum bed depth was significantly deeper at Salmon Beach than at Squaxin Island (-7.1 m and -3.0 m MLLW, respectively). Vegetation tends to grow deeper in areas with greater water clarity.
- Summer water temperatures at Squaxin Island greatly exceeded thresholds that are known to be deleterious to bull kelp (up to 20 °C). Nearshore water temperature was consistently significantly higher (1-4 °C) at Squaxin Island than at Salmon Beach between June and September. Additionally, strong gradients with temperatures up to 3°C higher were observed in the shallows at Squaxin (compared to 1°C at Salmon Beach).
- Nitrogen concentrations were significantly higher at Salmon Beach, with values above 10 µmol/L throughout the year. At Squaxin Island concentrations diverged from the Salmon Beach concentrations between May and August and dropped to 1-5 µmol/L, near or below the threshold for kelp growth. Kelp generally requires high nitrogen levels, although human-derived nutrients and associated conditions are deleterious to kelp.
- Northern kelp crabs (*Pugettia producta*) were abundant at Squaxin Island and rare at Salmon Beach in 2018. Bull kelp is known to be the preferred food for kelp crabs in this region. Grazing by kelp crabs could have influenced kelp abundance and condition at Squaxin Island.
- Levels of epiphyte and endophyte colonization on bull kelp bulbs and stipes were substantially higher at Squaxin Island than at Salmon Beach. The proportion of plants with physical damage to the bulb was also higher.

In a preliminary study of regional differences in bull kelp characteristics and environmental conditions, we compared SPS sites hypothesized to be in good (Salmon Beach) and poor (Squaxin Island) condition to a location in the eastern Strait of Juan de Fuca (North Beach) that we hypothesized to be in good condition. We found many similarities between the sites hypothesized to be in good condition (Salmon Beach and North Beach), despite their geographic separation.

- Salinity, temperature and nutrient levels at Salmon Beach were more similar to North Beach than Squaxin Island despite the geographic proximity of Salmon Beach to Squaxin Island in SPS.
- Morphometric sampling showed that bull kelp bulbs and blades at North Beach were the largest, followed by Salmon Beach and Squaxin Island. These results underscore the importance of local conditions, and perhaps genetics, on morphometrics.
- Levels of epiphyte/endophyte colonization and physical damage at North Beach were fairly low, and similar to levels at Salmon Beach.

Given the importance of bull kelp as a habitat and environmental indicator, the observed losses in SPS in recent years are a cause for concern.

We observed limited distribution of bull kelp in SPS and substantial losses over recent years. Similarly, other organizations have reported losses to bull kelp extent in the Salish Sea in recent years. Bull kelp is a critical habitat that forms the foundations of food webs that support healthy rockfish, salmon and orca populations. Research findings from other locations suggest that important candidate stressors include: recent warm climate conditions, kelp crab grazing, water quality degradation, trophic changes, and decreased spore availability. Investigations into stressors could elucidate the causes of observed bull kelp forest losses and poor condition in SPS and the Salish Sea. Information on trends in distribution over longer time periods could further inform our understanding of key stressors and impacts to species that rely on kelp habitats.

1 Introduction

1.1 Aquatic Vegetation Monitoring at DNR

The Washington Department of Natural Resources (DNR) manages 2.6 million acres of state-owned aquatic lands for the benefit of current and future citizens of Washington State. DNR's responsibilities include ensuring environmental protection while encouraging public access, water-dependent use and utilization of renewable resources (RCW 79.105.030).

As part of its stewardship responsibility, DNR's Nearshore Habitat Program monitors nearshore vegetation and other indicators of habitat health along Puget Sound's shorelines (https://www.dnr.wa.gov/programs-and-services/aquatics/aquatic-science). Research focuses on kelp and seagrass, two marine vegetation types with recognized ecological and management significance. The program inventories these resources in order to better understand distribution and to inform management decision-making. It also tracks trends over time to understand how critical habitats are changing in response to natural and human drivers. Information is used by managers, scientists and citizens from diverse organizations including government, academia, the tribes and non-governmental groups.

1.2 Kelp Biology and Ecology

Kelp refers to large brown seaweeds in the order Laminariales. Globally, 112 species (33 genera) occur in shallow coastal rocky habitats, primarily in temperate and Arctic regions (Bolton 2010). Kelp provides a range of important ecosystem services to nearshore environments, acting as a foundation for biodiverse communities and an engineer that alters water movement and light conditions within forest canopies (Bodkin 1986, Duggins et al. 1989, Krumhansl and Scheibling 2012, von Biela et al. 2016, Teagle et al. 2017). The high primary productivity of kelp forests is on par with that of tropical rainforests (Mann 1973) and may help ameliorate ocean acidification conditions and local nutrient pollution (Kim et al. 2015).

Larger kelp species with canopies that float on the surface can form dense stands, often referred to as kelp beds or kelp forests. Floating kelp species are some of the most iconic and well-studied — especially the giant kelp *Macrocystis pyrifera* (ie. Schiel and Foster 2015). A less extensively studied species, the bull kelp *Nereocystis luetkeana,* forms dense surface canopies on nearshore, rocky reefs from Baja California to Alaska (Gabrielson et al. 2012). Because it is the primary floating kelp species that occurs throughout the Salish Sea, it is the focus of considerable management and ecological interest in the region.

Often described as early successional, opportunistic and ruderal, bull kelp is an annual species that recruits in the spring and generally dies in the winter (Maxell and Miller 1996, Druehl and Clarkston 2016). Like other kelp species, the bull kelp lifecycle consists of two distinct phases: 1). Large (often > 10 m in length) adult sporophytes that produce 2). microscopic zoospores that germinate into male and female gametophytes (Mondragon and Mondragon 2003). Gametophytes reproduce sexually to produce the next generation of bull kelp sporophytes. The microscopic phase of the bull kelp lifecycle is often referred to as a “black box” due to the difficulties in observing and tracking microscopic life stages in subtidal marine environments (Dayton 1985). We know that some microscopic form (gametophyte or sporophyte) overwinters on the benthos before recruiting in the spring (Edwards 2000, Carney and Edwards 2006).

The persistence and health of bull kelp forests is influenced by both environmental conditions and biological interactions (Steneck et al. 2002). Changes to water temperature, nutrient availability and light quality can affect bull kelp recruitment timing, densities, growth rates and resiliency to other stressors (Connell et al. 2013, Wernberg et al. 2016, Rubin et al. 2017). Outside of abiotic conditions, competition for space with other seaweeds (macroalgae) and grazing from common invertebrates can exert significant influence on the persistence of kelp in general and annual species like bull kelp in particular (Duggins 1980, Suskiewicz 2010).

Rising ocean temperatures pose a documented threat to kelp forests worldwide (Harley et al. 2006). Recent marine heatwaves in California and Australia have decimated once thriving kelp forests with recovery slow to non-existent following the return of normal sea surface temperatures (Catton et al. 2016, Wernberg et al. 2016, Thomsen et al. 2019). Similarly, once abundant canopy species in Europe are retreating north to cooler waters in response to rising temperatures (Hawkins et al. 2009).

High temperatures alone can negatively affect adult bull kelp resiliency and spore germination rates (Tera Corp. 1982, Schiltroth et al. 2018). Bull kelp sporophytes can survive between -1.5 and 18 °C but, like other temperate kelp species, grows optimally between 5 and 17 °C (Lüning and Freshwater 1988, Bartsch et al. 2008). However, temperatures above 15 °C can negatively affect bull kelp’s ability to repair tissue damage leading to higher than expected mortality, even in the face of otherwise amenable conditions (Tera Corp. 1982). In laboratory investigations, bull kelp sporophytes quickly died at temperatures above 18 °C (Tera Corp. 1982). High temperatures also negatively impact spore germination rates which decrease from an average of 64% at 10 and 15 °C to 31% at 17.5 °C (Schiltroth et al. 2018).

Bull kelp photosynthesis, growth and reproduction also varies depending on light availability (Hurd et al. 2014). Sporophyte sorus production, spore germination, gametophyte reproduction and germling saprophyte growth may be delayed or impeded at low irradiances (Vadas 1972, Carney and Edwards 2006,). For example, Vadas (1972) found that only 1% of bull kelp gametophytes attained fertility when exposed to 2 µmol/m^2/s of light for three weeks. As light availability increases, photosynthetic rates peak at saturation irradiances – 150 to 250 µmol/m^2/s (Hurd et al. 2014). However, saturation levels may be significantly lower for microscopic lifestages. Above saturation irradiances, photostress and UV exposure can lead to cellular damage and tissue death (Swanson and Druehl 2000).

Light availability in the Puget Sound is highly seasonal, increasing 10-fold from December (minimum daylight hours) to July (maximum daylight hours) (Knapp n.d.). Spring and summer benthic light availability is further influenced by the timing of daily lower low tides which shift from night to day in late-winter (February/March). Several studies have observed that kelp recruitment in our region coincides with this late-winter tidal shift (Druehl and Hsiao 1977, Maxell and Miller 1996, Allen 2016).

Turbidity from high suspended sediment loads can block and delay spring recruitment by limiting the amount of light reaching the benthos (Rubin et al. 2017). Increased sediment loads can also be harmful to bull kelp microscopic lifestages (Deiman et al. 2012). Sediment accumulation can smother or block the attachment of microscopic lifestages to surrounding substrates while suspended sediments, if fine enough, can adhere to floating zoospores, suffocating them before they reach the benthos (Watanabe et al. 2016).

Like other marine and terrestrial autotrophs, bull kelp requires adequate nutrient availability to maintain growth rates (Jackson 1977). Bull kelp prefers nitrate over other nitrogen sources but biologically significant thresholds are not known (Ahn et al. 1998). Giant kelp in California requires a minimum of 1 to 2 µmol/L of inorganic nitrogen to maintain typical growth rates but populations in Chile declined at 17 °C despite nitrate concentrations of over 3 µmol/L (Schiel and Foster 2015).

Being a marine organism, all kelp species require salt water. Certain species can acclimate to a range of salinities down to 10 PSU (Fredersdorf et al. 2009). Druehl (1967, 1978) noted that salinity is often correlated with temperature and wave energy in the Salish Sea, and species such as *Macrocystis pyrifera* and *Saccharina. groenlandica* are found in high salinity areas while *Saccharina. latissima* occurs in lower salinity areas. High temperatures combined with diluted salinities may negatively impact spore germination rates.

In addition to environmental conditions, biological interactions play a large role in kelp forest dynamics. Intense grazing pressure can quickly deforest large stands of kelp (Steneck et al. 2002). Sea urchins (*Strongylocentrotus* spp.) are the most infamous kelp forest grazer and regularly create large areas of deforested "barrens" in California and Alaska (Estes and Duggins 1995, Catton et al. 2016). Generally, urchin barrens expand when top-level predators, most famously sea otters, are removed from the food web releasing invertebrate communities from top-down controls (Bertness et al. 2014). However alterations to food web dynamics can be complicated when multiple species from different trophic levels are removed from the system following harvest or displacement from invasive species (Steneck et al. 2013). In Puget Sound there is current interest in identifying changes in food web dynamics due to concerns about possible increased grazing pressure on local bull kelp forests. Rising ocean temperatures also alter trophic systems by increasing the range of many herbivorous invertebrates and fish species. This "tropicalization" of temperate ecosystems leads to increased barrens and reduced forest recovery following disturbances (Ling 2008, Vergés et al. 2016).

1.3 Kelp in Washington State

Washington State lies within the Temperate Northeast Pacific Marine Region, which harbors the greatest diversity of kelp species in the world and encompasses the marine waters extending from California to Alaska (Bolton 2010). More than 20 species of kelp occur in Washington waters (Gabrielson et al. 2012), with the greatest species diversity found on the exposed, rocky open coast and western Strait of Juan de Fuca. Understory kelp is more widely distributed than floating kelp (Fig. 1) but both occur throughout state waters where appropriate habitat conditions exist (DNR 2001). In greater Puget Sound, bull kelp is the primary species of kelp that forms a floating canopy (Mumford 2007).

Limited trend information exists for kelp in Washington State. In a global synthesis study, Krumhansl and others (2016) concluded that kelp abundance was stable along the exposed coastlines in a region spanning Oregon, Washington and Vancouver Island. However, the Salish Sea (Puget Sound and Georgia Strait) was excluded from analysis due to insufficient data.

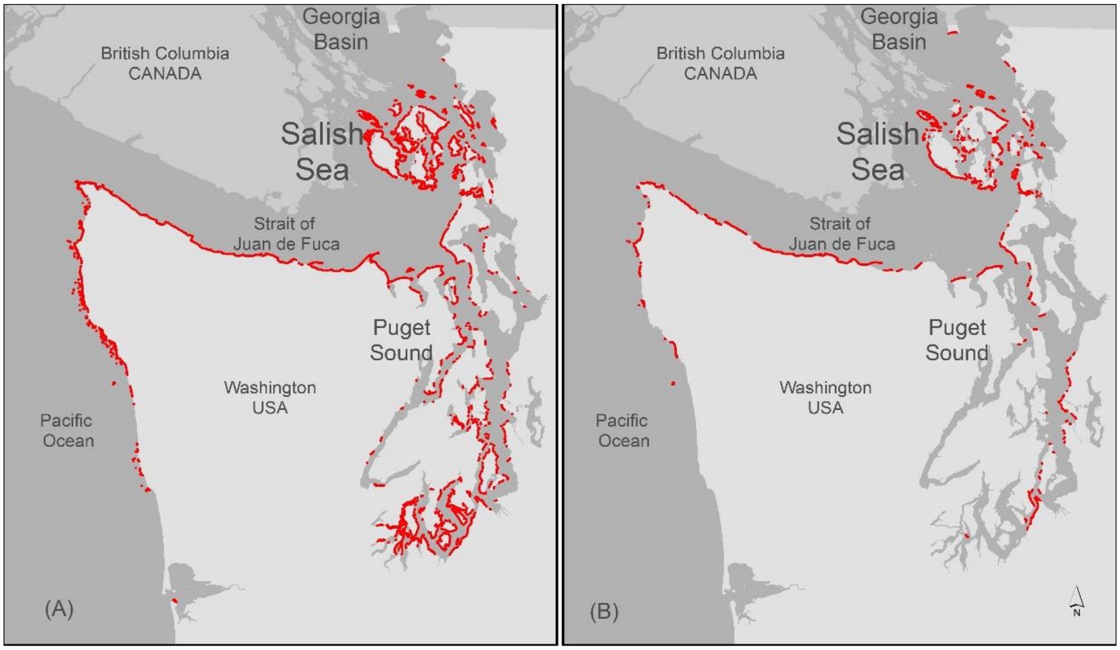


Figure 1. (A) understory and (B) floating kelp distribution in Washington State (Washington Department of Natural Resources 2001).

Pfister, Berry and Mumford (2017) found that floating kelp abundance was stable, with high year-to-year variability, along the open coast of Washington State and Strait of Juan de Fuca between 1989 and 2015. Further analysis showed that over the last century floating kelp abundance was generally stable along the Strait of Juan de Fuca, with possible minor decreases along far eastern portion of the strait, near Port Townsend. (Pfister et al. 2017).

Major concerns exist about floating kelp losses within the Puget Sound based on anecdotal observations and limited studies (ie., Berry et al 2014, Dunagan 2018, Berry and Shull 2018, Palmer-McGee 2019). The area is more urbanized, and naturally experiences elevated water temperatures and longer water residence times than the Strait of Juan de Fuca and open coast (Ebbesmeyer et al. 1988, Heery 2017). Dramatic losses along wave-sheltered shores on Vancouver Island following recent warm water events suggest that shorelines protected from the exposed coast may be especially sensitive to climate-related losses in kelp diversity and abundance (Starko et al. 2019). NOAA has funded conservation and recovery planning to address the impacts of kelp loss, especially to species listed under the Endangered Species Act such as rockfish, salmon and orcas (www.westcoast.fisheries.noaa.gov/protected_species/rockfish/kelp_conservation.html).

While the types of stressors affecting kelp in Puget Sound are broadly similar to other regions, local research has identified unique aspects.

Within Puget Sound, the introduced alga *Sargassum muticum* displaces native kelp species and is less palatable to invertebrates (Britton-Simmons 2004). While urchin grazing is often reported as a major control over kelp persistence in other areas, urchins are generally not observed in high densities in Puget Sound and extensive barrens have not been observed (Carson et al. 2016, H. Carson personal communication May 30, 2019). In contrast, anecdotal reports of increased Northern kelp crab (*Pugettia producta*) numbers have accompanied anecdotal observations of bull kelp declines. Northern kelp crabs (hereafter kelp crabs) are common Puget Sound herbivores that prefer bull kelp over other commonly occurring seaweed species in the area (Dobkowski 2017, Dobkowski et al. 2017). Furthermore, kelp crabs show a preference for younger bull kelp tissue, raising concerns over possible impacts to newly recruited plants in the early spring.

Recent genetic analysis has identified two distinct populations of bull kelp in Washington State. The first occurs along the Strait of Juan de Fuca and open coast while the second is isolated to Puget Sound proper (the waters inside of Admiralty Inlet) and the Strait of Georgia. Puget Sound sites exhibit low allelic richness, with lowest richness at Squaxin Island in SPS (Gierke et al. 2018). Whether this pattern reflects genetic adaptation to local conditions or the result of inbreeding depression in a small and isolated population is unknown.

Weigel and Pfister (2019) found that microbial community composition on bull kelp blades varied geographically and seasonally along the open coast and Puget Sound. They found lower microbial abundances in inner Puget Sound compared to the open coast and that Squaxin Island in the SPS had significantly different microbial communities than all other sites (Ramirez-Puebla et al. in prep). The kelp microbiome can be an indicator of overall host health but little is currently know regarding the relationship between different microbiomes and kelp health and fitness (Marzinelli et al. 2015).

1.4 Project Objectives

In 2017 and 2018, we conducted a series of linked studies to better understand bull kelp distribution and condition in South Puget Sound, an area of concern for bull kelp losses. Questions addressed and methods used:

- What is the current spatial extent of bull kelp in SPS?
    - Comprehensive survey of bull kelp presence/absence along the SPS shoreline in a small motorized boat.
- Has the areal extent of bull kelp beds changed in recent years?
    - Kayak surveys of bed perimeter and minimum/maximum bed depth.
    - Comparison to 2013 surveys.
- How does plant morphology and condition compare among healthy/unhealthy beds in SPS and other regions?
    - Comparison of morphometrics and indicators of condition at Squaxin Island (SPS, poor condition), Salmon Beach (SPS, good condition) and North Beach (eastern Strait of Juan de Fuca, good condition).
- How do water temperature, salinity, nutrient concentrations, light attenuation and suspended sediment concentrations (SSC) compare at current and historical sites in SPS? How do these measures compare in other regions in Puget Sound?
    - Analysis of existing Department of Ecology data from mid-channel stations near sites of interest.
    - Monthly monitoring of nearshore water temperature, salinity, nutrient concentrations, light attenuation and SSC at the deep edge of bull kelp beds (-6 m MLLW).
    - Supplemental sampling of salinity and temperature gradients within beds.

2 Methods

2.1 Study System

The South Puget Sound (SPS) is a 449 square km water body located at the extreme southern end of Puget Sound (Fig. 2). SPS is part of the larger Salish Sea fjord estuary complex and connects to the northeast Pacific Ocean through a network of sub-basins and sills (Burns 1985, Ebbesmeyer et al. 1988). While individual basins in Puget Sound exhibit distinct oceanographic properties (Moore et al. 2008), all are generally characterized by extended water residency, stratification, and strong primary production. These factors increase in intensity further within the fjord.

The most intense currents and mixing occur at the Tacoma Narrows, a narrow 1 km wide channel with a shallow sill (45 m) that connects SPS to the rest of Puget Sound. The greatest tidal range in Puget Sound occurs in SPS with a mean daily difference of 4.6 m in Olympia.

SPS has complex shorelines composed of islands, passages, and shallow inlets. The mean depth of SPS is 37 m (Burns 1985). Due to the area's glacial origin, gravel, sand, and mixed fine shorelines predominate in the intertidal and shallow subtidal zones (Dethier 1990, DNR 2001). Mixed coarse substrates are found along shorelines with strong currents and relatively long fetch, while mudflats are the predominant substrate at the heads of inlets and other shallow embayments (Burns 1985).

We conducted a comprehensive survey throughout SPS, and site-level studies at historical and current bull kelp bed locations (Fig. 2). We sampled one site outside SPS in order to compare plant and bed characteristics in SPS to those in the eastern Strait of Juan de Fuca, an area with greater oceanic influence. The North Beach (eastern Strait of Juan de Fuca) bull kelp forest is genetically distinct from other Puget Sound bull kelp populations (Gierke et al. 2018) and the site is monitored regularly by volunteers with the Jefferson County Marine Resources Committee.

