N A T U R A L R E S O U R C E S

Puget Sound Seagrass Monitoring Report

Monitoring Year 2018 - 2020

03/31/2022

PUGET SOUND ECOSYSTEM MONITORING PROGRAM

WASHINGTON STATE DEPARTMENT OF NATURAL RESOURCES
HILARY S. FRANZ | COMMISSIONER OF PUBLIC LANDS

DNR monitors abundance and depth distribution of native seagrasses to determine status and trends in greater Puget Sound through the Submerged Vegetation Monitoring Program (SVMP) (https://www.dnr.wa.gov/programs-and-services/aquatics/aquatic-science/nearshore-habitat-eelgrass-monitoring).

The Submerged Vegetation Monitoring Program is a component of the Puget Sound Ecosystem Monitoring Program (PSEMP) (https://www.psp.wa.gov/PSEMP-overview.php).

Cover Photo: Eelgrass (*Zostera marina*) at Joemma Beach State Park.

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The following document fulfills DNR's Eelgrass Monitoring Performance measure. It also fulfills tasks in the Puget Sound Partnership's Action Agenda by providing information on the status and trends of one of the selected indicators of environmental health in Puget Sound.

The authors of this report are Bart Christiaen, Lisa Ferrier, Pete Dowty, Jeff Gaeckle and Helen Berry. Lauren Johnson and Melissa Sanchez played a critical role in the video data collection and post-processing for the work summarized in this report.

The Nearshore Habitat Program would like to give special recognition to Ian Fraser of Marine Resources Consultants who continues to play a significant role in the success of the project. Marine Resources Consultants showed great dedication and logged many hours of sea time collecting data for the project.

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Copies of this report may be obtained from:

http://www.dnr.wa.gov/programs-and-services/aquatics/aquatic-science/nearshore-habitat-publications

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Contents

Executive summary

The Washington State Department of Natural Resources (DNR) manages 2.6 million acres of State-Owned Aquatic Lands for the benefit of current and future residents of Washington State. DNR's stewardship responsibilities include protection of native seagrasses, such as eelgrass (*Zostera marina*) and surfgrass (*Phyllospadix spp.*), important components of nearshore ecosystems in greater Puget Sound. DNR monitors abundance and depth distribution of native seagrasses to determine status and trends in greater Puget Sound through the Submerged Vegetation Monitoring Program (SVMP). Soundwide monitoring was initiated in 2000. The monitoring results are used by DNR for the management of State-Owned Aquatic Lands, and by the Puget Sound Partnership as one of 25 Vital Signs to track progress in the restoration and recovery of Puget Sound.

Key findings:

The San Juan Islands and Cypress Island emerges as a region of concern

- The San Juan Islands and Cypress Island has been identified as a region of concern, where sites with declines in eelgrass area significantly outnumber sites with increases, both over the long-term (2000-2020) and in recent years (2015-2020). Over the long-term there were 16 sites with eelgrass declines and 4 sites with increases (out of 89 sites sampled in the region). In recent years there were 15 sites with eelgrass declines and no sites with increases.
- Some of the largest eelgrass losses in the San Juan Islands have occurred in embayments. The most notable examples are Westcott Bay on San Juan Island (near total loss), Reef Net Bay on Shaw Island (> 60% loss), Shallow Bay on Sucia Island (~ 75% loss), and Swifts Bay on Lopez Island (~50% loss).
- Local declines are likely due to a variety of stressors, such as physical damage, local water quality impairments, and eelgrass wasting disease.

River delta eelgrass bed dynamics

- In both the Skokomish and the Nisqually deltas, eelgrass populations fluctuated by over 50% between 2000 and 2020.
- At a site in Skagit Bay (flats20), almost 200 ha of eelgrass was lost between 2004 and 2020. The rate of decline increased in recent years. Eelgrass loss at this location was likely due to erosion caused by an avulsion of the north fork of the Skagit River. The overall loss of eelgrass in Skagit Bay is more extensive, as adjacent sites were impacted as well.

Other site-level eelgrass bed dynamics

- Out of 214 randomly selected panel sites, a similar number of sites declined (n = 38) and increased (n = 33) between 2000 and 2020. However, in recent years (2015-2020) sites with declines in eelgrass area (n = 32) outnumbered sites with increases (n = 9).

- Between 2000 and 2020, there were more declines than increases at sites with small eelgrass beds, while there were more increases than declines at sites with medium and large beds. Between 2015 and 2020, there were more declines than increases regardless of bed size.

Soundwide eelgrass area was relatively stable over the long term (2000-2020)

- Soundwide eelgrass area increased between 2004 and 2016, but declined between 2016 and 2020. The magnitude of these changes was relatively small as compared to the total amount of eelgrass present in greater Puget Sound.
- Overall, eelgrass populations were relatively stable between 2000 and 2020. The relative stability sets Puget Sound apart from many other developed areas, where substantial system-wide declines are ongoing.
- The annual estimates of soundwide eelgrass area were 21,283 +/- 1,571[1] ha in 2018, 23,512 +/- 1,864 ha in 2019, and 22,845 +/- 1,864 ha in 2020. The 3-year soundwide average for 2018-2020 was approximately 22,100 +/- 1,100 ha (Figure A).
- The Puget Sound Eelgrass Vital Sign Indicator target of 20% increase in soundwide eelgrass area by 2020 has not been met. Stressors that affect seagrass in Puget Sound will likely need to be reduced to achieve significant soundwide gains in eelgrass area, depth distribution, and overall health.

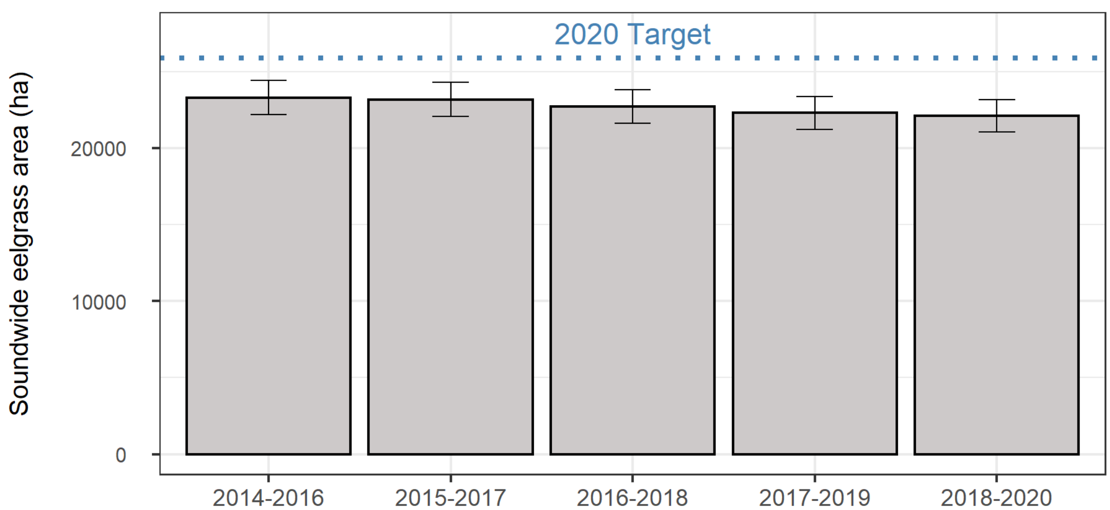

Figure A: Estimates of soundwide eelgrass area (ha) based on pooled 3-year samples in greater Puget Sound. The dotted blue shows the recovery target of a 20% increase in eelgrass area relative to a 2000-2008 baseline by 2020. Error bars are standard error.

[1] Mean +/- standard error

1 Introduction

1.1 The role of seagrass beds in nearshore environments

Seagrasses are flowering plants that grow submerged in marine environments. These plants flower, fertilize, and set seeds underwater, but often spread through vegetative growth (Cox 1998, Kendrick et al. 2012). There are approximately 60 species worldwide, which belong to 5 plant families (Den Hartog and Kuo 2006, Green and Short 2003). There are six species of seagrass in Washington State: *Zostera marina*, *Zostera japonica*, *Phyllospadix scouleri*, *Phyllospadix torreyi*, *Phyllospadix serrulatus*, and *Ruppia maritima*. Out of these, *Zostera marina* (eelgrass) is by far the most abundant and widespread in greater Puget Sound. Seagrass beds provide many important ecosystem services. They are refuge, foraging, and spawning habitat for a wide variety of organisms, including commercially important species such as salmonids, Pacific herring, and Dungeness crab (Fresh 2006, Pentilla 2007, Plummer and al. 2013). Seagrass beds are highly productive and export large quantities of organic matter to adjacent ecosystems, both in the form of detritus and as animal biomass (Heck et al. 2008). Because of their high primary productivity and the relatively low decomposition rates of organic matter in marine sediments, they are efficient carbon sinks (Mcleod et al. 2011). Seagrasses can improve water clarity by reducing resuspension of soft sediments, and have the potential to mitigate some effects of ocean acidification (de Boer 2007, Hendriks et al. 2013, Pacella et al. 2018). Seagrass beds can also limit algae blooms and remove harmful bacteria from the water column (Lamb et al. 2017, Inaba et al. 2017, Jacobs-Palmer et al. 2020, Reusch et al. 2021). In addition, they are valued hunting grounds and ceremonial foods for Native Americans and First Nations Peoples in the Pacific Northwest (Suttles 1951, Felger and Moser 1973, Kuhnlein and Turner 1991, Wyllie-Echeverria and Ackerman 2003).

1.2 Seagrass as indicator of ecosystem health

Seagrass ecosystems provide important ecosystem services, but they are sensitive to multiple pressures (Turschwell et al. 2021). Declines in seagrass extent are often attributed to light limitation by phytoplankton blooms, or overgrowth with epiphytes or green algae under eutrophic conditions (Burkholder et al. 2007). Excessive inputs of organic matter can lead to sulfide toxicity to the plants, which can be exacerbated by low light availability or low dissolved oxygen concentrations in the water column (Holmer and Bondgaard 2001, Plus et al. 2003, Holmer et al. 2005). Seagrass beds can be physically damaged by anchoring, prop scars, trawling and coastal development (Hemminga and Duarte, 2000). Other pressures include high water temperatures and eelgrass wasting disease (Sullivan et al. 2013, Thomson et al. 2015, Lefcheck et al. 2017, Graham et al. 2021). Because of their importance and sensitivity, seagrass species such as eelgrass are often used as indicators for the health of nearshore ecosystems (Krause-Jensen et al. 2005, Mumford 2007).

One potential complication of using seagrasses as bio-indicators of ecosystem health is that impacts of anthropogenic stressors can be difficult to distinguish from natural variability. Different seagrass species have different life history characteristics, and colonizing or opportunistic species tend to be more variable than persistent species (Kilminster et al. 2015). Eelgrass, the predominant seagrass in greater Puget Sound, can be characterized as an opportunistic species. It may appear stable on a regional scale, but shows significant variability at smaller spatial scales, with adjacent sites sometimes exhibiting opposite trends over time (Shelton et al. 2016, Christiaen et al. 2019). Eelgrass is variable on seasonal and interannual timescales, and can be impacted by long-term patterns in environmental drivers such as the El Nino Southern Oscillation (Thom et al. 2014). As such, it is important to consider multiple temporal and spatial scales when using eelgrass as an indicator for ecosystem health.

1.3 DNR’s seagrass monitoring program

The Washington State Department of Natural Resources (DNR) is steward of 2.6 million acres of State-Owned Aquatic Lands. DNR monitors native seagrasses (*Zostera marina* and *Phyllospadix spp*.) across the nearshore of greater Puget Sound as part of its stewardship responsibilities. Observations of the non-native seagrass *Zostera japonica* are recorded as part of monitoring but these are excluded when calculating SVMP area estimates because this species has a number of distinct resource management issues (Shafer et al. 2014).

DNR’s Submerged Vegetation Monitoring Program (SVMP) started in 2000 and uses towed underwater videography to estimate the area and depth distribution of native seagrass species in greater Puget Sound based on a probabilistic sample design. Monitoring is conducted on an annual basis. In addition to sampling with a probabilistic design, the SVMP has comprehensively surveyed entire stretches of shoreline through a number of projects with local governments and Tribes (Christiaen et al. 2018, Christiaen et al. 2020a, Christiaen et al. 2020b). Since 2014, these collaborations have become a major addition to the monitoring program.

In Washington State, the Puget Sound Partnership uses the soundwide eelgrass area estimates from DNR’s monitoring program as one of the indicators for the health of Puget Sound. This indicator provides valuable information on the overall status of eelgrass, but is not sensitive to changes in eelgrass area on smaller spatial scales. In 2022, the existing indicator will be complemented by a new *short and long-term eelgrass site status indicator*. This new site status indicator will allow us to highlight important declines in eelgrass in particular areas, such as the San Juan Islands, and in sensitive habitat types, especially protected embayments. Both indicators will be grouped in the new Beaches and Marine Vegetation Vital Sign.

This report summarizes the methods and key results from all SVMP data collected between 2000 and 2020, and is based on over 15 million data points, collected during 39,781 video surveys at 867 sites[2].

1.4 Data access

The SVMP monitoring database and a User Manual are available through the DNR GIS data download web page. The User Manual (Dowty et al. 2019) includes a more detailed description of project methods than are included in this report. The data is also accessible through an online data viewer. A summary of results can also be found on the webpage of the Eelgrass Vital Sign by the Puget Sound Partnership. These resources are available at the following locations:

https://www.dnr.wa.gov/programs-and-services/aquatics/aquatic-science/nearshore-habitat-eelgrass-monitoring

https://www.dnr.wa.gov/programs-and-services/aquatics/aquatic-science/puget-sound-eelgrass-monitoring-data-viewer

http://data-wadnr.opendata.arcgis.com

Vital Signs | Eelgrass (wa.gov)

[2] These numbers include some data collected by partner organizations such as the Island County MRC.

2 Methods

The Submerged Vegetation Monitoring Program (SVMP) is a regional monitoring program for native seagrass species in the US portion of the Salish Sea, here referred to as 'greater Puget Sound'. The goals of this program are to assess where different seagrass species grow, how much area is covered by native seagrass species (predominantly eelgrass or *Zostera marina*), and how these area estimates change over time (both at the site level and soundwide). To accomplish these goals, the SVMP uses a probabilistic sample design that has evolved over time. Here, we will give a short overview of the current methodology for the SVMP. A comprehensive presentation of SVMP methods is available in the User Manual distributed with the digital dataset (Dowty et al. 2019).

2.1 Sample design

The SVMP uses towed underwater videography to generate area estimates and depth distributions of native seagrass species in greater Puget Sound. This method makes it possible to assess the deep edge of native seagrass beds in areas where the water is too turbid to use aerial imagery, and allows for differentiating different types of marine vegetation. In order to employ this method on a large spatial scale (over 3000 km of shoreline), the SVMP uses probabilistic sampling according to a statistical framework to estimate eelgrass area at a set of randomly selected sites, and extrapolate results in different sub-regions and on a soundwide scale.

2.1.1 Site delineation

The statistical framework of the SVMP divides greater Puget Sound in 2,467 potential sample sites, spread over two sample frames based on geomorphological characteristics: flats sites (n= 74) and fringe sites (n = 2,393).

- The flats category includes embayments, tide flats and river deltas, potential habitat that is best represented as areal sample units. The potential eelgrass habitat for each flats site is calculated as the area between the shoreline and the -6.1 m (MLLW) depth contour.
- The fringe category contains potential habitat along a narrow band parallel to the shoreline, and is well represented by linear sample units. Fringe sites occupy 1000 m along the -6.1 m bathymetry line.

Each sample frame is further divided into 3 strata: the flats frame is divided into core (n = 4), persistent flats (n = 3), and rotational flats (n = 67), while the fringe frame is divided into core (n = 2), narrow fringe (n = 1,965) and wide fringe (n = 426).

2.1.2 Site selection

The SVMP encompasses several sample efforts, which are referred to as studies in this report. The largest studies in the dataset are the soundwide study (SVMPsw), and a number of recent studies that were conducted in partnership with local governments and Tribes.

2.1.2.1 Soundwide study (SVMPsw)

The soundwide study, which involves sampling 78-80 sites per year, is the core of the SVMP, and informs the eelgrass indicators of the new 'Beaches and Marine Vegetation Vital Sign' by the Puget Sound Partnership. This study is based on a statistical design that compromises between estimating the extent of native seagrass throughout greater Puget Sound (sample as many different sites as possible) and detecting trends over time (resample the same sites over time). Up to 2014, the soundwide study employed a design with 20% rotation, where sites were randomly selected within each stratum, and sampled for 5 consecutive years before being replaced with new randomly selected sites (Dowty et al. 2019). In 2015, the soundwide study switched to a new 3-rotating panel design that improves the ability to detect trends in soundwide eelgrass area, but limits the number of new sites over time. As part of the rotating panel design we revisit all sites sampled in either 2004, 2009 or 2014 on a 3-year basis; and use 3-year pooled samples to generate unbiased estimates of soundwide seagrass area in greater Puget Sound (Figure 1). The new 3-panel design covers 214 unique sites. Some of these sites are repeated in more than one panel, but the majority of sites are sampled once every 3 years.

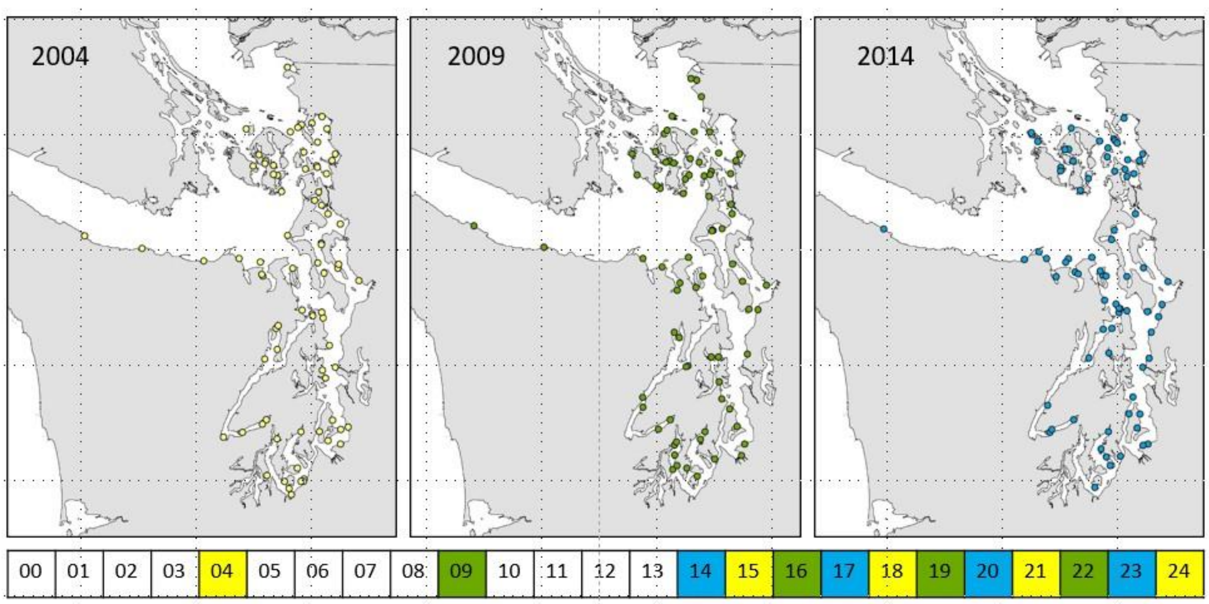

Figure 1: Site selection of the current 3-year rotating panel design. In 2015 and 2018, we resampled sites from the 2004 panel. In 2016 and 2019, we resampled sites from 2009, and in 2017 and 2020, we resampled sites from 2014.

2.1.2.2 *Partnerships with local governments and Tribes*

In recent years, collaborations with local governments and Tribes have become a major component of the SVMP. Between 2014 and 2020, DNR has sampled 378 sites in the central basin of Puget Sound with over 3800 transects perpendicular to shore, as part of a series of collaborations with the Suquamish Tribe (Christiaen et al. 2018, Christiaen et al. 2021), the City of Bainbridge Island (Christiaen et al. 2017), and King County (Christiaen et al. 2020a). This translates to over 378 km with one transect for approximately every 100 m of shoreline. More recently, DNR has collaborated with Snohomish County to survey eelgrass, understory kelp and other macroalgae along the shoreline of Snohomish County from Tulalip Bay to the King County line (Christiaen et al. 2020b, Christiaen et al. 2022). As a result of these collaborations, the number of sites visited by the SVMP has almost doubled in recent years (Figure 2).

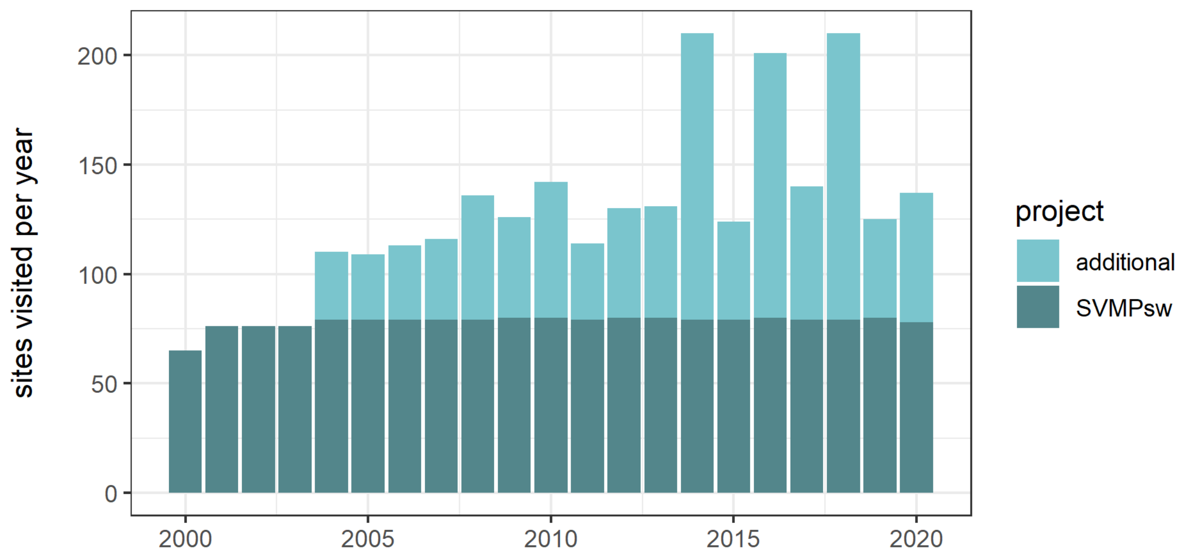

Figure 2: Sites visited by the SVMP between 2000 and 2020. The darker color indicates sites sampled as part of the soundwide study (SVMPsw). The lighter colors indicate sites sampled as part of collaborations with local governments and Tribes, or additional sites of interest.

2.1.2.3 *Sites of interest*

In addition to the soundwide study and the collaborations with local governments and Tribes, we also monitor a number of sites of interest. These additional sites are not used to generate an estimate of soundwide seagrass area. However, they do allow us to expand the footprint of the monitoring to new parts of Puget Sound, or focus on site level trends in areas of special interest. A recent example is a survey of eelgrass inside Kilisut Harbor in response to the Kilisut Harbor Restoration Project. In 2020, an earthen causeway on WA State Route 116 was replaced by a 440 ft. bridge, which restored the historical tidal connection between southern Kilisut Harbor and Oak Bay. We surveyed eelgrass beds along the entire shoreline of Kilisut Harbor, as well as several nearby control sites, in 2017, 2020, and 2021 (with an additional survey planned for 2023) to assess the potential beneficial impact of increased flushing on eelgrass beds inside Kilisut Harbor and along the northeastern shoreline of Oak Bay.

2.1.3 Site surveys

Sites are surveyed using a modified line-intercept technique (Norris et al. 1997). At each location, we tow an underwater video camera along a number of randomly selected transects that are oriented perpendicular to shore and span the entire depth range of native seagrass. To estimate eelgrass area, we multiply the fractions of transects covered by native seagrass (weighted by transect length) by the area of a sample polygon.

- Before 2016, transects were selected using new draw simple random sampling (SRS) in the area where eelgrass occurs at a site. For this, we delineated a sample polygon that includes all eelgrass at a site prior to sampling, and selected random transects perpendicular to shore throughout this polygon. Each time a site was visited, these transects were drawn anew.
- Starting in 2016, the SVMP transitioned to stratified random transects (STR), and repeat transect surveys[3]. Here, the sample polygon spans the entire length of the site regardless of the seagrass distribution. Stratified random transects are selected by dividing the -6.1 m bathymetry line into sections of equal length, and randomly selecting one transect that is perpendicular to shore in each of these sections. A small subset of sites has also been sampled using systematic transects (SYS). Here, -6.1 m bathymetry line is again divided into sections of equal length. There is one transect per section, but each transect is equidistant from each other.

2.2 Data collection and post-processing

2.2.1 Data collection

Field sampling is typically conducted between May and October, from the 11 m (36 ft.) research vessel, the *R/V* Brendan D II, operated by Marine Resources Consultants (Figure 3). The equipment used for sampling is listed in Table 1. During sampling, the vessel deploys a weighted towfish with an underwater video camera mounted in a downward-looking orientation (Figure 4). The towfish is deployed directly off the stern of the vessel using a cargo boom and hydraulic winch. During transect sampling, a technician adjusts the position of the towfish to fly the camera above the substrate. Parallel lasers mounted 10 cm apart on the towfish provide a scaling reference in the video image. A 500 watt underwater light provides illumination when needed.

Survey equipment simultaneously records the presence/absence of marine vegetation, position, depth, and time of day. Time and position data are acquired using a differential global positioning system (DGPS) with ability to utilize satellite based augmentation services

[3] Note that in 2016, 2017, and 2018 we started sampling sites for the SVMP soundwide study with both transects selected by SRS and STR. In practice, this means that we repeat transects from the corresponding panel year (2004, 2009, and 2014), and supplement these with additional transects based on a stratified design. By sampling using SRS we maintain backwards compatibility in the dataset. By supplementing the SRS transects with STR, we have a better ability to detect new seagrass patches at sites where sample polygons previously did not span the entire length of the site. From 2019 on, sites are sampled using repeat STR transects only, except at locations with very small seagrass beds.

(SBAS). The antenna is located on top of the cargo boom directly above the towfish and camera, ensuring that the position data reflect the geographic location of the camera (Figure 4). Depth is measured using a Garmin Fishfinder 250 and a BioSonics MX habitat echo sounder. Both are linked to the differential global positioning system (DGPS) so that collected depth data is location and time specific (Table 1).

A laptop computer equipped with a video overlay controller and data logger software integrates the DGPS data, user supplied transect information (transect number and site code), and the video signal at one second intervals. Video images with overlain DGPS data and transect information are simultaneously recorded on DVDs, and D/V hard drives. Date, time, position, and transect information are stored on the computer at one second intervals. A real-time plotting system integrates National Marine Electronic Association 0132 standard sentences produced by the DGPS, two depth sounders, and a user-controlled toggle switch to indicate presence of marine vegetation.

Table 1: Current equipment on the *R/V* Brandon D II (note that equipment has changed throughout the course of the monitoring program)

Equipment	Manufacturer/Model
Differential GPS Unit	*Hemisphere VS330 with Satellite Based Augmentation System (SBAS, sub-meter accuracy)*
Echosounders	*Primary: BioSonics Mx Habitat Echosounder / Secondary: Garmin Fishfinder 250*
Underwater Camera	*Ocean Systems Deep Blue SD (downward facing), Ocean Systems Deep Blue HD (forward facing)*
Underwater Light	*Deep Sea Power and Light Led SeaLite*
Lasers	*Deep Sea Power & Light (10 cm spread, red)*
DVD Recorder	*Sony RDR-GX7 + Intuitive Circuits TimeFrame Video Overlay Controller*
Image Recording	*3 Atomos Ninja 2 DV Recorders, ProRes format + VideoLogix Proteus II Video Overlay Controller*
Computer systems	*Rugged laptop with Microsoft Office and Hypack Max hydrographic software (capable of accepting ESRI ArcGIS files). HP 4480 Color printer*
Camera	*Nikon Coolpix waterproof camera*

2.2.2 Post processing

Video is reviewed, and each transect segment of nominal one-meter length (and one-meter width) is classified with respect to the presence of native (*Z. marina*, *Phyllospadix spp.*) and non-native seagrass species (*Z. japonica*). All presence and absence classification results are recorded with corresponding spatial information, and stored in an ArcGIS geodatabase. The fractional cover of eelgrass and surfgrass along transects is used to calculate site seagrass area. Depth information collected along each transect is used to estimate mean maximum and minimum depth of seagrass at each site. All measured depths are corrected to the MLLW datum by adding the transducer offset, subtracting the predicted tidal height for the site and adding the tide prediction error (calculated using measured tide data from the National Oceanic and Atmospheric Administration website http://co-ops.nos.noaa.gov/data_res.html). Corrected depth data are integrated with survey data information, so each video frame has an associated date/time, GPS position, and depth measurements corrected to MLLW datum.

Figure 3: All data were collected from the *R/V* Brendan D II, using towed underwater videography and depth sounding instrumentation.

Figure 4: The *R/V* Brendan D II is equipped with a weighted towfish that contains an underwater video camera mounted in a downward looking orientation, dual lasers for scaling reference, and underwater lights for night work (A). The towfish is deployed directly beneath the DGPS antenna attached to the A-frame cargo boom, ensuring accurate geographic location of the camera (B).

2.3 Data analysis

Data was analyzed with ArcGIS (Esri ArcGIS Desktop, Release 10.6.1, Redlands CA) and R (R Core Team 2018). We used several R-packages, including "broom" (Robinson and Hayes 2018), "dplyr" (Wickam et al. 2018), "ggplot2" (Wickam 2016), "tidyr" (Wickam and Henry 2018), and "weights" (Pasek et al. 2018).

2.3.1 Site area estimates

Eelgrass area[4] at each site was calculated using ArcGIS software and the site database file in the following sequential steps:

1. Calculate the area within the sample polygon;
2. Calculate the fraction of native seagrass (predominantly eelgrass) along each random line transect within the sample polygon;
3. Calculate the mean fraction and associated variance, weighed by transect length[5];
4. Estimate the overall eelgrass area and variance at the site by extrapolating the mean fraction along random transects over the sample polygon area.

2.3.2 Site level trends

We used a hybrid approach to assess trends in eelgrass area at individual sites. We determined trends over two periods of time: (1) all data collected between 2000 and 2020 (long-term trends), and (2) data collected between 2015 and 2020 (recent trends). At each site we calculate inverse variance weighted regressions of site eelgrass area estimates over time (which includes all samples taken at a site), as well as pairwise T-tests of the vegetated fraction of repeat transects for sites that were sampled using the same set of transects at different points in time (the exact years of the repeat samples vary depending on the site[6]).

These two analyses highlight different aspects of changes in eelgrass distribution. The regressions are based on all area estimates at a site. These analyses are less precise but they use all available information and encompass the entire time series at each site. Paired transect analyses are more precise since they compare changes in cover at the transect level, but they are limited to comparing 2 years from the entire time series. To determine overall trends in eelgrass area at each site, we combined results from the regression and all paired transect analyses, and confirmed any potential trends ($p < 0.05$) based on visual assessment of the change in spatial distribution over time in ArcGIS.

[4] Throughout this report we use the term 'eelgrass area' instead of 'native seagrass area', as eelgrass is by far the most abundant species in our region, and the main target for DNR's seagrass monitoring program. However, our area estimates are based on both *Z. marina* and *Phyllopadix sp*.

[5] We calculate variance for stratified random samples using the textbook variance estimator. This formula may overestimate actual variance for stratified random samples and systematic samples, and is thus a conservative estimator of variance for these sampling schemes (McGarvey et al. 2016).

[6] These are mainly the panel sites. At these sites we resampled transects in 2004/2018, 2009/2016, 2014/2017, 2016/2019, and 2017/2020.

2.3.3 Site depth distribution

Eelgrass depth characteristics for each site were estimated using descriptive statistics (i.e., the 2.5^{th}, 10^{th}, 25^{th}, 50^{th}, 75^{th}, 90^{th}, and 97.5^{th} percentile) for all eelgrass observations along all SRS, STR, or SYS transects at a site (all years pooled).

To calculate a depth distribution, eelgrass observations were binned according to their depth relative to MLLW in 0.5 m bins. The number of observations in each depth bin was divided by the total number of eelgrass observations at the site. This fraction was multiplied by the estimated eelgrass area at the site to estimate the area of eelgrass in each depth bin at the site. We used the following formula to estimate eelgrass area in each depth bin at each site:

$$a_{jk} = A_j \frac{c_{jk}}{\sum_{k=1}^{n} c_{jk}} \qquad (1)$$

Where a_{jk} is eelgrass area in each histogram bin (k) at site (j), c_{jk} is the count of observations in depth bin k, and A_j is average eelgrass area at site j.

2.3.4 Soundwide area estimates

The estimator for eelgrass area within a stratum takes one of three forms depending on whether it is a fringe stratum subject to probabilistic sampling (with linear extrapolation), a flats stratum subject to probabilistic sampling (with areal extrapolation) or a stratum that is subject to complete census (no extrapolation, such as core and persistent flats). For a stratum with *N* sites that is subject to complete census, eelgrass area within the stratum (*B*) is estimated by (2) with the associated variance estimator (3), where $\hat{X}_i$ is the estimated average eelgrass area at the i^{th} site in the stratum and $Var_{\hat{X}_i}$ is the associated variance at the site.

$$\hat{B} = \sum_{i=1}^{N} \hat{X}_i \qquad (2)$$

$$Var_{\hat{B}} = \sum_{i=1}^{N} Var_{\hat{X}_i} \qquad (3)$$

For a fringe stratum with *N* sites, where each site is represented by a 1000 m line segment on the -6.1 m isobath, the estimator for eelgrass area in the stratum is given by (4): n is the number of sites actually surveyed in the stratum, L_N is the total linear length of sample units in the stratum (i.e., the sampled population,) and L_T is the total length of the target population which includes orphan segments that are shorter than 1000 m but otherwise meet the criteria for inclusion in the stratum. The estimator for the associated variance is (5).

$$\hat{B} = \left(\frac{L_T}{L_N}\right) \left[\frac{N}{n} \sum_{i=1}^{n} \hat{X}_i\right] \qquad (4)$$

$$Var_{\hat{B}} = \left(\frac{L_T}{L_N}\right)^2 \left[N^2 \frac{\left(1-\frac{n}{N}\right)}{n} s^2{}_{\hat{X}_i} + \frac{N}{n} \sum_{i}^{n} Var_{\hat{X}_i}\right],$$

where $s^2{}_{\hat{X}_i} = \frac{\sum_{i=1}^{n}(\hat{X}_i - \bar{\hat{X}})^2}{(n-1)}$ and $\bar{\hat{X}} = \frac{\sum_{i=1}^{n}\hat{X}_i}{n}$ (5)

For the rotational flats stratum, we estimate eelgrass area with a ratio estimator (Cochran 1977) of the form (6), where a_i is the area of the *i*th flats site and $A = \sum_{i=1}^{n} a_i$ is the total flats area within the stratum. The estimator for the variance of this estimate was derived by Skalski (2003) and is given by (7).

$$\hat{B} = A \left[\frac{\sum_{i=1}^{n}\hat{X}_i}{\sum_{i=1}^{n}a_i}\right] \quad (6)$$

$$Var_{\hat{B}} = N^2 \left(1 - \frac{n}{N}\right) \frac{\sum_{i=1}^{n}(X_i - a_i\,\hat{R})^2}{n\,(n-1)} + \frac{N\,\sum_{i=1}^{n} Var_{\hat{X}_i}}{n} ,$$

where $\hat{R} = \frac{\sum_{i=1}^{n}\hat{X}_i}{\sum_{i=1}^{n}a_i}$ (7)

2.3.5 Multiyear estimates of soundwide eelgrass area

Our annual estimates of soundwide eelgrass area are based on a sample of 78 to 80 sites (depending on the panel year). It is possible to increase the sample size for this calculation by combining sites from multiple panels. By combining data from 3 panels the sample size increases from ~78-80 to 214. Calculations are similar to the soundwide eelgrass area estimate as described in section 2.3.4. In addition to the soundwide estimate, we also calculate estimates in 3 sub-regions of greater Puget Sound: Central Puget Sound and Hood Canal (CPS/HDC), Northern Puget Sound and Saratoga Whidbey Basin (NPS/SWH), and the San Juan Islands and Strait of Juan de Fuca (SJS).

Several sites (mainly core and persistent flats) are sampled each year. For these sites we calculate the mean eelgrass area and pooled variance by (8) and (9), where X_i = mean eelgrass area at a site for year i; k = the number of years a site is sampled, n_i = the sample size for year i, and Var_i is the sample variance the site for year i.

$$\overline{X} = \frac{\sum_{i=1}^{k} Xi}{k} \quad (8)$$

$$Pooled\ Var = \frac{\sum_{i=1}^{k}(n_i - 1)\,Vari}{\sum_{i=1}^{k}(n_i - 1)} \quad (9)$$

2.3.6 Soundwide area change estimates

To estimate change in soundwide eelgrass area between two years represented by the same sample of sites, we first calculate change in eelgrass area for each individual site by subtracting the year1 estimate from the year2 estimate. We propagate the uncertainty around the change estimate by (10), where $se_{\widehat{Diff_i}}$ is the standard error for the difference between the average eelgrass area estimates for both years at site i.

$$se_{\widehat{Diff_i}} = \sqrt[2]{se_{\widehat{year2_i}}^{\,2} + se_{\widehat{year1_i}}^{\,2}} \qquad (10)$$

We then use the change in eelgrass area and uncertainty around the change estimate as input for an extrapolation per stratum, similar to the soundwide eelgrass area estimate in section 2.3.4.

2.3.7 Soundwide depth distribution

In order to calculate the depth distribution of eelgrass on a soundwide scale and in each of 3 sub-regions of greater Puget Sound, we use the following approach:

- First we estimate eelgrass area in 0.5-m depth bins for each individual site in the 3-year rotating panel using the calculations described in section 2.3.3.
- In a second step, we complete the depth distribution for each site by adding 0 values for each depth bin with no eelgrass present (so there is one value for each 0.5-m depth bin between +1.5 and -15 m MLLW at each location).
- We then estimate the mean eelgrass area per depth bin for the core, persistent flats, rotational flats, narrow fringe, and wide fringe strata by extrapolating these values using the calculations described in section 2.3.4.

The regional/soundwide depth distribution is calculated by adding the estimates for the different strata per habitat type (flats vs. fringe), and expressing them as the % of eelgrass area per depth bin and per habitat type.

3 Results

3.1 Where seagrass is found

3.1.1 Seagrass species in the Salish Sea

There are six species of seagrass in Washington State: *Zostera marina*, *Zostera japonica*, *Phyllospadix serrulatus*, *Phyllospadix scouleri*, *Phyllospadix torreyi*, and *Ruppia maritima* (Figure 5). Only four species have been documented by the SVMP (*Z. marina*, *Z. japonica*, *P. serrulatus*, and *P. scouleri*) in greater Puget Sound. Along Washington's shorelines, *P. torreyi* is mostly limited to the outer coast. *R. maritima* (widgeongrass) is an opportunistic species that prefers brackish waters. This species has been reported in several estuaries in greater Puget Sound, but it tends to grow too shallow for our research vessel.

Eelgrass (*Z. marina*) is typically found on sandy and muddy substrates in the lower intertidal and shallow subtidal along beaches and tide flats. This species was present at approximately 80% of all sites sampled by DNR's monitoring program. Eelgrass does not grow in the most southern parts of Puget Sound, and is sparse or absent in Dyes Inlet, Liberty Bay, Sinclair Inlet, and the heavily developed shorelines of Commencement Bay and Elliott Bay (Figure 6).

The non-native *Z. japonica* tends to grow in areas that are less exposed to intense wave action, and is commonly found along the shorelines on Central Puget Sound and Hood Canal. This species is rare along the exposed shorelines of the Strait of Juan de Fuca and the San Juan Islands (Figure 6). *Z. japonica* grows higher up in the intertidal (shallower than 0 m, MLLW), and is generally smaller than *Z. marina.* It has been documented at over 25% of all sites sampled in greater Puget Sound. However, his may be an underestimate as this species sometimes grows too shallow for our research vessel.

Surfgrasses (*Phyllospadix sp.*) thrive in exposed environments. They grow attached to rocks and are often exposed at low tide. Surfgrasses have an opposite spatial pattern as compared to *Z. japonica*. They are commonly found along the Strait of Juan de Fuca and the San Juan Islands. They are present in Admiralty Inlet but are not found in Hood Canal, Central Puget Sound, the Saratoga Whidbey Basin or Northern Puget Sound. Surfgrass has been documented at less than 5% of all sites sampled by DNR's monitoring program. As with *Z. japonica*, this is probably an underestimate, as it tends to grows in shallow rocky environments that are not safe to survey using the SVMP methods.

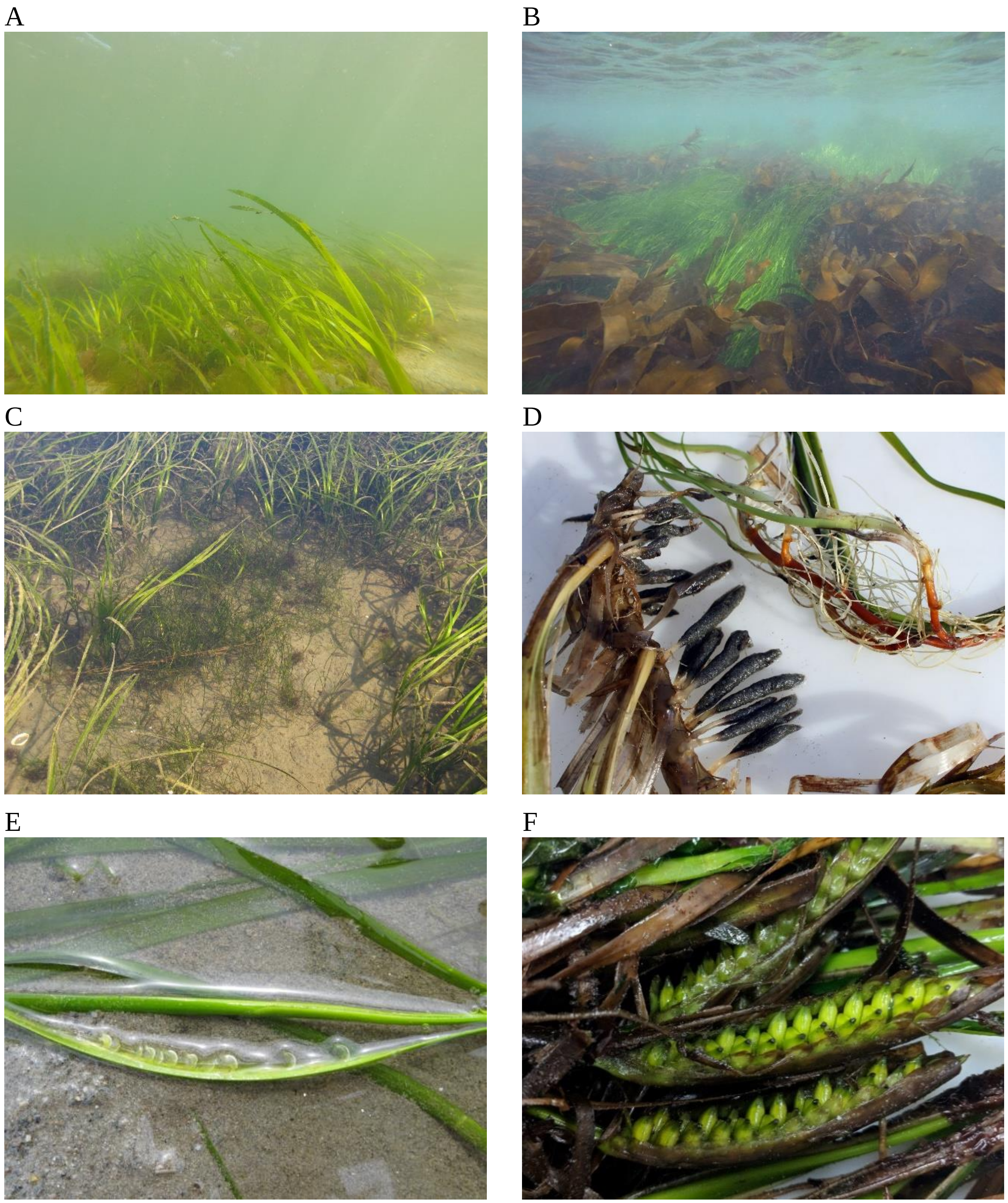

Figure 5: A. Eelgrass (*Zostera marina*) growing on soft sediments near Joemma Beach State Park, WA. B. *Phyllospadix scouleri* growing interspersed with understory kelp on rocky substrate near Tongue Point, WA. C. Intertidal *Zostera japonica* surrounded by larger *Z. marina* plants near Sunset Beach, Hood Canal, WA. D. Roots and rhizome of *Phyllospadix serrulatus* (bottom) and *Zostera marina* (top), picture by Tiffany Stephens. E. Male flowering shoot (spathe) of *Z. marina*. F. Maturing seeds of *Phyllospadix sp*.

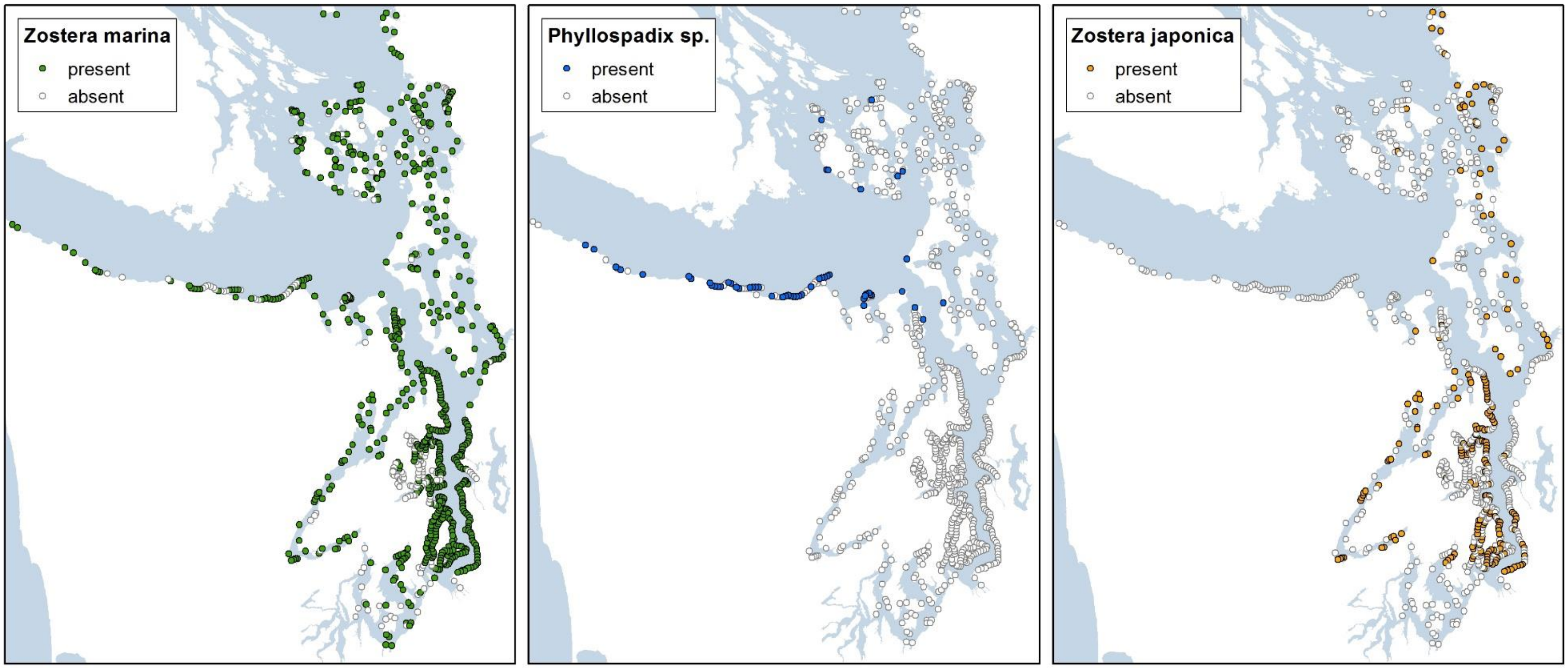

Figure 6: Seagrass species at sites sampled by DNR's Submerged Vegetation Monitoring Program (SVMP). The presence of *Z. marina* is indicated in green (left), *Phyllospadix sp.* in blue (middle) and *Z. japonica* in orange (right). *Z. marina* has been documented at over 80% of sites sampled for the SVMP. It does not grow at the most southern parts of Puget Sound (southwest of Dana and Pickering Passages), and is sparse or absent in Dyes Inlet, Liberty Bay, Sinclair inlet, and the heavily developed shorelines of Commencement Bay and Elliott Bay.

3.1.2 *Spatial distribution of eelgrass*

DNR estimates the areal extent of native seagrass beds (*Z. marina* and *Phyllospadix sp.*) at individual sites, and extrapolates these values to estimate soundwide eelgrass area[7]. Based on the most recent 3-year pooled soundwide eelgrass area estimate, there are approximately 22,100 +/- 1,100 ha (mean +/- se) of eelgrass in greater Puget Sound. Eelgrass area is not uniformly distributed throughout the study area. In Northern Puget Sound and the Saratoga Whidbey Basin (NPS/SWH), the majority of eelgrass grows on large tidal flats. In Central Puget Sound and Hood Canal (CPS/HDC) as well as in the San Juan Islands and the Strait of Juan de Fuca (SJS), the majority of eelgrass grows as a narrow band along the shoreline, which we refer to as fringe habitat (Figure 7).

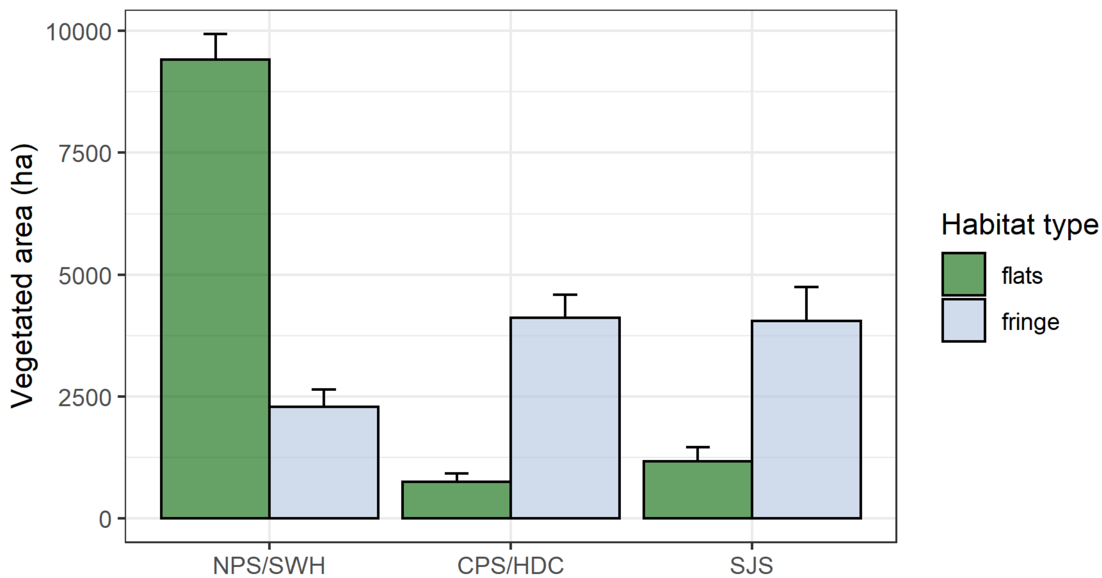

Figure 7: Eelgrass area in each of 3 sub-regions of Puget Sound, split by habitat type (flats or fringe). In Northern Puget Sound & the Saratoga Whidbey Basin (NPS/SWH) the majority of eelgrass is found on tidal flats. In Central Puget Sound and Hood Canal (CPS/HDC), as well as the San Juan Islands and Strait of Juan de Fuca (SJS), the majority of eelgrass is found as a narrow band along the shore (fringe).

Figure 8 shows the size of individual eelgrass beds at sites sampled by DNR. The size of these eelgrass beds is partly determined by the amount of available substrate in the depth band where eelgrass can grow at these locations (which in turn depends on water clarity and tidal range). The largest eelgrass beds are located in Padilla Bay (core001, > 3500 ha), Samish Bay (flats11 and flats12, > 2000 ha combined) and Jamestown (core003, ~ 400 ha). Other sites with substantial eelgrass beds are Salmon Bank, Drayton Harbor, Birch Bay, Lummi Bay, Skagit Bay, Port Susan, the Snohomish delta, Cultus Bay, Quilcene Bay, Dosewallips flats, and Lynch Cove. At the vast majority of sites (over 77% of sites with eelgrass present), individual eelgrass beds are less than 10 ha in size.

[7] Throughout this report we use the term 'eelgrass area' instead of 'native seagrass area', as eelgrass is by far the most abundant species in our region, and the main target for DNR's seagrass monitoring program.

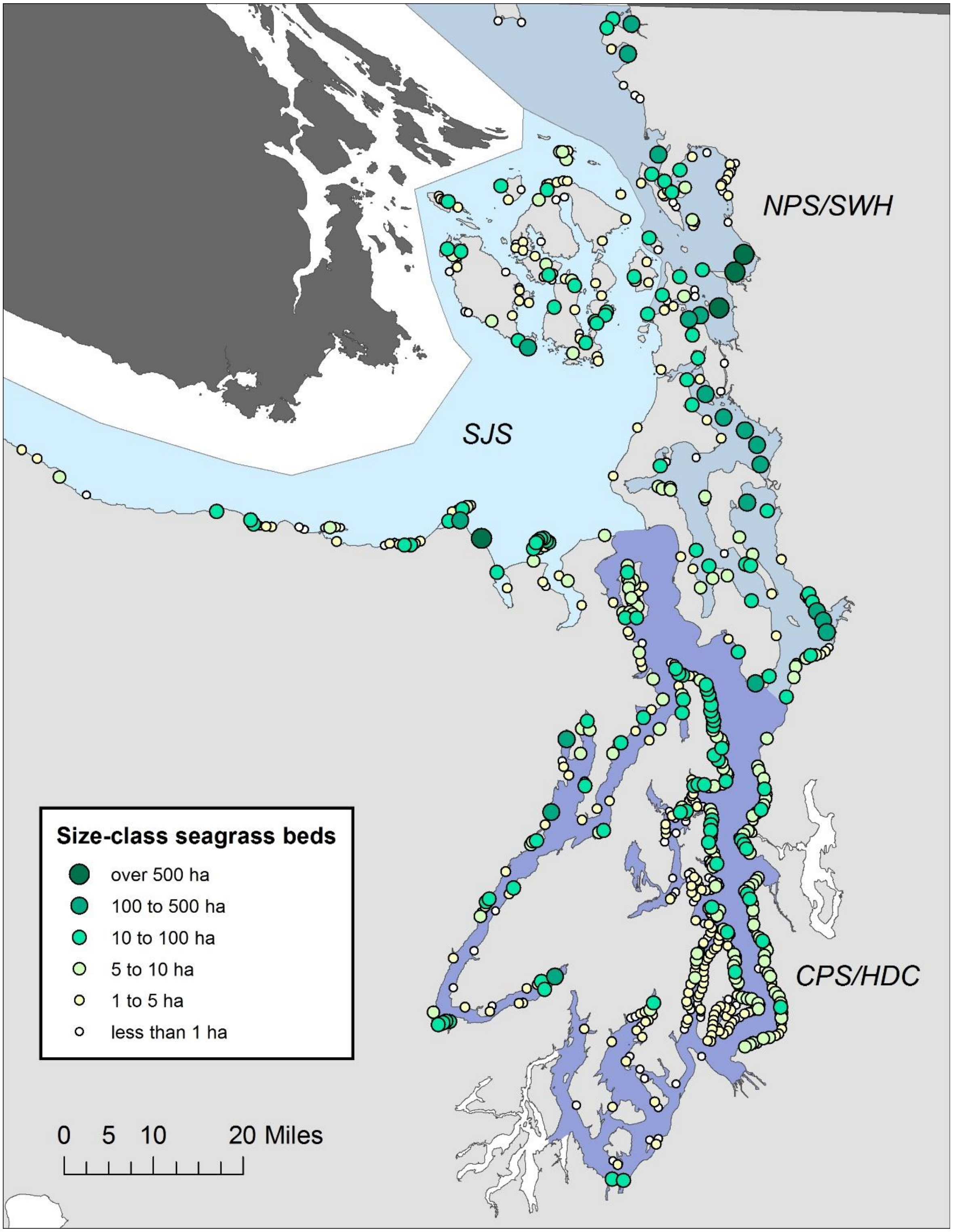

Figure 8: Eelgrass area at individual sites. Larger symbols and darker colors indicate larger eelgrass beds. The largest contiguous eelgrass beds are located in Padilla Bay (core001), Samish Bay (flats11 & flats12), Skagit Bay (flats20, flats21, flats70 & flats71) and Jamestown (core003). The vast majority of sites has less than 10 ha of eelgrass present.