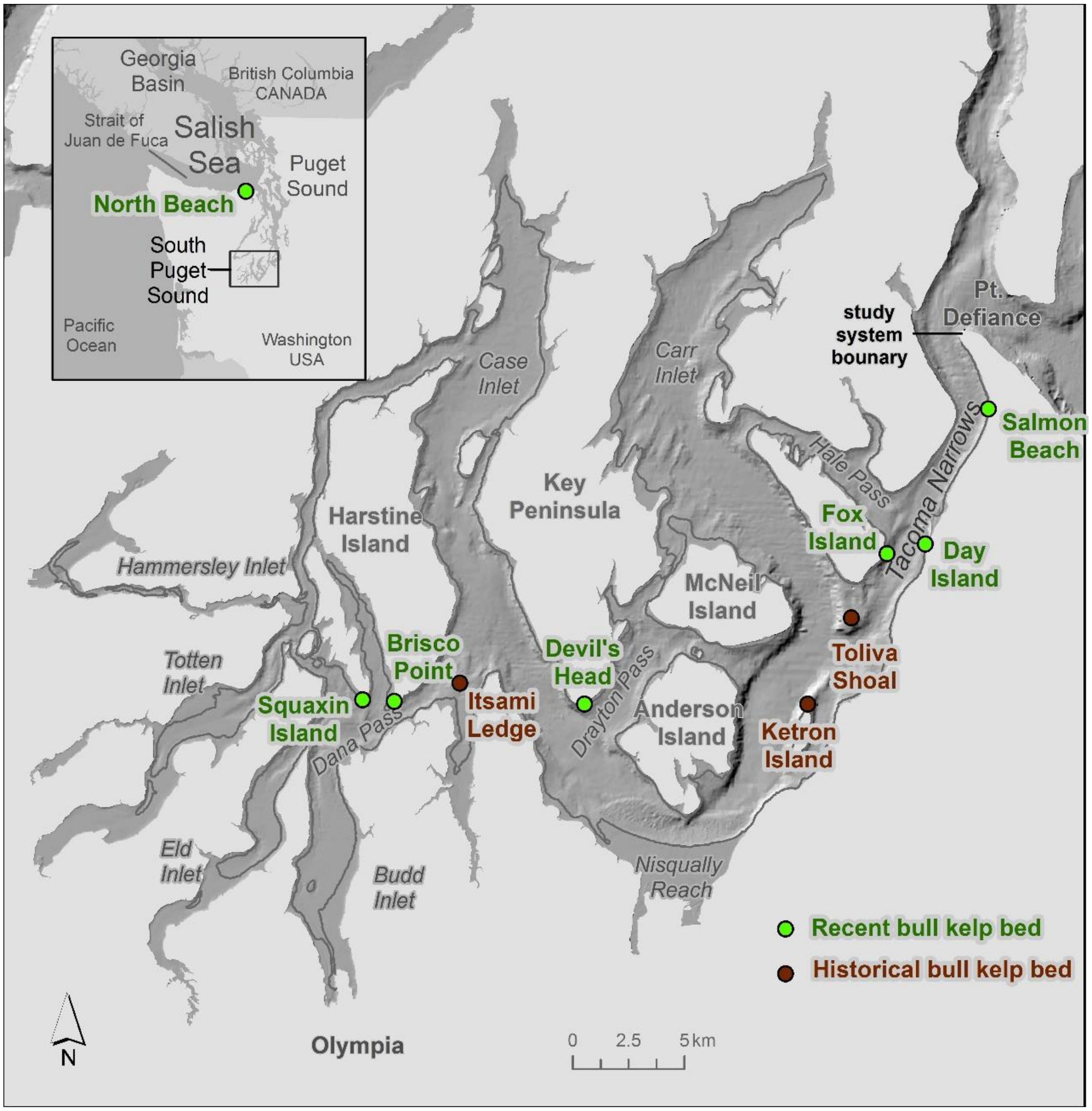


Figure 2. The South Puget Sound (SPS) study system, bounded by Pt. Defiance at the northeastern outlet of the Tacoma Narrows. The inset map depicts the regional location of SPS, at the southern extreme of Puget Sound. Puget Sound and Georgia Basin make up the Salish Sea. We sampled one site in the Strait of Juan de Fuca (North Beach) for regional comparison to SPS.

2.2 Field Work Overview

The following sections describe the methods used to monitor bull kelp distribution, condition and related factors.

All assessments of bull kelp were completed during the hour preceding and following negative (below MLLW) lower low tides in order to minimize the effects of water level and currents on bull kelp detection (Britton-Simmons et al. 2009). Because bull kelp is known to grow during the summer months, annual measures were collected between mid-July and early September for greatest comparability. Some metrics were also monitored for seasonal patterns. Table 1 summarizes the dates of kelp bed area, depth distribution and morphology assessments. Other variables that were not dependent on negative low tides, such as salinity and temperature profiles, light attenuation and suspended sediment concentrations (SSC) were monitored during low tides of lower amplitude.

Some components of the field work are not included here and will be reported in future analyses, including:

- Seasonal and observer-based differences in bed area.
- Density estimation.
- Continuous underwater monitoring of temperature and depth at Squaxin Island.
- Drone-based remote sensing at Squaxin Island and North Beach.

2.3 Spatial Extent of Bull Kelp in South Puget Sound

In 2017, we mapped the linear shoreline extent of bull kelp beds in SPS by slowly meandering along the shoreline between depths of -1 m and -6 m (MLLW), the approximate depth range for bull kelp beds in SPS in recent years (Berry 2017). Shallow mud and sand embayments, such as the heads of Case, Totten, Hammersley and Eld Inlets, were excluded from the survey because they lacked appropriate kelp habitat.

All surveys were completed in a 5.2 m Boston Whaler with at least two observers, one posted at the bow and the other mid-ship. Boat speeds were slow, generally 1 m/s, in order to navigate safely in the shallows and to identify submerged and floating vegetation types. Surveys took place between July and early September, during weather conditions of negligible water surface roughness, within one hour of a negative lower low tide (MLLW). Handheld Garmin GPSMap 62s or 78sc units recorded the location of the boat during navigation. The spatial accuracy during surveys was ± 2.1 – 2.7 m (estimated by instruments based on satellite coverage at the time). Scientists classified kelp as present or absent along the navigation track. The threshold for bull kelp presence was a single bulb that was attached to substrate. In areas with bull kelp beds, the boat navigated along the deep edge of the bed, staying 1-3 m waterward of the deepest plants, which were often submerged.

In the office, a linear model of bull kelp bed distribution throughout SPS was created by transferring presence/absence information to a common isobath in a geospatial database using ArcGIS 10.4 (ESRI 2011). We selected the -6.1 m (MLLW) isobath line because it represents a general maximum depth of bull kelp beds in SPS based on field observations (Berry 2017), and a relatively high-quality digital isobath exists for this depth. The digital isobath was derived from gridded bathymetric data (Nysewander et al. 2005). The minimum linear extent of a bull kelp observation was set at 4 m, based on rough estimates of the distance a single stipe and blades streamed in both directions along the shoreline during incoming and outgoing tidal currents. Gaps between plants needed to span more than 20 m (4 boat lengths) in order to be classified as absent. These precise thresholds should not belie the approximate nature of these data. While the survey detected single bulbs, it remains a generalized depiction of the linear extent of bull kelp beds along SPS shorelines.

While we noted differences between the 2017 and 2013 extents of bull kelp, the datasets are not directly comparable because the 2013 survey excluded much of the Tacoma Narrows. Changes that we noted over time in linear extent were evident in higher resolution surveys of areal extent.

2.4 Bed Area at Sites

Scientists mapped bull kelp bed area at selected sites from single, sit-on-top kayaks (4.3 m length) or a stand-up paddleboard (3.2 m length). Handheld Garmin GPSMap 62s or 78sc units recorded the location of the boats during navigation. The spatial accuracy during surveys was ± 2.1 – 2.7 m (reported by instruments based on satellite coverage at the time). The delineation method used a ‘connect-the-dots’ approach of kayaking around the perimeter of the bed, from the outermost bulb to outermost bulb, along the deep and shallow edges and the alongshore boundaries. Both floating and submerged bull kelp plants were included.

Any bulbs within 20 m were grouped within the same bed. We adopted a fairly large distance threshold based on experimentation with a variety of thresholds. We found that, in areas with low bull kelp density, surveys that employed smaller distance thresholds and defined multiple beds were less consistent (Berry 2017). The 20 m threshold encompassed the distance that was easily observed from a slow-moving kayak or paddleboard.

Whenever time allowed, we mapped the perimeter at least three times in order to allow for statistical comparisons. Most annual estimates were based on replicates from the same day in summer, between late June and mid-August. In one case, schedule constraints led to combining two separate dates within the seasonal window (Table 1). After observing early seasonal die back of the bull kelp at Squaxin Island in mid-July in 2016 (Berry 2017), we prioritized surveys in areas with warmer water temperatures earlier in the summer.

We compared bed area estimates from 2017 and 2018 to previous surveys collected with similar methods. Previous surveys existed from 2013 at four sites. Surveys were also completed at Squaxin Island in 2014 and 2016 (Berry 2017).

2.1 Depth Distribution

Minimum and maximum bed depth were measured at 13 regularly placed across-shore transects by lowering a weighted tape measure to the benthos at the shallowest and deepest plant along the transect. Transect boundaries began and ended where kelp occurred within 5 m in either alongshore direction. Measured depths were subsequently corrected in the office using data from NOAA’s verified tide levels (https://tidesandcurrents.noaa.gov/).

We monitored minimum and maximum bed depth at Squaxin Island in 2017 and 2018. In 2018, we also compared minimum and maximum depth at Squaxin to Salmon Beach (in Tacoma Narrows) and North Beach (near Port Townsend). The two latter sites were hypothesized to be healthier based on visual observation of adult bull kelp individuals, which we later quantified with morphometric and condition measures (below).

We tested for differences in mean minimum and mean maximum depth between sites with a Welch's ANOVA that does not require equality of variance or equal sample size (R Core Team 2019). If significant differences existed, we used a Games-Howell post hoc test to describe pairwise differences in minimum and maximum depth extent between sites ($p < 0.05$) (Peters 2018).

Table 1. Dates of bull kelp bed assessment, including surveys reported in this study and previous surveys that are compared in this report.

Year	Date	Site	Replicates
2013	8/05	Squaxin Island	1
2013	8/05	Brisco Pt.	1
2013	8/06	Devil's Head	1
2013	8/06	Fox Island	1
2016	7/18	Squaxin Island	3
2017	6/23	Squaxin Island	2
2017	7/12	Squaxin Island	1
2017	7/22	Brisco Pt.	1
2017	8/22	Fox Island	1
2017	8/22	Devil's Head	2
2018	7/23	Squaxin Island	4
2018	7/25	Brisco Pt.	1
2018	7/26	Salmon Beach	5
2018	7/28	Devil's Head	1
2018	8/27	Fox Island	3
2018	9/05	Squaxin Island	2
2018	9/06	Salmon Beach	2

2.2 Morphometrics and Condition

In 2018, we collected morphometric measurements and noted the condition of individual bull kelp individuals at three sites: Squaxin Island, Salmon Beach and North Beach. Measurements were collected along 13 equidistant, across-shore transects that spanned the full length of the bull kelp forest. Two points were randomly selected along each transect for measurement, and the closest individual to each point was measured. At Salmon Beach and North Beach, we assessed 13 transects (n = 13 minimum and maximum measurements for each site). At Squaxin Island, two transects were excluded because no bull kelp occurred along them (n = 11 for each variable). Individuals were measured in late July at Squaxin and Salmon Beach because plants in SPS have been observed to die back at some sites by August. Measurements took place in early September at North Beach, where the bull kelp bed has been observed to expand into the fall by Jefferson County MRC volunteers.

For each kelp individual, we counted the number of blades and measured the bulb diameter and sub-bulb diameter (15 cm below the bottom of the bulb) to the nearest millimeter using calipers, and blade length using a measuring tape. To assess the condition of each individual, we counted the number of crabs present on the individual. We also visually classified the level of endophyte/epiphyte cover and the severity of physical damage (absent, low, medium and high). Visual classification levels were based on proportion of damage and endophyte/epiphyte cover of the bulb and blades (low = up to 1/3 coverage, medium = 1/3 to 2/3 coverage and high > 2/3 coverage). Mechanisms for physical damage were not identified but we include both grazer activity and herbivory as potential causes. We then weighed the blades of the kelp plant non-destructively by placing the blades on a mesh tray on a hand-held fish scale. We tested for site differences using a Welch's ANOVA (R Core Team 2019). If significant differences existed, we used a Games-Howell post-hoc test to determine which sites differed from one another (Peters 2018).

2.3 Kelp Crab Size at Squaxin Island

At Squaxin Island in 2017, we observed a narrow range of kelp crab sizes on bull kelp individuals that increased markedly over the summer. In 2018, we explored this apparent trend by measuring kelp crab size during three sampling occasions over the growing season. We measured the carapace width of the first crab we encountered at random points in the shallow, middle and deep portions of the bed on 11 across shore transects through the bull kelp bed. Carapace width increased over the summer and seemed similar throughout bed. (Note that in June, we only sampled the western half of the bed (n = 15), while in other months we sampled all transects (n = 33).

2.4 Mid-channel Salinity, Temperature and Nutrient Levels

We characterized water column salinity, temperature and nutrient concentrations at mid-channel stations sampled monthly by the Washington State Department of Ecology (WDOE) near four of our monitoring sites (Table 2). Temperature and salinity data were restricted to the top five meters of the water column. Surface nutrient samples were collected at 0.5 m depth. Mean monthly values were fit to a curve for data visualization. Patterns in the data were interpreted with caution because sample sizes varied among sites, with an order of magnitude fewer samples at NRR001 (Table 2).

Table 2. Water column sample sizes (data from the Washington State Department of Ecology, Marine Water Monitoring Program)

Water Sampling Site	Nearby Bull Kelp Site	Count of monthly data points	Number of years
ADM002	North Beach (current bed)	324	27
NRR001	Salmon Beach (current bed)	36	3
GOR001	Ketron Island (historical bed)	252	21
DNA001	Squaxin Island (current bed)	336	28

2.5 Nearshore Salinity, Temperature, Light Attenuation, Suspended Sediment Concentrations and Nutrient Levels

Monitoring of salinity, temperature and light attenuation occurred monthly at seven sites in the South Puget Sound: Squaxin Island, Itsami Ledge, Devil's Head, Ketron Island, Fox Island, Day Island and Salmon Beach from September 2017 to August 2018 (Figure 2). All sites were surveyed on the same day, within two hours before and after solar noon. Three sampling stations were established along the offshore side of each bed at -6 m depth (MLLW), one in the center of each site and one at each edge. Between one and three stations were sampled at each site during each sampling event, depending on time available. Four sites were prioritized for collection of three samples whenever time and weather allowed (Squaxin Island, Devil's Head, Day Island, Salmon Beach) in order to support more detailed site studies (Calloway 2019).

Temperature and salinity were measured using a weighted *SonTek Castaway®-CTD* instantaneous data sonde. The sonde was cast from the side of the boat and data was quality checked in the field using the sonde's real-time data display. The sonde uses flow-through electrodes to ensure rapid and accurate readings which require sonde casts to be quick and uniform. After each cast, depth profiles were reviewed and, in cases of non-uniform profiles, the sonde was recast to ensure data quality. Review of cast data also allowed for in-field assessment of any water column stratification due to temperature.

Light attenuation was measured by recording simultaneous ambient and in-water photosynthetically available radiation (PAR) measured in quantum flux ($\mu m/m^2/s$). Ambient levels were monitored using a *Li-Cor LI-190R* 2π quantum sensor mounted to the boat. In-water light was measured using a *Li-Cor LI-192* 2π quantum sensor paired with an *Onset HOBO Water Level* data logger attached to a weighted anchor. The anchor assembly was cast in one-meter intervals down to a depth of 5 m, beginning 1 to 3 cm below the water surface. The data logger was kept at each depth interval for a minimum of 30 seconds.

Light attenuation was calculated from pressure corrected depths using Beer's Law:

$$Kd = \ln [Ed_{(z2)} / ED_{(z1)}] \times (z_1 - z_2)^{-1}$$

Where *Ed* refers to the corrected, measured irradiances at depths (z_1 and z_2). In the case of this study z_2 always refers to corrected surface (0 m) depth and z_1 refers to the corrected, deepest depth reached during the cast.

Monthly field filtered water samples were collected from the central monitoring point for nitrate concentrations at four sites (Squaxin Island, Devil's Head, Day Island, Salmon Beach). An acid washed 60 mL syringe with an attached 0.45 µm cellulose acetate filter was filled with water directly from a Van Dorn sampler. A small amount of water was filtered through the syringe to rinse the syringe and syringe filter before rinsing an acid washed 60 mL high density polyethylene (HDPE) bottle with filtrate. The bottle was then filled with filtrate before being placed immediately in a cooler on ice and transported to the Evergreen State College laboratory where they were frozen (-10˚ C) for later transport to the University of Washington's Marine Chemistry Lab for total dissolved nutrient analysis using spectrophotometric methods.

In March, only one sample was taken from each site from a depth of 4 m due to the relatively low water temperatures observed during sampling. From May to September two samples were taken from each site at 0.25 m and 4 m depths in order to assess possible water stratification during late-spring and summer.

For water temperature and salinity data, we calculated the mean integrated water column value for each monthly sampling event. For nutrient concentration data, we calculated the mean of surface and 4 m samples because concentrations at the two depths were similar (Calloway 2019). We tested for differences among sites and months using Welch's ANOVA (Peters 2018, R Core Team 2019). We tested for differences between mean annual light attenuation at all sites using a Welch's ANOVA and a Games-Howell post hoc test for pairwise differences between sites (Peters 2018, R Core Team 2019). We did not analyze monthly light attenuation between sites due to large monthly variation both within and between sites with no clear pattern.

Suspended sediment concentration (SSC) was monitored monthly from December 2017 to September 2018 at Squaxin Island, Devil's Head and Day Island. Salmon Beach SSC was monitored in July and September while SSC concentrations were only monitored at North Beach in September. SSC (mg/L seawater) was assessed by collecting water samples and determining dry weight of sediments after vacuum filtration in the lab.

During site visits, water samples were taken from midwater (4 m depth) at all three casting sites using a Van Dorn sampler. Sampling bottles (1 L Nalgene) were rinsed with water from the Van Dorn before being filled and capped. Bottles were filled from a single cast of the Van Dorn. All bottles were immediately placed in a cooler on ice, transported to the Evergreen State College and processed within four hours of sampling.

Prior to sample processing, 0.45 µm 47 mm cellulose acetate filters were dried for a minimum of 24 hours at 60 °C in individual plastic petri dishes and transferred to a desiccation chamber to cool to room temperature before being weighed on an analytical scale. Any possible static electricity on the filters was dissipated using a *Milty Zerostat 3* anti-static gun before weighing. Prefiltered weight (g) was recorded on each filter's petri dish and set aside until sample processing.

After field collection, water samples were poured from field containers into a 1,000 mL graduated cylinder before being filtered. In the event that less than 1,000 mL was available, the total volume filtered was recorded. After the entire sample volume was filtered, the resulting sediment laden filter was removed using clean forceps and placed back into its individual petri dish. Filter drying and weighing followed the same procedure used in obtaining prefiltered weight.

Suspended sediment concentrations (mg/L) were calculated using the following equation:

$$SSC\ (mg\ /\ L) = \frac{(Prefilter\ weight\ (g) - filtered\ weight\ (g)) * 1{,}000}{water\ volume\ (mL) *\ 0.001}$$

We tested for differences in average SSC between sites using a Welch's ANOVA and a Games-Howell post hoc test for pairwise differences between (Peters 2018, R Core Team 2019). We focused on the overall site averages instead of monthly differences due to inconsistencies in sampling events at some sites over the sampling period.

2.6 Salinity and Temperature Patterns Within Kelp Beds

We collected water temperature and salinity measurements at points within the bull kelp bed at Squaxin Island and Salmon Beach in 2018 to assess temperature and salinity patterns within the sites. To characterize conditions at Squaxin Island, a grid of points with 40 m spacing was placed over the site map and points were chosen from the grid that fell within the bed. At each point, we used a YSI temperature and salinity probe to measure water temperature and salinity at the surface and at two meters deep around the low tide. Since the bull kelp bed at Salmon Beach is very narrow, temperature and salinity measurements were taken at three points along 13 transects; the shallow and deep extent of kelp at each transect, and a mid-point.

We plotted these data in ArcMap to visualize the salinity and temperature gradient present at Squaxin Island and Salmon Beach.

3 Results

3.1 Spatial Extent of Bull Kelp in 2017

In 2017, bull kelp occurred along 11 km of shoreline in SPS, approximately 2% of the entire 452 km shoreline. The vast majority (87%, 9.5 km) of bull kelp occurred in the Tacoma Narrows and along its southerly approaches at Day Island and Fox Island (Fig. 3). Three other beds occurred at Chambers Creek, south of Day Island, and at Devil's Head and Squaxin Island, two sites distant from the Tacoma Narrows. The Squaxin Island bed comprised 11% (1.2 km) of total linear extent of bull kelp in SPS. The Chambers Creek and Devil's Head beds were tiny and low density; they spanned 0.06 and 0.15 km of shoreline respectively and comprised 2% of total linear extent of bull kelp in SPS.

3.2 Bed Area at Sites

In 2017 and 2018, we repeated bed area surveys at four bull kelp sites within South Puget Sound that were surveyed in 2013 (Berry 2017). All showed pronounced declines between 2013 and recent years (Fig. 4). At Brisco Point, the bull kelp bed was small in 2013 (0.12 ha) and absent in both 2017 and 2018. At Devil's Head, bed area declined in 2017 to 3% of the 2013 extent and was absent in 2018. At Fox Island, bed area declined in 2017 to 5% of the 2013 extent. In 2018, only scattered plants remained at Fox Island.

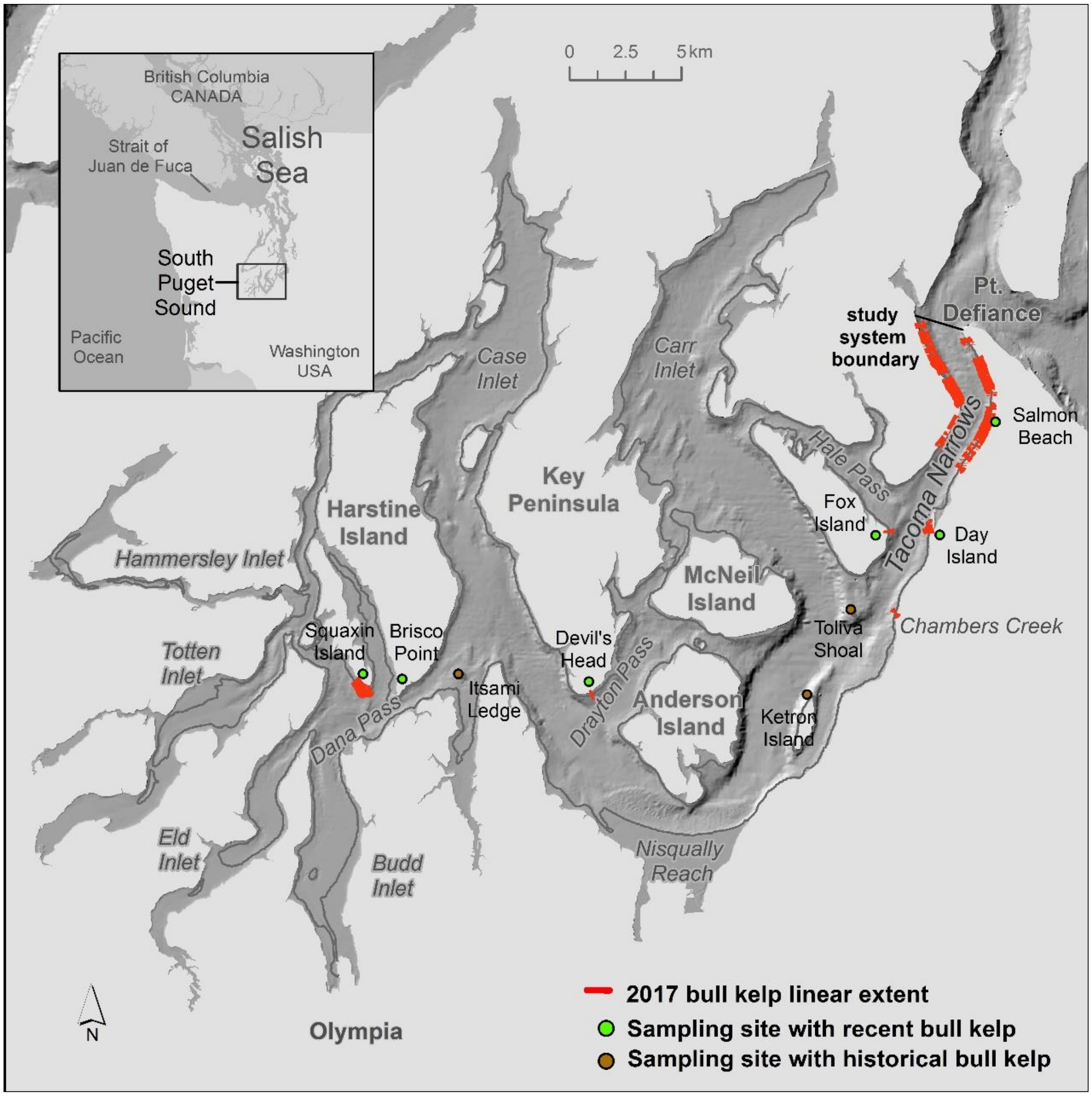


Figure 3. Linear extent of bull kelp in 2017, visualized along the -6 m bathymetry line (MLLW). Red lines denote presence (buffered to be visible at map scale), grey lines denote absence. Shorelines north of the study system boundary were not surveyed.

Our most detailed bed area surveys exist at Squaxin Island. Between 2013 and 2017, bed area decreased to approximately one-fifth of its original size (Table 3). In 2018, it increased to a size similar to 2016, one-third of the 2013 bed area.

Relative to 2013, the Squaxin Island bed contracted in 2017 and 2018 along both the waterward edge and alongshore boundary (Fig. 4). The spatial pattern of loss along the waterward edge coincided with shallowing of the maximum bed depth (Section 3.3).

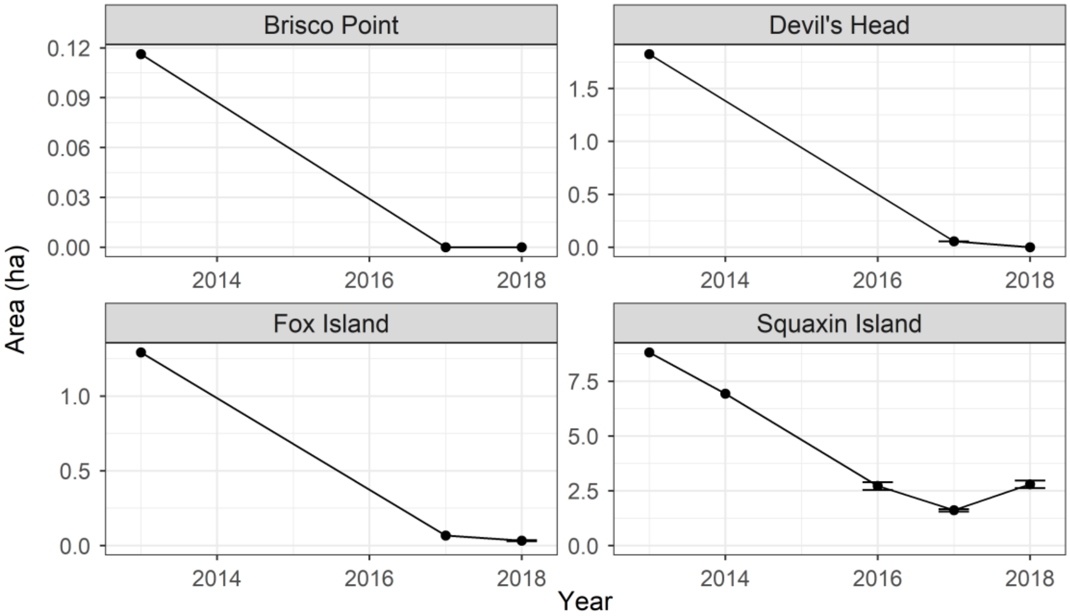


Figure 4. Bull kelp bed area at four sites between 2013 and 2018. Error bars (SD) are included for estimates with three or more surveys in high summer. Note that y-axis range varies among sites.

Table 3. Annual bull kelp bed area at Squaxin Island (2013-2018 except 2015)

year	**area (ha) ± SD**	**% of 2013 area**
2013	8.8	100%
2014	6.9	78%
2016	2.7 ± 0.18	31%
2017	1.6 ± 0.06	18%
2018	2.8 ± 0.18	32%

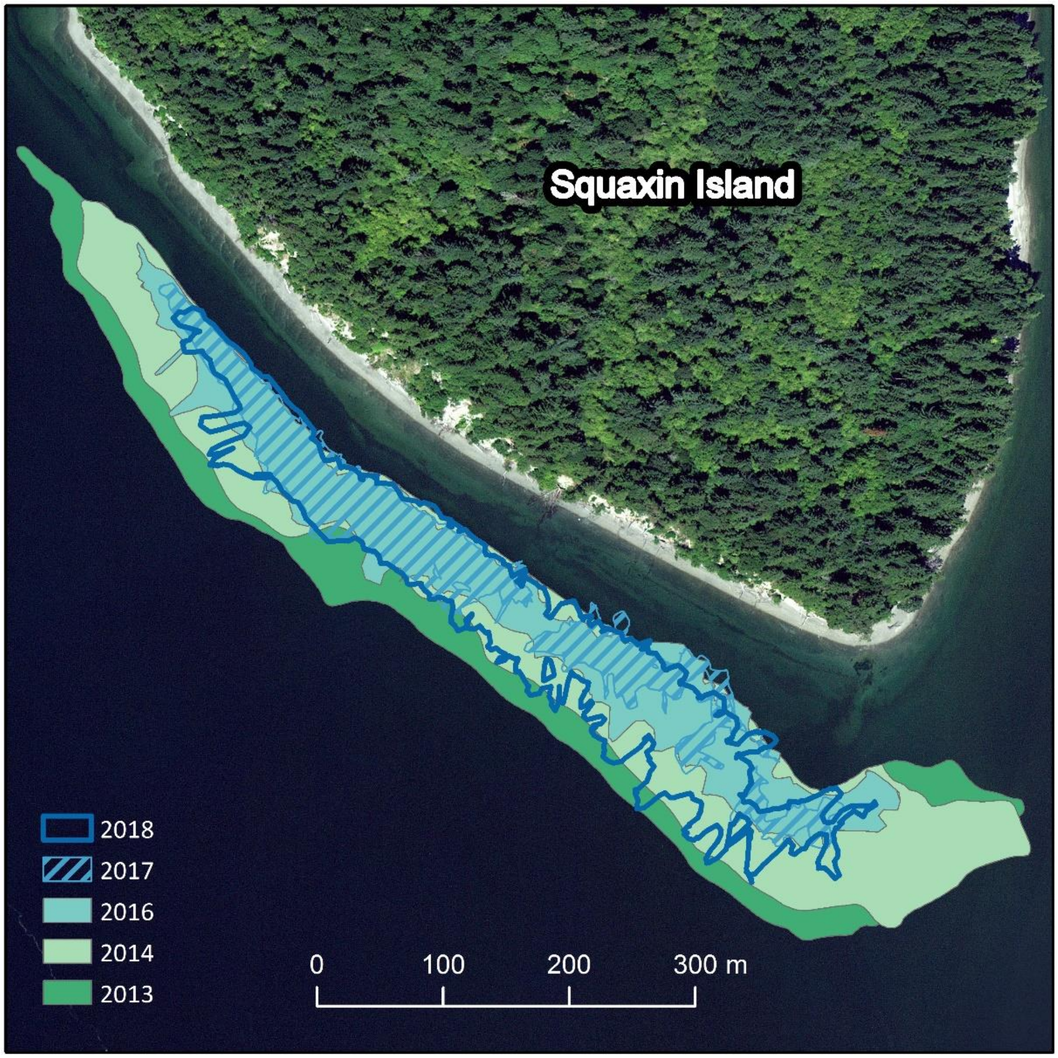


Figure 4. Bull kelp bed perimeter at Squaxin Island between 2013 and 2018 (except 2015).

3.3 Depth Distribution

At Squaxin Island, mean maximum bed depth was deepest in 2013 (mean ± SD = -4.4 ± 0.9 m, Fig. 5) and steadily shoaled to its shallowest estimate in 2017 (mean ± SD = -2.3 ± 0.5 m). Mean maximum depth at Squaxin Island was significantly shallower in 2016, 2017 and 2018 relative to 2013, the first year of monitoring. In 2018, it deepened slightly (mean ± SD = -3.0 ± 0.7 m), an estimate that is not significantly different from the shallower values in 2016 and 2017. Mean minimum depth was not significantly different in all three years it was measured, with mean minimum depth of -1.3 ± 0.4 m in 2018 (± SD) and similar values in previous years.

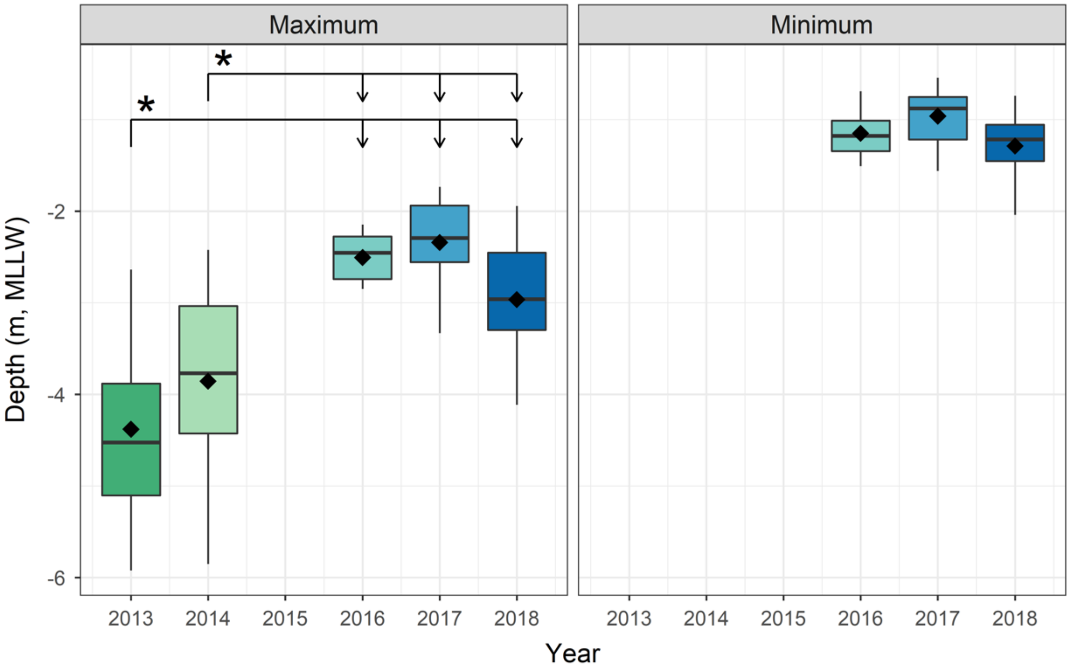


Figure 5. Minimum and maximum depth of bull kelp bed at Squaxin Island between 2013 and 2018 (n=11 for each metric). Diamonds represent the mean. The lines and asterisks indicate significant differences between a single measure and other measures identified by arrows (Welches ANOVA, $F_{4, 31} = 26.51$, $p < 0.001$; Games-Howell post hoc test, $p < 0.05$).

In 2018, we compared maximum depth at Squaxin to two beds in areas that were hypothesized to have healthier bull kelp (Fig. 6). The Squaxin Island bed was the shallowest bed and had a mean maximum depth of -3.0 m (range -2.0 to -4.1 m). At Salmon Beach, mean maximum depth was more than twice the depth at Squaxin Island (mean -7.1 m, range -4.1 to -10.8 m). The kelp bed at North Beach had the deepest mean maximum depth (mean -8.1 m, range -6.5 to -9.7 m).

Maximum bed depth was significantly different at the three sites (Welches ANOVA, $F_{2, 21} = 100.09$, $p < 0.0001$). A Games-Howell post-hoc test identified that Squaxin was significantly shallower than both Salmon Beach ($p < 0.001$) and North Beach ($p < 0.0001$). Maximum depth at Salmon Beach and North Beach were not significantly different ($p = 0.23$). The lack of a significant difference in mean maximum depth at North Beach and Salmon Beach may reflect sample size (n=13), given the generally lower maximum depth values and lower median maximum depth at North Beach (Figure 6).

Minimum bed depth ranged from the shallowest at Squaxin (mean ± SD = -1.3 ± 0.4 m) to the deepest at North Beach (mean ± SD = -2.8 ± 1.1 m). Significant differences in minimum bed depth existed among the three sites (Welches ANOVA, $F_{2, 20} = 16.31$, $p < 0.0001$). A Games-Howell post-hoc test identified that minimum depth at North Beach was significantly deeper than both Salmon Beach ($p < 0.001$) and Squaxin ($p = 0.001$). Minimum depth at Salmon Beach and Squaxin were not significantly different ($p=0.253$).

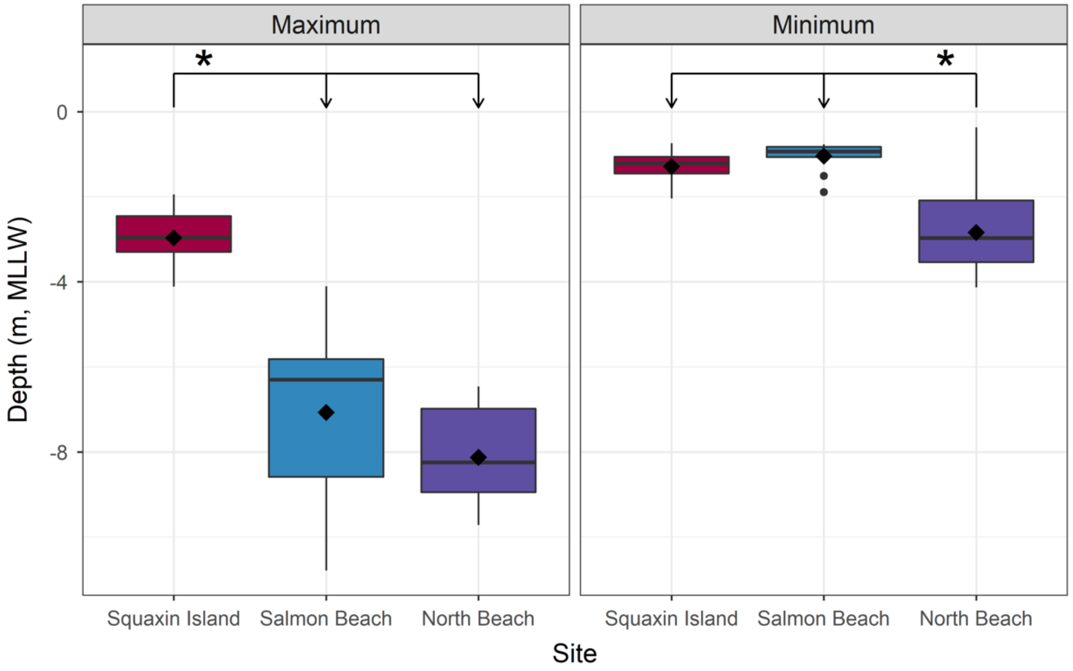


Figure 6. Minimum and maximum depth of bull kelp bed at three sites in 2018 (for each metric, n=11 at Squaxin Island and n=13 at the other sites). Diamonds represent the mean. The lines and asterisks indicate significant differences between a single measure and other measures identified by arrows (Welches ANOVA, $p < 0.001$; Games-Howell post hoc test, $p < 0.05$).

3.4 Morphometrics and Condition

Overall, North Beach had larger bull kelp individuals compared with Salmon Beach and Squaxin Island (Fig. 7). Mean blade length was 6.6 m (range 0.6 to 11.6 m) at North Beach, 3.3 m (range 0.6 to 5.5 m) at Salmon Beach and 1.29 m (range 0.2 to 2.7 m) at Squaxin Island. Mean blade length differed significantly across sites (Welch's ANOVA, $F_{2, 45} = 68.64$, $p < 0.0001$). A Games-Howell post-hoc test identified that blade length was significantly longer at North Beach than Salmon Beach ($p < 0.001$) and Squaxin Island ($p < 0.001$) and Salmon Beach had longer blades than Squaxin Island ($p < 0.001$, Fig. 7A).

Mean blade weight was 9.4 kg (range 0.3 to 31.8 kg) at North Beach, 1.6 kg (range 0.1 to 4.0 kg) at Salmon Beach and 0.6 kg (range 0.02 to 1.6 kg) at Squaxin Island. Blade weight differed significantly across sites (Welch's ANOVA, $F_{2, 41} = 30.28$, $p < 0.001$). A Games-Howell post-hoc test identified that North Beach kelp blades weighed significantly more than blades at Salmon Beach ($p < 0.001$) and Squaxin Island ($p < 0.001$) and Salmon Beach kelp blades weighed significantly more than blades at Squaxin Island ($p < 0.001$, Fig. 7B).

Mean blade number was 24.7 (range 6 to 38) at North Beach, 18.4 (range 5 to 27) at Salmon Beach and 12.2 (range 2 to 18) at Squaxin Island. Blade number also differed significantly across sites (Welch's ANOVA, $F_{2, 47} = 21.97$, $p < 0.001$). A Games-Howell post hoc test identified that the North Beach bull kelp had significantly more blades than Salmon Beach ($p = 0.007$) and Squaxin Island ($p < 0.001$) and Salmon Beach plants had significantly more blades than Squaxin Island ($p = 0.001$, Fig. 7E).

Mean bulb diameter was 77.5 mm (range 57.8 to 97.0 mm) at North Beach, 48.5 mm (range 35.1 to 58.4 mm) at Salmon Beach and 49.4 mm (range 38.6 to 58.5 mm) at Squaxin Island. Bulb diameter differed significantly across sites (Welch's ANOVA, $F_{2, 45} = 95.68$, $p < 0.001$). A Games-Howell post hoc test found that the bulb diameter at North Beach was significantly larger than Salmon Beach ($p < 0.001$) and Squaxin Island ($p < 0.001$). Bulb diameter was not significantly different between Salmon Beach and Squaxin Island ($p = 0.87$, Fig. 7C).