3.1.3 Depth distribution of eelgrass

In greater Puget Sound, eelgrass has been observed as shallow as +1.4 m and as deep as -14.3 m relative to MLLW. The majority of eelgrass occurs between 0 and -4 m relative to MLLW (Figure 9). One exception is the Strait and the San Juan Islands. Here eelgrass tends to grow deeper as compared to the other regions of greater Puget Sound. The average depth of eelgrass beds is similar between flat and fringe sites in Central Puget Sound, Hood Canal, Northern Puget Sound, and the Saratoga Whidbey Basin, but the deepest eelgrass observations occur at greater depths at fringe sites (as indicated by the tail of the distribution in Figure 9). The San Juan Islands and the Strait of Juan de Fuca show a slightly different pattern. Here, eelgrass beds at flats sites show a bi-modal depth distribution. This is partly due to large eelgrass bed at Salmon Bank (flats73). Approximately 55% of all eelgrass in greater Puget Sound grows below the extreme low tide line (-4.5 ft relative to MLLW, -1.4 m relative to MLLW).

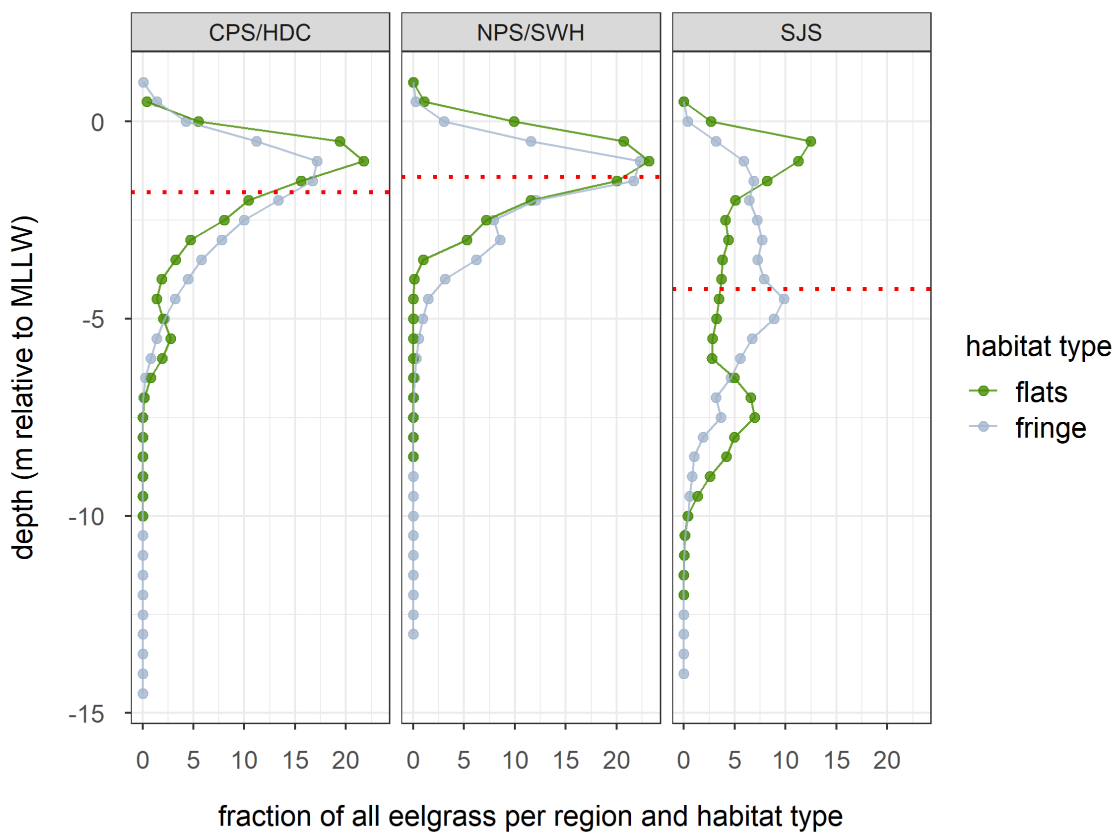

Figure 9: Depth distribution of eelgrass in greater Puget Sound, calculated as the % of total eelgrass observations per 0.5-m depth bins, split per region and per habitat type. Points that are close to zero indicate the presence of small amounts of eelgrass in the corresponding depth bins. The red dotted line indicates the mean depth of eelgrass in each of the 3 regions of greater Puget Sound.

The general shape of the depth distribution in Figure 9 is partly due to geomorphology of greater Puget Sound. Eelgrass beds are abundant on tide flats, and on underwater platforms that were cut by waves in the steep walls of the basins on Puget Sound (Finlayson 2006). These tend to be located between 0 and -4 m relative to MLLW. The deep edge of eelgrass beds (calculated as the 2.5th percentile of all eelgrass observations) often coincides with the edge of the platform, where there is an abrupt change in the slope of the bottom (Figure 10). At most sites, eelgrass beds become more patchy seawards of this break in slope, with individual plants growing down to -14.3 m depending on water clarity and tidal range.

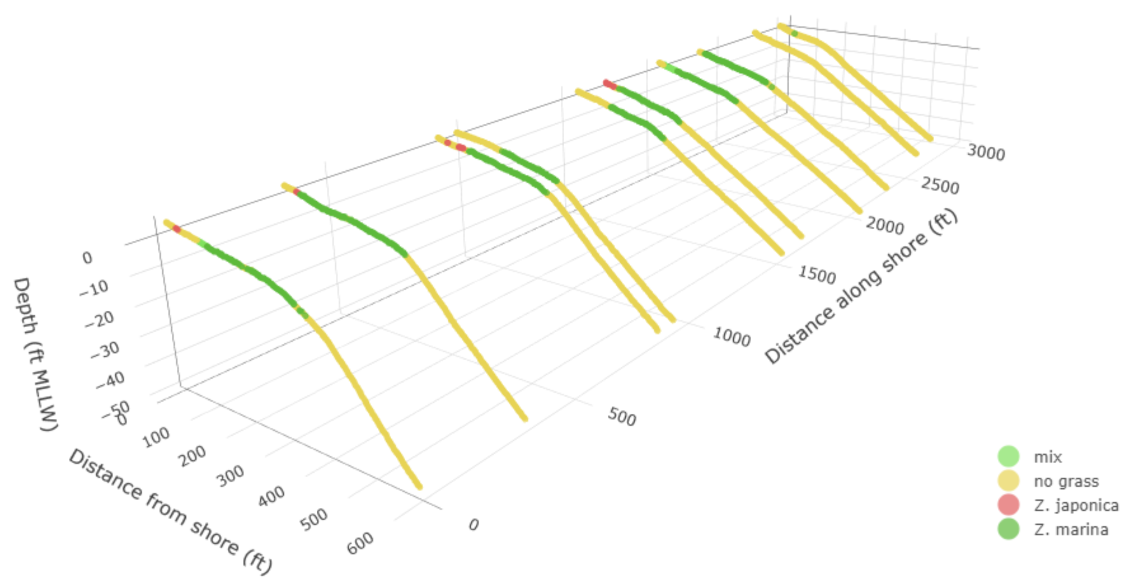

Figure 10: 3D representation of transects sampled at cps1758 in 2018. The dark green color indicates where eelgrass (*Z. marina*) is present along the tracks. Depth relative to MLLW is shown on the vertical axis (not to scale). Note the change in slope, which coincides with the deep edge of the eelgrass bed at this location.

Figure 11 and Figure 12 show the deep and shallow edge of eelgrass beds at all sites sampled between 2004 and 2020. There is a clear spatial pattern in eelgrass depth data throughout greater Puget Sound. Eelgrass grows to deeper extents near the Strait of Juan de Fuca and the San Juan Islands and grows less deep further south in Central Puget Sound and Hood Canal. Eelgrass grows less deep at the end of inlets and in enclosed embayments, such as lower Hood Canal, Case Inlet, Carr Inlet, Quartermaster Harbor, Sinclair Inlet, Port Orchard, and Kilisut Harbor. The deep edge of eelgrass beds is also limited at locations with strong riverine influence, which includes most sites in Northern Puget Sound and the Saratoga Whidbey Basin. Other sites with strong riverine influence are the Skokomish Delta, the Nisqually Delta, and a number of sites North of Federal Way (East Passage) that appear to be impacted by the Puyallup River plume.

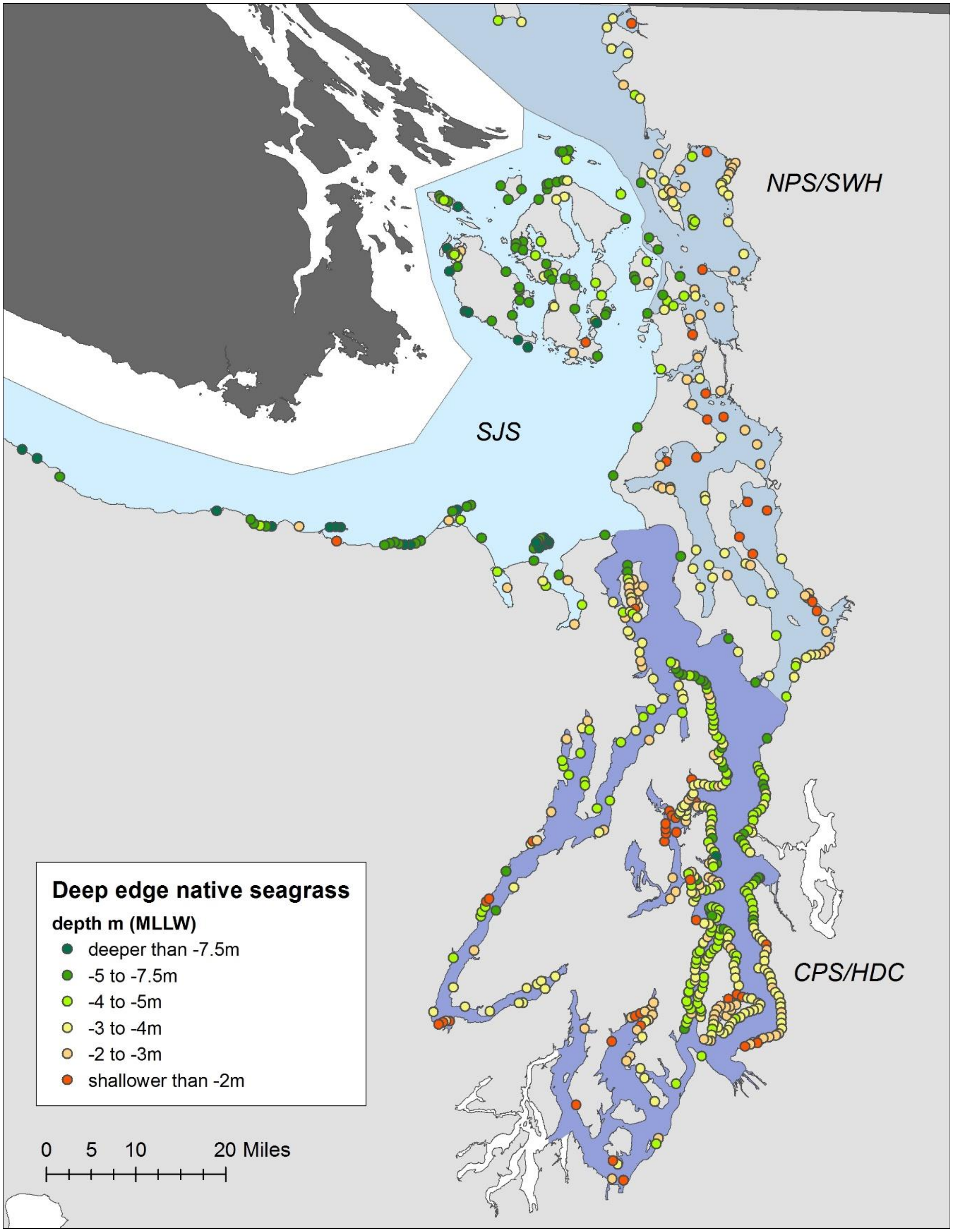

Figure 11: Deep edge of eelgrass beds (calculated as the 2.5th percentile of all depth observations) at all sites sampled between 2004 and 2020. Dark green indicates that eelgrass beds extend to deeper depths, red indicates a shallow deep edge.

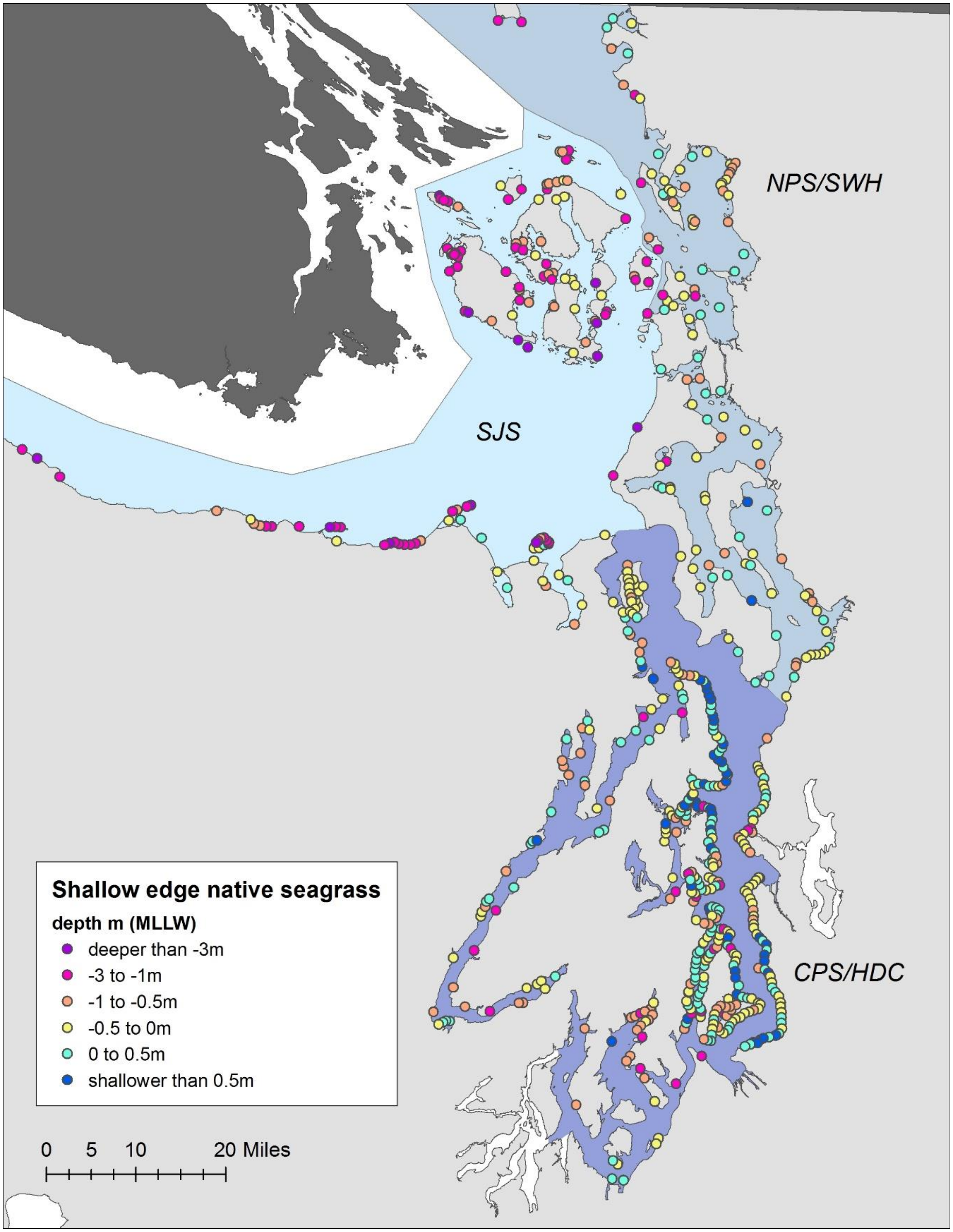

Figure 12: Shallow edge of eelgrass beds (calculated as the 97.5th percentile of all depth observations) at all sites sampled between 2004 and 2020. Dark blue indicates that eelgrass beds extend to further into the intertidal, purple indicates a deep shallow edge.

Eelgrass grows to its shallowest extent in Central Puget Sound, particularly along the upper Kitsap Peninsula, on the northern part of Bainbridge Island, North of Federal Way, West of SeaTac and on the East side of Vashon Island. Results from a previous analysis of sites in the Central Basin of Puget Sound suggests that intertidal eelgrass was more frequently found in drift cells with long fetch, gentle intertidal slope, and lower tidal range (Christiaen et al. 2022, in prep.). One possible explanation is that in Central Puget Sound, the upper edge of eelgrass beds is in part limited by desiccation at low tides. At sites with gentle slopes and more exposure, there can be more pooling of water at low tides, which can protect the plants from desiccation and allow them to grow higher up the intertidal. On a soundwide scale, the shallow edge of eelgrass beds tends to be deeper at sites with a high fetch or with rocky shorelines, such as near the Strait of Juan de Fuca and the San Juan Islands. Note that some species of surfgrass are found at relatively shallow depths in this region.

3.2 How seagrass is changing

3.2.1 Soundwide eelgrass area

Figure 13 shows the annual estimates of soundwide eelgrass area between 2009 and 2020, relative to a baseline (2000-2008) and the 2020 target for the soundwide indicator of the PSP eelgrass Vital Sign (https://vitalsigns.pugetsoundinfo.wa.gov/VitalSignIndicator/Detail/10). The annual estimates show some variability, but do not exhibit a clear trend between 2009 and 2020. This suggests that on a soundwide basis, eelgrass area remained relatively stable over this period of time. The 2020 target of a 20% increase in soundwide eelgrass area relative to the 2000-2008 baseline has not been met.

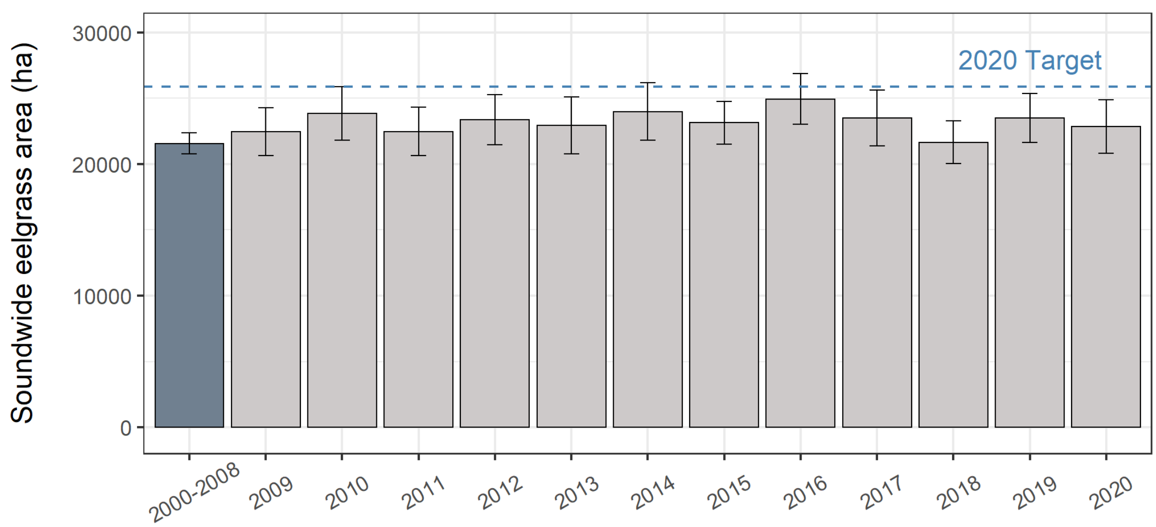

Figure 13: Annual estimates of soundwide eelgrass area (ha) in greater Puget Sound. The dark grey bar represents the 2000-2008 baseline, light grey bars are annual estimates of soundwide eelgrass area, and the dashed line shows the recovery target of a 20% increase in eelgrass area relative to the 2000-2008 baseline by 2020. Error bars are standard error.

Note that there is some uncertainty around the annual estimates, as they are based on a relatively small sample of sites in greater Puget Sound (78-80 sites). There is also interannual variability in the soundwide estimates. This variability is partially caused by environmental drivers, but also reflects random variation associated with sampling.

As referenced in the methods, the design of the soundwide monitoring program has evolved. Up to 2014, we sampled using 20% rotation, which means that the annual estimates from 2000-2014 are partially based on the same sample of sites across adjacent years. During this period of time, estimates that are one year apart have 80% overlap in sample sites, while estimates that are 2 years apart have 60% overlap in sample sites. This makes it challenging to assess potential trends in soundwide eelgrass area during this period of time. Since 2015, we are sampling with a 3-year rotating panel design: we sample 3 random sets of sites (originally sampled in 2004, 2009, and 2014) on a 3-year basis[8]. This induces a periodicity in the annual estimates, which is visible on Figure 13, but allows us to calculate soundwide area based on pooled 3-year samples of 214 randomly selected sites that give better estimates of the current status of eelgrass, as well as more accurate trends in soundwide eelgrass area over time.

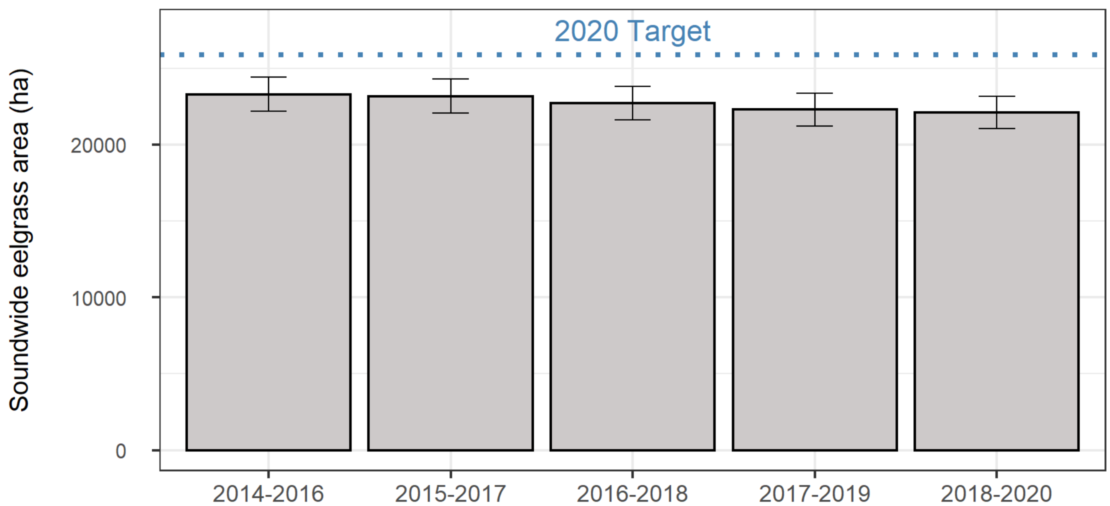

Figure 14: Estimates of soundwide eelgrass area (ha) based on pooled 3-year samples in greater Puget Sound. The dotted blue shows the recovery target of a 20% increase in eelgrass area relative to the 2000-2008 baseline by 2020. Error bars are standard error.

Figure 14 shows soundwide estimates based on pooled 3-year samples between 2014 and 2020. Note that the error-bars are smaller, as these estimates are based on a larger sample size (214 sites for the 3 panels combined). According to the most recent estimate (2018-2020), there is approximately 22,100 +/- 1,100 ha (mean +/- se) of eelgrass in greater Puget Sound. Since 2014, the 3-year average has declined by approximately 1,200 ha (which is in the same order of magnitude as the standard error around these estimates).

[8] Sample years 2004, 2009, and 2014 represent 3 new-draw random samples of sites from greater Puget Sound. The sites from 2004 were resampled in 2015 and 2018, the sites from 2009 were resampled in 2016 and 2019, and the sites from 2014 were resampled in 2017 and 2020. See methods section for more information.

Resampling the same panels of sites allows us to compare soundwide eelgrass area between particular years in the dataset. By extrapolating the differences in area at individual sites between two years, we are able to assess if soundwide eelgrass area changed between years sampled with the same set of sites. Figure 15 shows the annual estimates of soundwide eelgrass area for individual panel-years. Estimates that have the same color are based on the same set of 79-80 sites (depending on the panel). In panels a (red) and b (green) soundwide eelgrass area initially increased over time, but subsequently declined. In panel c (blue), the soundwide estimates gradually declined between 2014 and 2020. This would suggest that the period between 2004 and 2016 was generally conducive for eelgrass in greater Puget Sound. After 2016 overall eelgrass area declined. However, this decline is within the same order of magnitude as the earlier increases noted in the time series. Overall, eelgrass area did not appear to change between 2004 and 2020.

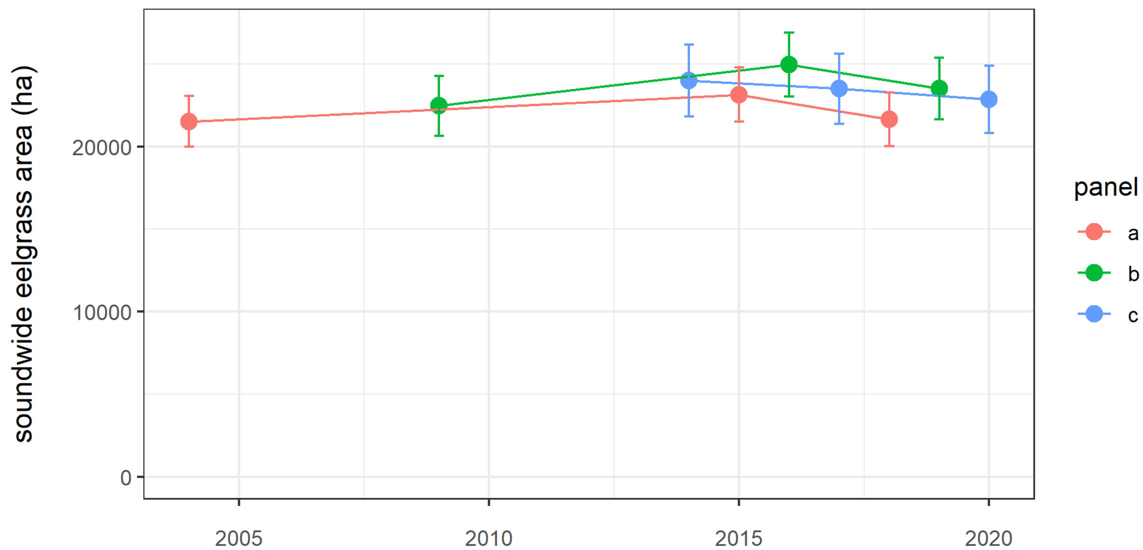

Figure 15: Change in soundwide eelgrass area estimates for the 3-year rotating panel design (panel a: 2004-2015-2018, panel b: 2009-2016-2019, panel c: 2014-2017-2020).