Mean sub-bulb diameter was 53.7 mm (range 25.5 to 68.5 mm) at North Beach, 31.2 mm (range 15.5 to 43.5 mm) at Salmon Beach and 24.8 mm (range 15.6 to 37.6 mm) at Squaxin Island. Sub-bulb diameter differed significantly across sites (Welch's ANOVA, $F_{2, 46} = 75.36$, $p < 0.001$). A Games-Howell post hoc test identified that North Beach plants had significantly larger sub-bulb diameters than Salmon Beach ($p < 0.001$) and Squaxin Island ($p < 0.001$). Sub-bulb diameter was significantly larger at Salmon Beach than at Squaxin Island ($p = 0.002$, Fig. 7D).

All bull kelp individuals at Squaxin Island had some epiphyte/endophyte colonization, with over half the plants having moderate (59%) or high (14%) colonization (Fig. 8). At Salmon Beach, 42% of kelp individuals had no colonization, 54% of plants had low colonization and only 4% had moderate colonization. At North Beach, levels of colonization were more evenly distributed: 31% of kelp individuals had no epiphyte colonization, 23% had low colonization, 31% had moderate colonization and 15% had high colonization.

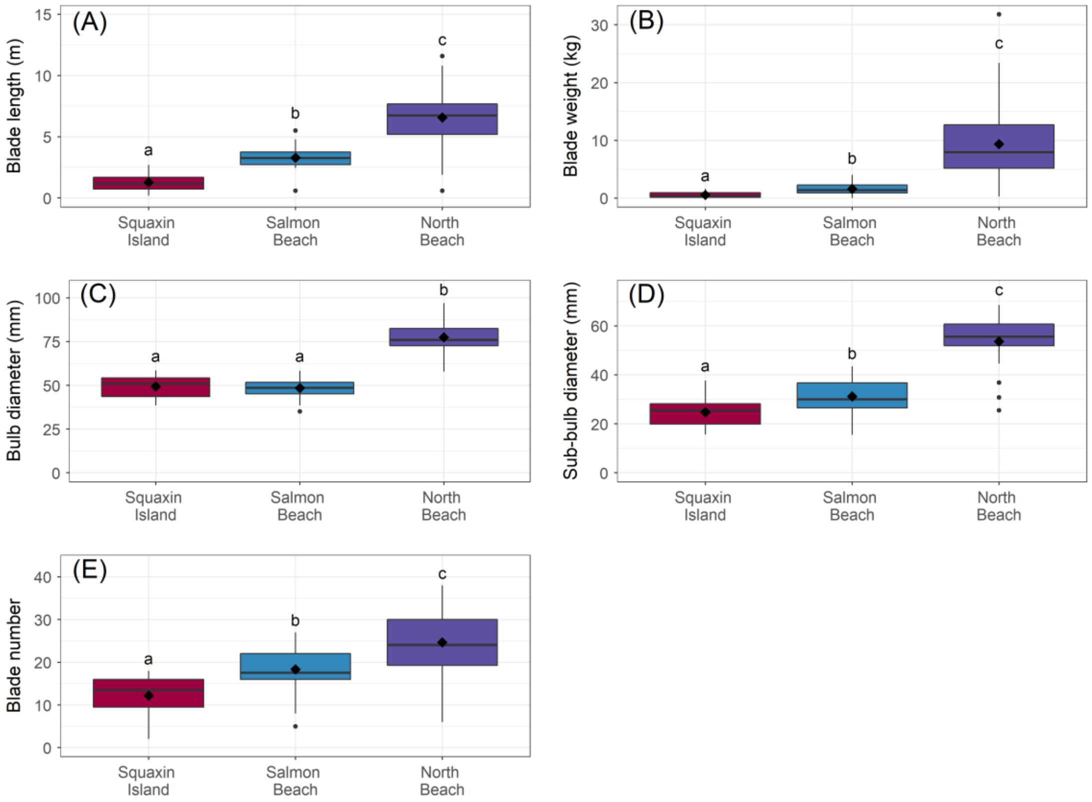


Figure 7. Bull kelp morphometrics at three sites in Puget Sound in 2018, (A) blade length, (B) blade weight, (C) bulb diameter, (D) sub-bulb diameter and (E) blade number. Diamonds represent the mean. Lower case letters indicate significant differences (Welch's ANOVA, $p < 0.001$; Games-Howell post hoc test, $p < 0.05$).

The highest rates of physical damage were recorded at Squaxin Island with 41% of the individuals showing low damage and 32% showing moderate physical damage (Fig. 9). At Salmon Beach, only one individual (4%) was damaged and no individuals showed physical damage at North Beach.

Squaxin Island bull kelp had the greatest number of kelp crabs per individual (mean ± SD = 2.3 ± 3.3) with a maximum of 14 crabs on one individual (Fig. 10). Salmon Beach had no kelp crabs present on individuals measured and North Beach had almost no crabs present (mean ± SD = 0.12 ± 0.3). Two individuals had one crab each.

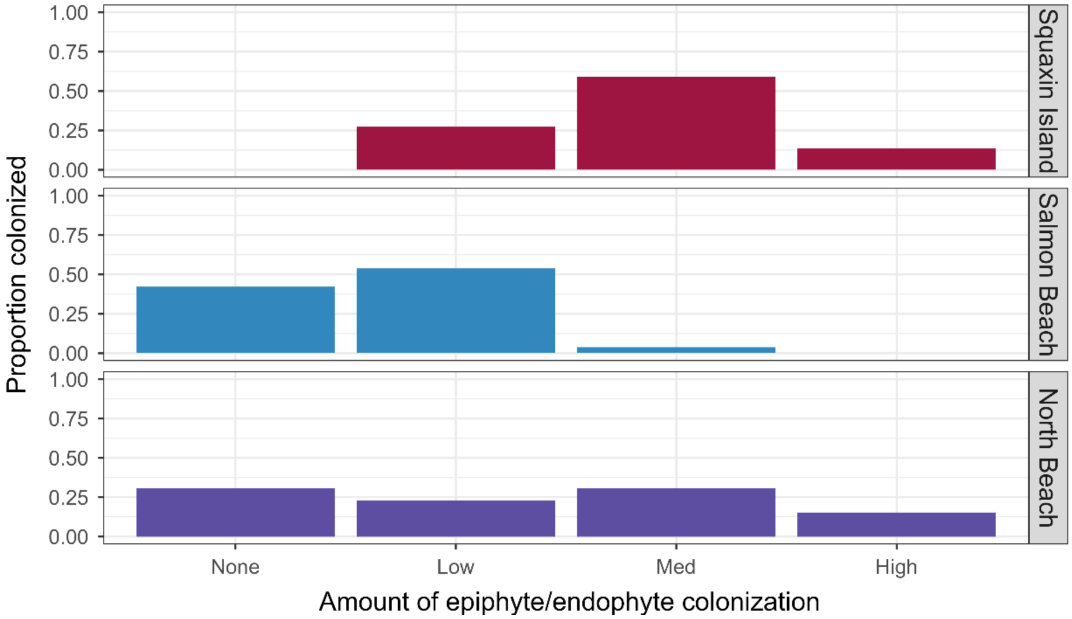


Figure 8. Proportion of bull kelp individuals with epiphyte/endophyte colonization at three sites in Puget Sound in 2018.

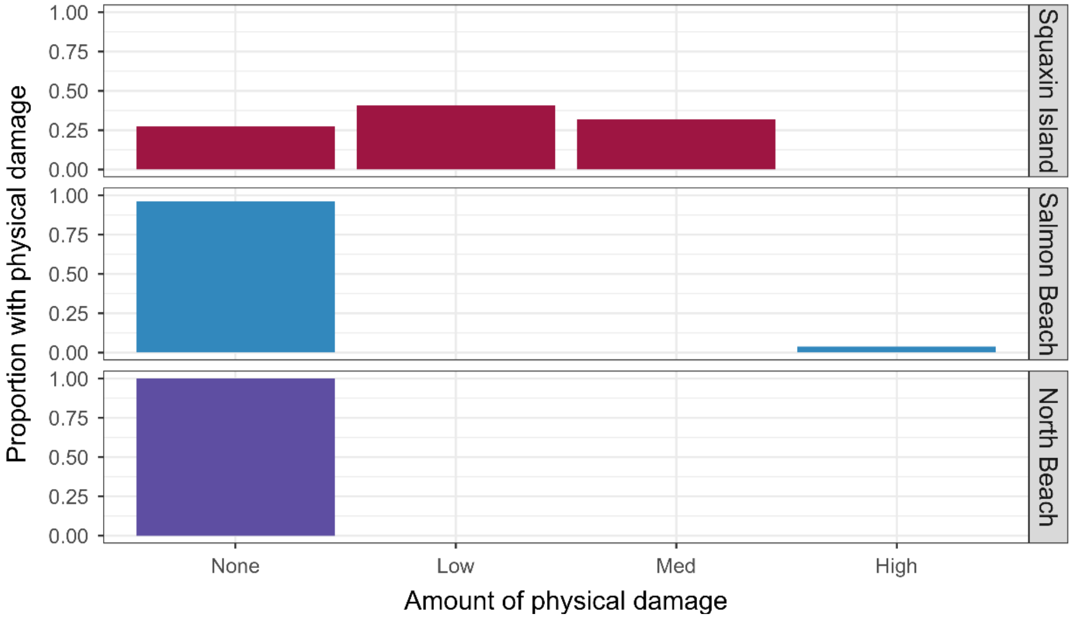


Figure 9. Proportion of bull kelp individuals with physical damage at 3 sites in Puget Sound in 2018.

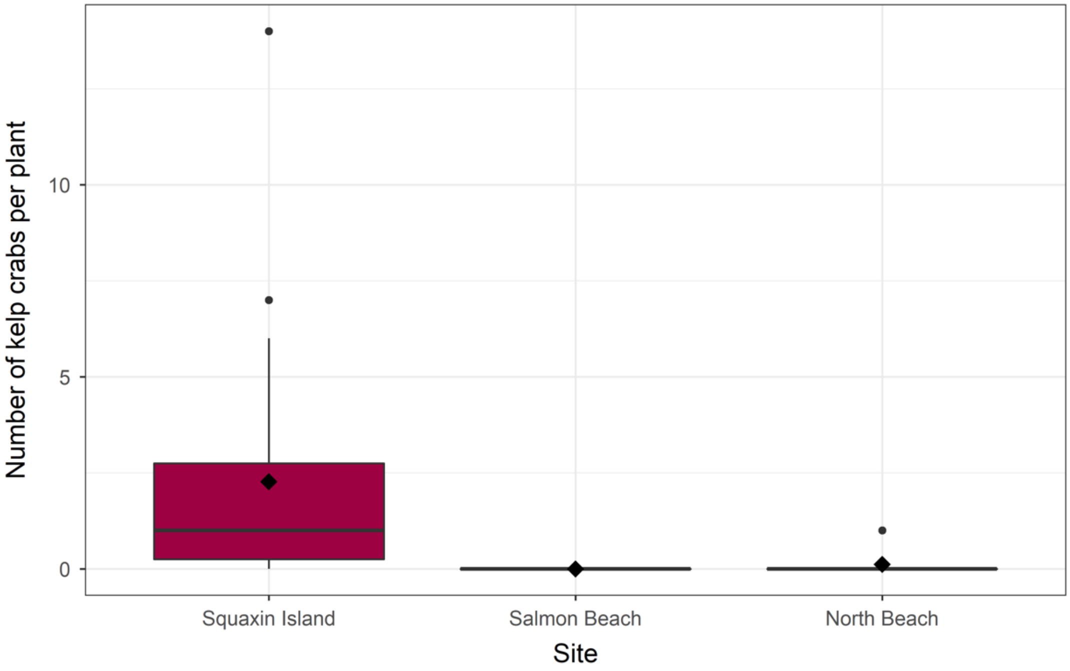


Figure 10. Number of kelp crabs per bull kelp individual at three sites in Puget Sound in late summer, 2018. Diamonds represent the mean.

3.5 Northern Kelp Crab Size at Squaxin Island

Kelp crab carapace width was relatively small in the spring and increased through the summer (Fig. 11). Mean kelp crab carapace width was 18.2 ± 4.4 mm (± SD) on June 12, increased to 39.3 ± 7.3 mm (± SD) on July 24, and further increased to 53.4 ± 9.0 mm (± SD) on September 5. The size ranges at each sampling event were significantly different (Welch's ANOVA, $F_{2,48} = 177.3$, $p < 0.0001$, Games-Howell post hoc test, $p < 0.001$). The September carapace widths were similar to the reported size range of mature kelp crabs (52 to 72 mm; Hines 1982). This pattern strongly suggests the crabs living on the upper portions of the bull kelp plants were an annual population of the same age class. There was no spatial pattern to crab size relative to inner/outer bed or alongshore location. In contrast, Hines (1982) documented multiple age classes in a giant kelp bed, and a gradient of larger crabs in the outer portions of the bed.

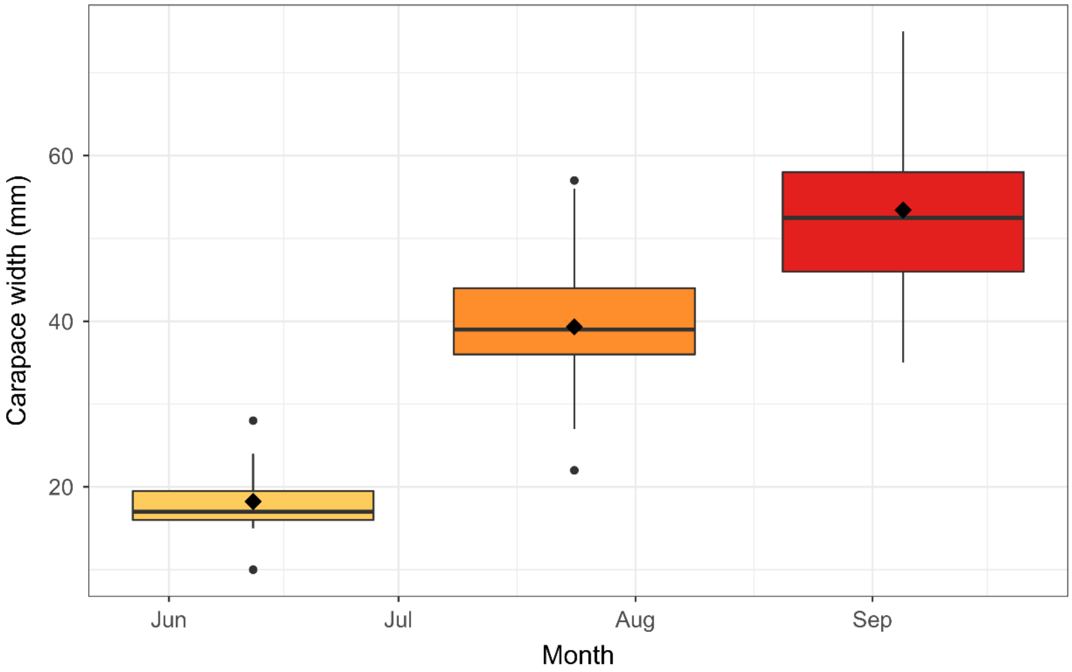


Figure 11. Carapace width of northern kelp crabs on bull kelp stipes and blades at Squaxin Island between June and September 2018. Diamonds represent the mean. (Welch's ANOVA, $F_{2,48} = 177.3$; $p < 0.0001$, Games-Howell post hoc test, $p < 0.001$). Note that only transects 1-5 were sampled on June 12 (n=11 in June and n=33 in July and September).

3.6 Mid-channel Salinity, Temperature, and Nutrients

Long-term monitoring data from the Washington State Department of Ecology (WDOE) showed pronounced spatial differences and seasonal patterns in temperature, salinity and nitrate-nitrite at sites (Fig. 12). Surface water temperature exhibited similar low values in winter months before diverging and peaking in the summer months. The warmest water was measured at Dana Passage, near Squaxin Island while Admiralty Inlet had consistently cooler water temperatures than the other sites throughout the year. The Tacoma Narrows water temperatures were similar to Admiralty Inlet in winter and spring, rising to several degrees warmer during summer and fall. Gorman Point water temperatures were midway between Squaxin Island and Tacoma Narrows values in the summer.

Salinity was higher in the summer and late fall when rainfall and other freshwater inputs were low. Similar to the water temperature pattern, Admiralty Inlet and Dana Passage exhibited the highest and lowest salinity values respectively. The Tacoma Narrows salinity values exhibited high month to month variation rather than a smooth curve, this may reflect the limited number of data points from that site (Table 2), or large fluctuations in salinity caused by freshwater input.

Surface nitrate-nitrite concentrations were high in the winter months, with little difference among sites. Concentrations among sites diverged throughout the spring, with pronounced differences between sites by mid-summer. Nitrate-nitrite concentrations were lowest in the summer, with a gradient of decreasing concentrations into SPS and the most extreme drawdown of nutrients at Dana Passage.

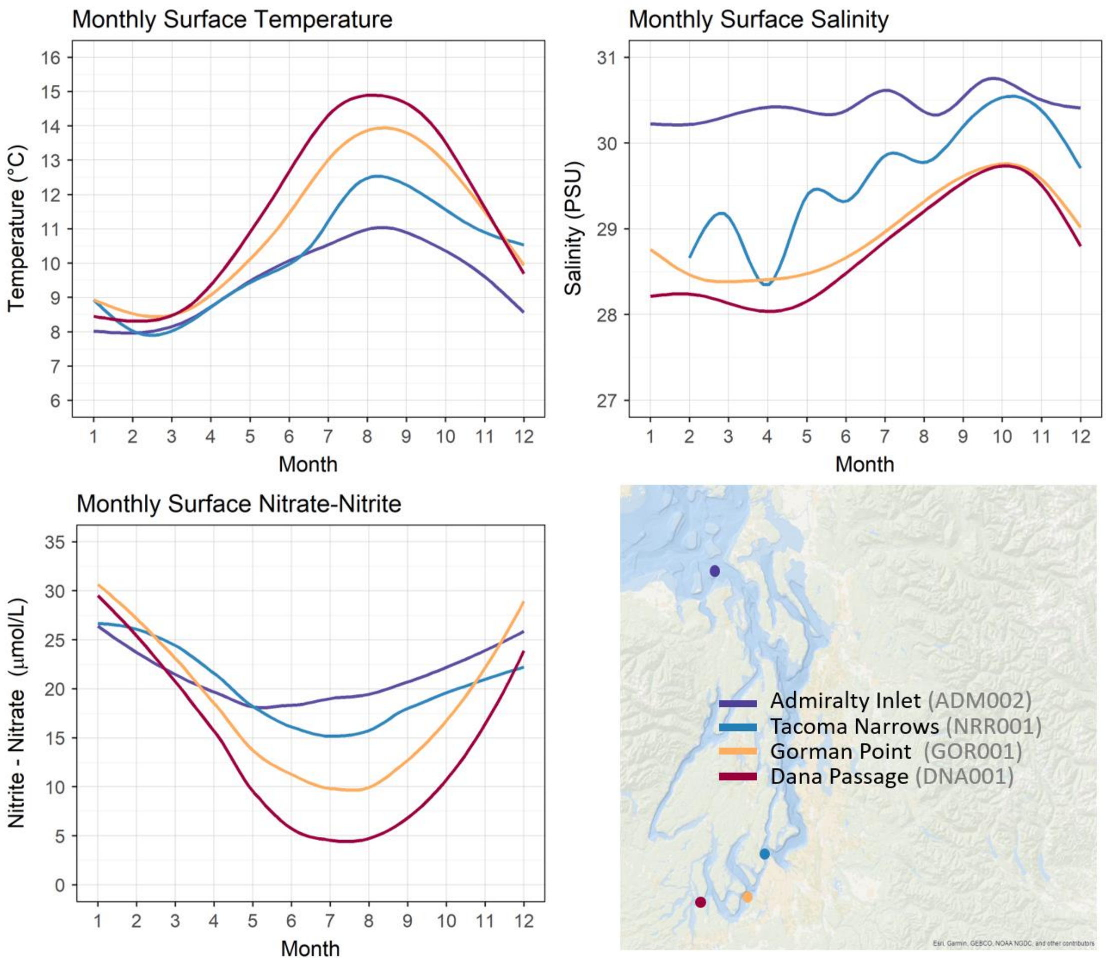


Figure 12. Monthly surface temperature, salinity and nitrate-nitrite at four stations, based on Department of Ecology long-term monitoring data collected at mid-channel stations. Lines represent a curve fit to mean values.

3.7 Nearshore Salinity, Temperature, Light Attenuation, Suspended Sediment Concentrations and Nutrients

Nearshore temperature, salinity and nutrient concentrations showed similar patterns to WDOE’s mid-channel data. However, temperatures were consistently 1 to 2 °C warmer and nitrate-nitrite levels were lower by several µmol/L at nearshore stations (Fig. 13, Appendix A). Salinity was similar in range to WDOE data.

Average water column temperature differed significantly across sites over the entire time period (Welch's ANOVA, $F_{6, 1951} = 25.88$, $p < 0.0001$) and month to month (Welch's ANOVA, $F_{83, 1191} = 691900$, $p < 0.0001$). Water temperature displayed a clear geographic gradient with higher temperatures in more southerly regions (more distant from the Tacoma Narrows) (Fig. 13, Appendix A). Squaxin Island, the southernmost site, consistently had the highest recorded integrated water column temperatures during the spring and summer (May through September, $p < 0.01$) as determined by a Games-Howell post hoc test. Squaxin Island also had the highest nearshore water temperature recorded (16.2 °C in July) in this study. Salmon Beach, the most northerly site, had the lowest spring and summer temperatures observed and temperatures never exceeded 14 °C. Temperatures at Salmon Beach reached a summer high of 13.9 °C in August, reaching peak summertime temperatures at the same time as all other sites except Squaxin Island. Water temperature was similar art all sites during fall and winter (October to March).

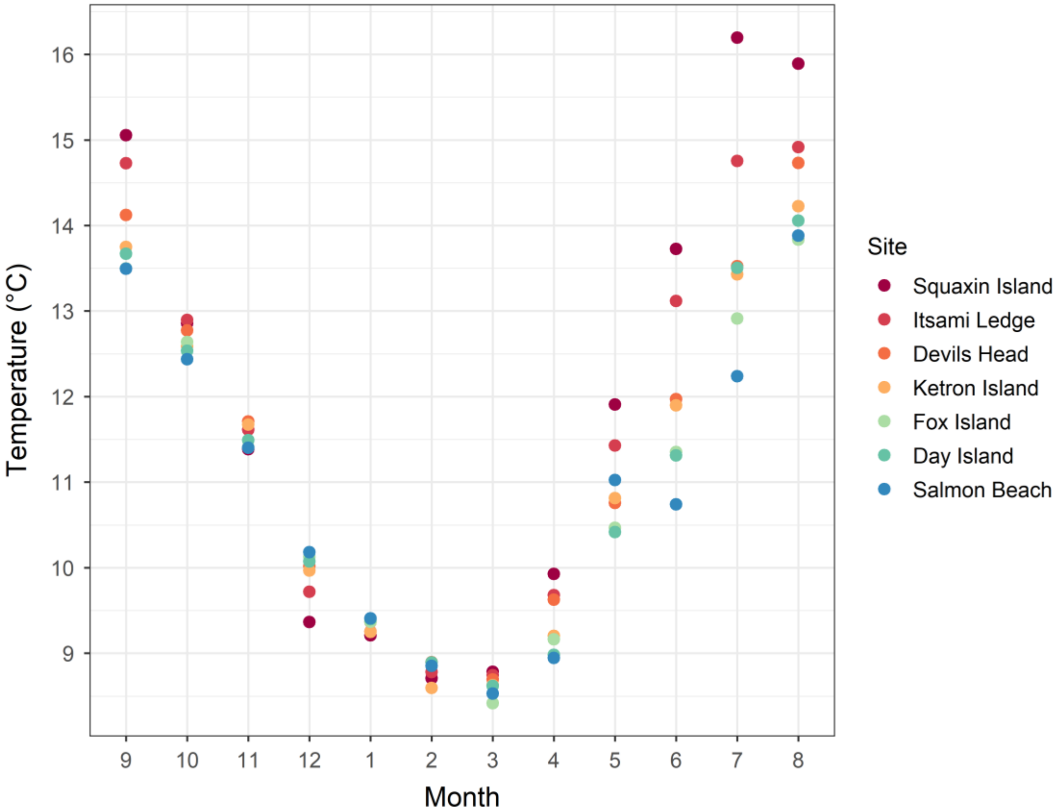


Figure 13. Monthly integrated water column temperature from September 2017 to August 2018 at seven sites in South Puget Sound (-6 m MLLW depth).

The water column in July was well-mixed, even in July and August during peak annual temperatures (Fig. 14). Devil's Head displayed the greatest vertical range (13.2 to 14.0 °C). Day Island showed a similar total range (13.1 to 14.0 °C). However, the greatest differences at Day Island were among areas within the site (distinct casts). We attribute this within-site difference to the broad shoal and spit which shelters a portion of the site. Squaxin Island and Salmon Beach showed vertical ranges for three casts of 16.1 to 16.3 °C and 12.1 to 12.4 °C, respectively.

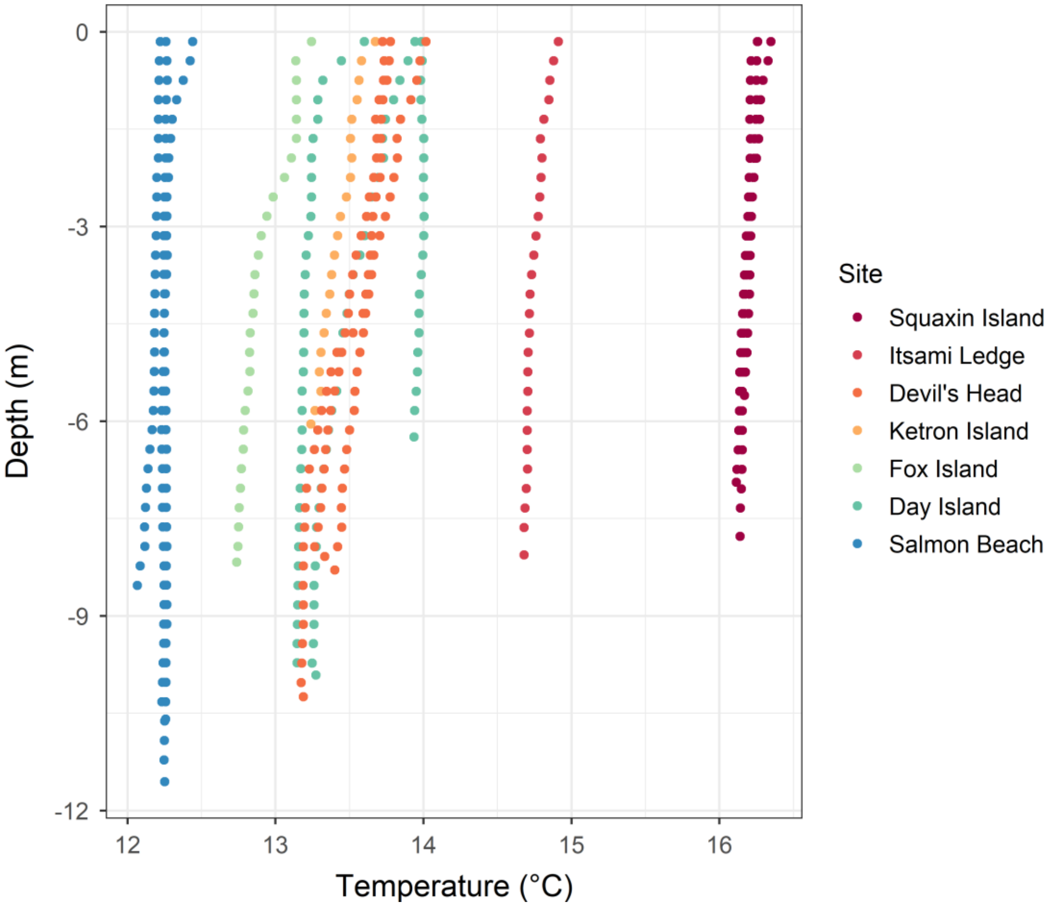


Figure 14. Nearshore temperature profiles in July 18, 2018 at seven sites in SPS. Three casts were collected along the alongshore boundaries and center of the site at Squaxin Island, Devil's Head, Day Island and Salmon Beach. Single casts were collected at other sites.

Salinity also displayed a distinct geographic gradient, decreasing with distance from the Tacoma Narrows. Salinity differed between sites across the growing season (Welch's ANOVA, $F_{6, 1935} = 233.54$, $p < 0.0001$) and from month to month (Welch's ANOVA, $F_{83, 1195} = 60427$, $p < 0.0001$). Salinity rose at all sites over the course of the spring, summer and fall, then dropped in December, most likely due to heavy winter rainfall (Fig. 15, Appendix B). The waters around Squaxin Island had the lowest salinity of any site monitored reaching 27.1 PSU in February and never rose above 30 PSU at any time during sampling. In contrast, salinity at Salmon Beach never dropped below 28 PSU and peaked in November at 30.3 PSU, the highest salinity observed in this study.

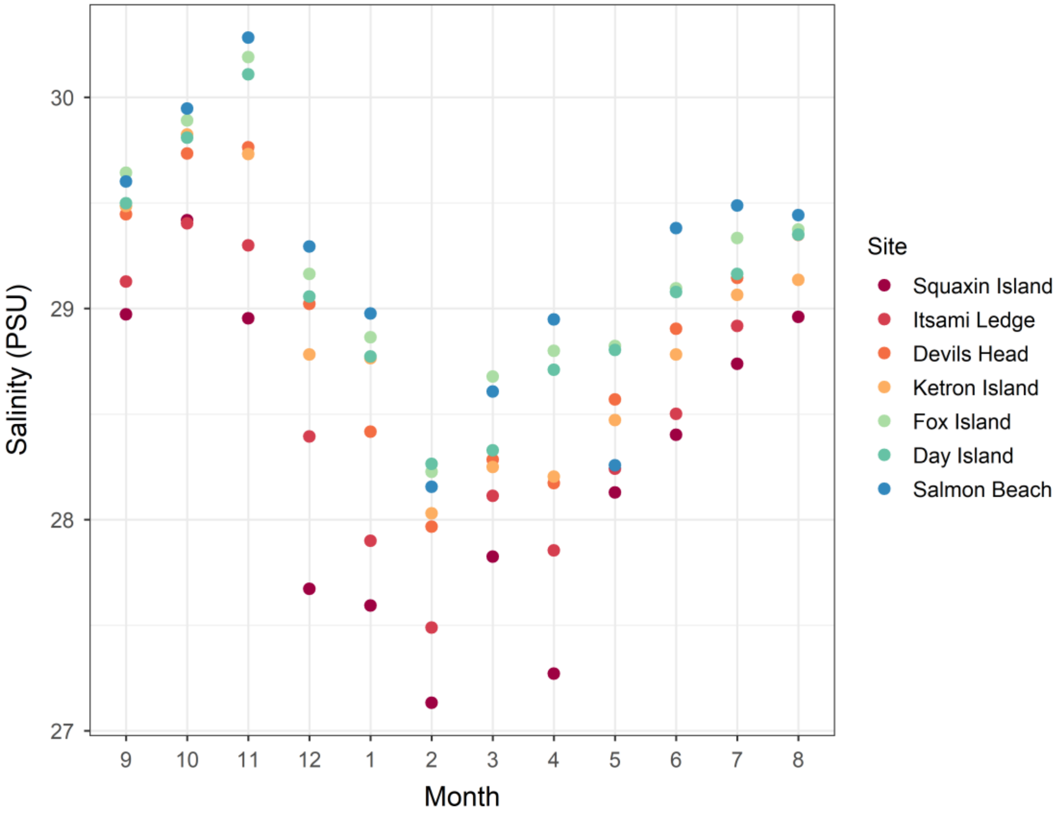


Figure 15. Monthly integrated water column salinity from September 2017 to August 2018 at 7 sites in South Puget Sound (-6 m MLLW depth).

Nitrate-nitrite concentrations were similar at all sites sampled during March at around 25 µmol/L (Fig. 16, Appendix C). By April, nitrogen concentrations had decreased at all sites but remained similar at Devil's Head (23.2 µmol/L) and Day Island (22.5 µmol/L). In contrast, Squaxin Island nitrogen concentrations decreased strongly, in April dropping to 18.8 µmol/L. Concentrations at all South Sound sites except Squaxin Island hovered between approximately 10 and 15 µmol/L from May until September. Squaxin Island nitrogen concentration continued to decrease between April and May, dropping below 5 µmol/L in June and July and reaching the lowest concentrations measured at any site in August (0.59 µmol/L). Late summer (September) nitrogen concentrations at North Beach (23.0 µmol/L) were considerably higher than those recorded at Salmon Beach (15.0 µmol/L) or Day Island (15.4 µmol/L). It is unclear whether nitrogen levels at North Beach in September mark an increase from summer time lows, as was the case at Salmon Beach and Day Island due to a lack of observations at this site.

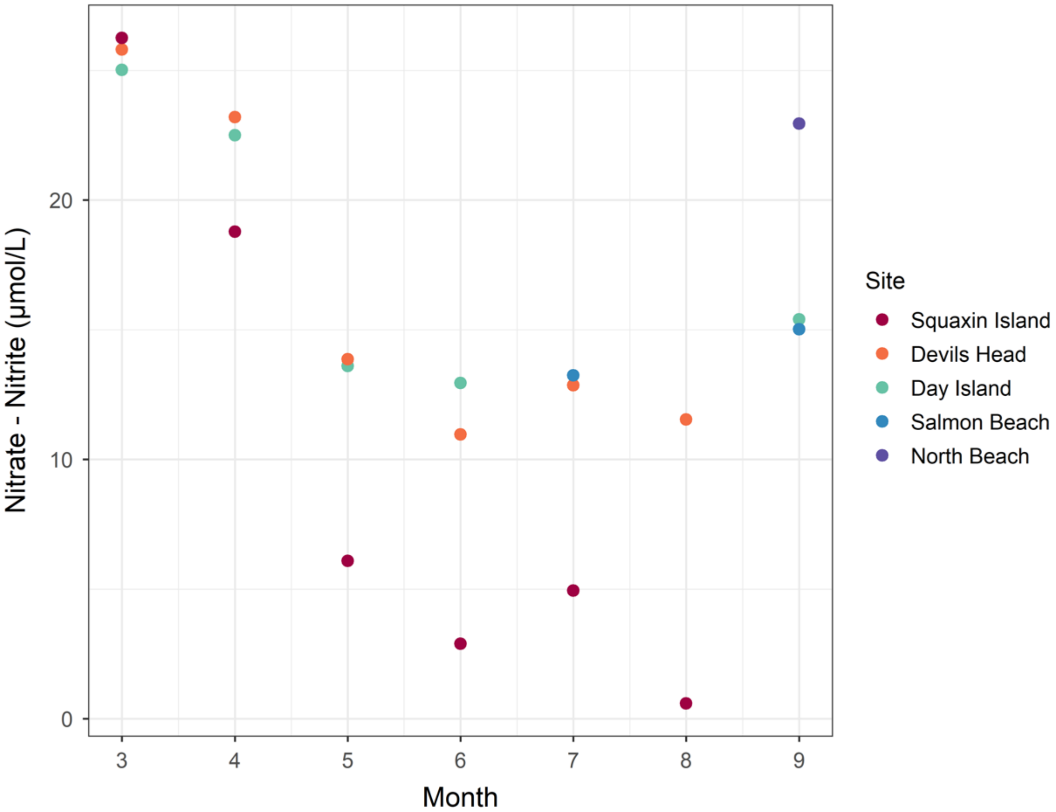


Figure 16. Monthly integrated water column nitrate-nitrite concentration from March to September 2018 at sites in South Puget Sound and one site in the Strait of Juan de Fuca. Note that a subset of sites were sampled in most months.

The mean suspended sediment concentration (SSC) at Squaxin Island was 2.15 ± 0.7 mg/L (± SD). This SSC was significantly more than mean SSC at all other sites (Welch's ANOVA, $F_{4, 13.54} = 15.12$, $p < 0.0001$, Games-Howell post hoc test, $p < 0.05$, Appendix D, Fig. 17). Mean SSC decreased with distance north in the SPS with the lowest concentrations of 0.83 ± 0.12 mg/L (± SD) and 1.14 ± 0.29 mg/L (± SD) observed at Salmon Beach and North Beach respectively. Whether this pattern stems from inherent environmental differences in sub-basin reaches or from anthropogenic influence is unknown due to a lack of baseline and long-term monitoring data.

Mean annual light attenuation at sites followed a similar pattern to SSC, generally decreasing with distance north in the SPS (Welch's ANOVA, $F_{6, 65.54} = 5.99$, $p < 0.0001$, Games-Howell post hoc test, $p < 0.05$, Appendix E, Fig. 18). Light attenuation at Squaxin Island was 0. 42 ± 0.11 (± SD) and significantly higher than at all other sites. Mean attenuation at all other sites ranged between 0.28 ± 0.06 (± SD) at Fox Island to 0.37 ± 0.08 (± SD) at Itsami Ledge. These values are generally consistent with published values for the Salish Sea (Kirk 1994, Glover et al. in review).

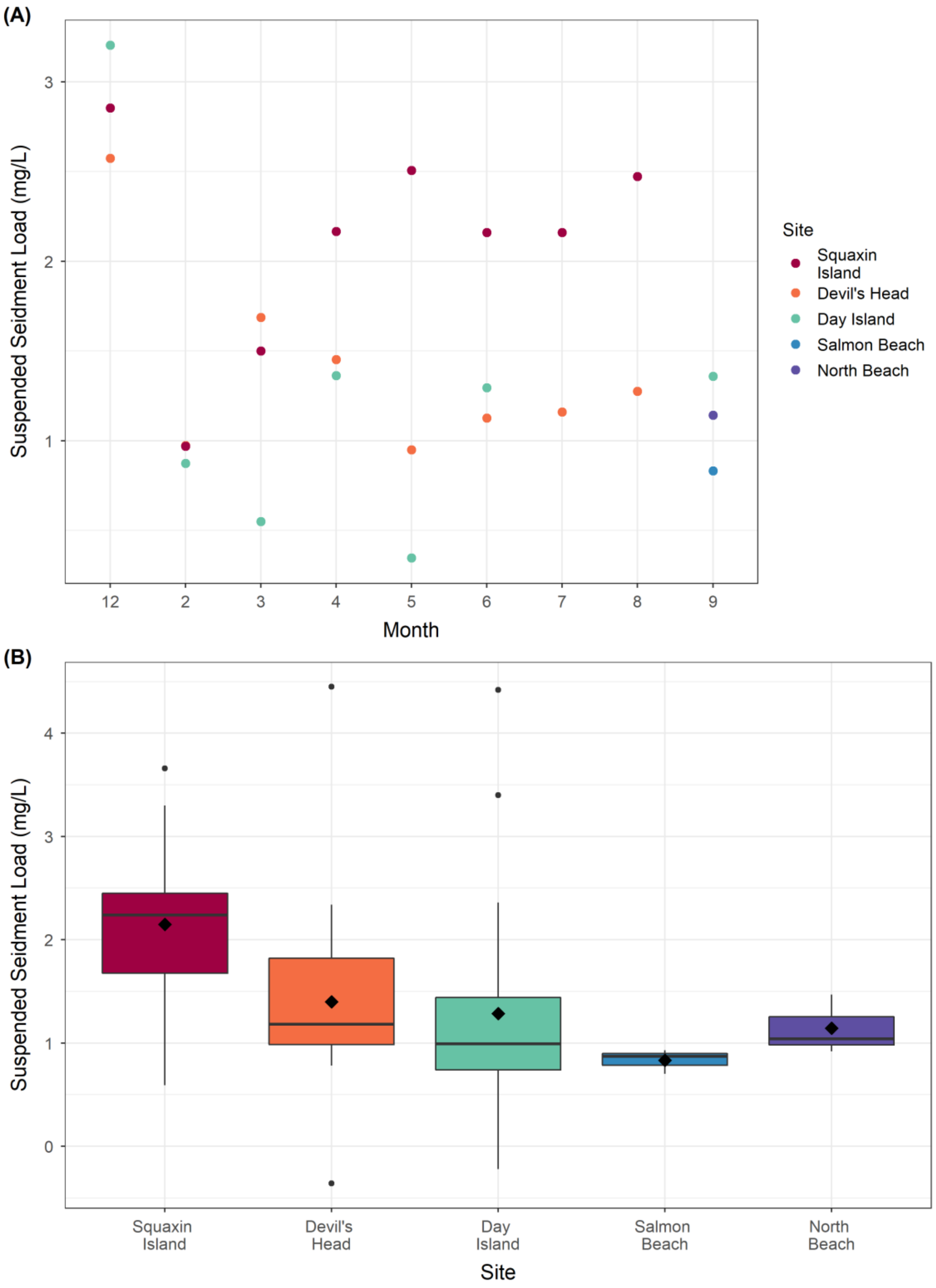


Figure 17. Mean suspended sediment concentrations (A) from December 2017 to September 2018 (note that a subset of sites were sampled in most months) and (B) by site.