The visual assessment of Figure 15 is backed by the statistical assessment of changes in eelgrass area between pairs of years (Table 2). When extrapolating the differences in eelgrass area between years to the soundwide level, there was a significant increase over time from 2004 to 2015 and from 2009 to 2016. There were significant declines in eelgrass area from 2014 to 2020, 2015 to 2018, and 2016 to 2019. There was no significant difference in eelgrass area from 2004 to 2018 and from 2009 to 2019, which suggests that there was no apparent long-term trend in soundwide eelgrass area over the last two decades.

Table 2: Pairwise comparisons in soundwide eelgrass area between years that were sampled with the same panel of sites in greater Puget Sound. The mean difference and standard error were calculated by extrapolating site-level differences in eelgrass area similar to the soundwide calculation. For most years we used the SRS estimates as input for the calculation. In 2019 and 2020 we used the STR estimates (as indicated by *).

Panel	Comparison	Mean difference +/- 95% CI	Interpretation
a	2015 - 2004	1,621 +/- 995 ha	2015 > 2004
a	2018 - 2015	-1,481 +/- 940 ha	2018 < 2015
a	2018 - 2004	140 +/- 914 ha	No difference
b	2016 - 2009	2,494 +/- 1,025 ha	2016 > 2009
b	2019 – 2016*	-1,437 +/- 764 ha	2019 < 2016
b	2019 – 2009*	1,057 +/- 1,143 ha	No difference
c	2017 - 2014	-496 +/- 799 ha	No difference
c	2020 – 2017*	-649 +/- 819 ha	No difference
c	2020 – 2014*	-1,145 +/- 1,124 ha	2020 < 2014

3.2.2 Local change at panel sites

While eelgrass is relatively stable soundwide, there is significant variability at the site level. We analyzed the 214 sites sampled as part of the 3-year rotating panel design (Figure 1) to assess regional patterns of change at smaller spatial scales. These sites represent a random sample of 2,467 potential sample sites in greater Puget Sound. We looked at both long-term trends (based on all data) and recent trends (based on data from 2015 to 2020). Data were summarized over 3 regions: the San Juan Islands and the Strait of Juan de Fuca (SJS), Northern Puget Sound and the Saratoga Whidbey Basin (NPS/SWH), and Central Puget Sound and Hood Canal (CPS/HDC). The regions are mapped in Figure 8, Figure 11, and Figure 12.

Sites were classified as insufficient data (not enough data to assess a potential trend during the period of interest), no grass (no native seagrass present), trace (very small amount of native seagrass at the site, not enough to get an accurate area estimate), no trend, increase or decline. These assessments were based on two methods:

- inverse variance weighted regressions of site eelgrass area estimates over time (which includes all samples taken at a site);
- pairwise *t*-tests of the vegetated fraction of transects for sites that were sampled using the same set of transects over multiple years (the exact years of the repeat samples varies depending on the site[9]).

All potential trends were confirmed based on visual inspection of point features in ArcGIS. Trends were excluded if there was potential confusion in species identification between years (*Z. marina* vs *Z. japonica*), or if there was an issue with sampling in a particular year. The trend assessments for all panel sites are included in Appendix 1. Plots of eelgrass area at individual panel sites are included in Appendix 2. Summary plots for CPS/HDC, NPS/SWH, and SJS are included in Appendix 3.

[9] All panel sites have been sampled at least once using repeat transects

Both on the long-term and in recent years, the majority of panel sites had substantial amounts of eelgrass present (n = 178 over the entire time series, n = 175 for the most recent 6 years of data). When considering the entire monitoring period, 40% of panel sites with eelgrass present experienced significant changes in eelgrass area (Figure 16). The number of sites with increase (n = 33) was similar to the number of sites with declines (n = 38). When looking at data collected between 2015 and 2020, eelgrass area changed significantly over time at approximately 23% of sites with eelgrass present. However, sites with declines (n = 32) outnumber sites with increases (n = 9). This result is in line with the outcome of the pairwise comparisons in soundwide eelgrass area, that suggest that that eelgrass overall declined between 2016 and 2020.

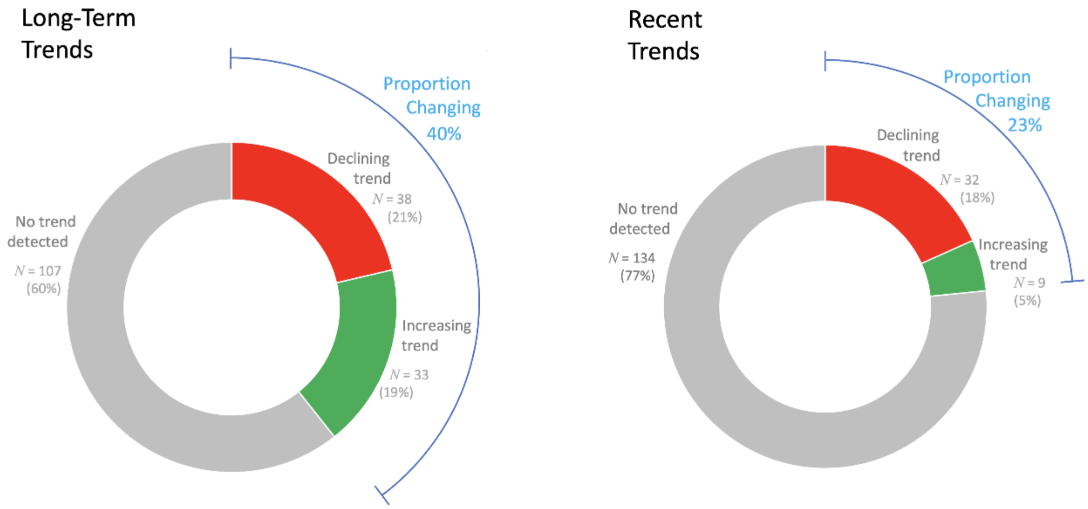

Figure 16: Trend assessment at vegetated panel sites in greater Puget Sound. Long-term trends are based on data collected between 2000 and 2020, while recent trends are based on data collected between 2015 and 2020.

Figure 17 and Figure 18 show long-term trends and recent trends for the 3-year panel sites in greater Puget Sound. Sites with declines are indicated in different shades of red, sites with increases are shown in different shades of green, sites without trends are shown in blue, and sites with no grass or trace are indicated in brown and yellow. These maps suggest that there is a spatial pattern in the overall number of sites with increases and declines in greater Puget Sound. On longer time scales, sites with increases are evenly spread over Puget Sound, but in recent years they are mostly limited to Hood Canal, southern Puget Sound, and the Saratoga Whidbey Basin. On longer time scales, there are clusters of declines in the Strait of Juan de Fuca, near the San Juan Islands and Cypress Island, and in South-Central Puget Sound. In recent years, sites with declines are most abundant in the San Juan Islands and Cypress Island. Here, the ratio of increasing vs. declining sites is lower than would be expected by chance, both for long-term trends and in recent years. On the long-term, sites with declines outnumber sites with increases 4:1. In recent years, almost 40% of sites with eelgrass showed declines. We did not document any recent increases (Figure 19).

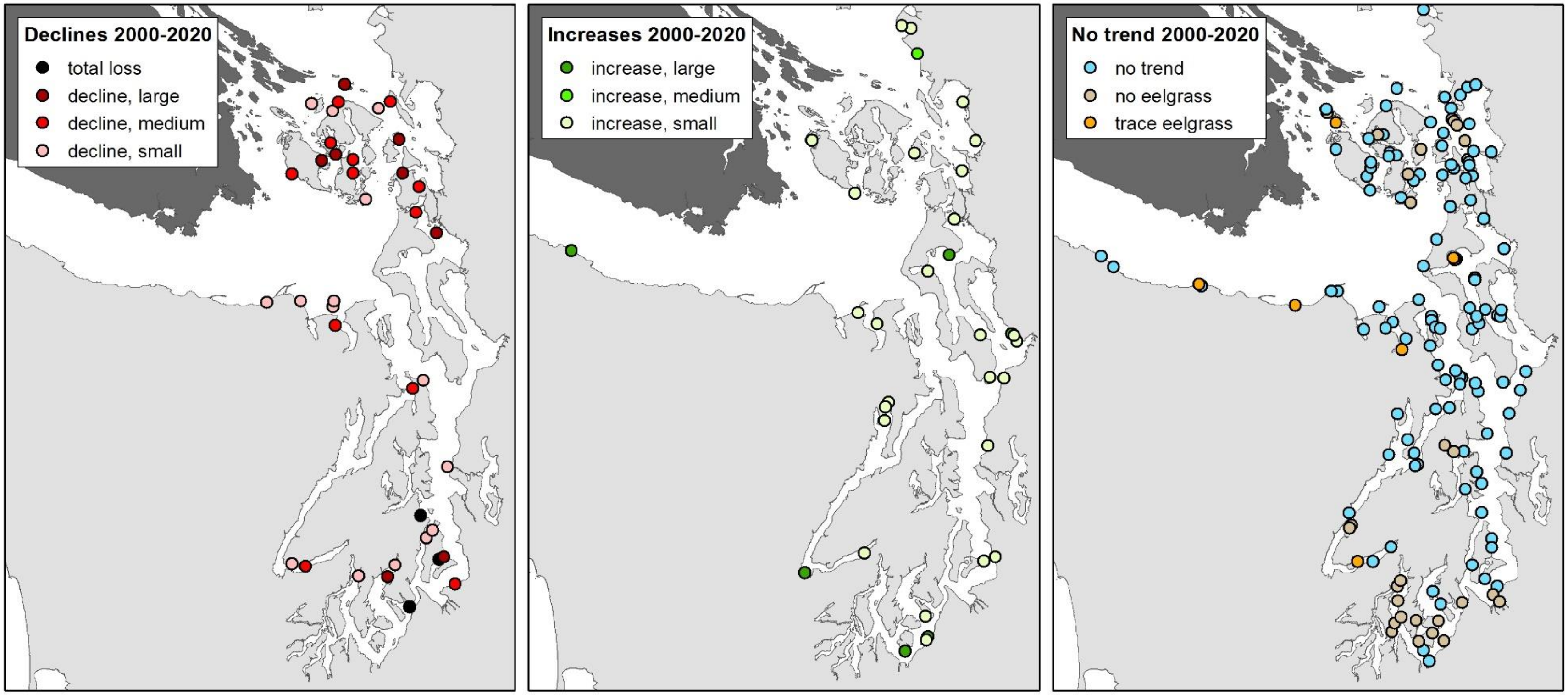

Figure 17: Long-term trends at 214 panel sites based on data collected between 2000 and 2020. These sites represent a stratified random sample of greater Puget Sound. Declines are indicated in different shades of red (left), increases in different shades of green (center) and sites with no trend are indicated in blue (right). The map on the right also shows sites where eelgrass was present in trace amounts or absent.

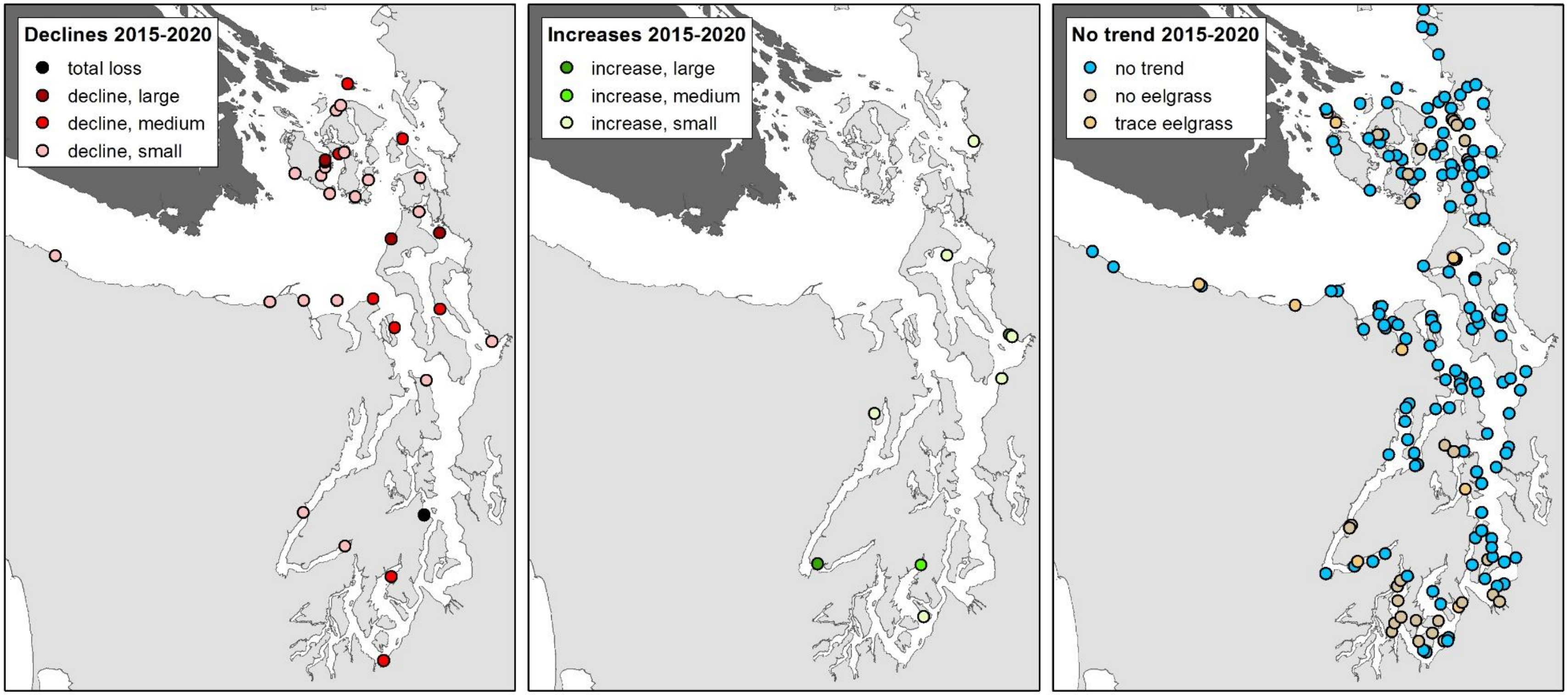

Figure 18: Recent trends at 214 panel sites based on data collected between 2015 and 2020. These sites represent a stratified random sample of greater Puget Sound. Declines are indicated in different shades of red (left), increases in different shades of green (center) and sites with no trend are indicated in blue (right). The map on the right also shows sites where eelgrass was present in trace amounts or absent.

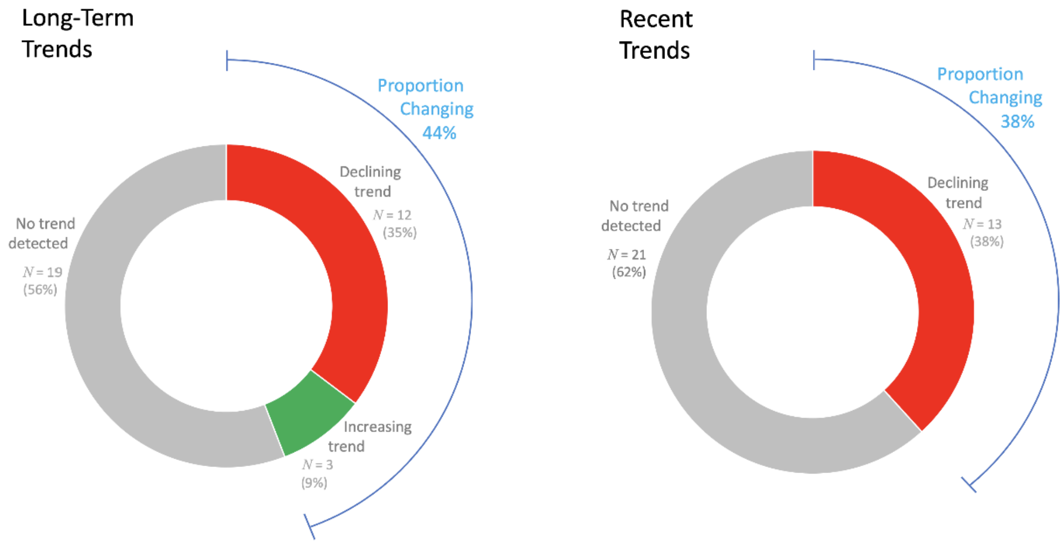

Figure 19: Trend assessment at vegetated panel sites in the San Juan Islands and Cypress Island. Long-term trends are based on all data collected between 2000 and 2020, while recent trends are based on all data collected between 2015 and 2020.

We apply two approaches for visualizing the size of these site level increases and declines. Figure 20 shows the number of panel sites with increases and declines (expressed as % of all panel sites) classified by the magnitude of the relative change at these sites. Trends were considered large if the final area estimate was more than 2 times larger or smaller than the mean, medium if the final estimate was between 1.5 and 2 times larger or smaller than mean, and small if the final estimate was less than less than 1.5 times larger or smaller than the mean. Both long-term trends and recent trends show a similar pattern. There is a higher percentage of sites where the relative magnitude of the declines is small or medium as compared to large or total loss. On the long-term, sites with relatively small increases outnumber sites with large relative increases. This pattern is less pronounced for recent trends.

Figure 21 shows the number of 3-year panel sites with increases and declines, classified by the maximum size of the eelgrass beds at these locations. In both the long-term and recent timeframes, there were more sites with declines than increases in the smallest size class (less than 10 ha of eelgrass present). In the long term, there were more declines than increases at small sites, but more increases than declines at sites with larger seagrass beds. In recent years this changed, and there were more declines than increases in each size category. Both ways to visualize the data support the results from the soundwide eelgrass area estimations, which suggest that eelgrass increased between 2004 and 2016, but declined soundwide between 2016 and 2020.

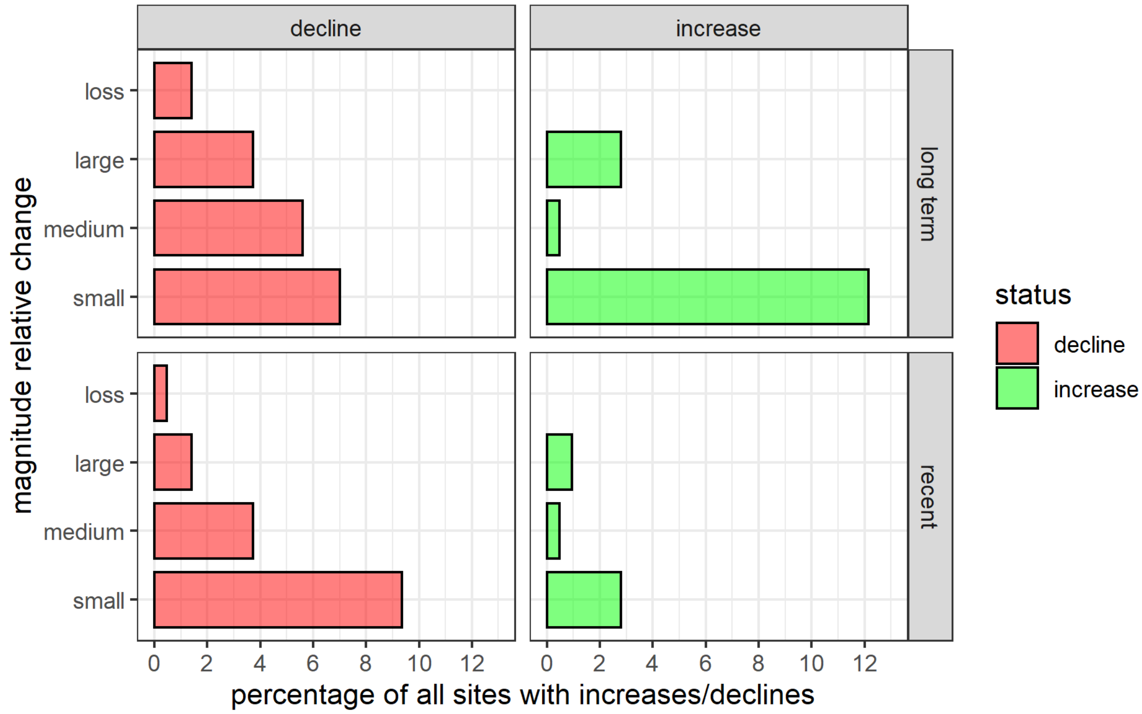

Figure 20: The percentage of all panel sites with increases and declines, classified by the relative magnitude of change at the site, for both long-term and recent trends.

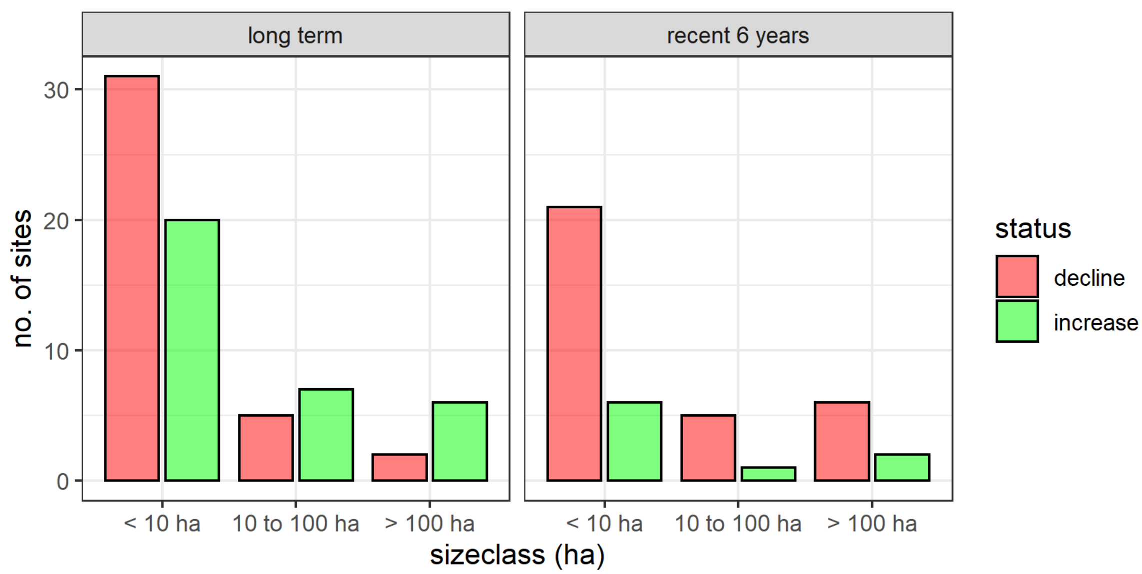

Figure 21: The number of increasing and declining panel sites, classified by the maximum size of eelgrass beds at these locations.

3.3 Region of concern: the San Juan Islands

Based on the analysis of 214 random panel sites, the San Juan Islands has been identified as a region of concern, where sites with declines significantly outnumber sites with increases. Additional data collected between 2000 and 2020 confirms this pattern. Throughout the course of the SVMP, DNR has visited 89 sites near the San Juan Islands and Cypress Island. Out of these 89 sites, there were 4 sites with long-term increases, 16 sites with long-term declines and 45 sites with no long-term trend detected. In recent years this pattern has become even more pronounced. Between 2015 and 2020 there were no sites with increases, 15 sites with declines, 22 sites with no trend detected. At the other sites there was either no grass or trace eelgrass present, or there was insufficient data to determine trends (Figure 22).

Table 3 and Table 4 show details for all sites with increases and declines near the San Juan Islands, including an estimate of the relative size of the increases/declines (calculated as the final area estimate over the mean), and some observations of where the increases/declines occurred within each site.

Out of the 4 sites with long-term increases, there was only one site with a substantial increase over time: East of White Point on San Juan Island (sjs2853). Here, eelgrass area increased from approximately 1 ha in 2008 to almost 3 ha in 2019. At the other 3 locations, the increase in eelgrass area was relatively small as compared to the mean size of the eelgrass bed at these locations. However, the 3 sites with long-term increases had large eelgrass beds present (Table 3), so even if the increases were small, they represent a significant amount of eelgrass gained at these locations.

Out of the 16 sites with long-term declines, there were 12 sites with substantial eelgrass declines (relative change classified as medium or large). There is no consistent pattern on where losses occurred at each site. Eelgrass was lost either at the shallow edge, at the deep edge or throughout the entire eelgrass bed at these locations. There is also no clear pattern in when these declines occurred. At some sites, eelgrass was lost relatively early in the monitoring period (for example flats53 & sjs0635), but at other sites declines occurred later on (flats64, flats66, sjs0191, sjs0682, sjs0683). At some locations, such as core002, eelgrass area fluctuated over time.

Some of the largest eelgrass losses (both in terms of the relative and the absolute amount of eelgrass lost) have occurred in enclosed embayments (Figure 23). The most notable examples are Westcott Bay on San Juan Island (flats53), Reef Net Bay on Shaw Island (flats64), Shallow Bay on Sucia Island (flats66), and Swifts Bay on Lopez Island (flats62):

- Westcott Bay (flats53) has been sampled repeatedly since 2000. In the first two years of monitoring, eelgrass was present throughout the embayment between approximately -0.5 and -2 m relative to MLLW. Between 2002 and 2003, most eelgrass was lost at this location (over 15 ha). Since then there has only been a trace amount of eelgrass at the site, in the form of a small patch near Bell Point. Despite multiple restoration efforts, eelgrass has not recolonized the perimeter of the Bay.
- Reef Net Bay (flats64 – formerly named Squaw Bay) has been sampled multiple times since 2006. At this location, over half of the eelgrass bed disappeared between 2016 and 2019. The declines were most pronounced at the shallow edge of the embayment.

Note that nearby Picnic Cove (core002) also experienced a large decline in eelgrass over the same period of time.

- In Shallow Bay (flats66), eelgrass area has steadily declined over time from approximately 5 ha in 2003 to slightly over 1 ha in 2019. Paired transect comparisons show clear loss in eelgrass cover between 2009 and 2016, and 2016 and 2019. Declines were throughout the entire site.
- In Swifts Bay (flats62), eelgrass in the inner portion of the bay has declined steadily since 2001. At this point only the deep bed at the mouth of the bay remains. Note that the area estimate from 2000 is an underestimate, as we did not sample the entire deep edge of the bed in that year.

There have also been substantial declines in eelgrass at fringe sites (Figure 24). While individual eelgrass beds at fringe sites are typically small, they comprise the majority of eelgrass habitat in the San Juan Islands and the Strait of Juan de Fuca. The largest relative declines in eelgrass area occurred near Brown Island (sjs0683) and on the northern side of Orcas Island (sjs0454, outf456 & outf457):

- At sjs0683 (NE Brown Island), there was a large decline in eelgrass cover between 2004 and 2018. Most of the eelgrass bed disappeared between 2015 and 2018. There are a few patches remaining near the edges of the site. A more recent survey has shown a decline in eelgrass in the adjacent site (sjs0682) between 2015 and 2020.
- At outf456 (northern side of Orcas Island, West of airport) and outf457 (northern side of Orcas Island, near outfall) eelgrass area has steadily declined since 2013. At both sites, the losses are most pronounced near the shallow and the deep edge of the bed. The center of these beds has remained relatively stable over time.
- At sjs0454 (northern side of Orcas Island) there was a substantial decline in eelgrass between 2014 and 2017. Eelgrass area remained low in 2020. Here the declines were mainly located near the deep edge of the site.