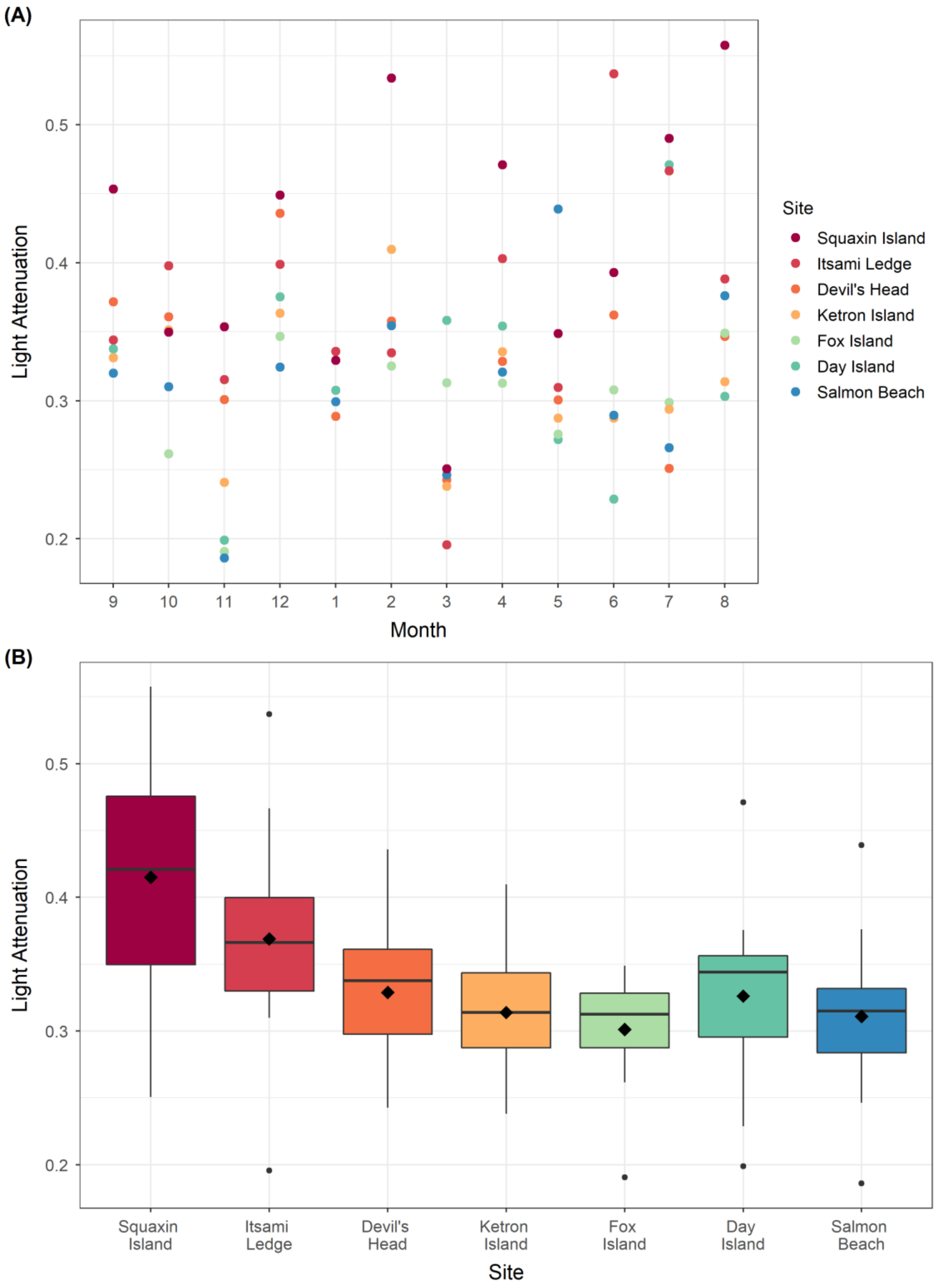


Figure 18. Mean light attenuation (A) monthly from September 2017 to August 2018 and (B) by site.

3.8 Salinity and Temperature Patterns Within Kelp Beds

A 3 to 4°C surface water temperature gradient was present within the Squaxin Island bull kelp bed in summer 2018 (Fig. 19). In late July, the surface water temperature at Squaxin Island ranged from 16.9 to 20.1 °C with the highest temperatures measured in shallow water close to shore where sunlight can more effectively elevate water temperature. By early September, the surface water temperature had cooled and ranged from 15.9 to 17.2 °C—again with the highest temperature measured in the shallow water close to shore. During both sampling periods, the temperature within the kelp bed at Squaxin Island was similar to, and slightly higher than, the nearshore measurement. In comparison to Squaxin Island, the surface water temperature in early September at Salmon Beach was 1 to 2°C lower and had a smaller range (13.9 to 14.5 °C). This small temperature range could be attributed to the strong tidal mixing present in the Tacoma Narrows and the narrow shoreline, which drops off to deep water quickly.

Salinity at Salmon Beach in early September varied little and ranged from 29.4 to 29.9 PSU (Fig. 18). The salinity range at Squaxin Island was slightly larger than at Salmon Beach and varied little between late July (28.0 to 29.2 PSU) and early September (28.0 to 29.5 PSU). Salinity in the southern half of the Squaxin Island kelp bed appeared to be slightly lower in late July and higher in early September.

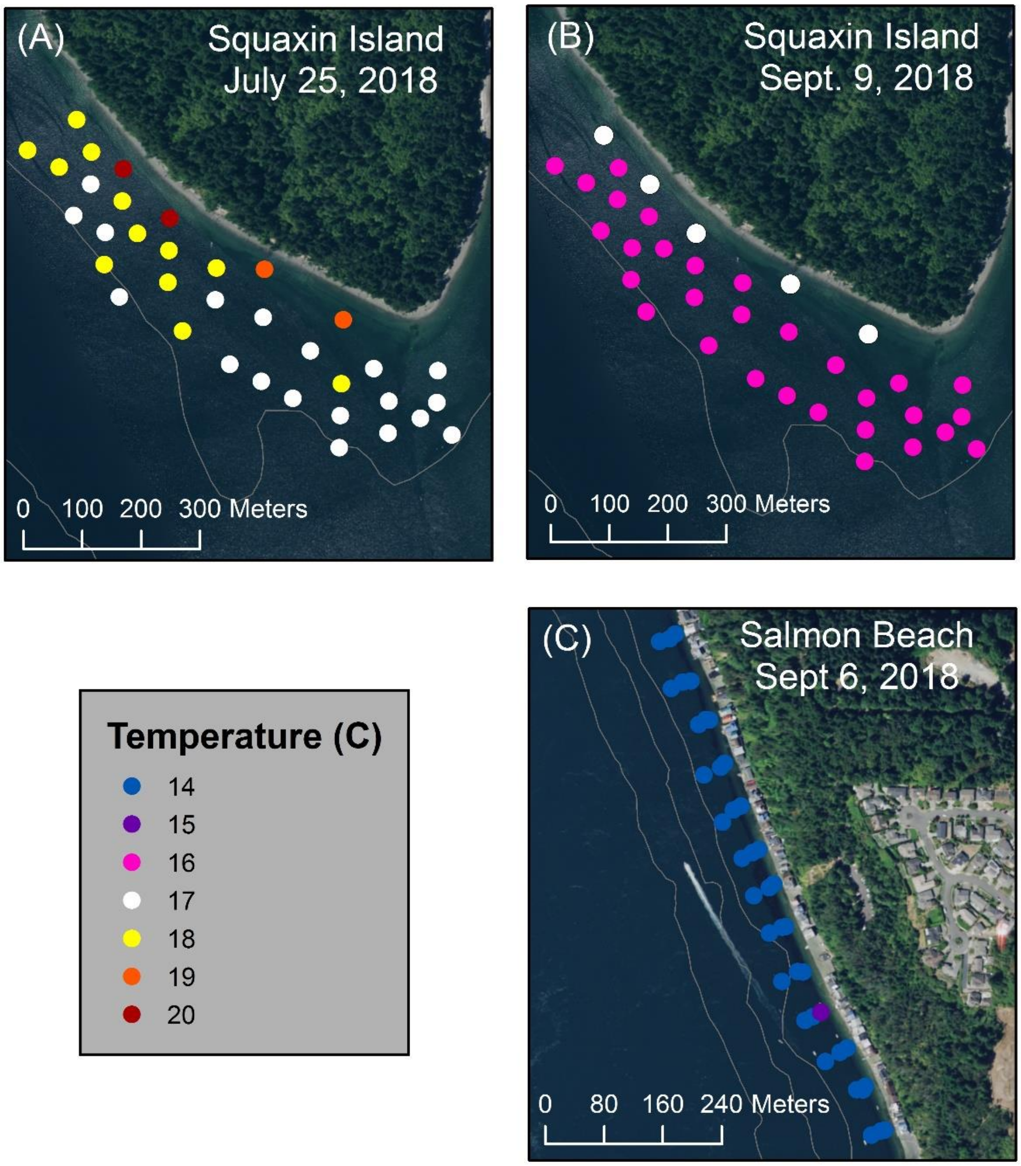


Figure 19. Surface temperature within the bull kelp bed at Squaxin Island on (A) July 25, 2018 and (B) September 9, 2018 and (C) Salmon Beach on September 6, 2018.

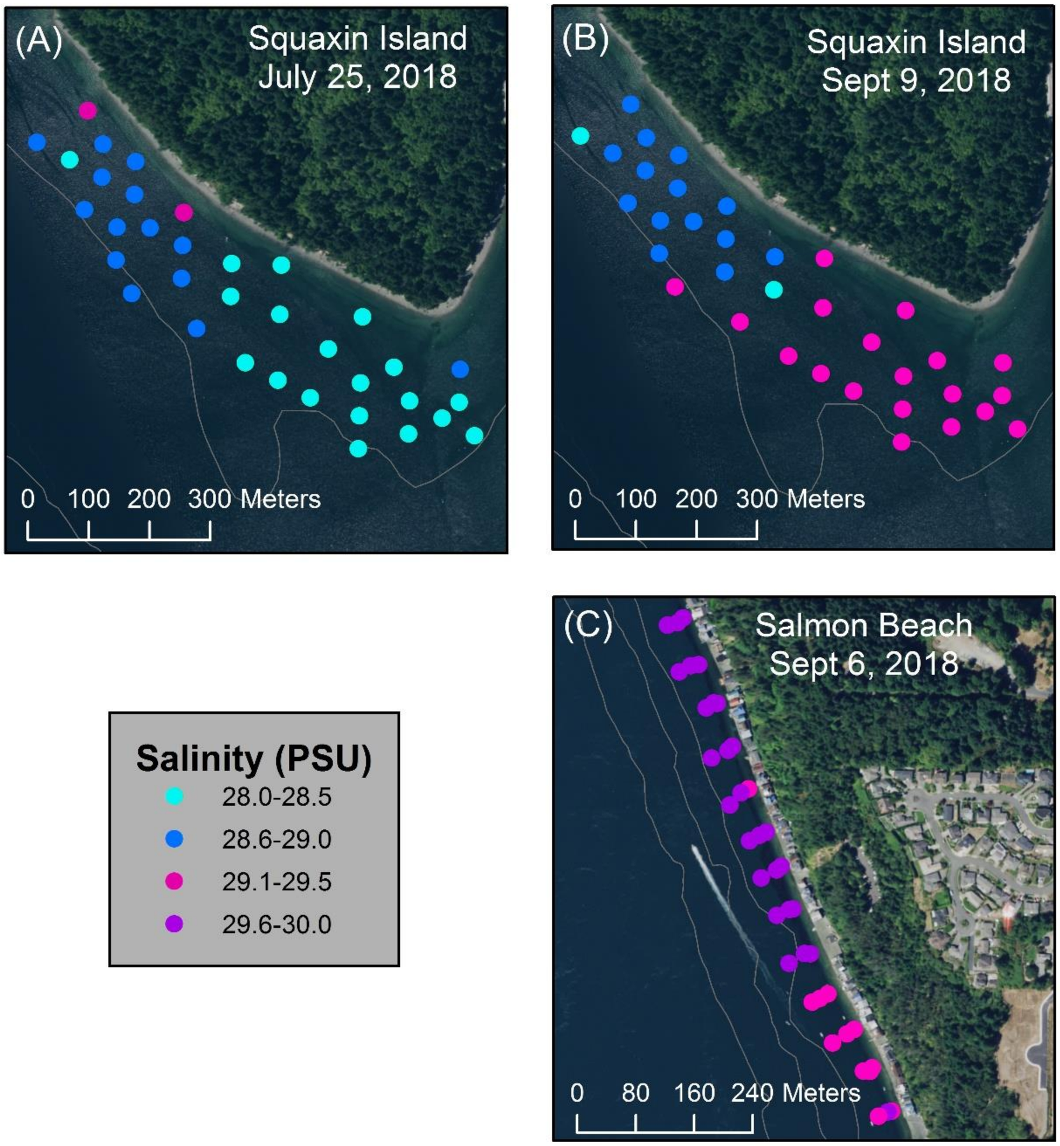


Figure 20. Surface salinity gradient at Squaxin Island on (A) July 25, 2018 and (B) September 9, 2018 and (C) Salmon Beach on September 6, 2018.

4 Discussion

4.1 Recent Status and Condition of Bull Kelp in SPS

In 2017, bull kelp had limited distribution in South Puget Sound (SPS) with beds occurring along only 2% of shorelines. The vast majority of shorelines with bull kelp were located along the Tacoma Narrows and its approaches. Major declines occurred between 2013 and 2017/2018 at all four beds that were re-surveyed. Two of the beds (Brisco Point and Devil's Head) were lost, and the remaining two beds (Squaxin Island and Fox Island) declined significantly.

In 2018, we established a new monitoring site at Salmon Beach in the Tacoma Narrows because bull kelp was common along shorelines in that area and plants appeared to be in healthier condition (Fig. 21). While we did not have bed area estimates from 2013 for comparison, bull kelp beds extended to significantly deeper maximum depths in 2018 at Salmon Beach compared to Squaxin Island. Additionally, the length, weight and number of blades on individuals at Salmon Beach were significantly greater than at Squaxin Island. Robust blades persisted at Salmon Beach into winter, whereas at Squaxin Island almost all plants lacked blades for photosynthesis or reproduction by August.

This study documented a clear geographic gradient in known kelp stressors, with intensity increasing from the Tacoma Narrows to Squaxin Island. Stressors included decreasing nutrient concentrations, and increasing temperature, SSC, kelp crab abundances and epiphyte/endophyte colonization. While this study cannot draw causal relationships, the stressors monitored in this study are likely to be related to recent Puget Sound bull kelp losses and declines. While some stressors monitored in the study were within known ranges to cause negative impacts to bull kelp, stressors often interact synergistically resulting in more severe negative impacts to species and habitats than would be predicted from single stressors alone (Crain, Kroeker, & Halpern, 2008).

Given the gradient of increasing stressor intensity within SPS and the observed losses in intermediate locations (Brisco Point, Devils Head and Fox Island), the question emerges, why has the bull kelp bed persisted at Squaxin Island? It is the most distant bed from the Tacoma Narrows and it experienced more extreme water temperatures than sites where bull kelp no longer occurs (eg. Ketron Island, Devils Head, Itsami Ledge). We suspect that individual site characteristics play a strong role. The site has a relatively extensive shallow subtidal bench and long fetch that is exposed to predominantly southwesterly winds. These attributes may provide more potential shallow subtidal habitat and regular disturbance by waves to support bull kelp persistence.

The relative health of the Salmon Beach bull kelp bed is likely related, in part, to strong currents and intense tidal mixing at the Tacoma Narrows. Salinity, temperature, nutrient levels and SSC at Salmon Beach were more similar to North Beach than Squaxin Island, which suggests that water mixing dynamics can be more important than geographic proximity for kelp productivity. Morphometric sampling showed that bull kelp blades and bulbs at North Beach were the largest, followed by Salmon Beach and Squaxin. In contrast to Squaxin Island, levels of epiphyte/endophyte colonization and physical damage at North Beach and Salmon Beach were low. These results suggest that, in addition to genetics (Gierke et al. 2018), local conditions play an important role in shaping the morphology and longevity of adult sporophytes.

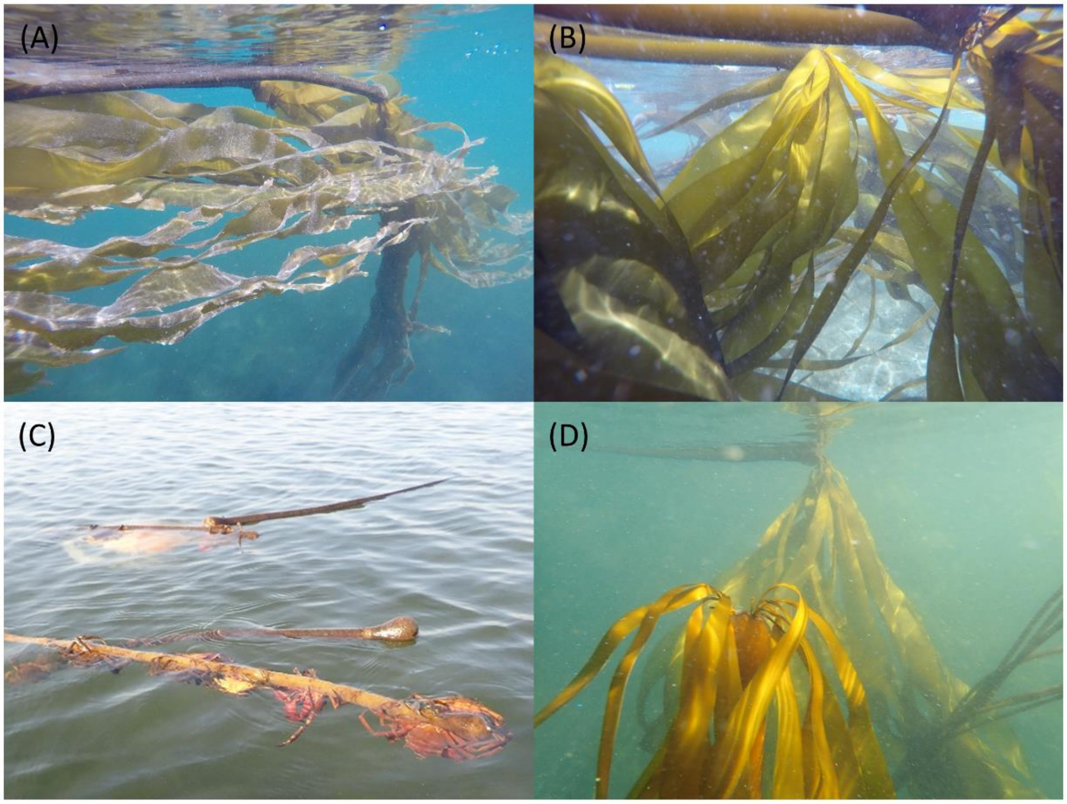


Figure 21. Comparative condition of bull kelp at Squaxin Island (left, A & C) and Salmon Beach (right, B & D) on July 23-27, 2018 (top, A & B) and September 5-6, 2018 (bottom, C & D).

4.2 Candidate Stressors

Research from other regions and basic information on conditions in SPS suggest a short list of candidate stressors of prime concern for bull kelp in SPS (Fig. 22).

In recent years especially, warm climate conditions made water temperature a primary candidate for kelp impacts in SPS. Elevated water temperature is correlated with floating kelp declines elsewhere in the world including California, Australia and Tasmania where a majority of kelp canopies have been lost in recent years due, in part, to marine heatwaves (Catton et al. 2016, Wernberg et al. 2016). The first year of surveys considered here (2013) pre-dates the onset of "the Blob" and other warm climate events (PSEMP Marine Waters Workgroup 2018), while recent surveys (2017 and 2018) occurred after consecutive years of warm water conditions. In 2018, summer water temperatures at Squaxin Island (up to 20 °C) exceeded known bull kelp stress thresholds (17 °C) by several degrees. Significantly lower water temperatures were measured at Salmon Beach, where bull kelp condition appeared to be healthier.

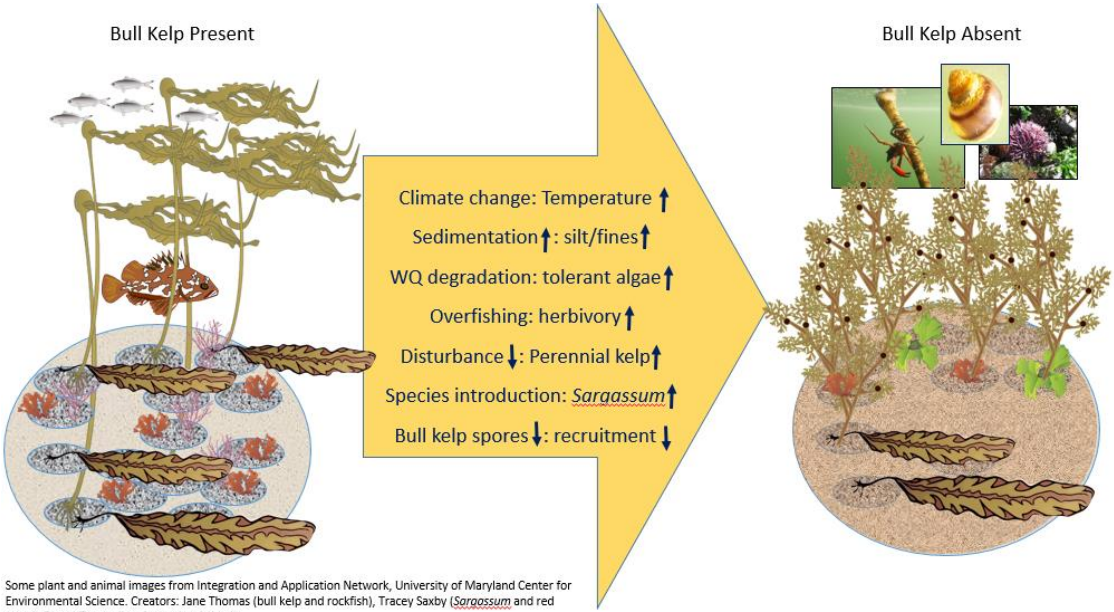
While short term decreases in bull kelp occurred in tandem with warm water conditions in recent years, major concerns regarding losses in inner basins existed prior to 2013 (Berry et al. 2014, Berry 2017). Therefore, we hypothesize that recent warm weather conditions further exacerbated ongoing losses. DNR is currently synthesizing distribution data from this study and historic surveys to explore the long-term pattern of change in bull kelp distribution prior to recent warm water events (Berry et al. in prep).