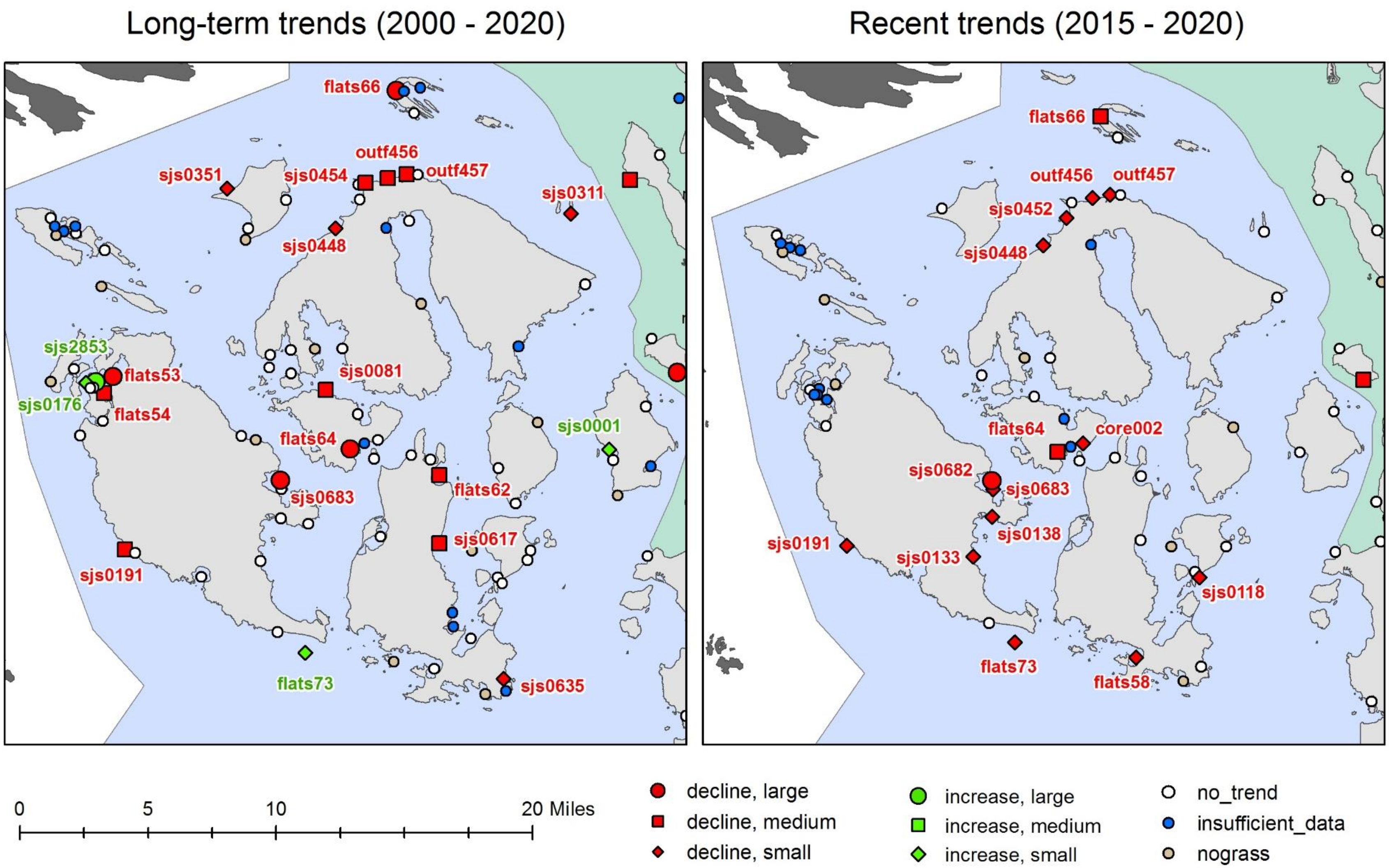

Figure 22: The location of sites with long-term and recent increases (green) and declines (red) near the San Juan Islands and Cypress Island. Declines outnumber increases on both long-term and recent timescales.

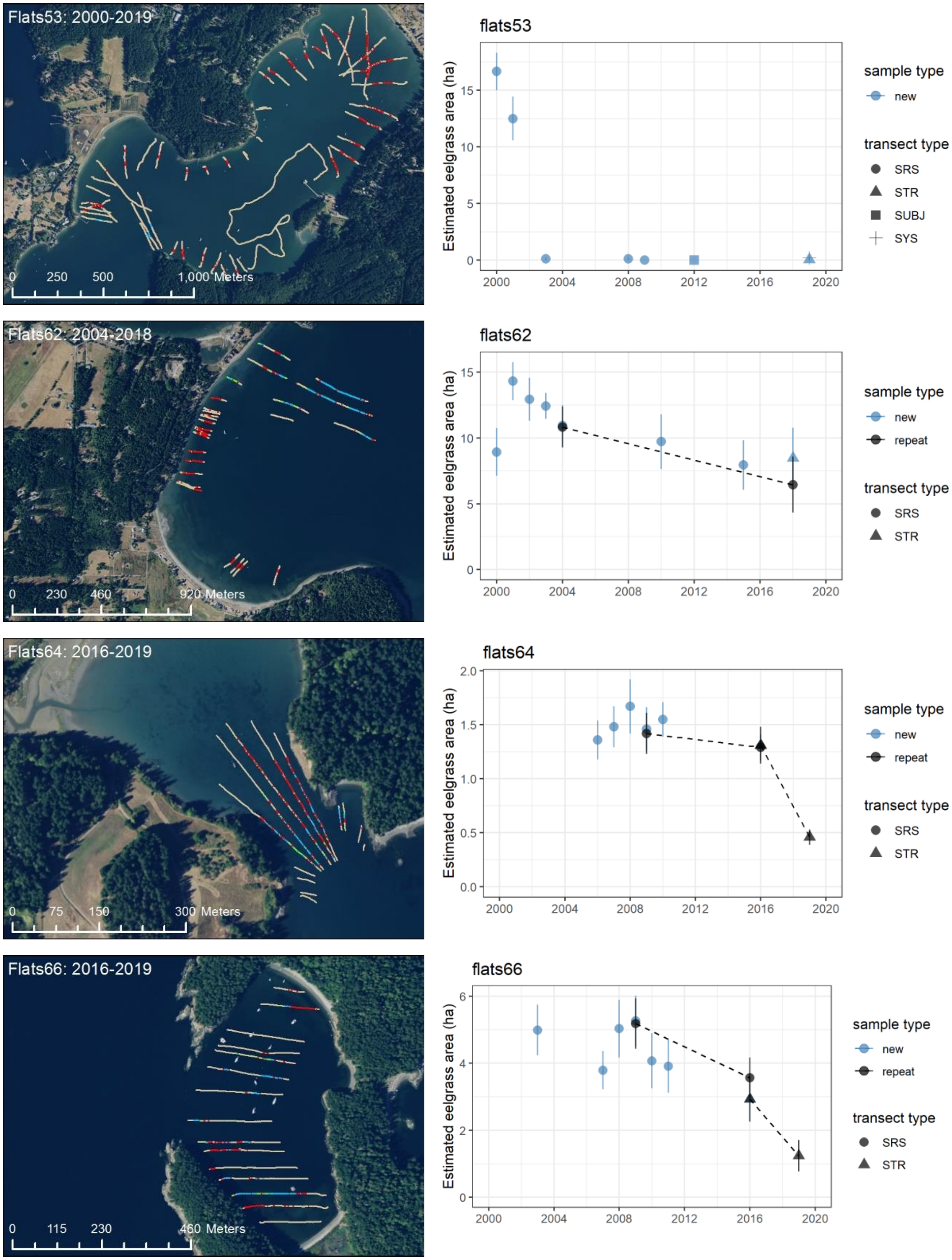

Figure 23: Examples of declines in eelgrass area in embayments on the San Juan Islands: flats53 (Westcott Bay), flats62 (Swifts Bay), flats64 (Reef Net Bay) and flats66 (Shallow Bay). The maps on the left show where eelgrass persisted (blue), where it became newly established (green), and where it has been lost along repeat sampled tracks (red). The plots on the right shows loss of eelgrass area over time at each of the sites.

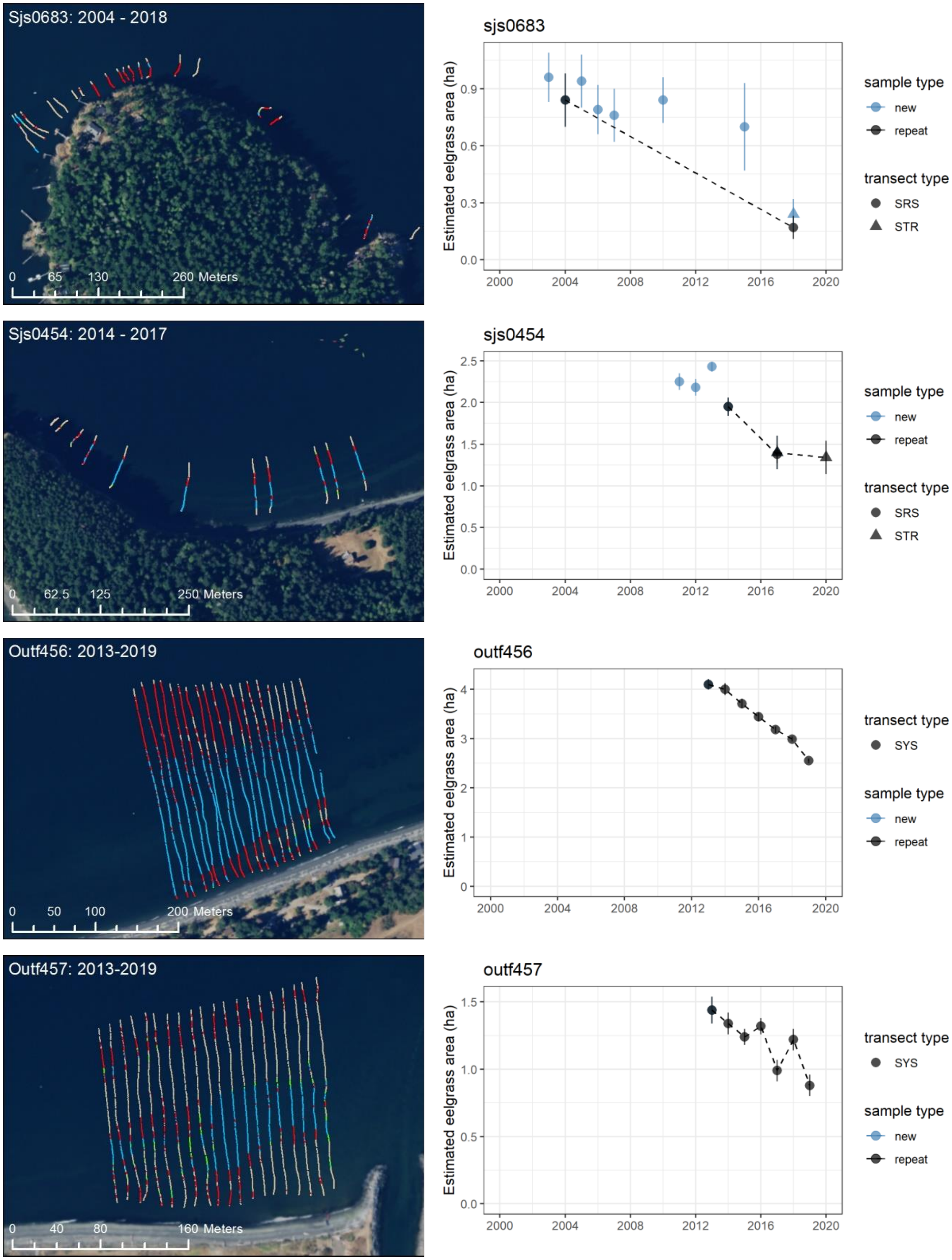

Figure 24: Examples of declines in eelgrass area at fringe sites on the San Juan Islands: sjs0683, sjs0454, outf456 and outf457. The maps on the left show where eelgrass persisted (blue), where it became newly established (green), and where it has been lost along repeat sampled tracks (red). The plots on the right shows loss of eelgrass area over time at each of the sites.

Table 3: Sites with long-term trends (2000-2020) near the San Juan Islands and Cypress Island. Out of 89 sites sampled, there 4 sites with long-term increases and 16 sites with long-term declines in eelgrass area. Start, end, and # years indicate the first year, last year, how often the site was surveyed within this period of time. Location indicates where along repeat sample tracks a change in eelgrass was observed.

Site code	start	end	# years	size class	long-term trend	relative change	location
flats73	*2003*	*2019*	*8*	*100 to 500 ha*	*increase*	*small*	*throughout*
sjs0001	*2007*	*2019*	*7*	*10 to 100 ha*	*increase*	*small*	*throughout*
sjs0176	*2008*	*2019*	*8*	*5 to 10 ha*	*increase*	*small*	*throughout*
sjs2853	*2008*	*2019*	*4*	*1 to 5 ha*	*increase*	*large*	*throughout*
flats53	*2000*	*2019*	*7*	*10 to 100 ha*	*decline*	*large*	*throughout*
flats54	*2003*	*2019*	*5*	*less than 1 ha*	*decline*	*medium*	*inner bay (east)*
flats62	*2000*	*2018*	*8*	*10 to 100 ha*	*decline*	*medium*	*inner bay*
flats64	*2006*	*2019*	*7*	*1 to 5 ha*	*decline*	*large*	*throughout*
flats66	*2003*	*2019*	*8*	*5 to 10 ha*	*decline*	*large*	*throughout*
sjs0081	*2000*	*2018*	*11*	*1 to 5 ha*	*decline*	*medium*	*deep edge (east)*
sjs0191	*2009*	*2019*	*7*	*less than 1 ha*	*decline*	*medium*	*deep edge*
sjs0311	*2000*	*2018*	*8*	*1 to 5 ha*	*decline*	*small*	*throughout*
sjs0351	*2001*	*2018*	*8*	*10 to 100 ha*	*decline*	*small*	*shallow & deep edge*
sjs0448	*2006*	*2019*	*7*	*5 to 10 ha*	*decline*	*small*	*deep edge*
sjs0454	*2011*	*2020*	*6*	*1 to 5 ha*	*decline*	*medium*	*deep edge*
outf456	*2013*	*2019*	*7*	*1 to 5 ha*	*decline*	*medium*	*shallow & deep edge*
outf457	*2013*	*2019*	*7*	*1 to 5 ha*	*decline*	*medium*	*shallow & deep edge*
sjs0617	*2002*	*2018*	*7*	*1 to 5 ha*	*decline*	*medium*	*throughout*
sjs0635	*2003*	*2018*	*11*	*1 to 5 ha*	*decline*	*small*	*shallow edge*
sjs0683	*2003*	*2018*	*8*	*less than 1 ha*	*decline*	*large*	*throughout*

Table 4: Sites with short-term trends (2015-2020) near the San Juan Islands and Cypress Island. Out of 89 sites sampled, there no sites with short-term increases and 15 sites with short-term declines in eelgrass area. Start, end, and # years indicate the first year, last year, how often the site was surveyed within this period of time. Location indicates where along repeat sample tracks a change in eelgrass was observed.

Site code	start	end	# years	size class	long-term trend	relative change	location
core002	*2015*	*2020*	*6*	*1 to 5 ha*	*decline*	*small*	*throughout*
flats58	*2003*	*2020*	*2*	*5 to 10 ha*	*decline*	*small*	*shallow edge*
flats64	*2006*	*2019*	*2*	*1 to 5 ha*	*decline*	*medium*	*throughout*
flats66	*2003*	*2019*	*2*	*5 to 10 ha*	*decline*	*medium*	*throughout*
flats73	*2003*	*2019*	*2*	*100 to 500 ha*	*decline*	*small*	*throughout*
sjs0118	*2006*	*2019*	*3*	*10 to 100 ha*	*decline*	*small*	*deep edge*
sjs0133	*2009*	*2019*	*2*	*1 to 5 ha*	*decline*	*small*	*throughout*
sjs0138	*2004*	*2020*	*2*	*1 to 5 ha*	*decline*	*small*	*throughout*
sjs0191	*2009*	*2019*	*2*	*less than 1 ha*	*decline*	*small*	*deep edge*
sjs0448	*2006*	*2019*	*2*	*5 to 10 ha*	*decline*	*small*	*deep edge*
sjs0452	*2006*	*2019*	*2*	*10 to 100 ha*	*decline*	*small*	*throughout*
outf456	*2013*	*2019*	*5*	*1 to 5 ha*	*decline*	*small*	*shallow & deep edge*
outf457	*2013*	*2019*	*5*	*1 to 5 ha*	*decline*	*small*	*shallow & deep edge*
sjs0682	*2013*	*2020*	*2*	*1 to 5 ha*	*decline*	*small*	*throughout*
sjs0683	*2003*	*2018*	*2*	*less than 1 ha*	*decline*	*large*	*throughout*

3.4 Change in eelgrass beds near river deltas

Over the course of our monitoring program, we have documented high variability in eelgrass area near the mouth of river deltas. These areas offer expansive eelgrass habitat in the form of large sloping sand flats, but are also subject to environmental stressors such as variable salinity, desiccation at low tide, burial and erosion due to wave action and river flow.

At the Nisqually delta (flats34 & flats35), eelgrass area has fluctuated significantly over time. Eelgrass estimates were highest near 2003/2004 and 2012. Values were lowest near 2007/2008 and after 2017. Between 2015 and 2017 there was a large loss of eelgrass near the delta front (~ 50 ha lost), particularly on the western side of the delta (Figure 25). These losses were confirmed by a series of acoustic surveys by USGS in 2012, 2014, and 2017 (Stevens et al. 2020). Note that at the Nisqually delta, a large dike removal project restored tidal processes to over 300 ha of former freshwater wetlands in 2009 (http://nisquallydeltarestoration.org). It is unclear if this project has influenced eelgrass habitat. The large declines between 2015 and 2017 were likely caused by erosion or sediment deposition from winter storms.

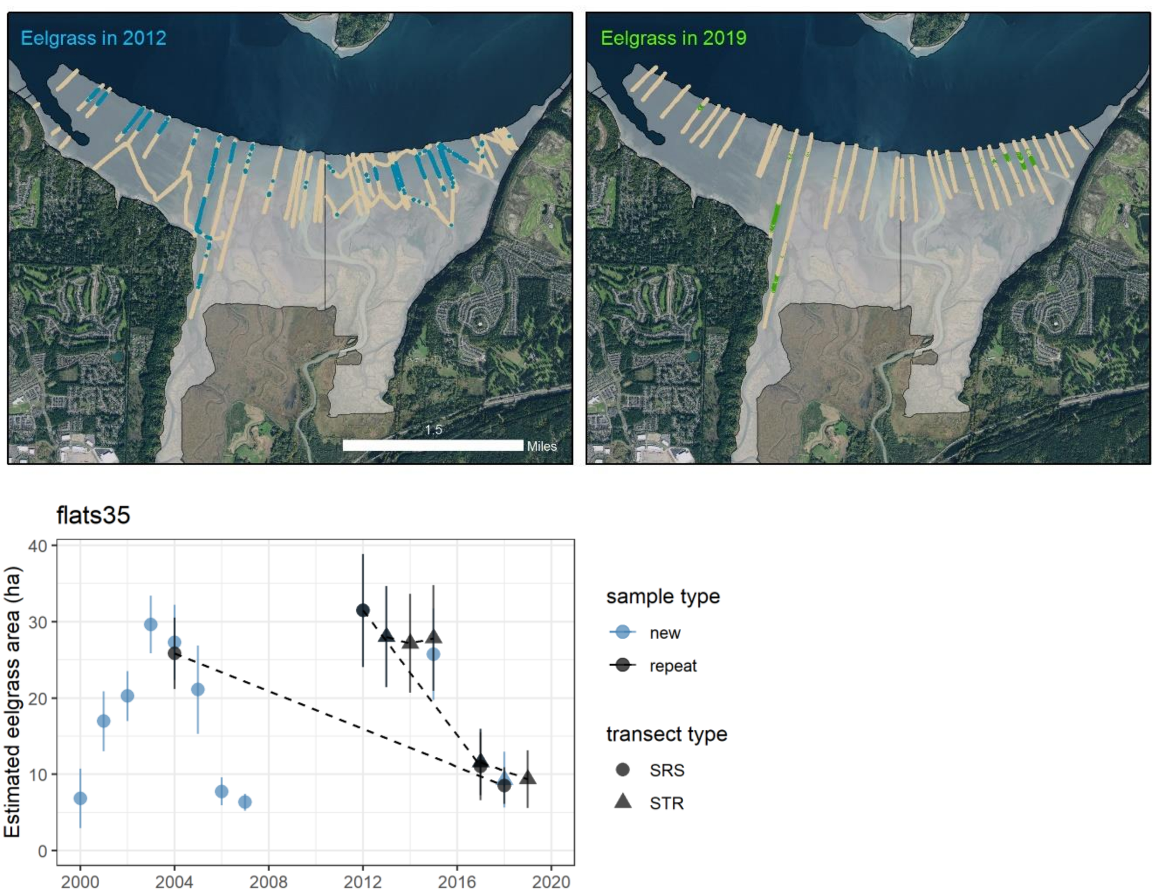

Figure 25: Decline in eelgrass area between 2005 and 2019 flats34 and flats35 (Nisqually delta).

At three sites near the Skokomish delta in lower Hood Canal (hdc2380, hdc2381 and hdc2383), eelgrass beds expanded shoreward by over 80 ha between 2005 and 2013 (Figure 26). Between 2013 and 2016, the shallow edge experienced significant loss (approximately 50 ha), only to expand again between 2016 and 2019. As of 2019, we estimate that there is approximately 137 ha of eelgrass at these 3 locations. Similar to the Nisqually delta, there has been large restoration projects near the Skokomish delta (initiated in 2006). In contrast to Nisqually, eelgrass was most variable at the shallow edge of the bed. In greater Puget Sound, the shallow edge of eelgrass beds is partially determined by desiccation at low tide. The high variability in intertidal eelgrass at this location may be due to complex interactions between environmental drivers such as tidal stage, sun exposure, and river flow and sediment deposition over the delta flat. The abundance of the non-native *Z. japonica* appears to have increased between 2013 and 2019 at both hdc2380 and hdc2381. Further data is needed to confirm that this is a real trend and not a sampling artifact.

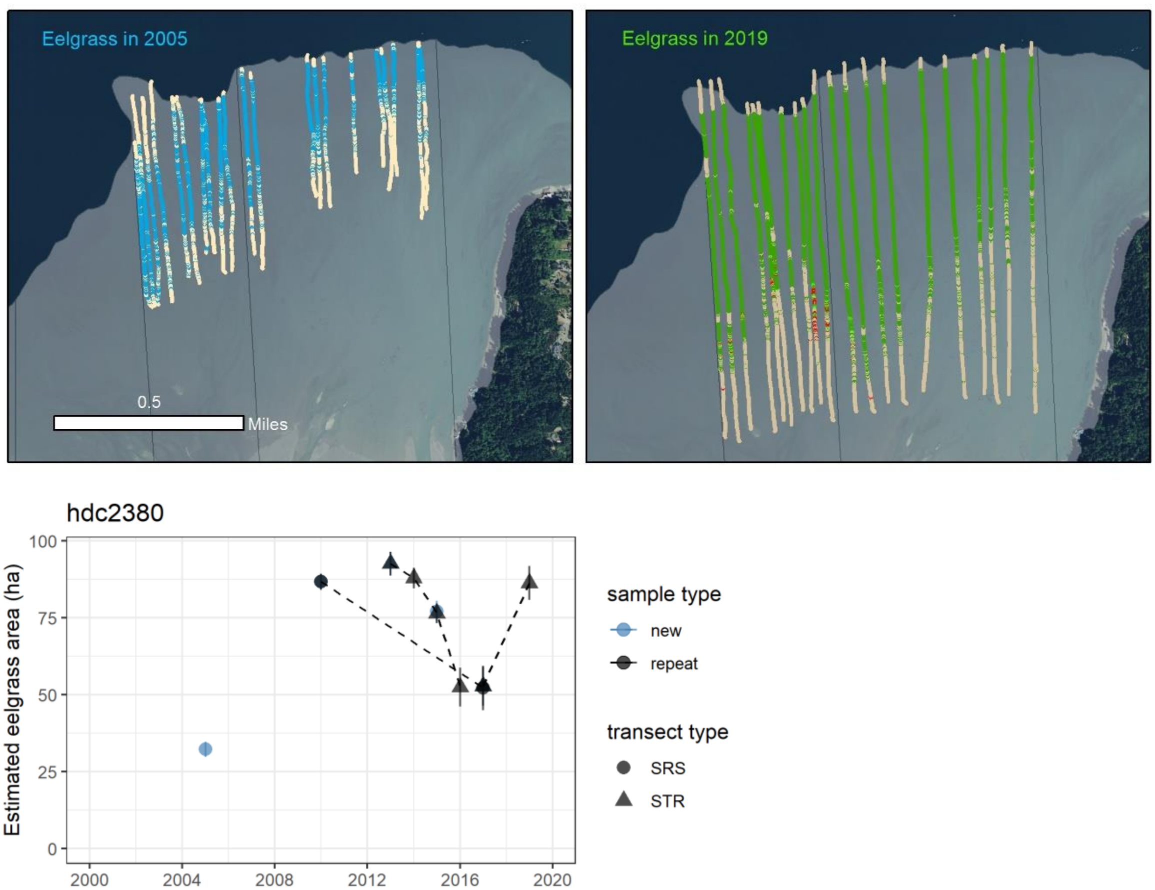

Figure 26: Increase in eelgrass area between 2005 and 2019 at hdc2380 and hdc2381 (Skokomish delta).

At flats20 (N side of the Skagit delta) eelgrass area has continuously declined between 2004 and 2020 (Figure 27). The rate of decline appears to increase between 2016 and 2020. The loss of eelgrass is likely due to erosion. At the end of 2014, a large fraction of the flow from the north fork of the Skagit River was rerouted through a newly formed channel, created by an avulsion of the river through a coastal wetland, 1.5 miles SE of the river mouth. Since then, a series of drainage channels has progressively cut into a contiguous eelgrass meadow at the center of Skagit Bay leading to an approximate 200 ha decline in eelgrass at flats20 between 2004 and 2020. While the adjacent site, flats21, has not been surveyed by the SVMP since 2011, a recent survey by USGS shows a large decline along the northwestern edge of this location (Andrew Stevens, personal communication). We previously hypothesized that the lower flow rate could have a positive impact on eelgrass growing at the former river mouth of the north fork near La Conner (northwestern edge of flats20; Christiaen et al. 2016). The current data do not support this hypothesis.

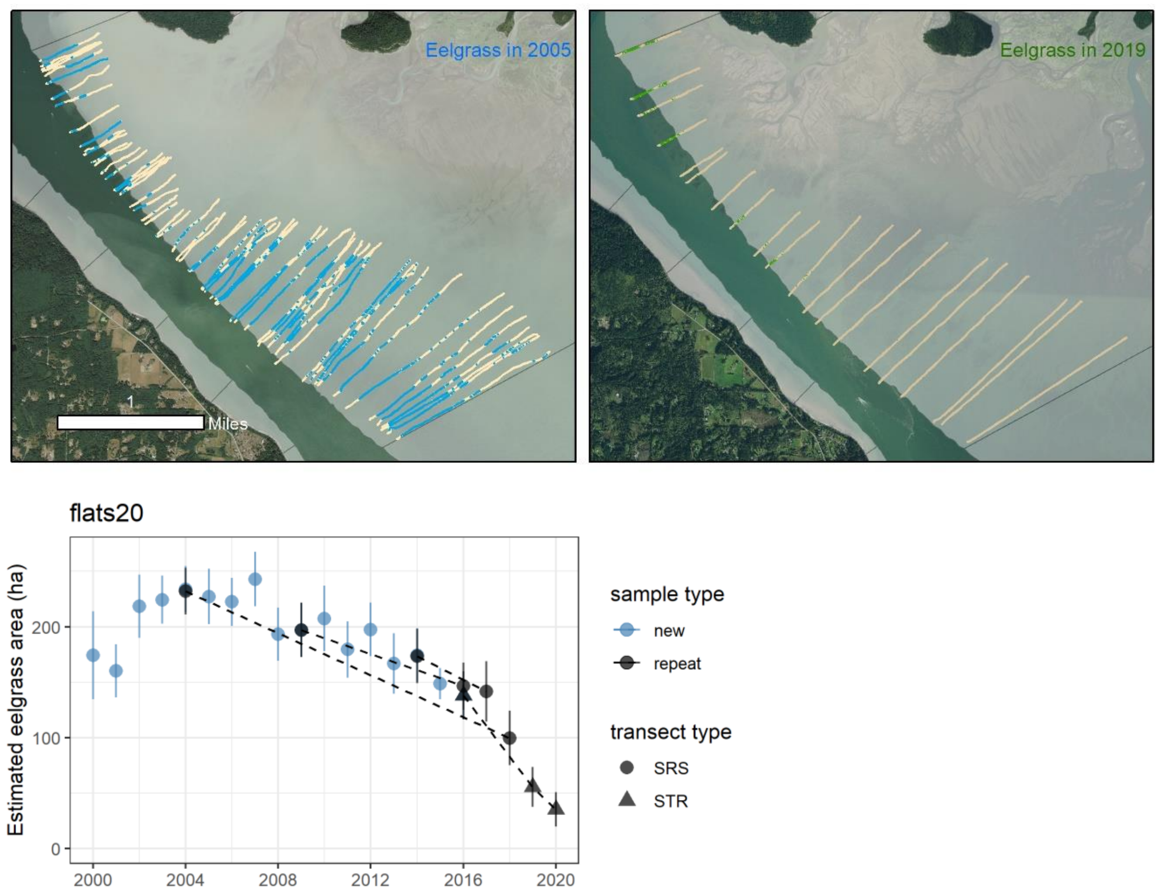

Figure 27: Large loss in eelgrass area between 2004 and 2020 at flats20 (N. Skagit Bay).

4 Discussion

4.1 Status and trends of eelgrass in greater Puget Sound

Z. marina (eelgrass) is the most abundant seagrass in the Salish Sea. Eelgrass is typically found on sandy and muddy substrates in the lower intertidal and shallow subtidal along beaches and tide flats, where it forms dense underwater meadows (Phillips 1972). It grows well at salinities between 10 and 35 psu, and has an optimal water temperature between 10 and 20 degrees C (Nejrup &Pedersen 2008). In the Salish Sea, most of the eelgrass beds are perennials but plants are mostly dormant during winter. The growth season stretches from spring to early fall, with a maximum standing stock in mid to late summer (Thom & Albright 1990). The plants are morphologically plastic: canopy height ranges from less than 40 cm all the way up to 2 m or longer, depending on the depth and the location in Puget Sound.

4.1.1 Area and depth distribution

According to the most recent 3-year pooled soundwide area estimate (2018-2020), there is ~ 22,100 +/- 1,100 ha (mean +/- se) of eelgrass in greater Puget Sound. Approximately half of all eelgrass grows on tidal flats, while the other half occurs as small fringing beds along the shoreline. In Northern Puget Sound and the Saratoga Whidbey Basin the majority of eelgrass grows on tidal flats, while in Central Puget Sound, Hood Canal and the San Juan Islands and the Strait, the majority of eelgrass grows on fringe sites. The size of individual eelgrass beds varies from less than 1 ha to over 3,500 ha for flats sites and from less than 1 ha to over 90 ha for fringe sites, and is partly determined by the amount of suitable substrate in the depth band where eelgrass can grow.

Eelgrass mostly grows between 0 and -4 m relative to MLLW, but has been found as shallow as +1.4 m and as deep as -14.3 m in greater Puget Sound. Approximately half of all eelgrass grows deeper than the extreme low tide line, which forms the boundary between tidelands and bedlands for a large part of Puget Sound. Since most of all bedlands and approximately 29% of all tidelands are owned by the State (Ivey 2014), more than half of all eelgrass in greater Puget Sound grows on State-Owned Aquatic Lands.

There is a clear regional pattern in the depth distribution of eelgrass beds. Eelgrass tends to grow to greater depths in Admiralty Inlet, the San Juan Islands, and the Strait of Juan de Fuca. The deep edge of eelgrass beds becomes shallower the more south one goes into Puget Sound. The deep edge of eelgrass beds is also shallower at the end of inlets or in enclosed embayments, such as Quartermaster Harbor, and near locations with strong riverine influence such as Skagit Bay. These patterns are likely driven by a combination of different factors, including regional changes in water clarity, a north-to-south gradient in tidal range, and localized water quality impairments.

Seagrasses have high light requirements as compared to other marine vegetation, as they need to meet the respiratory demands of their belowground tissues (Lee et al. 2007). The maximum depth to which they grow depends on the amount of light that filters through the water column, and is influenced by water clarity and tidal range. On average, eelgrass requires 5 to 6 hours of light-saturated photosynthesis per day to maintain a positive carbon balance (Alcoverro 1999). In the Pacific Northwest this translates to an average of 3 mol quanta m^{-2} day^{-1} for long-term survival of the plants (Thom et al. 2008).

The shallower maximum depth of eelgrass beds near Bellingham Bay and in the Saratoga Whidbey Basin are partly due to turbidity from glacially fed rivers such as the Nooksack, and the Skagit rivers. The north to south gradient in maximum eelgrass depth corresponds to a large scale gradient in tidal range, which varies from 2 m near the mouth of the Strait of Juan de Fuca to 4.4 m in South Puget Sound[10]. The high tidal range in South Puget Sound limits the amount of light that reaches plants at the deep edge, which could limit the maximum depth of eelgrass at this location (Koch & Beer 1996).

Embayments tend to have lower flushing rates and are more sensitive to water quality impairments due to nutrient over enrichment (Ahmed et al. 2017). This may be one factor contributing to the shallower depth or lack of eelgrass in embayments such as Liberty Bay, Dyes Inlet, or Quartermaster Harbor. Seagrasses are sensitive to nutrient over enrichment, because they are less able to compete for light compared to phytoplankton, epiphytes, or green macroalgae when nitrogen limitation is lifted (Valiela et al. 1997). The presence of fine sediments with high organic matter content could be another contributing factor. High amounts of organic matter are often associated with high sulfide concentrations in sediment pore-waters, which negatively impact seagrass photosynthesis, metabolism, and growth (Holmer et al. 2001, Plus et al. 2003, Holmer et al. 2005).

Over the course of the monitoring program, we have detected several locations with large blooms of green macroalgae. Examples include Yukon Harbor (cps2104-cps2107), Tramp Harbor (cps1159-cps1161), and Seahurst (cps1739). Green algae blooms are often associated with eutrophication, and can have negative impacts on eelgrass and other biota in nearshore habitats (Nelson and Lee 2001, Burkholder et al. 2007). However, they are highly seasonal (Nelson et al. 2003), and more dynamic than eelgrass beds. Our data are a snapshot of one point in time. More information is needed to assess if green algae blooms are a recurring phenomenon at these locations. Efforts are underway to assess multiple years of SVMP footage for several marine vegetation types, including understory kelp, *Sargassum*, and green algae. These data will give a better overview of the relative abundance of different vegetation types in different regions of greater Puget Sound.

4.1.2 Trends in eelgrass abundance

On February 17, 2011, the Leadership Council for the Puget Sound Partnership approved an ecosystem recovery target of 20% increase in soundwide eelgrass area relative to a 2000-2008 baseline by 2020. This target has not been met. Both the annual soundwide

[10] Calculated as the difference between MHHW and MLLW (VDATUM)

eelgrass area estimates and the individual site level trends indicate that soundwide eelgrass area did not substantially increase between 2009 and 2020. There is interannual variability between annual area estimates, which could be due to differing climatic influences, such as precipitation, temperature, and the amount of light available to the plants.

Some of the differences in annual estimates are probably due to sampling. Up to 2014, the soundwide area was based on a stratified random sample of sites with 20% rotation. This means that every subsequent year, 20% of sites were replaced by new randomly selected sites within their stratum. Individual sites were sampled for 5 years before being replaced. Since 2015, we are sampling with a 3-year rotating panel design: we sample 3 random sets of sites (originally sampled in 2004, 2009, and 2014) on a 3 year basis. This allows us to calculate a 3-year pooled average based on 214 random sites that gives a better estimate of soundwide eelgrass area, as well as trends in soundwide eelgrass area over time. However, it does cause some periodicity in the annual estimates.

While there was no overall change in eelgrass area between 2009 and 2020, there is some evidence of trends in eelgrass area within this period of time. The 3-year pooled average declined from 23,285 +/- 1,122 ha between 2014 and 2016, to 22,101 +/- 1,074 ha between 2018 and 2020. Pairwise comparisons between years sampled with the same panel of sites also suggest that there may have been a systematic change over time. Eelgrass area appeared to increase between 2004 and 2016, and declined between 2016 and 2020. Taken together, these results suggest that environmental conditions between 2016 and 2020 were not as favorable for seagrass growth as compared to the period between 2004 and 2016. One potential explanation is that greater Puget Sound experienced two marine heat waves in 2015-16 and 2019 (PSEMP Marine Waters Workgroup, 2020). Warmer water temperatures can increase the respiratory burden of seagrasses, increasing their light requirements (Marsh et al. 1986, Lee et al. 2007). This can lead to lower growth rates in light limited environments, such as the deep edge of seagrass beds, nutrient rich areas with high phytoplankton or epiphyte biomass, or turbid river deltas.

While eelgrass in greater Puget Sound appears stable on a regional scale, there is significant variability at smaller spatial scales. This is consistent with a previous study on eelgrass in the herring spawn areas in Puget Sound (Shelton et al. 2016). Approximately 40% of vegetated sites from the 3-year rotating panel had a significant long-term trend, while 23% of vegetated sites had a significant trend in recent years. Out of these changing sites, there was a similar number of long-term increases and declines. However, in recent years declines outnumbered increases. This pattern seems to confirm the results of the pairwise comparisons in soundwide eelgrass area, which suggest that that eelgrass overall declined between 2016 and 2020. Note that there appears to be a regional pattern in the number of sites with increases and declines. Near the San Juan Islands and the Strait, sites with declines outnumber sites with increases, both on the long-term and in recent years. Eelgrass in enclosed embayments seems particularly vulnerable.

4.2 Eelgrass near the San Juan Islands

Based on the random sample of 214 sites, there is a higher number of declines than increases near the San Juan Islands than would be expected by chance. Site-level trends in the larger SVMP dataset seem to confirm this pattern. Of the 89 sites with enough data for trend analysis near the San Juan Islands, there were 4 sites with long-term increases and 16 sites with long-term declines. This pattern becomes more pronounced when you look at data between 2015 and 2020 (no sites with increases, 15 sites with declines). Eelgrass was lost either at the shallow edge, at the deep edge, or throughout the entire eelgrass bed at these locations. This suggests that these declines were due to a variety of stressors, depending on the location.

One possible stressor is eelgrass wasting disease, caused by the protist *Labyrinthula zosterae*. This disease causes necrotic lesions on the leaves, which limits plant growth, reduces belowground carbon accumulation, and lowers survival of the plants (Graham et al. 2021). Wasting disease is ubiquitous in many areas in the San Juan Islands and is commonly found in eelgrass beds (Watson and Ordal 1951), but has the ability to cause large die-offs when the host plants are under stress (Burge et al. 2013). The prevalence and severity of this disease depends on both environmental factors and characteristics of the plants. Eelgrass wasting disease tends to increase with temperature and salinity, and decline with increasing depth (Jakobsson-Thor et al. 2018, Groner et al. 2021). Older, longer leaves with high levels of fouling have a higher probability of being infected (Groner et al. 2014, Groner et al. 2016). This suggests that mature eelgrass beds and shallow eelgrass beds are more susceptible, while deeper beds or eelgrass in low salinity areas can act as refuge against the disease. Recent studies have found a high prevalence of eelgrass wasting disease in eelgrass beds near the San Juan Islands (Groner et al. 2014, Groner et al. 2016). After the 2015-2016 marine heat wave, an increase in prevalence and severity of eelgrass wasting disease was associated with declines in shoot density in this area (Groner et al. 2021). This suggests that wasting disease may be responsible for some of the recent declines near the San Juan Islands at sites sampled by the SVMP.

Some of the largest eelgrass losses near the San Juan Islands (both in terms of the relative and the absolute amount of eelgrass lost) have occurred in enclosed embayments. However, the timing of these declines has varied. At some embayments, such as Westcott Bay (flats53) and Watmough Bay (sjs0635), declines happened relatively early in the time series. In Swifts Bay (flats62) the declines were gradual over time, while in Reef Net Bay (flats64) and Shallow Bay (flats66) the loss was most pronounced after 2016.

Several factors may have contributed to these declines. Enclosed embayments tend to have lower flushing, which makes them more sensitive to water quality impairments. This could potentially lead to higher turbidity and lower water clarity, which could impact eelgrass through light limitation. In 2007 and 2008, DNR measured light availability across a gradient from the mouth to the head of Westcott Bay, and only found a reduction of 20% in terms of mean daily PAR (photosynthetically active radiation) during the growing season when comparing the head of Westcott Bay (where eelgrass disappeared) to Mosquito Pass (where eelgrass is thriving). The reduced daily average PAR at the head of the bay was above the minimum threshold for *Z. marina* in the Pacific Northwest (Thom et al. 2008, Dowty and Ferrier 2009). This suggests that light limitation was not the primary controlling factor for the loss of *Z. marina* at this location.

Another factor could be the general pattern of warmer water temperatures at the head of shallow embayments during summer. For example, data from the DOH commercial shellfish data viewer show that summer water temperatures are on average more than 3 degrees Celsius higher at the head of the Westcott Bay as compared to Mosquito Pass. There could be a possible interactive effect between light availability and water temperature, since light requirements to maintain a positive carbon balance in eelgrass typically increase with increasing temperatures (Staehr and Borum 2011). As previously stated, warmer water temperatures are also associated with increased eelgrass wasting disease (Groner et al. 2021).

Several embayments with declines in eelgrass populations are also popular destinations for recreational boaters during summer months. Data from marinetraffic.com shows high vessel densities in embayments with documented eelgrass declines, such as Westcott Bay, Garrison Bay, and Shallow Bay. Vessels can cause physical damage to eelgrass beds through anchoring and propeller scars, and may contribute to water quality degradation through resuspension of sediment due to excessive wakes and if vessel black water is discharged instead of using a pump-out station (Orth et al. 2001, Unsworth et al. 2017). Recent efforts by local resource managers to create voluntary no-anchor zones in sensitive embayments may reduce some of the impact on eelgrass at these locations.

While the absolute amount of eelgrass lost at individual sites in the San Juan Islands is relatively small compared to the large losses in for example Skagit Bay, the cumulative loss in the region could be substantial and have trophic level implications. This is a reason for concern, as eelgrass provides valuable habitat for a wide range of organisms, including salmon, forage fish, and waterfowl.

The San Juan Islands are an important feeding and rearing environment for out-migrating juvenile salmon from northern Puget Sound and British Columbia (WRIA 2 Salmon Recovery Plan, 2022). Juvenile chinook, pink and chum salmon arrive in spring, and are often found along the shorelines of Waldron Island, President Channel, and the SW part of Rosario Strait. Pocket beaches appears to be a particularly important habitat, especially or for juvenile chinook (Beamer and Fresh 2012). We have documented eelgrass declines at several sites along these shorelines, particularly on the northern side of Orcas Island. We have also documented several eelgrass declines at sites with pocket beaches, such as Shallow Bay, Reef Net Bay, Watmough Bay, and more recently Picnic Cove.

The San Juan Islands provide significant spawning and rearing area for forage fish, an important source of prey for juvenile salmon (Daly et al. 2009, Chamberlin et al. 2021). Historically, there have been two separate herring stocks that spawn in this region: the ‘Northwest San Juan Island Herring Stock’ and the ‘Interior San Juan Islands Herring Stock’. The ‘Northwest San Juan Island Stock’ was a small stock with spawning grounds primarily in Westcott Bay and Garrison Bay on San Juan Island. This stock disappeared after 2001, which coincides with the large loss of eelgrass in the embayments. Limited surveys up to 2012 did not detect any spawning activity at these sites (Sandell et al. 2019). Herring spawn for the Interior San Juan Islands Stock has been documented in West Sound and East Sound (Orcas Island), Mud Bay (Lopez Island), and Blind Bay (Shaw Island), but spawn deposition was only observed in East Sound in recent years (Sandell et al. 2019). This stock is currently considered in critical condition (92% below the 25-year mean).

4.3 Eelgrass near deltas

River deltas in the Pacific Northwest are often home to expansive eelgrass beds. The large sloping sand flats create large amounts of potential habitat, but are also subject to a range of environmental stressors such as variable salinity, desiccation at low tide, and deposition and erosion due to wave action and river flow. We have documented high variability in eelgrass area at 3 river deltas that have been extensively sampled.

In both the Skokomish and the Nisqually deltas, eelgrass populations fluctuated by over 50% throughout the course of our monitoring program. Near the Skagit delta, eelgrass populations have declined by over 200 ha over the last 15 years. At the Skokomish and the Nisqually deltas, the variability in eelgrass area may be due to complex interactions between environmental drivers. At the Skagit delta, the large loss at flats20 is likely due to erosion, after an avulsion at the North fork of the Skagit River. Since 2014, a series of newly formed drainage channels has progressively cut into a contiguous eelgrass bed at the center of Skagit Bay.

These changes could have an impact on local salmon populations. Juvenile chinook and chum salmon make extensive use of nearshore and estuarine environments during their early marine rearing phase (Duffy et al. 2005, Moore et al. 2016). These species are often found in high abundances in eelgrass beds at the outer edge of river deltas during their outmigration period (Hodgson et al. 2016, Rubin et al. 2018). A large loss, such as in Skagit Bay, could limit available habitat for out-migrants, which rely on eelgrass for forage habitat (Kennedy et al. 2018) and refuge from predation (Semmens 2008). The large declines in eelgrass at the Skagit delta may also have an impact on other fish species, such as Shiner Perch and Pacific Herring, which tend to have higher abundance in eelgrass beds than nearby unvegetated habitat (Rubin et al. 2018).

4.4 Conclusions

- Soundwide seagrass area has remained relatively stable since 2000. This is reassuring and sets Puget Sound apart from many other developed areas, where substantial system-wide declines are ongoing.
- There is evidence that soundwide eelgrass area increased between 2004 and 2016, and subsequently declined between 2016 and 2020. The recent declines correspond with 2 large marine heatwaves in the region (2015-2016 and 2019).
- The PSP goal of 20% increase in seagrass area by 2020 has not been met. Stressors that affect seagrass in Puget Sound will likely need to be reduced to see significant soundwide gains in seagrass area, depth distribution and overall health.
- While eelgrass area in greater Puget Sound appears stable on a regional scale, there is significant variability at smaller spatial scales. There appears to be a pattern in the spatial distribution of sites with increases and declines.
- The San Juan Islands have been identified as a region of concern. Here, sites with declines outnumber sites with increases, both on longer time scales and in recent years. Declines were likely due to a variety of stressors, depending on the location.
- We have noted high variability in eelgrass area at sites near river deltas. At flats20 near the Skagit delta, over 200 ha of eelgrass has been lost to erosion after a large amount of flow from the N fork of the Skagit River was rerouted due to a natural avulsion.

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6 Appendix 1: Site-level trends

Table 5: Short and long-term trends at randomly selected sites in greater Puget Sound (3-year rotating panel, n = 214). At some locations, apparent trends were rejected because of potential misclassification between Z. marina and Z. japonica in different years. The notes column indicates if samples may not be representative of the site (for example, if the research vessel not able to survey the shallow edge of eelgrass bed due to the tide)

site_code	size class	start	end	years sampled	long-term trend	recent trend	notes
core001	over 500 ha	2001	2020	20	increase	no_trend	
core002	1 to 5 ha	2000	2020	21	no_trend	decline	
core003	over 500 ha	2000	2020	21	decline	decline	2000 & 2001 values inaccurate
core004	100 to 500 ha	2000	2020	21	no_trend	decline	
core005	1 to 5 ha	2000	2020	21	decline	no_trend	
core006	5 to 10 ha	2000	2020	21	decline	increase	2000 & 2001 values inaccurate
cps0041	5 to 10 ha	2013	2020	4	no_trend	no_trend	
cps0221	no grass	2002	2018	7	nograss	nograss	
cps0224	no grass	2007	2019	7	nograss	nograss	
cps1035	less than 1 ha	2005	2019	8	no_trend	trace	
cps1054	1 to 5 ha	2008	2019	8	no_trend	no_trend	potential misidentification Zm/Zj
cps1069	10 to 100 ha	2003	2018	8	no_trend	no_trend	
cps1113	5 to 10 ha	2014	2020	4	no_trend	no_trend	
cps1128	1 to 5 ha	2002	2018	7	no_trend	no_trend	potential misidentification Zm/Zj
cps1137	1 to 5 ha	2011	2020	6	decline	no_trend	
cps1141	1 to 5 ha	2008	2019	8	decline	no_trend	
cps1153	5 to 10 ha	2012	2020	5	no_trend	no_trend	
cps1156	5 to 10 ha	2002	2018	7	no_trend	no_trend	
cps1160	1 to 5 ha	2006	2019	9	decline	no_trend	potential misidentification Zm/Zj
cps1164	5 to 10 ha	2002	2018	7	increase	no_trend	
cps1175	1 to 5 ha	2003	2018	9	no_trend	no_trend	
cps1194	no grass	2006	2019	7	nograss	nograss	
cps1204	1 to 5 ha	2013	2020	4	increase	increase	
cps1215	less than 1 ha	2012	2020	5	no_trend	no_trend	
cps1245	no grass	2000	2018	7	nograss	nograss	
cps1277	1 to 5 ha	2003	2018	7	increase	no_trend	
cps1278	less than 1 ha	2010	2020	7	no_trend	no_trend	
cps1282	no grass	2000	2018	7	nograss	nograss	
cps1289	no grass	2007	2019	7	nograss	nograss	
cps1296	no grass	2000	2018	7	nograss	nograss	
cps1663	5 to 10 ha	2007	2020	8	no_trend	no_trend	
cps1676	5 to 10 ha	2005	2019	8	increase	no_trend	
cps1678	10 to 100 ha	2013	2020	5	no_trend	no_trend	
cps1686	5 to 10 ha	2000	2020	14	decline	no_trend	
cps1750	5 to 10 ha	2004	2018	8	increase	no_trend	
cps1764	5 to 10 ha	2012	2020	5	no_trend	no_trend	
cps1770	no grass	2014	2020	3	nograss	nograss	
cps1777	no grass	2008	2019	7	nograss	nograss	

site_code	size class	start	end	years sampled	long-term trend	recent trend	notes
cps1820	*less than 1 ha*	*2004*	*2018*	*7*	*increase*	*no_trend*	
cps1821	*1 to 5 ha*	*2003*	*2018*	*10*	*increase*	*no_trend*	
cps1951	*no grass*	*2005*	*2019*	*7*	*nograss*	*nograss*	
cps1954	*no grass*	*2007*	*2019*	*7*	*nograss*	*nograss*	
cps1967	*1 to 5 ha*	*2004*	*2018*	*11*	*decline*	*no_trend*	*non-linear decline*
cps1983	*no grass*	*2006*	*2019*	*7*	*nograss*	*nograss*	
cps1999	*no grass*	*2008*	*2019*	*7*	*nograss*	*nograss*	
cps2038	*1 to 5 ha*	*2008*	*2019*	*9*	*decline*	*decline*	
cps2047	*less than 1 ha*	*2013*	*2020*	*4*	*no_trend*	*no_trend*	
cps2068	*less than 1 ha*	*2009*	*2019*	*7*	*decline*	*trace*	*non-linear decline*
cps2070	*no grass*	*2012*	*2020*	*5*	*nograss*	*nograss*	
cps2105	*1 to 5 ha*	*2009*	*2019*	*7*	*decline*	*decline*	
cps2182	*no grass*	*2008*	*2019*	*7*	*nograss*	*nograss*	
cps2201	*5 to 10 ha*	*2003*	*2018*	*8*	*no_trend*	*no_trend*	
cps2218	*5 to 10 ha*	*2002*	*2018*	*8*	*no_trend*	*no_trend*	*potential misidentification Zm/Zj*
cps2221	*10 to 100 ha*	*2002*	*2018*	*9*	*no_trend*	*no_trend*	
cps2223	*5 to 10 ha*	*2011*	*2020*	*7*	*decline*	*decline*	
cps2226	*10 to 100 ha*	*2014*	*2020*	*4*	*no_trend*	*no_trend*	
cps2227	*10 to 100 ha*	*2012*	*2020*	*5*	*no_trend*	*no_trend*	
cps2230	*less than 1 ha*	*2011*	*2020*	*6*	*no_trend*	*no_trend*	
cps2544	*5 to 10 ha*	*2013*	*2020*	*4*	*no_trend*	*no_trend*	
cps2552	*10 to 100 ha*	*2007*	*2020*	*9*	*no_trend*	*no_trend*	
cps2565	*1 to 5 ha*	*2007*	*2020*	*12*	*no_trend*	*decline*	
cps2573	*5 to 10 ha*	*2002*	*2018*	*7*	*no_trend*	*no_trend*	
flats03	*100 to 500 ha*	*2009*	*2019*	*7*	*increase*	*no_trend*	
flats08	*10 to 100 ha*	*2003*	*2018*	*7*	*no_trend*	*no_trend*	
flats09	*1 to 5 ha*	*2008*	*2020*	*5*	*no_trend*	*no_trend*	
flats10	*less than 1 ha*	*2002*	*2018*	*7*	*no_trend*	*no_trend*	
flats11	*over 500 ha*	*2001*	*2020*	*20*	*increase*	*increase*	
flats12	*over 500 ha*	*2004*	*2020*	*17*	*no_trend*	*no_trend*	
flats14	*100 to 500 ha*	*2010*	*2020*	*7*	*no_trend*	*no_trend*	
flats15	*100 to 500 ha*	*2008*	*2019*	*8*	*no_trend*	*decline*	
flats16	*10 to 100 ha*	*2008*	*2019*	*8*	*decline*	*no_trend*	
flats17	*1 to 5 ha*	*2008*	*2019*	*7*	*no_trend*	*no_trend*	
flats18	*10 to 100 ha*	*2000*	*2018*	*13*	*no_trend*	*no_trend*	
flats19	*100 to 500 ha*	*2003*	*2018*	*8*	*increase*	*no_trend*	*2011 value inaccurate*
flats20	*100 to 500 ha*	*2000*	*2020*	*21*	*decline*	*decline*	
flats26	*100 to 500 ha*	*2005*	*2020*	*8*	*increase*	*decline*	
flats30	*100 to 500 ha*	*2014*	*2020*	*3*	*no_trend*	*no_trend*	*potential misidentification Zm/Zj*
flats33	*less than 1 ha*	*2004*	*2020*	*5*	*decline*	*nograss*	
flats35	*10 to 100 ha*	*2000*	*2019*	*15*	*no_trend*	*decline*	
flats37	*10 to 100 ha*	*2002*	*2020*	*9*	*no_trend*	*no_trend*	
flats39	*no grass*	*2007*	*2020*	*7*	*nograss*	*nograss*	
flats41	*100 to 500 ha*	*2004*	*2020*	*9*	*no_trend*	*no_trend*	*potential misidentification Zm/Zj*
flats42	*100 to 500 ha*	*2005*	*2019*	*9*	*no_trend*	*increase*	
flats43	*10 to 100 ha*	*2000*	*2018*	*9*	*increase*	*no_trend*	
flats46	*10 to 100 ha*	*2007*	*2020*	*5*	*no_trend*	*no_trend*	
flats49	*100 to 500 ha*	*2011*	*2020*	*6*	*no_trend*	*no_trend*	
flats50	*10 to 100 ha*	*2011*	*2020*	*6*	*no_trend*	*no_trend*	
flats55	*1 to 5 ha*	*2003*	*2019*	*9*	*no_trend*	*no_trend*	
flats58	*5 to 10 ha*	*2003*	*2020*	*10*	*no_trend*	*decline*	
flats62	*10 to 100 ha*	*2000*	*2018*	*8*	*decline*	*no_trend*	*2000 value inaccurate*
flats64	*1 to 5 ha*	*2006*	*2019*	*7*	*decline*	*decline*	
flats66	*5 to 10 ha*	*2003*	*2019*	*8*	*decline*	*decline*	
flats67	*5 to 10 ha*	*2003*	*2019*	*9*	*no_trend*	*no_trend*	*heterogeneous bed*
flats69	*1 to 5 ha*	*2010*	*2020*	*6*	*no_trend*	*no_trend*	