The SPS is naturally characterized by higher temperatures and lower nutrient concentrations than regions of the Salish Sea that are more proximate to oceanic influence. The negative effects of temperature stress on kelp in general and bull kelp in particular are well documented, and decrease overall kelp resilience to additional stress (Tera Corp. 1982, Wernberg et al. 2010, Schiltroth et al. 2018). Temperature stress can increase the damage and reproductive consequences of otherwise normal grazing pressure and result in kelp declines even when nutrient concentrations are otherwise adequate for normal growth (Rothäusler et al. 2009, Schiel and Foster 2015). High temperatures are also known to negatively impact spore production and germination rates, potentially reducing subsequent recruitment and delaying recovery from disturbance. Thus, conditions in SPS may provide managers and scientists insight into the effects of future climate change on bull kelp forests in the Salish Sea as climate effects intensify.

High summer temperatures also coincide with large ephemeral algal blooms known to deplete otherwise plentiful surface nitrogen concentrations in several Puget Sound subbasins. These algal blooms reduce surface nitrogen concentrations faster than can be replenished from deeper, oceanic inflows (Khangaonkar et al. 2018). Land-based nitrogen inputs into the Puget Sound account for less than 2% of the total nitrogen delivered by deep water oceanic inflows. However waste water treatment plant effluent accounts for 59% of total land-based nitrogen inputs and 81% of summer land-based nutrient inputs into the Puget Sound (Mohamedali et al. 2011). These substantial inputs likely alter the frequency and severity of algal blooms leading to increased instances of hypoxia and nutrient depletion as surface waters are fertilized by anthropogenic nutrient sources (Khangaonkar et al. 2018).

While observations of low nitrogen concentrations (< 5 µmol/L) were rare in this study, low nutrient concentrations may be further decreasing kelp resilience at Squaxin Island. In California, giant kelp requires 1 to 2 µmol/L in order to sustain average growth rates, but high temperatures can exacerbate nutrient stress. For example, giant kelp populations in Chile have declined in the face of temperatures exceeding 17 °C even when nutrient concentrations were above 3 µmol/L (Schiel and Foster 2015). Low nutrient concentrations have also been documented to delay gametophyte reproduction and germling sporophyte growth in giant kelp but the effects of low nutrient availability on bull kelp are unknown (Hurd et al. 2014).

Squaxin Island nutrient concentrations hovered between 4.94 and 0.59 µmol/L from May to August, coinciding with peak summer temperatures. In tandem, high temperatures and low nutrient concentrations interfere with the ability of bull kelp to repair damaged tissue. The high densities of kelp crabs observed at Squaxin Island, especially in the face of stressful temperatures and nutrient concentrations, may have resulted in more damage than would be otherwise expected from such grazer activity. This could explain the more frequent observations of adult bull kelp with high damage ratings at Squaxin Island as compared to Salmon Beach and North Beach.



Some plant and animal images from Integration and Application Network, University of Maryland Center for Environmental Science. Creators: Jane Thomas (bull kelp and rockfish), Tracey Saxby (*Sargassum* and red algae), Dieter Tracey (herring and red algae).

Figure 22. Major factors hypothesized to drive bull kelp losses in SPS.

Water quality degradation in the form of increased sediment can also have negative consequences for kelp forests (Deiman et al. 2012, Falkenberg et al. 2013). Increased SSCs and associated increases in turbidity can reduce light availability and delay spring recruitment (Airoldi 2003, Rubin et al. 2017) and may explain the shoaling of the Squaxin Island bed. Light attenuation values observed in this study were similar to averages reported elsewhere for the Puget Sound (Kirk, 1994, Glover et al. in review). However, light availability thresholds for kelp are not well understood, either alone or in conjunction with other conditions.

In addition to decreasing available light, increased sediment transport to kelp habitats can lead to mortality of microscopic lifestages by blocking the settlement of spores on available substrate and smothering attached gametophytes and germling sporophytes (Deiman et al. 2012). Whether sediment related mortality of bull kelp microscopic lifestages in the Puget Sound is an issue is unknown. Futhermore, little is known regarding changes in sediment transport to the Puget Sound as a result of land-use change and development but both are known to increase sediment and nutrient delivery to estuarine systems.

Other factors are likely playing a role in the decline of SPS bull kelp given historic and current human activities. In Europe and Australia, increased nutrient pollution in combination with warmer water temperatures and increased carbon availability resulting from anthropogenic emissions alter competitive interactions between large kelp and weedy turf algae. Increased nitrogen concentrations in concert with increased carbon availably due to anthropogenic emissions allow turf to monopolize available substrate more quickly and efficiently than native kelp species (Falkenberg et al. 2013). Once turf algae gain a foothold, they increase sediment accumulation, blocking and delaying recovery following disturbance. While the distribution and extent of turf algae in the Puget Sound is unknown, turf algae have been correlated to urban development in Europe and Australia and there is some evidence of increased turf abundance in proximity to the Seattle (Heery 2017).

Changes to the biotic community in and around kelp forests can also impact kelp persistence and distribution. Overexploitation of marine fauna, especially of high-level predators (such as sea otters, salmon, rockfish) can lead to increases in grazer populations that deforest large swaths of kelp forests (Steneck et al. 2002). Kelp crabs, which are known to graze preferentially on bull kelp in Puget Sound (Dobkowski 2017), were rare at Salmon Beach and abundant at Squaxin Island. Copper rockfish are known predators of kelp crabs in SPS, and their populations have declined substantially in recent decades (D. Lowry, personal communication, Oct. 16, 2018). However, no information exists regarding trophic controls on kelp crab populations in the Puget Sound. Nor do we fully understand the impacts of herbivory from kelp crabs and other common macro- and mesograzers on Puget Sound kelp forests. Despite our gaps in knowledge, the proportion of individuals with physical damage to the bulb and stipe was larger at Squaxin Island than Salmon Beach suggesting that kelp crabs may cause significant damage to bull kelp individuals when found in sufficient densities and combined with additional stressors.

Too little grazer activity can also alter kelp assemblages by decreasing disturbances that early successional species like bull kelp rely on for continued persistence (Duggins 1980). Without regular, intermediate disturbance, perennial and longer-lived annual kelp species can decrease bull kelp recruitment by blocking spore attachment to available substrate and decreasing the amount of light reaching bull kelp microscopic life stages. Invasive macroalgae, such as the locally abundant *Sargassum muticum* can exert similar effects on local kelp assemblages and have been documented to decrease native kelp recruitment in the Puget Sound (Britton-Simmons 2004). *Sargassum muticum* was abundant at Squaxin Island and uncommon at Salmon Beach and North Beach.

Finally, bull kelp in SPS may be recruitment limited due to a low availability of spores as persistent beds become smaller and reduced connectivity between beds as losses continue. Giant kelp spore dispersal distances have been shown to depend on local current regimes and the height of spore dispersal in the water column (Gaylord et al. 2002). Spores dispersed closer to the surface can be distributed several kilometers. Considering that bull kelp produces all of its spores in the top few meters of water, it is likely that spore dispersal distances may be larger than those of giant kelp which produce all sorus material close to the benthos. If this is the case, lost forests in closer proximity to currently persistent forests (for example Brisco Point close to Squaxin Island) may have the highest potential for recovery compared to more isolated forests that have been lost (for example Devil's Head or Ketron Island). However, there is currently no understanding of bull kelp population connectivity in the Puget Sound. Furthermore, there is no data on average sorus production rates or spore densities per sorus for bull kelp. Information on estimated spore production as a function of bed density and size may offer insight into thresholds for bed persistence and provide valuable information on the scale of restoration out-planting necessary to recover persistent bull kelp beds. A better understanding of population dynamics and connectivity could help prioritize sites for conservation and recovery actions by targeting sites found to contribute the most to spore dispersal and site connectivity.

4.3 In Closing

Given the importance of bull kelp as a habitat and environmental indicator, the observed losses in SPS constitute a cause for concern. Other regional studies and anecdotal observations report losses throughout the Salish Sea, suggesting that trends observed in the SPS are not unique. A variety of monitoring efforts are underway by volunteers, the tribes, government and non-governmental organizations to provide baseline information regarding current bull kelp abundance, distribution and trends.

The diversity of oceanographic conditions found within the Puget Sound makes it likely that stressors negatively impacting bull kelp differ among basins. Some of these stressors (anthropogenic nutrient pollution, increased runoff from urban development) may be reduced through mitigation actions, while others stemming from anthropogenic climate change will be more difficult to address.

Bull kelp in the Salish Sea is a critical habitat that forms the trophic foundations necessary to support healthy rockfish, salmon and orca populations. Despite the recent documented losses to bull kelp canopies in the SPS and elsewhere, bull kelp beds still exists in all major Salish Sea basins. Continued monitoring of bull kelp condition, extent and associated stressors will be essential for future management of this critical habitat and the protection of some of the state's most iconic marine species.

5 References

Ahn, O., R.J. Petrell, and P.J. Harrison. 1998. Ammonium and nitrate uptake by *Laminaria saccharina* and *Nereocystis luekeana* originating form a salmon sea cage farm. Journal of Applied Phycology 10:333-340.

Airoldi, L. 2003. The effects of sedimentation on rocky coast assemblages. Oceanography and Marine Biology: An Annual Review 41:161–236.

Allen, B. 2016. Sampling for Kelp Production and Biomass at the Hood Head Mariculture (HCM) Kelp Farm: Methods for Estimating Net Productivity in Sugar Kelp at Hood Head. Puget Sound Restoration Fund.

Bartsch, I., C, Wiencke, K. Bischof, C.M. Buchholz, B.H. Buck, A. Eggert, P. Feuerpfeil, D. Hanelt, S. Jacobsen, R. Karez, U. Karsten, M. Molis, M.Y. Roleda, H. Schubert, R. Schumann, K. Valentin, F. Weinberger, and J. Wiese (2008). The genus Laminaria sensu lato: recent insights and developments. European Journal of Phycology 43(1):1-86. https://doi.org/10.1080/09670260701711376

Berry, H.D. 2017. Assessment of bull kelp at Squaxin Island in 2013, 2014 and 2016. Nearshore Habitat Program. Washington State Department of Natural Resources, Olympia, WA.

Berry, H.D., T.F. Mumford, P. Dowty, and B. Christiaen. in preparation. Long term changes in kelp forests in an inner basin in the Salish Sea.

Berry, H., T. Mumford, and K. Sherman. 2014. An historical study of change in floating kelp beds in Puget Sound and the Strait of Juan de Fuca. Salish Sea Ecosystem Conference, Seattle, WA.

Berry, H. and S. Shull. 2018. Warm conditions associated with declines in floating kelp. In Puget Sound Marine Waters: 2017 overview. S.K. Moore, R. Wold, K. Stark, J. Bos, P. Williams, N. Hamel, S. Kim, A. Brown, C. Krembs, and J. Newton (Eds). pp 51. URL: www.psp.wa.gov/PSmarinewatersoverfiew.php.

Bertness, M.D., J.F. Bruno, B.R. Silliman and J.J. Stachowicz (Eds). 2014. Marine Community Ecology and Conservation. Sinauer Associates, Inc., Sunderland, MA.

Bodkin, J.L. 1986. Fish assemblages in *Macrocystis* and *Nereocystis* kelp forests off Central California. Fishery Bulletin 84:799–808.

Bolton, J.J. 2010. The biogeography of kelps (Laminariales, Phaeophyceae): a global analysis with new insights from recent advances in molecular phylogenetics. Helgoland Marine Research 64:263-279.

Burns, R. 1985. The shape and form of Puget Sound. University of Washington Press, Seattle, WA.

Britton-Simmons, K.H. 2004. Direct and indirect effects of the introduced alga *Sargassum muticum* on benthic, subtidal communities of Washington State, USA. Marine Ecology Progress Series 277:61-78.

Britton-Simmons, K.H., G. Foley, D. Okamoto. 2009. Spatial subsidy in the subtidal zone: utilization of drift algae by a deep subtidal sea urchin. Aquatic Biology 5:233-243.

Calloway, M.D. 2019. *Nereocystis luetkeana* (bull kelp) in South Puget Sound: stressor impacts on the health of native floating kelp canopies. Masters Thesis. The Evergreen State College, Olympia, WA, USA.

Cameron, F.K. 1915. Potash from Kelp. USDA Report No. 100. Washington D.C. 122 p.

Carney, L. and M.S. Edwards. 2006. Cryptic processes in the sea: a review of delayed development in the microscopic life stages of marine macroalgae. Algae 21(2):161-168.

Carson, H.D., M. Ulrich, D. Lowry, R.E. Pacunski, R. Sizemore. 2016. Status of the California sea cucumter (*Parastichopus californicus*) and red sea urchin (*Mesocentrotus franciscanus*) commercial dive fisheries in the San Juan Islands, Washington State, USA. Fisheries Research. 179:179-190.

Catton, C., L. Rogers-Bennett and A. Amrhein. 2016. “Perfect storm” decimates northern California kelp forests. California Department of Fish and Wildlife Marine Management News. URL: https://cdfwmarine.wordpress.com/2016/03/30/perfect-storm-decimates-kelp/

Connell, S.D., K.J. Kroeker, K.E. Fabricius, D.I. Kline and B.D. Russell. 2013. The other ocean acidification problem: CO2 as a resource among competitors for ecosystem dominance. Philosophical Transactions of the Royal Society B: Biological Sciences 368(1627):1-9.

Crain, C. M., Kroeker, K., & Halpern, B. S. (2008). Interactive and cumulative effects of multiple human stressors in marine systems. *Ecology Letters*, *11*(12), 1304–1315. https://doi.org/10.1111/j.1461-0248.2008.01253.x

Dayton, P.K. 1985. Ecology of kelp communities. Annual Review of Ecology and Systematics 19:261-280.

Deiman, M., K. Iken, and B. Konar. 2012. Susceptibility of *Nereocystis luetkeana* (Laminariales, Ochrophyta) and *Eualaria fistulosa* (Laminariales, Ochrophyta) spores to sedimentation. Algae 27(2):115–123. https://doi.org/10.4490/algae.2012.27.2.115

Department of Natural Resources. 2001. The Washington State ShoreZone Inventory. Nearshore Habitat Program, Washington State Department of Natural Resources, Olympia, WA.

Dethier, M.N. 1990. A marine and estuarine habitat classification system for Washington State. Washington Natural Heritage Programs, Dept. Natural Resources.

Dobkowski, K. 2017. The role of kelp crabs as consumers in bull kelp forests-evidence form laboratory feeding trials and field enclosures. PeerJ 5:e3372; DOI 10.7717/peerj.3372.

Dobkowski, K.A., J. Kobelt, S. Brentin, K.L. Van Alstyne, M.N. Dethier. 2017. Picky *Pugettia*: a tale of two kelps. Marine Biology 164:210.

Druehl, L.D. 1967. Distribution of two species of Laminia as related to some environmental factors. Journal of Phycology 3:103-108.

Druehl, L.D. 1978. The distribution of *Macrocystis integrifolia* in British Columbia as related to environmental parameters. Canadian Journal of Botany 56:69-79.

Druehl, L.D., and B.E. Clarkston. 2016. Pacific seaweeds: A guide to common seaweeds of the West Coast. Harbour Publishing, Madeira Park, BC.

Druehl, L.D., and S.I.C. Hsiao. 1977. Intertidal kelp response to seasonal environmental changes in a British Columbia inlet. Journal of Fisheries Research Board of Canada 34(8):1207–1211.

Duggins, D.O. 1980. Kelp beds and sea otters: an experimental approach. Ecology 61(3):447-453. https://doi.org/10.2307/1937405

Duggins, D. O., C.A. Simenstad, and J.A Estes. 1989. Magnification of secondary production by kelp detritus in coastal marine ecosystems. Science 245:170–173. https://doi.org/10.1126/science.245.4914.170

Dunagan, C. 2018. Kelp continues steady decline in Puget Sound. Encyclopedia of Puget Sound. https://www.eopugetsound.org/magazine/ssec2018/kelp

Ebbesmeyer, C.C., J.Q. Word, and C.A. Barnes. 1988. Puget Sound: a fjord system homogenized with water recycled over sills by tidal mixing. In B. Kjerfve (Ed). Hydrodynamics of Estuaries Volume 2. CRC Press, Boca Raton, pp 17-29.

Edwards, M.S. 2000. The role of alternate life-history stages of a marine macroalga: a seed bank analogue? Ecology 81(9):2404-2415. https://doi.org/10.2307/177463

ESRI 2011. ArcGIS Desktop: Release 10. Redlands, CA: Environmental Systems Research Institute.

Estes, J.A. and D.O. Duggins. 1995. Sea otters and kelp forests in Alaska: generality and variation in a community ecological paradigm. Ecological Monographs 56(1):75-100.

Falkenberg, L. J., B. D. Russell, and S.D. Connell. 2013. Contrasting resource limitations of marine primary producers: implications for competitive interactions under enriched CO 2 and nutrient regimes. Oecologia 172(2):575–583.

Fredersdorf, J., R. Müller, S. Becker, C. Wiencke, and K. Bischof. 2009. Interactive Effects of Radiation, Temperature and Salinity on Different Life History Stages of the Arctic Kelp Alaria esculenta (Phaeophyceae). Oecologia 160(3):483–492.

Gabrielson, P.W., S.C Linstrom, and C.J. O'Kelly. 2012. Keys to the Seaweeds and Seagrasses of Southeast Alaska, British Columbia, Washington and Oregon. Phycological Contribution No. 8, PhycoID, Hillsborough, NC. Iv + 192 pp.

Gaylord, B., D. C. Reed, P.T. Raimondi, L. Washburn, and S.R. McLean. 2002. A physically based model of macroalgal spore dispersal in the wave and current-dominated nearshore. Ecology 83(5):1239–1251. https://doi.org/10.1890/0012-9658(2002)083[1239:APBMOM]2.0.CO;2

Gierke, L.G., R.A. San Miguel, T. Mumford, F.A. Alberto. 2018. Genetic structure of the bull-kelp *Nereocystis luetkeana* in the Salish Sea. Salish Sea Ecosystem Conference, Seattle, WA.

Glover, H.E, A.S. Ogston, I.M. Miller, E.F. Eidam, S.P. Rubin, H.D. Berry. in review. Impacts of suspended sediment on nearshore benthic light availability following dam removal in a small mountainous river: *in-situ* observations and statistical modeling.

Harley, C.D.G., A.R. Hughes, K.M. Hultgren, B.G. Miner, C.J. Sorte, C.S. Thornber, L.F. Rodriguez, L. Tomanek, S.L. Williams. 2006. The impacts of climate change in coastal marine systems. Ecology Letters 9(2):228-241.

Harvey, B. P., D. Gwynn-Jones, and P.J. Moore, P. J. 2013. Meta-analysis reveals complex marine biological responses to the interactive effects of ocean acidification and warming. Ecology and Evolution *3*(4):1016–1030. https://doi.org/10.1002/ece3.516

Hawkins. S.J., H.E. Sugden, N. Mieszkowska, P.J. More, E. Poloczanska, R. Leaber, R.J.H. Herbert, M.J. Genner, P.S. Moschella, R.C. Thompson, S.R. Jenkins, A.J. Sourthward, M.T. Burrows. 2009. Consequences of climate-driven biodiversity changes for ecosystem functioning of North European rocky shores. Marine Ecology Progress Series 396:245-259. https://doi.org/10.3354/meps08378

Heery, E. C. 2017. Benthic subtidal assemblages and ecological processes in urbanized seascapes of Puget Sound, Washington, USA. (Dissertation). University of Washington, Seattle, WA.

Hurd, C.L, P.J. Harrison, K. Bischof, C.S. Lobban. 2014. Seaweed Ecology and Physiology, Cambridge University Press, Cambridge, UK, 551 pp.

Jackson, G.A. 1977. Nutrients and production of giant kelp, *Marcocystis pyrifera*, off southern California. Limnology and Oceanography 22(6):979-995.

Kim, J.K., G.P. Kramer, C. Yarish. 2015. Use of sugar kelp aquaculture in Long Island Sound and the Bronx River Estuary for nutrient extraction. Marine Ecology Progress Series 531:155-166. DOI: 10.3354/meps11331

Kirk, John T.O. 1994. *Light and Photosynthesis in Aquatic Ecosystems*. 2nd ed. Cambridge: Cambridge University Press.

Khangaonkar, T., A. Nugraha, W. Xu, W. Long, L. Bianucci, A. Ahmed, T. Mohamedali, and G. Pelletier. 2018. Analysis of Hypoxia and Sensitivity to Nutrient Pollution in Salish Sea. Journal of Geophysical Research: Oceans 123(7):4735–4761. https://doi.org/10.1029/2017JC013650

Knapp, C. L. (n.d.). Insolation Data Manual: Long-Term Monthly Averages of Solar Radiation, Temperature, Degree-Days, and Global Kt for 248 National Weather Service Stations. 295.

Krumhansl, K., and R. Scheibling. 2012. Production and fate of kelp detritus. Marine Ecology Progress Series 467:281–302. https://doi.org/10.3354/meps09940

Krumhansl, K.A., D.K. Okamoto, A. Rassweiler, M. Novak, J.J. Bolton, K.C. Cavanaugh, S.D. Connell, C.R. Johnson, B. Konar, S.D. Ling, F. Micheli, K.M. Norderhaug, A. Perez-Matus, I. Sousa-Pinto, D.C. Reed, A.K. Salomon, N.K. Shears, T. Wernberg, R.J. Anderson, N.S. Barrett, A.H. Buschmann, M.H. Carr, J.E. Caselle, S. Derrien-Courtel, G.J. Edgar, M. Edwards, J.A. Estes, C. Goodwin, M.C. Kenner, D.J. Kushner, F.E. Moy, J. Nunn, R.S. Steneck, J. Vasquez, J. Watson, J.D. Witman, and J.E.K Byrnes. 2016. Global pattern of kelp forest change over the past half-century. Proceedings of the National Academy of Science 113(48):13785-13790.