site_code	size class	start	end	years sampled	long-term trend	recent trend	notes
flats70	*100 to 500 ha*	*2004*	*2018*	*8*	*no_trend*	*no_trend*	
flats73	*100 to 500 ha*	*2003*	*2019*	*8*	*increase*	*decline*	
hdc2237	*1 to 5 ha*	*2010*	*2020*	*7*	*no_trend*	*no_trend*	
hdc2239	*5 to 10 ha*	*2002*	*2018*	*8*	*decline*	*no_trend*	
hdc2259	*5 to 10 ha*	*2014*	*2020*	*3*	*no_trend*	*no_trend*	
hdc2283	*10 to 100 ha*	*2006*	*2019*	*7*	*no_trend*	*no_trend*	*potential misidentification Zm/Zj*
hdc2284	*5 to 10 ha*	*2005*	*2019*	*7*	*no_trend*	*no_trend*	*potential misidentification Zm/Zj*
hdc2320	*no grass*	*2014*	*2020*	*3*	*nograss*	*nograss*	
hdc2321	*no grass*	*2007*	*2019*	*7*	*nograss*	*nograss*	
hdc2338	*1 to 5 ha*	*2000*	*2018*	*12*	*decline*	*increase*	
hdc2344	*1 to 5 ha*	*2003*	*2020*	*12*	*decline*	*no_trend*	
hdc2346	*no grass*	*2010*	*2020*	*7*	*trace*	*trace*	
hdc2359	*10 to 100 ha*	*2000*	*2018*	*13*	*increase*	*no_trend*	
hdc2364	*1 to 5 ha*	*2005*	*2019*	*7*	*no_trend*	*no_trend*	
hdc2383	*10 to 100 ha*	*2004*	*2019*	*9*	*increase*	*no_trend*	
hdc2408	*5 to 10 ha*	*2009*	*2019*	*9*	*no_trend*	*decline*	
hdc2460	*5 to 10 ha*	*2007*	*2019*	*7*	*increase*	*no_trend*	
hdc2465	*5 to 10 ha*	*2004*	*2018*	*7*	*increase*	*no_trend*	
hdc2479	*10 to 100 ha*	*2004*	*2018*	*8*	*no_trend*	*no_trend*	
hdc2492	*1 to 5 ha*	*2012*	*2020*	*5*	*no_trend*	*no_trend*	
hdc2511	*1 to 5 ha*	*2010*	*2020*	*8*	*no_trend*	*no_trend*	
hdc2529	*5 to 10 ha*	*2000*	*2018*	*9*	*no_trend*	*no_trend*	
nps0059	*less than 1 ha*	*2000*	*2018*	*9*	*decline*	*decline*	
nps0064	*10 to 100 ha*	*2014*	*2020*	*3*	*no_trend*	*no_trend*	
nps0522	*1 to 5 ha*	*2002*	*2018*	*7*	*no_trend*	*no_trend*	
nps0550	*no grass*	*2006*	*2019*	*7*	*nograss*	*nograss*	
nps0652	*less than 1 ha*	*2009*	*2020*	*10*	*no_trend*	*no_trend*	
nps0654	*10 to 100 ha*	*2002*	*2018*	*7*	*no_trend*	*no_trend*	
nps0669	*no grass*	*2000*	*2018*	*7*	*nograss*	*nograss*	
nps0670	*less than 1 ha*	*2004*	*2018*	*7*	*no_trend*	*no_trend*	
nps0671	*less than 1 ha*	*2014*	*2020*	*3*	*no_trend*	*no_trend*	
nps1320	*10 to 100 ha*	*2003*	*2018*	*7*	*no_trend*	*no_trend*	*potential misidentification Zm/Zj*
nps1328	*1 to 5 ha*	*2007*	*2019*	*7*	*increase*	*no_trend*	
nps1344	*less than 1 ha*	*2005*	*2019*	*7*	*increase*	*no_trend*	
nps1363	*1 to 5 ha*	*2000*	*2018*	*8*	*decline*	*no_trend*	
nps1372	*no grass*	*2011*	*2020*	*6*	*nograss*	*nograss*	
nps1373	*no grass*	*2014*	*2020*	*3*	*nograss*	*nograss*	
nps1375	*no grass*	*2011*	*2020*	*6*	*nograss*	*nograss*	
nps1387	*1 to 5 ha*	*2006*	*2019*	*8*	*no_trend*	*no_trend*	
nps1392	*10 to 100 ha*	*2004*	*2018*	*7*	*no_trend*	*no_trend*	
nps1433	*1 to 5 ha*	*2004*	*2018*	*7*	*increase*	*no_trend*	
nps1461	*10 to 100 ha*	*2014*	*2020*	*3*	*no_trend*	*no_trend*	
nps1487	*1 to 5 ha*	*2008*	*2019*	*8*	*decline*	*no_trend*	
sjs0001	*10 to 100 ha*	*2007*	*2019*	*7*	*increase*	*no_trend*	
sjs0081	*1 to 5 ha*	*2000*	*2018*	*11*	*decline*	*no_trend*	
sjs0099	*10 to 100 ha*	*2012*	*2020*	*5*	*no_trend*	*no_trend*	
sjs0114	*10 to 100 ha*	*2008*	*2019*	*7*	*no_trend*	*no_trend*	
sjs0118	*10 to 100 ha*	*2006*	*2019*	*10*	*no_trend*	*decline*	
sjs0133	*1 to 5 ha*	*2009*	*2019*	*7*	*no_trend*	*decline*	
sjs0138	*1 to 5 ha*	*2004*	*2020*	*7*	*no_trend*	*decline*	
sjs0176	*5 to 10 ha*	*2008*	*2019*	*8*	*increase*	*no_trend*	
sjs0191	*less than 1 ha*	*2009*	*2019*	*7*	*decline*	*decline*	
sjs0205	*10 to 100 ha*	*2005*	*2019*	*7*	*no_trend*	*no_trend*	
sjs0311	*1 to 5 ha*	*2000*	*2018*	*8*	*decline*	*no_trend*	
sjs0318	*no grass*	*2013*	*2020*	*4*	*nograss*	*nograss*	
sjs0330	*1 to 5 ha*	*2014*	*2020*	*3*	*no_trend*	*no_trend*	

site_code	size class	start	end	years sampled	long-term trend	recent trend	notes
sjs0351	*10 to 100 ha*	*2001*	*2018*	*8*	*decline*	*no_trend*	
sjs0417	*less than 1 ha*	*2010*	*2020*	*7*	*no_trend*	*no_trend*	*STR not comparable to SRS*
sjs0427	*no grass*	*2011*	*2020*	*6*	*nograss*	*nograss*	
sjs0448	*5 to 10 ha*	*2006*	*2019*	*7*	*decline*	*decline*	
sjs0452	*10 to 100 ha*	*2006*	*2019*	*7*	*no_trend*	*decline*	
sjs0454	*1 to 5 ha*	*2011*	*2020*	*6*	*decline*	*no_trend*	
sjs0473	*1 to 5 ha*	*2010*	*2020*	*7*	*no_trend*	*no_trend*	
sjs0488	*no grass*	*2006*	*2019*	*7*	*nograss*	*nograss*	
sjs0526	*no grass*	*2010*	*2020*	*7*	*trace*	*trace*	
sjs0544	*1 to 5 ha*	*2008*	*2019*	*7*	*no_trend*	*no_trend*	
sjs0600	*1 to 5 ha*	*2006*	*2019*	*7*	*no_trend*	*no_trend*	
sjs0617	*1 to 5 ha*	*2002*	*2018*	*7*	*decline*	*no_trend*	
sjs0635	*1 to 5 ha*	*2003*	*2018*	*11*	*decline*	*no_trend*	*non-linear decline*
sjs0639	*no grass*	*2005*	*2019*	*7*	*nograss*	*nograss*	
sjs0649	*less than 1 ha*	*2002*	*2018*	*7*	*no_trend*	*no_trend*	
sjs0682	*1 to 5 ha*	*2013*	*2020*	*4*	*no_trend*	*decline*	
sjs0683	*less than 1 ha*	*2003*	*2018*	*8*	*decline*	*decline*	
sjs0695	*no grass*	*2000*	*2018*	*8*	*nograss*	*nograss*	
sjs0819	*1 to 5 ha*	*2001*	*2018*	*8*	*no_trend*	*no_trend*	
sjs0829	*1 to 5 ha*	*2009*	*2019*	*7*	*no_trend*	*decline*	
sjs0983	*10 to 100 ha*	*2009*	*2020*	*7*	*decline*	*decline*	*non-linear decline*
sjs0987	*10 to 100 ha*	*2009*	*2020*	*5*	*no_trend*	*no_trend*	
sjs0989	*5 to 10 ha*	*2003*	*2018*	*9*	*decline*	*no_trend*	*Low sample size 2009/2011*
sjs1004	*1 to 5 ha*	*2013*	*2020*	*4*	*no_trend*	*no_trend*	
sjs1492	*10 to 100 ha*	*2007*	*2019*	*7*	*no_trend*	*no_trend*	
sjs2605	*5 to 10 ha*	*2009*	*2020*	*11*	*no_trend*	*decline*	
sjs2620	*1 to 5 ha*	*2012*	*2020*	*5*	*no_trend*	*no_trend*	
sjs2622	*5 to 10 ha*	*2011*	*2020*	*6*	*increase*	*no_trend*	
sjs2628	*1 to 5 ha*	*2009*	*2019*	*9*	*no_trend*	*no_trend*	*potential misidentification Zm/Zj*
sjs2632	*no grass*	*2009*	*2019*	*7*	*trace*	*trace*	
sjs2645	*less than 1 ha*	*2004*	*2018*	*7*	*no_trend*	*no_trend*	
sjs2646	*1 to 5 ha*	*2000*	*2018*	*7*	*decline*	*no_trend*	
sjs2652	*5 to 10 ha*	*2007*	*2019*	*7*	*increase*	*no_trend*	
sjs2688	*1 to 5 ha*	*2006*	*2020*	*4*	*decline*	*decline*	*2006 value inaccurate*
sjs2695	*less than 1 ha*	*2003*	*2018*	*4*	*trace*	*trace*	
sjs2741	*10 to 100 ha*	*2000*	*2018*	*11*	*no_trend*	*no_trend*	
sjs2742	*no grass*	*2005*	*2019*	*7*	*trace*	*trace*	
sjs2775	*5 to 10 ha*	*2003*	*2018*	*7*	*no_trend*	*no_trend*	
sjs2781	*1 to 5 ha*	*2011*	*2020*	*6*	*no_trend*	*decline*	
sjs2784	*1 to 5 ha*	*2009*	*2019*	*7*	*increase*	*no_trend*	
swh0713	*less than 1 ha*	*2006*	*2019*	*7*	*no_trend*	*no_trend*	
swh0848	*10 to 100 ha*	*2000*	*2020*	*13*	*decline*	*decline*	
swh0869	*less than 1 ha*	*2007*	*2019*	*7*	*increase*	*increase*	
swh0881	*no grass*	*2007*	*2019*	*7*	*nograss*	*nograss*	
swh0882	*no grass*	*2008*	*2019*	*7*	*nograss*	*nograss*	
swh0883	*less than 1 ha*	*2011*	*2020*	*6*	*trace*	*trace*	
swh0901	*5 to 10 ha*	*2013*	*2020*	*4*	*increase*	*no_trend*	
swh0918	*10 to 100 ha*	*2004*	*2018*	*7*	*no_trend*	*no_trend*	
swh0926	*5 to 10 ha*	*2013*	*2020*	*4*	*no_trend*	*no_trend*	
swh0940	*5 to 10 ha*	*2003*	*2018*	*7*	*no_trend*	*no_trend*	
swh0943	*10 to 100 ha*	*2001*	*2018*	*7*	*no_trend*	*no_trend*	
swh0955	*10 to 100 ha*	*2005*	*2019*	*11*	*increase*	*no_trend*	
swh0973	*10 to 100 ha*	*2006*	*2019*	*7*	*increase*	*no_trend*	
swh1556	*5 to 10 ha*	*2000*	*2018*	*9*	*no_trend*	*no_trend*	
swh1557	*1 to 5 ha*	*2004*	*2018*	*7*	*no_trend*	*no_trend*	
swh1568	*less than 1 ha*	*2005*	*2019*	*7*	*no_trend*	*decline*	

site_code	size class	start	end	years sampled	long-term trend	recent trend	notes
swh1574	*10 to 100 ha*	*2012*	*2020*	*5*	*no_trend*	*no_trend*	
swh1575	10 to 100 ha	2001	2018	6	no_trend	no_trend	
swh1593	*5 to 10 ha*	*2000*	*2018*	*8*	*no_trend*	*no_trend*	
swh1625	*1 to 5 ha*	*2000*	*2020*	*9*	*increase*	*increase*	*non-linear increase*
swh1626	*10 to 100 ha*	*2010*	*2020*	*8*	*increase*	*increase*	
swh1646	*1 to 5 ha*	*2011*	*2020*	*8*	*no_trend*	*no_trend*	
swh1649	*5 to 10 ha*	*2005*	*2019*	*7*	*increase*	*increase*	
swh1653	*10 to 100 ha*	*2012*	*2020*	*5*	*no_trend*	*no_trend*	

7 Appendix 2: Site eelgrass area

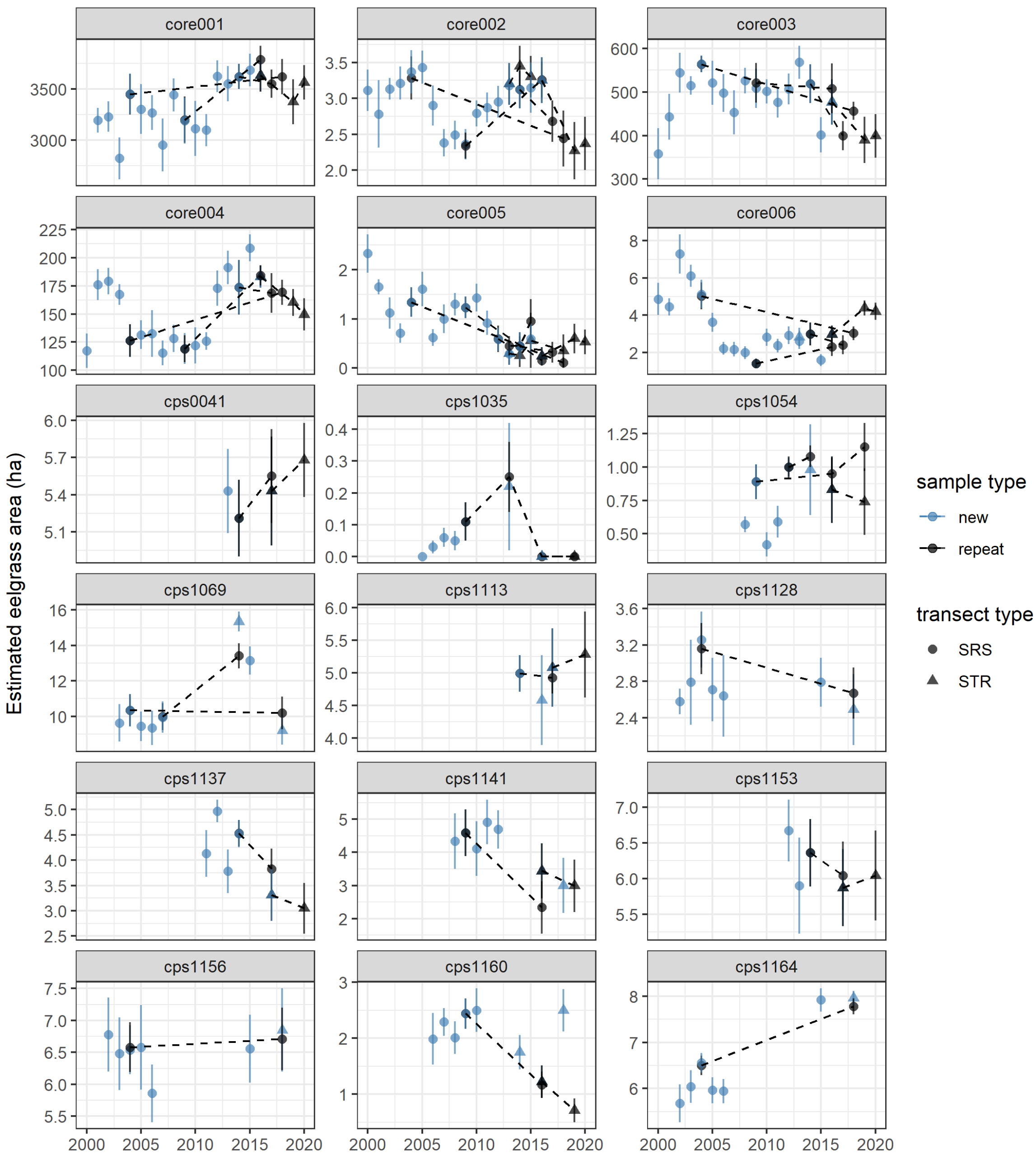

Figure 28: Estimated eelgrass area over time at individual panel sites (ha). Blue indicates new draw random samples, while black symbols represent transects that are resampled over time. SRS are simple random samples, and STR are stratified random samples.

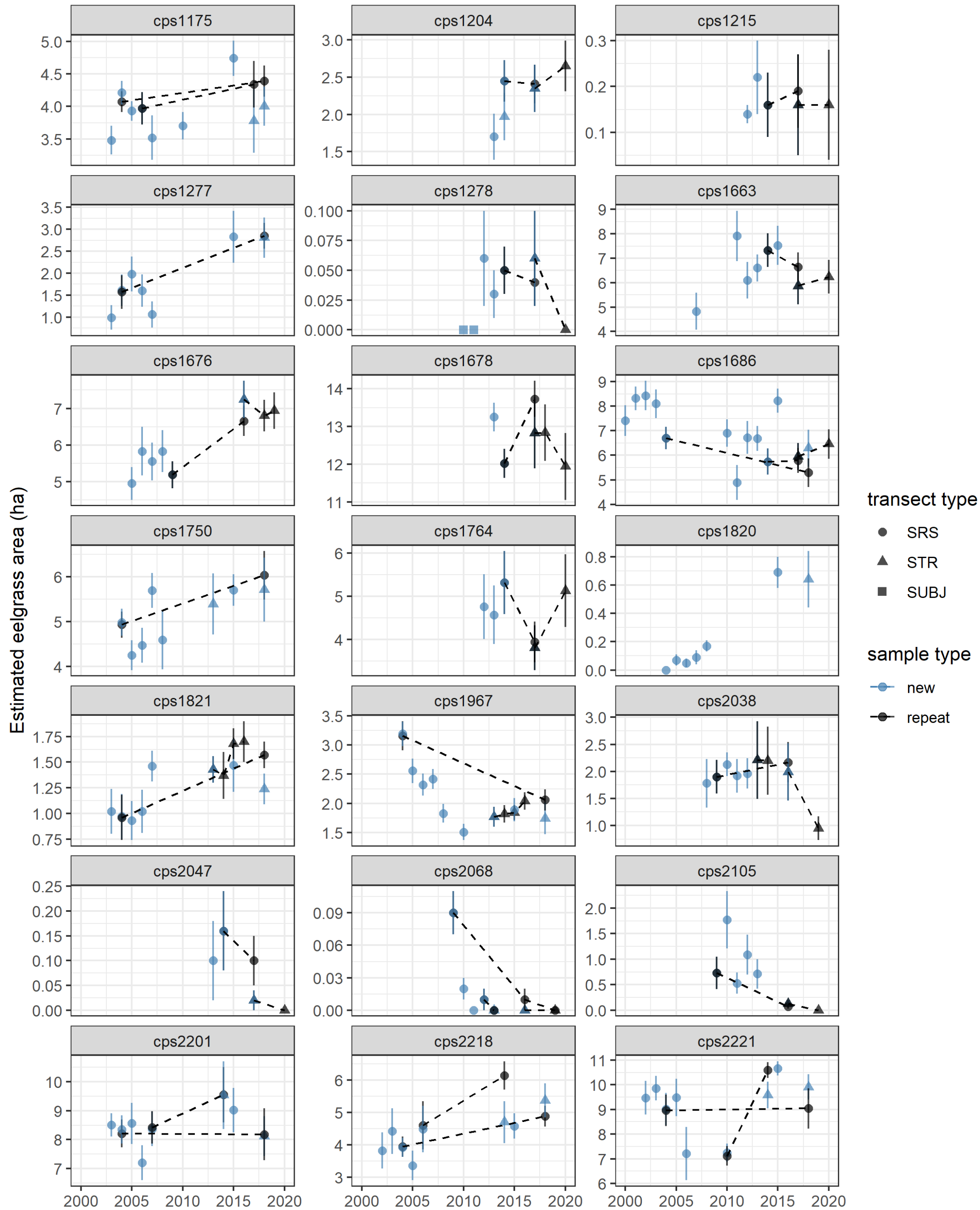

Figure 29: Estimated eelgrass area over time at individual panel sites (ha). Blue indicates new draw random samples, while black symbols represent transects that are resampled over time. SRS are simple random samples, and STR are stratified random samples.

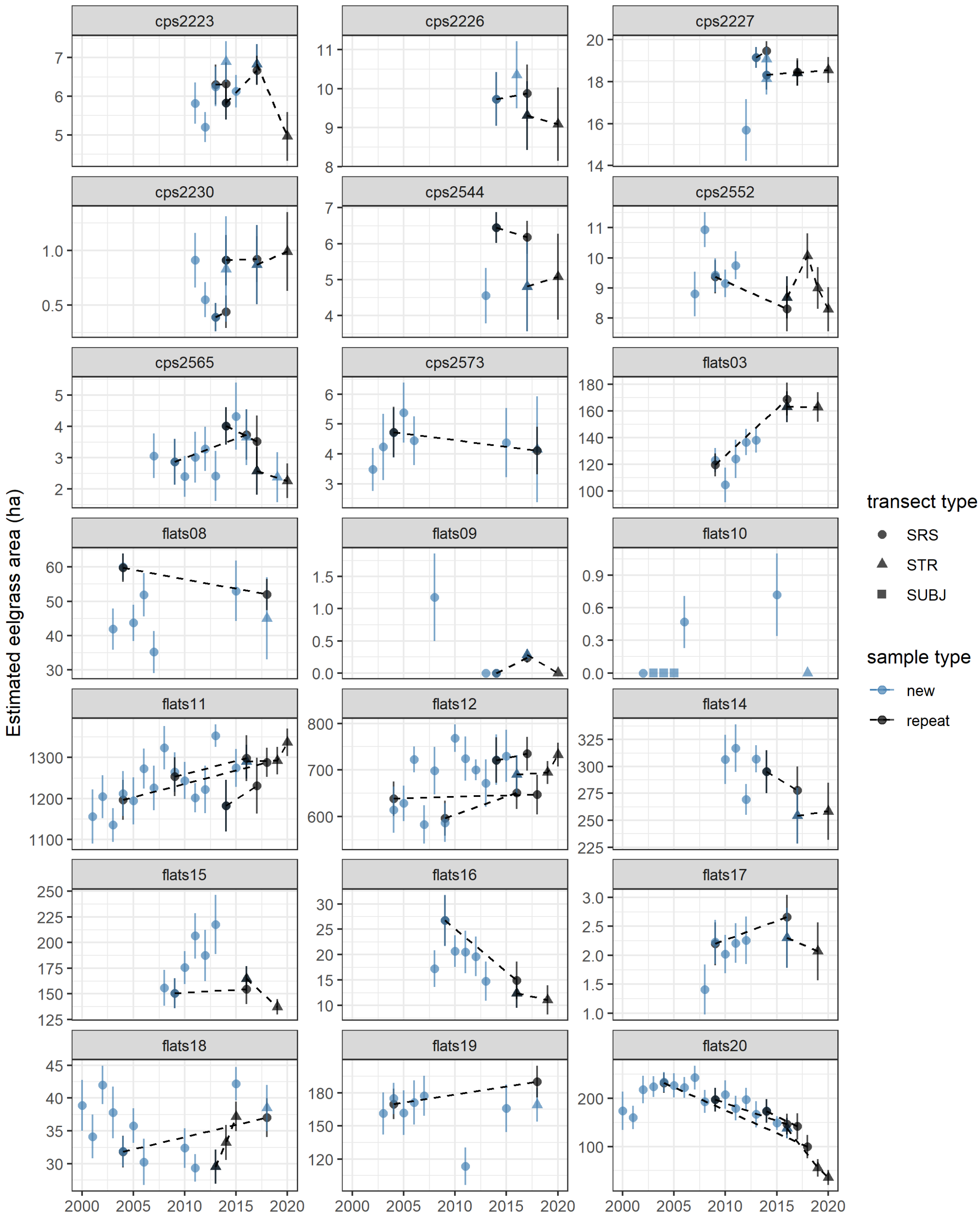

Figure 30: Estimated eelgrass area over time at individual panel sites (ha). Blue indicates new draw random samples, while black symbols represent transects that are resampled over time. SRS are simple random samples, and STR are stratified random samples.

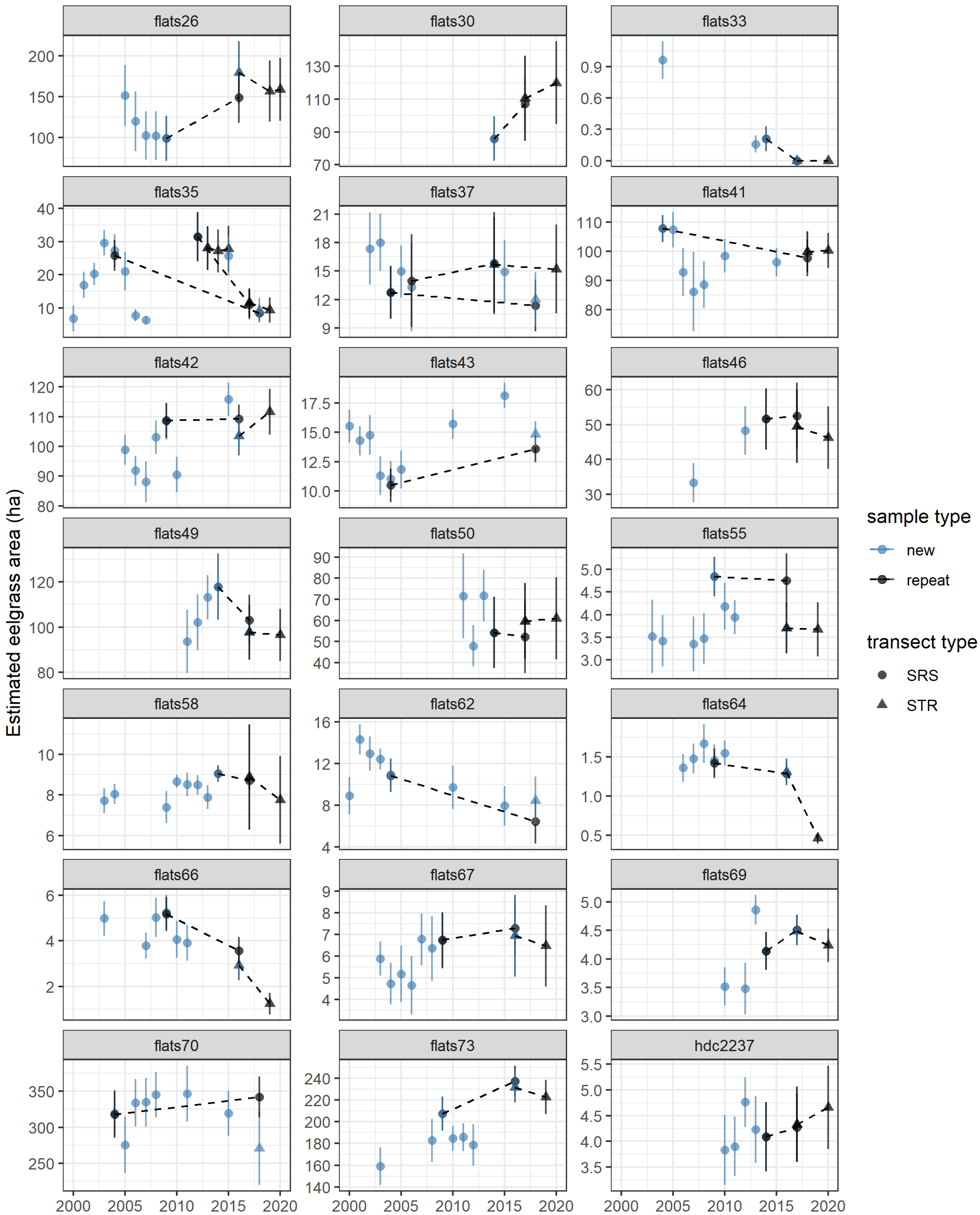

Figure 31: Estimated eelgrass area over time at individual panel sites (ha). Blue indicates new draw random samples, while black symbols represent transects that are resampled over time. SRS are simple random samples, and STR are stratified random samples.

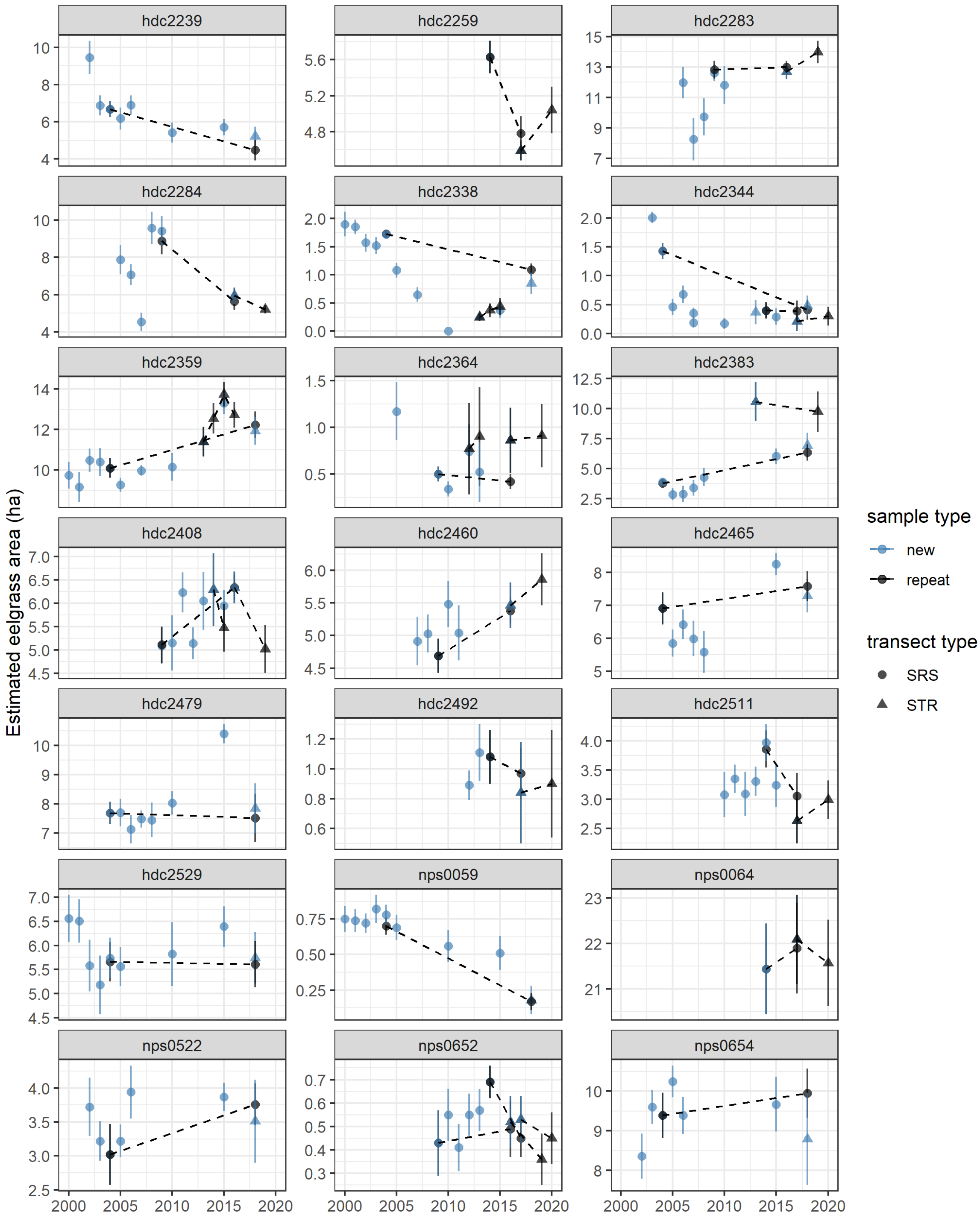

Figure 32: Estimated eelgrass area over time at individual panel sites (ha). Blue indicates new draw random samples, while black symbols represent transects that are resampled over time. SRS are simple random samples, and STR are stratified random samples.

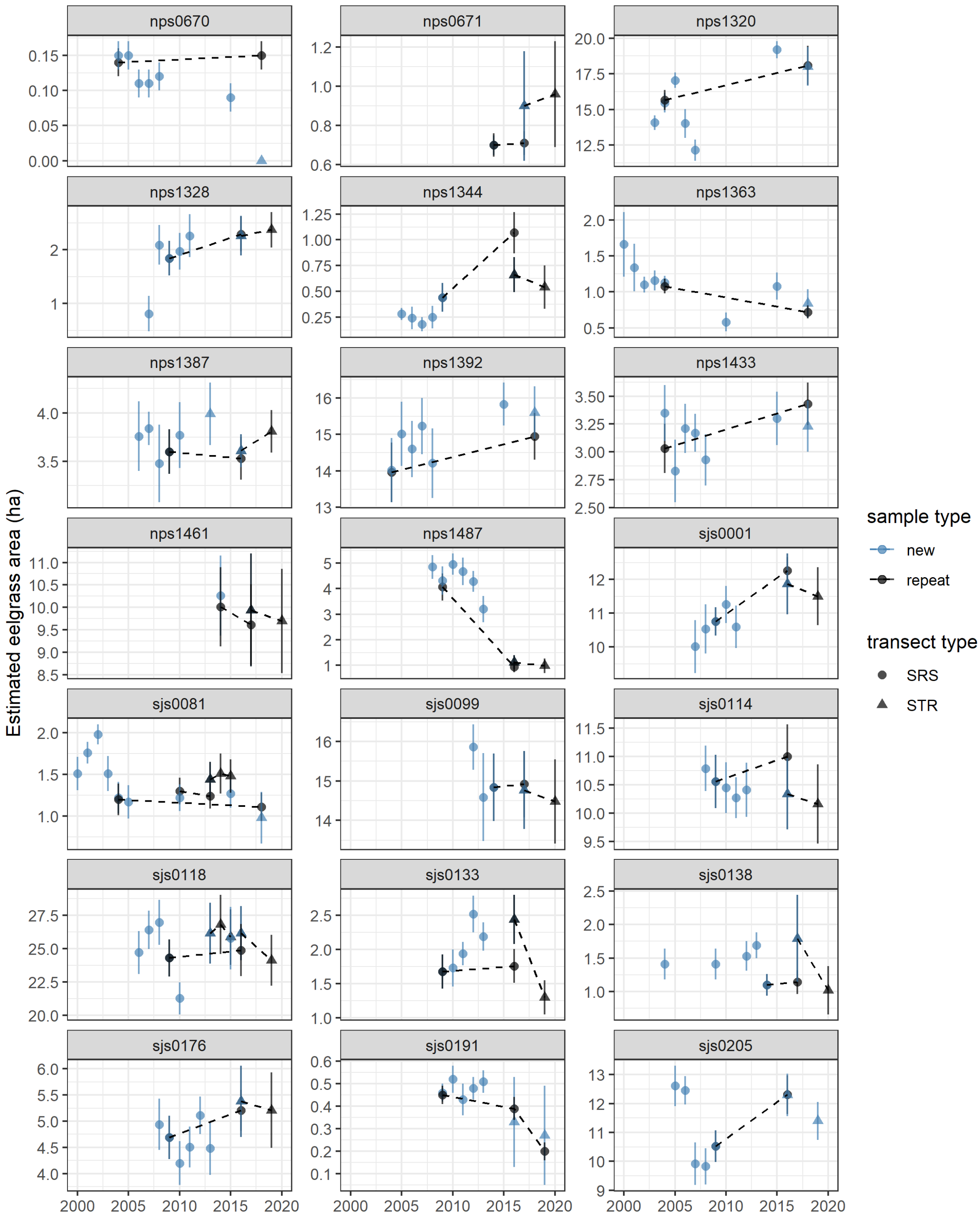

Figure 33: Estimated eelgrass area over time at individual panel sites (ha). Blue indicates new draw random samples, while black symbols represent transects that are resampled over time. SRS are simple random samples, and STR are stratified random samples.

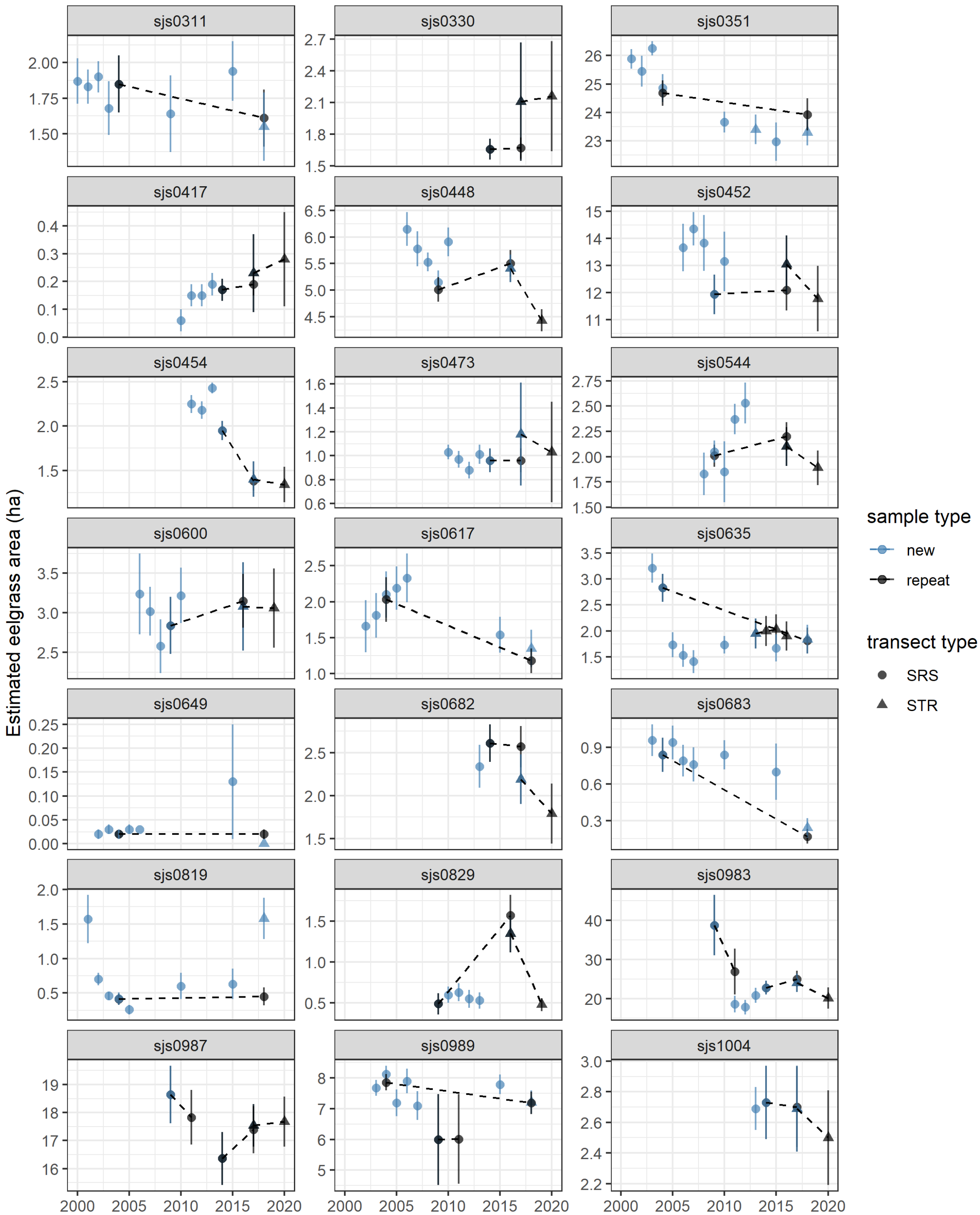

Figure 34: Estimated eelgrass area over time at individual panel sites (ha). Blue indicates new draw random samples, while black symbols represent transects that are resampled over time. SRS are simple random samples, and STR are stratified random samples.

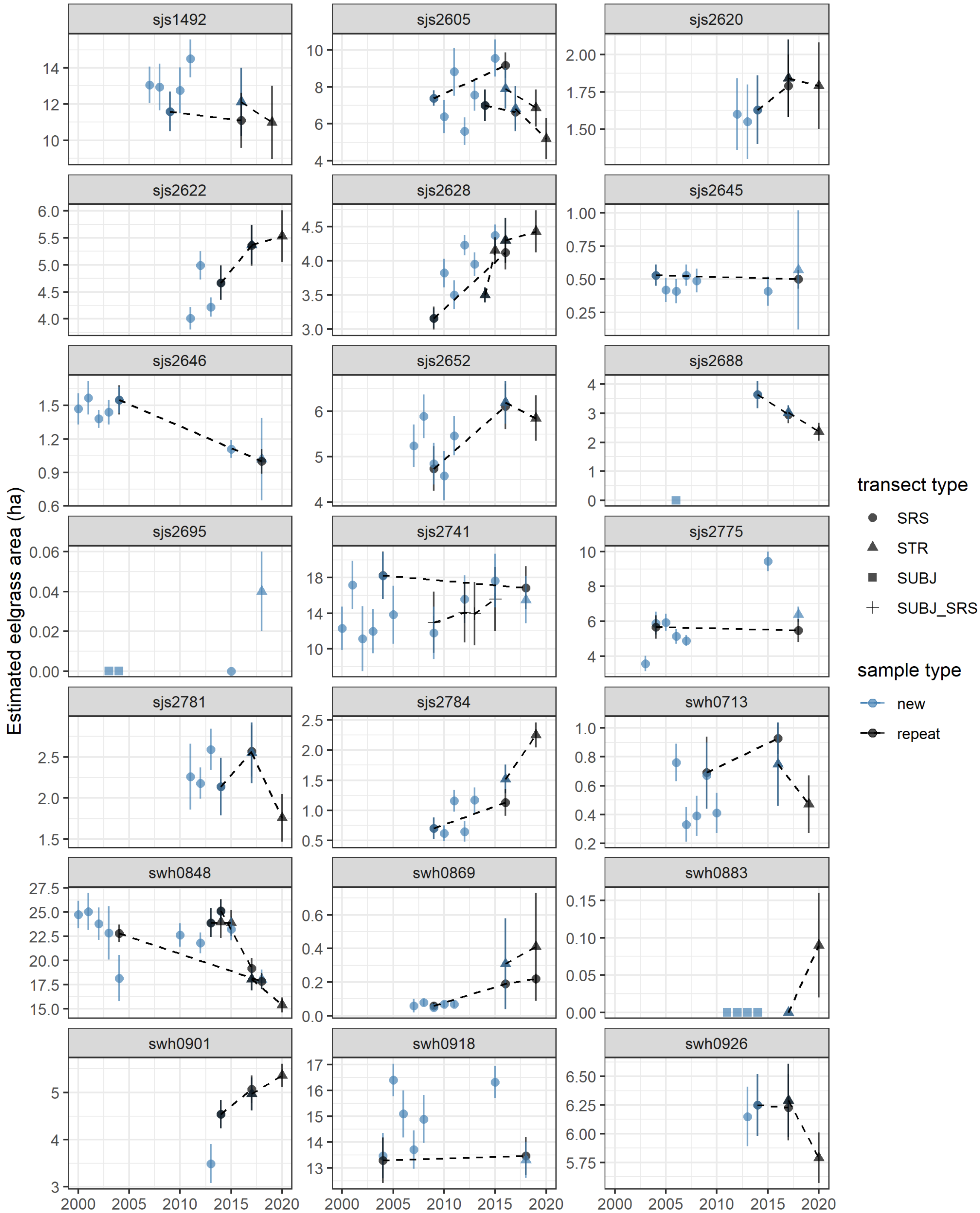

Figure 35: Estimated eelgrass area over time at individual panel sites (ha). Blue indicates new draw random samples, while black symbols represent transects that are resampled over time. SRS are simple random samples, and STR are stratified random samples.

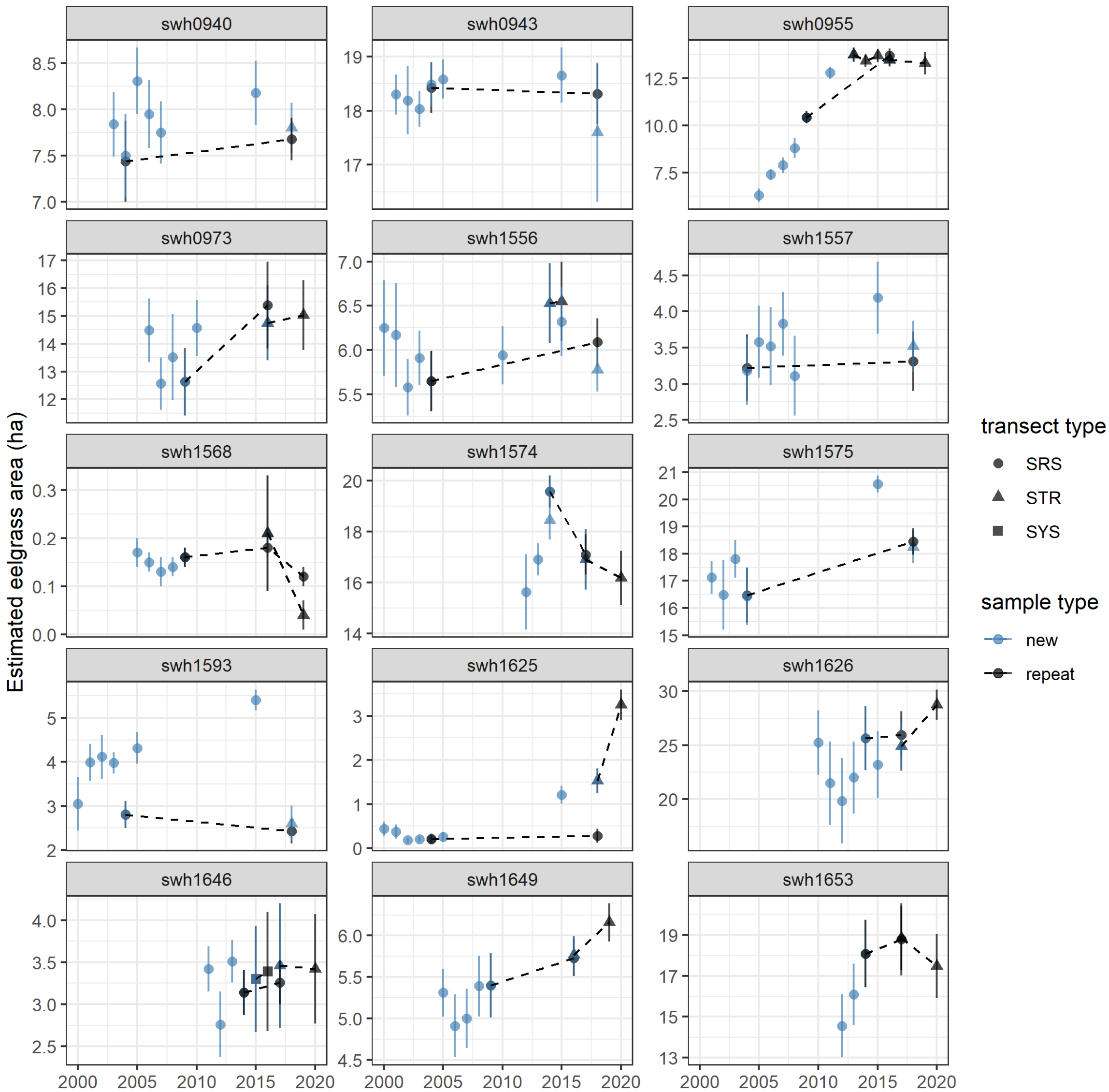

Figure 36: Estimated eelgrass area over time at individual panel sites (ha). Blue indicates new draw random samples, while black symbols represent transects that are resampled over time. SRS are simple random samples, and STR are stratified random samples.

8 Appendix 3: Regional summary

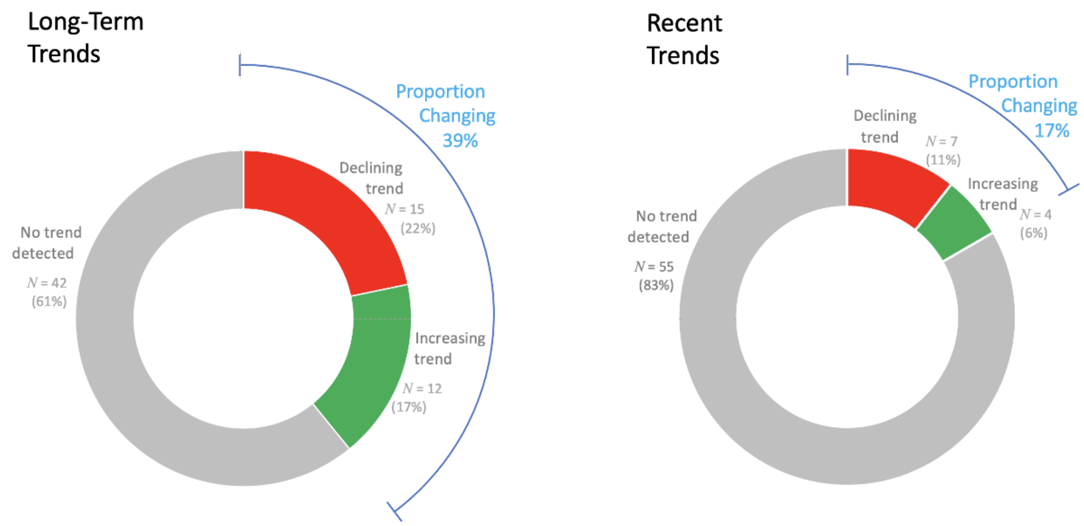

Figure 37: Trend assessment at vegetated panel sites in Central Puget Sound (CPS) and Hood Canal (HDC). Long-term trends are based on all data collected between 2000 and 2020, while recent trends are based on all data collected between 2015 and 2020.

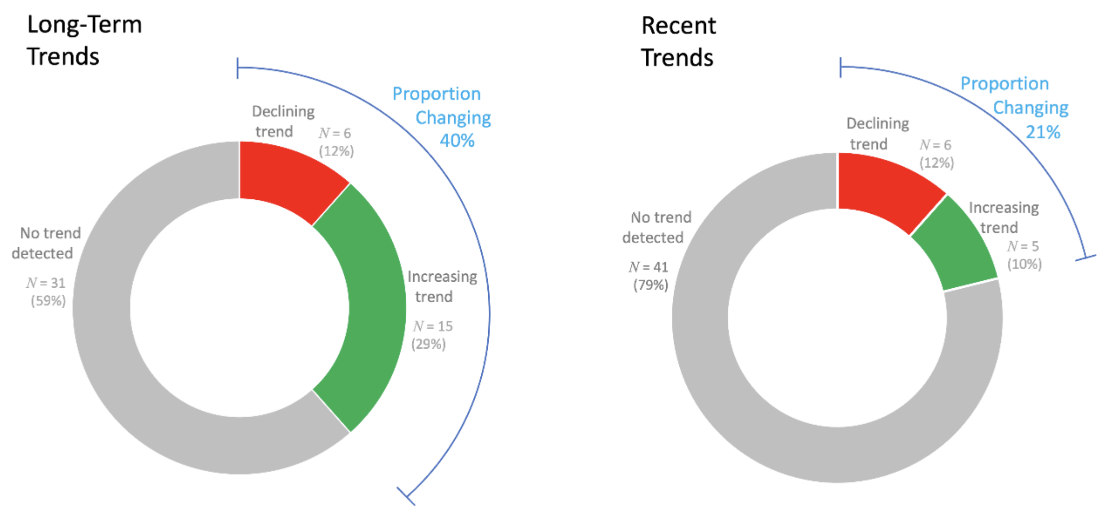

Figure 38: Trend assessment at vegetated panel sites in Northern Puget Sound (NPS) and the Saratoga Whidbey Basin (SWH). Long-term trends are based on all data collected between 2000 and 2020, while recent trends are based on all data collected between 2015 and 2020.

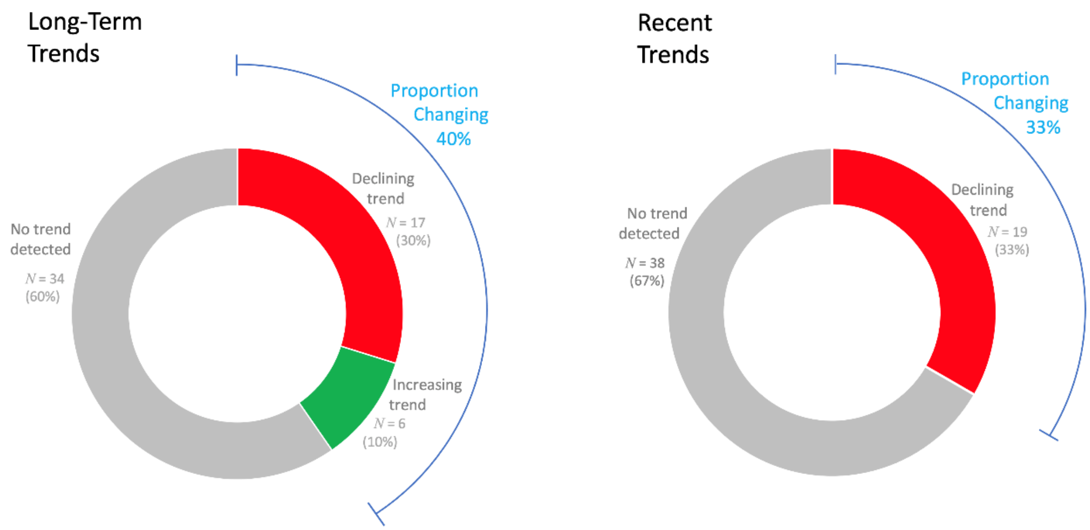

Figure 39: Trend assessment at vegetated panel sites in the San Juan Islands and the Strait of Juan de Fuca (SJS). Long-term trends are based on all data collected between 2000 and 2020, while recent trends are based on all data collected between 2015 and 2020.