Ling, S.D. 2008. Range expansion of a habitat-modifying species leads to loss of taxonomic diversity: a new and impoverished reef state. Oecologia 156:883-894. https://doi.org/10.1007/s00442-008-1043-9

Lüning, K. and W. Freshwater. 1988. Temperature tolerance of Northeast Pacific marine algae. Journal of Phycology 24(3):310-315.

Mann, K. H. 1973. Seaweeds: Their productivity and strategy for growth. Science, 182, 975–981. https://doi.org/10.1126/science.182.4116.975

Marzinelli, E.M., A.H. Campbell, E. Zozaya Valdes, A. Vergés, S. Nielson, T. Wernberg, T. de Dettignies, S. Bennett, J.G. Caporaso, T. Thomas, and P.D. Steinberg, 2015. Continental-scale variation in seaweed host-associated bacterial communities is a function of host condition, not geography. Environmental Microbiology 17(10):4078-4088.

Maxell, B.A. and K.A. Miller. 1996. Demographic studies of annual kelps *Nereocystis leutkeana* and *Costaria costata* (Laminariales, Phaeophyta) in Puget Sound, Washington. Botanica Marina 39:479-489. https://doi.org/10.1515/botm.1996.39.1-6.479

Mondragon, J. and J. Mondragon (R. Rasmussen, Ed). 2003. Seaweeds of the Pacific Coast: Common Marine Algae from Alaska to Baja California. Sea Challengers. Monterey, CA.

Mohamedali, T., M. Roberts, B. Sackmann, and A. Kolosseus. 2011. Puget Sound dissolved oxygen model nutrient load summary for 1999-2008 (No. 11-03–057, p. 172). Washington State Department of Ecology.

Moore, S.K., N.J. Mantua, J.P. Kellogg, and J.A. Newton. 2008. Local and large-scale climate forcing of Puget Sound oceanographic properties on seasonal to interdecadal timescales. Limnology and Oceanography 53(5):1746-1758.

Mumford, T.F., 2007. Kelp and Eelgrass in Puget Sound. Puget Sound Nearshore Partnership Technical Report 2007-05. Published by Seattle District, U.S. Army Corps of Engineers, Seattle, Washington. http://www.pugetsoundnearshore.org/technical_papers/kelp.pdf

Nysewander, D.R., J.R. Evenson, B.L. Murphie, and T.A. Cyra. 2005. Report of Marine Bird and Marine Mammal Component, Puget Sound Ambient Monitoring

Program, for July 1992 to December 1999 Period. Washington State Department of Fish and Wildlife. Olympia, Washington.

Palmer-McGee, C. 2019. A decade of disappearance: Kelp canopies in Samish traditional territory. Retrieved March 19, 2019, from https://www.arcgis.com/apps/MapJournal/index.html?appid=b9690e508ac74af7acb0d9ee0976e27b#map

Peters, G. 2018. userfriendlyscience: Quantitative analysis made accessible. doi: 10.17605/osf.io/txequ. URL: https://doi.org/10.17605/osf.io/txequ), R package version 0.7.2. URL: https://userfriendlyscience.com.

Pfister, C.A., H.D. Berry, and T. Mumford. 2017. The dynamics of kelp forests in the Northeast Pacific Ocean and the relationship with environmental drivers. Journal of Ecology 00:1-14.

PSEMP Marine Waters Workgroup. 2018. Puget Sound marine waters: 2017 overview. S. K. Moore, R. Wold, K. Stark, J. Bos, P. Williams, N. Hamel, S. Kim, A. Brown, C. Krembs, and J. Newton (Eds). URL: www.psp.wa.gov/PSmarinewatersoverview.php.

Ramirez-Puebla, S.T., B.L. Weigel, L. Jack, C. Schlundt, C.A. Pfister, and J.L. Mark Welch. In prep. Spatial structure of microbial biofilms on the canopy-forming kelp Nereocystis luetkeana.

R Core Team 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Rustria. URL http://www.R-project.org

Rothäusler, E., I. Gómez, I.A. Hinojosa, U. Karsten, F. Tala, and M. Thiel. 2009. Effect of temperature and grazing on growth of *Macrocystis* spp. (Phaeophyceae) along a latitudinal gradient. Journal of Phycology, 45(3):547–559. https://doi.org/10.1111/j.1529-8817.2009.00676.x

Rubin, S.P., I.M. Miller, M.M. Foley, H.D. Berry, J.J. Duda, B. Hudson, N.E. Elder, M.M. Beirne, J.A. Warrick, M.L. McHenry, A.W. Stevens, E.F. Eidam, A.S. Ogston, G. Gelfenbaum, R. Pedersen. 2017. Increased sediment load during a large-scale dam removal changes nearshore subtidal communities. PLoS ONE 12(12):e0187742. https://doi.org/10.1371/journal.pone.0187742

Schiel, D.R and M.S. Foster. 2015. The Biology and Ecology of Giant Kelp Forests. University of California Press, Oakland, CA.

Schiltroth, B., S. Bisgrove, and B. Heath. 2018. Effects of warm ocean temperatures on bull kelp forests in the Salish Sea. Salish Sea Ecosystem Conference, Seattle, WA.

Starko, S., L.A. Bailey, E. Creviston, K.A. James, A. Warren, M.K. Brophy, A. Danasel, M.P. Fass, J.A. Townsend, C.J. Neufeld. 2019. Environmental heterogeneity mediates scale-dependent declines in kelp diversity of intertidal rocky shores. PLoS ONE 14(3):e0213191. https://doi.org/10.1371/journal.pone.0213191

Steneck, R.S., M.H. Graham, B.J. Bourque, D. Corbett, J.M. Erlandson, J.A. Estes, and M.J. Tegner. 2002. Kelp forest ecosystems: biodiversity, stability, resilience and future. Environmental Conservation 29(4):436-459. https://doi.org/10.1017/S0376892902000322

Steneck, R. S., A. Leland, D.C. McNaught, and J. Vavrinec. 2013. Ecosystem Flips, Locks, and Feedbacks: the Lasting Effects of Fisheries on Maine's Kelp Forest Ecosystem. Bulletin of Marine Science 89(1):31–55. https://doi.org/10.5343/bms.2011.1148

Suskiewicz, M. 2010. Effects of competition and dispersal on the recruitment of the annual kelp *Nereocystis luetkeana*. (Masters Thesis) San Jose State University, San Jose, California, U.S.A.

Swanson, A.K. and L.D. Druehl. 2000. Differential meiospore size and tolerance of ultraviolet light stress within and among kelp species along a depth gradient. Marine Biology 136(4):657-664.

Teagle, H., S. J. Hawkins, P.J. Moore, D.A. Smale. 2017. The role of kelp species as biogenic habitat formers in coastal marine ecosystems. Journal of Experimental Marine Biology and Ecology 492:81–98. https://doi.org/10.1016/j.jembe.2017.01.017

Tera Corp. 1982. Compendium of thermal effects laboratory studies: Diablo Canyon Power Plant (No. Volume 1). Tera Corporation, San Francisco, CA.

Thomsen, M.D., L. Mondardini, T. Alestra, S. Gerrity, L. Tait, P.M. South, S.A. Lilley, D.R. Schiel. 2019. Local extinction of bull kelp (*Durvillaea* spp.) due to a marine heatwave. Frontiers in Marine Science 6:84. doi: 10:3389/fmars.2019/00084.

Vadas, R.L. 1972. Ecological implications of culture studies on *Nereocystis luetkeana*. Journal of Phycology 8(2):196-203.

Vergés, A., C. Doropoulos, H.A. Malcolm, M. Skye, M. Garcia-Piza, E.M. Marzinelli, A.H. Campbell, E. Ballesteros, A.S. Hoey, A. Vila-Concejo, Y. Bozec, P.D. Steinberg. 2016. Long-term empirical evidence of ocean warming leading to tropicalization of fish communities, increased herbivory, and loss of kelp. Proceedings of the National Academy of Science 113(48):13791-13796. https://doi.org/10.1073/pnas.1610725113 von Biela, A., S.D. Newsome, J.L. Bodkin, G.H. Kruse, C.E. Zimmerman. 2016. Widespread kelp-derived carbon in pelagic and benthic nearshore fishes suggested by stable isotope analysis. Esuarine, Coastal and Shelf Science 181:364-374.

Watanabe, H., M. Ito, A. Matsumoto, and H. Arakawa. 2016. Effects of sediment influx on the settlement and survival of canopy-forming macrophytes. Scientific Reports, 6:18677. https://doi.org/10.1038/srep18677

Wernberg, T., S. Bennett, R.C. Babcock, T. de Bettignies, K. Cure, M. Depczynski, F. Dufois, J. Fromont, C.J. Fulton, R.K. Hovey, E.S. Harvey, T.H. Holmes, G.A. Kendrick, B. Radford, J. Santana-Garcon, B.J. Saunders, M.A. Vanderklift, S. Wilson. 2016. Climate-driven regime shift of a temperate marine ecosystem. Science 535(6295):169-172.

Wernberg, T., M.S. Thomsen, F. Tuya, F., G.A. Kendrick, P.A. Staehr, and B.D. Toohey. 2010. Decreasing resilience of kelp beds along a latitudinal temperature gradient: potential implications for a warmer future: Climate and resilience of kelp beds. Ecology Letters, 13(6):685–694. https://doi.org/10.1111/j.1461-0248.2010.01466.x

Weigel, B. and C.A Pfsiter. 2019. Successional dynamics and seascape-level patterns of microbial communities on the canopy-forming kelps *Nereocystis luetkeana* and *Marcocystis pyrifera*. Frontiers in Microbiology 10:346. doi:10.3389/fmicb.2019.00346

6 Appendices

Appendix A. Average integrated water column temperatures as a function of survey month and site with average differences [a]

		n	M	SD	1	2	3	4	5	6
April	1. Squaxin Island	91	9.93	0.27						
	2. Istami Ledge	25	9.68	0.31	0.25					
	3. Devil's Head	98	9.63	0.53	0.30**	0.05				
	4. Ketron Island	32	9.20	0.09	0.73***	-0.48***	-0.43***			
	5. Fox Island	23	9.16	0.07	0.76***	0.52***	-0.46***	0.04		
	6. Day Island	93	8.98	0.07	0.95***	0.70***	0.65***	0.22***	0.18	
	7. Salmon Beach	114	8.94	0.07	0.98***	-0.74***	-0.68***	-0.26***	-0.23	-0.04
May	1. Squaxin Island	91	11.91	0.39						
	2. Istami Ledge	32	11.43	0.17	0.48***					
	3. Devil's Head	91	10.76	0.24	1.15***	0.67***				
	4. Ketron Island	24	10.81	0.21	1.10***	-0.61***	0.06			
	5. Fox Island	24	10.47	0.05	1.44***	0.96***	-0.29***	0.35***		
	6. Day Island	89	10.42	0.36	1.49***	1.00***	0.34***	0.39***	0.05	
	7. Salmon Beach	83	11.02	0.34	0.88***	0.40***	0.27***	0.21	0.56***	0.61***
June	1. Squaxin Island	79	13.72	0.37						
	2. Istami Ledge	26	13.12	0.13	0.61***					
	3. Devil's Head	73	11.97	0.42	1.76***	1.15***				
	4. Ketron Island	29	11.90	0.13	1.83***	-1.22***	-0.07			
	5. Fox Island	19	11.35	0.03	2.37***	1.77***	-0.62***	0.55***		
	6. Day Island	88	11.31	0.16	2.41***	1.81***	0.66***	0.59***	0.04	
	7. Salmon Beach	81	10.74	0.06	2.98***	-2.38***	-1.23***	-1.16***	-0.61***	-0.57***
July	1. Squaxin Island	69	16.20	0.05						
	2. Istami Ledge	27	14.75	0.07	1.44***					
	3. Devil's Head	91	13.53	0.22	2.67***	1.23***				
	4. Ketron Island	21	13.43	0.12	2.77***	-1.32***	-0.1			
	5. Fox Island	28	12.91	0.16	3.28***	1.84***	-0.61***	0.51***		
	6. Day Island	88	13.50	0.33	2.69***	1.25***	0.02	-0.07	-0.59***	
	7. Salmon Beach	104	12.24	0.05	3.96***	-2.52***	-0.29***	-1.19***	-0.68***	-1.27***
August	1. Squaxin Island	81	15.87	0.11						
	2. Istami Ledge	34	15.11	0.03	0.76***					
	3. Devil's Head	75	14.80	0.29	1.06***	0.31***				
	4. Ketron Island	16	14.40	0.09	1.47***	-0.71***	-0.40***			
	5. Fox Island	15	13.94	0.05	1.92***	1.17***	-0.86***	0.46***		
	6. Day Island	68	14.22	0.06	1.64***	0.89***	0.58***	0.18***	-0.28***	
	7. Salmon Beach	113	13.91	0.06	1.96***	-1.20***	-0.90***	-0.49***	-0.03	-0.31***
September	1. Squaxin Island	52	15.06	0.03						
	2. Istami Ledge	16	14.73	0.01	0.33***					

3. Devil’s Head	25	14.12	0.05	0.93***	0.60***				
4. Ketron Island	72	13.75	0.08	1.31***	-0.98***	-0.37***			
5. Fox Island	25	13.50	0.01	1.56***	1.23***	-0.63***	0.25***		
6. Day Island	60	13.67	0.03	1.39***	1.06***	0.45***	0.08***	-0.17***	
7. Salmon Beach	25	13.49	0.03	1.56***	-1.23***	-0.63***	-0.26***	-0.0003	-0.18***

* <0.05, ** <0.01, *** <0.001, [a] Games-Howell post hoc test on Welch’s ANOVA used to determine significance of pairwise comparisons.

Appendix B. Average integrated water column salinity as a function of survey month and site with average differences [a]

		n	M	SD	1	2	3	4	5	6
April	1. Squaxin Island	91	27.27	0.47						
	2. Istami Ledge	25	27.85	0.13	-0.58***					
	3. Devil’s Head	98	28.17	0.30	-0.99***	-0.32***				
	4. Ketron Island	32	28.20	0.09	-0.93***	0.35***	0.03			
	5. Fox Island	23	28.80	0.03	-1.53***	-0.95***	0.63***	-0.59***		
	6. Day Island	93	28.71	0.05	-1.44***	-0.86***	-0.54***	-0.51***	0.09***	
	7. Salmon Beach	114	28.95	0.03	-1.68***	-1.68***	0.78***	0.74***	0.15***	0.24***
May	1. Squaxin Island	91	28.13	0.09						
	2. Istami Ledge	32	28.24	0.04	-0.11***					
	3. Devil’s Head	91	28.57	0.05	-0.44***	-0.33***				
	4. Ketron Island	24	28.47	0.07	-0.34***	0.23***	-0.10***			
	5. Fox Island	24	28.82	0.02	-0.69***	-0.58***	0.25***	-0.35***		
	6. Day Island	89	28.80	0.13	-0.67***	-0.56***	-0.23***	-0.33***	0.02	
	7. Salmon Beach	83	28.26	0.19	-0.13***	0.02	-0.31***	-0.21***	-0.56***	-0.55***
June	1. Squaxin Island	79	28.40	0.06						
	2. Istami Ledge	26	28.50	0.03	-0.10***					
	3. Devil’s Head	73	28.90	0.08	-0.50***	-0.40***				
	4. Ketron Island	29	28.78	0.04	-0.38***	0.28***	-0.12***			
	5. Fox Island	19	29.10	0.01	-0.69***	-0.59***	0.19***	-0.31***		
	6. Day Island	88	29.08	0.04	-0.68***	-0.58***	-0.17***	-0.30***	0.02	
	7. Salmon Beach	81	29.38	0.02	-0.98***	0.88***	0.48***	0.60***	0.29***	0.03***
July	1. Squaxin Island	69	28.74	0.02						
	2. Istami Ledge	27	28.92	0.02	-0.18***					
	3. Devil’s Head	91	29.14	0.03	-0.41***	-0.23***				
	4. Ketron Island	21	29.06	0.03	-0.33***	0.15***	-0.08***			
	5. Fox Island	28	29.33	0.04	-0.60***	-0.42***	0.19***	-0.27***		
	6. Day Island	88	29.16	0.03	-0.43***	-0.25***	-0.02*	-0.10***	0.17***	
	7. Salmon Beach	104	29.49	0.02	-0.75***	0.57***	0.34***	0.42***	0.15***	0.32***
August	1. Squaxin Island	81	29.12	0.03						
	2. Istami Ledge	34	29.27	0.01	-0.15***					
	3. Devil’s Head	75	29.44	0.03	-0.32***	-0.17***				
	4. Ketron Island	16	29.46	0.01	-0.34***	0.19***	0.02			
	5. Fox Island	15	29.55	0.01	-0.43***	-0.28***	0.11***	-0.09***		
	6. Day Island	68	29.51	0.02	-0.39***	-0.24***	-0.07***	-0.05***	0.04***	
	7. Salmon Beach	113	29.59	0.03	-0.47***	0.32***	0.15***	0.13***	0.04***	0.08***
September	1. Squaxin Island	52	28.97	0.02						
	2. Istami Ledge	16	29.13	0.01	-0.16***					

3. Devil’s Head	25	29.45	0.01	-0.47***	-0.32***				
4. Ketron Island	72	29.49	0.09	-0.51***	0.36***	0.04			
5. Fox Island	25	29.64	0.01	-0.67***	-0.52***	0.20***	-0.16***		
6. Day Island	60	29.50	0.03	-0.53***	-0.37***	-0.05***	-0.01	0.14***	
7. Salmon Beach	25	29.60	0.01	-0.63***	0.47***	0.15***	0.15***	-0.04***	0.10***

* <0.05, ** <0.01, *** <0.001, [a] Games-Howell post hoc test on Welch’s ANOVA used to determine significance of pairwise comparisons.

Appendix C. Average integrated water column Nitrate-Nitrite as a function of survey month and site with average differences [a]

		n	M	1	2	3	4
April	1. Squaxin Island	1	18.78				
	2. Devil's Head	1	23.20	-4.42			
	3. Day Island	1	22.50	-3.72	0.7		
	4. Salmon Beach	--	--	--	--	--	
	5. North Beach	--	--	--	--	--	--
May	1. Squaxin Island	1	6.09				
	2. Devil's Head	1	13.86	-7.78			
	3. Day Island	1	13.61	-7.52	0.26		
	4. Salmon Beach	--	--	--	--	--	
	5. North Beach	--	--	--	--	--	--
June	1. Squaxin Island	1	2.89				
	2. Devil's Head	1	10.96	-8.07			
	3. Day Island	1	12.95	-10.06	-1.99		
	4. Salmon Beach	--	--	--	--	--	
	5. North Beach	--	--	--	--	--	--
July	1. Squaxin Island	1	4.94				
	2. Devil's Head	1	12.86	-7.92			
	3. Day Island	--	--	--	--		
	4. Salmon Beach	1	13.24	-8.30	0.385	--	
	5. North Beach	--	--	--	--	--	--
August	1. Squaxin Island	1	0.59				
	2. Devil's Head	1	11.54	-10.95			
	3. Day Island	--	--	--	--		
	4. Salmon Beach	--	--	--	--	--	
	5. North Beach	--	--	--	--	--	--
September	1. Squaxin Island	--	--				
	2. Devil's Head	--	--	--			
	3. Day Island	1	15.40	--	--		
	4. Salmon Beach	1	15.02	--	--	-0.39	
	5. North Beach	1	22.95	--	--	7.55	-7.94

[a] Games-Howell post hoc test on Welch's ANOVA used to determine significance of pairwise comparisons.

Appendix D. Average suspended sediment concentrations (mg/L) as a function of site with average differences [a]

	n	M	SD	1	2	3	4
1. Squaxin Island	23	2.15	0.70				
2. Devil's Head	24	1.40	0.86	– 0.75 *			
3. Day Island	21	1.28	1.05	– 0.86 *	– 0.12		
4. Salmon Beach	3	0.83	0.12	– 1.31 ***	– 0.57 *	– 0.45	
5. North Beach	3	1.14	0.29	– 1.00 *	– 0.26	– 0.14	0.31

* <0.05, ** <0.01, *** <0.001, [a] Games-Howell post hoc test on Welch's ANOVA used to determine significance of pairwise comparisons.

Appendix E. Average annual light attenuation as a function of site with average differences [a]

	n	M	SD	1	2	3	4	5	6
1. Squaxin Island	36	0.42	0.11						
2. Itsami Ledge	15	0.37	0.08	-0.05					
3. Devil's Head	33	0.32	0.06	– 0.09**	– 0.05				
4. Ketron Island	17	0.31	0.05	– 0.1 ***	– 0.06	– 0.01			
5. Fox Island	15	0.28	0.06	– 0.13 ***	– 0.09 *	– 0.04	– 0.03		
6. Day Island	36	0.33	0.10	– 0.09 **	– 0.04	< 0.01	0.01	0.05	
7. Salmon Beach	33	0.31	0.07	– 0.1 ***	– 0.06	– 0.01	– 0.01	0.03	– 0.02

* <0.05, ** <0.01, *** <0.001, [a] Games-Howell post hoc test on Welch's ANOVA used to determine significance of pairwise comparisons.