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Evaluating the effectiveness of long-term monitoring to detect declines in a large subarctic eelgrass meadow

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ABSTRACT

Eelgrass (*Zostera marina*) forms extensive seagrass meadows in coastal embayments at northern latitudes globally and provides habitat for a wide range of species. Eelgrass abundance has been monitored with a point-sampling survey at Izembek and Kinzarof lagoons, in southwest Alaska, since 2007, to provide annual estimates of above-ground biomass and detect change over time. We conducted a power analysis to determine if the current sampling effort would provide sufficient statistical power to have an 80% chance of detecting a ≥25% decline in lagoon-wide mean eelgrass biomass over 5 yr. We found that the current monitoring scheme is unlikely to meet that goal, but the same rate of decline (–5.6% per year) could be detected in 10 yr. We also found that additional sampling per year would not help detect a trend in eelgrass-when-present. Results from other studies suggest that eelgrass presence can change substantially from year to year, and thus continuing to monitor the current sample size (∼100 points in Izembek Lagoon, 50 in Kinzarof Lagoon) would be useful to monitor changes in presence. Further evaluation of satellite imagery, when available, could also inform changes in the sampling frame if the extent of eelgrass meadows in each lagoon changes. Continued monitoring of eelgrass in this area could inform management of eelgrass itself or of other species of interest that rely on eelgrass, such as Pacific black brant (*Branta bernicla nigricans*), under continuing environmental or ecological changes.

Key words: Alaska, monitoring design, power analysis, seagrass, *Zostera marina*.

INTRODUCTION

Long-term ecological monitoring programs strive to detect meaningful changes in key components of an ecosystem over time (Murray and Krebs 2024). Effective and useful monitoring programs not only track changes in the main components of an ecosystem, but have clear goals and objectives, appropriate study design to adequately detect change, and self-assessments to ensure goals and objectives are achievable over time (Holland et al. 2012). Ecosystems that are highly dynamic, and in areas with greater natural and human stressors, may require more research to maintain the effectiveness of a monitoring program (Munsch et al. 2023).

Eelgrass (*Zostera marina* L.) is a seagrass species that provides important habitat and food source for numerous species, including migratory birds (Ward et al. 2005, Plummer et al. 2013) and commercially valuable fish (Johnson and Thedinga 2005). Because of the importance of eelgrass within coastal ecosystems, its distribution, health, and abundance have been tracked through a variety of ground-based monitoring and remote sensing approaches (Ward et al. 1997, Hogrefe et al. 2014, Balsby et al. 2017, Christiaen et al. 2018, Bartenfelder et al. 2022, Munsch et al. 2023, Douglas et al. 2024). Detecting trends in eelgrass can be challenging due to high interannual variation in eelgrass abundance and distribution (Balsby et al. 2017, Bartenfelder et al. 2022, Munsch et al. 2023). Careful evaluation of the effectiveness of long-term monitoring programs can help ensure that monitoring objectives will be met in the face of these challenges.

In Alaska, eelgrass is the dominant seagrass in coastal areas below about 67°N latitude, extending from the Bering Sea south into the Gulf of Alaska and southeast Alaska (Wyllie-Echeverria and Ackerman 2003). Assessments of eelgrass abundance and distribution have been conducted in many of these regions of Alaska, either as short-term, singular site efforts (Ward et al. 2022) or through regular monitoring of a site to evaluate changes over time (Harris et al. 2012, Ward 2021). One of the largest eelgrass meadows in the world is in Izembek Lagoon (55.3°N, –162.9°W, at the tip of the Alaska Peninsula), with a spatial extent of nearly 170 km^2 (Green and Short 2003, Hogrefe et al. 2014). The longest-running monitoring program for eelgrass in Alaska has been in Izembek Lagoon (340 km^2) and neighboring Kinzarof Lagoon (18 km^2; Ward 2021) (Figure 1). This monitoring program is motivated largely by the importance of eelgrass as a fall staging and overwintering habitat and food source for nearly the entire population of Pacific black brant (*Branta bernicla nigricans* Lawrence; Ward and Amundson 2019). The abundance of black brant that overwinter in these lagoons has increased from under 4,000 in 1981 to over 60,000 birds in 2022 (U.S. Fish and Wildlife Service 2022), which appears to be connected to declining ice cover (Ward et al. 2009) and has raised questions about potential for brant to overgraze eelgrass during its winter dormancy period.

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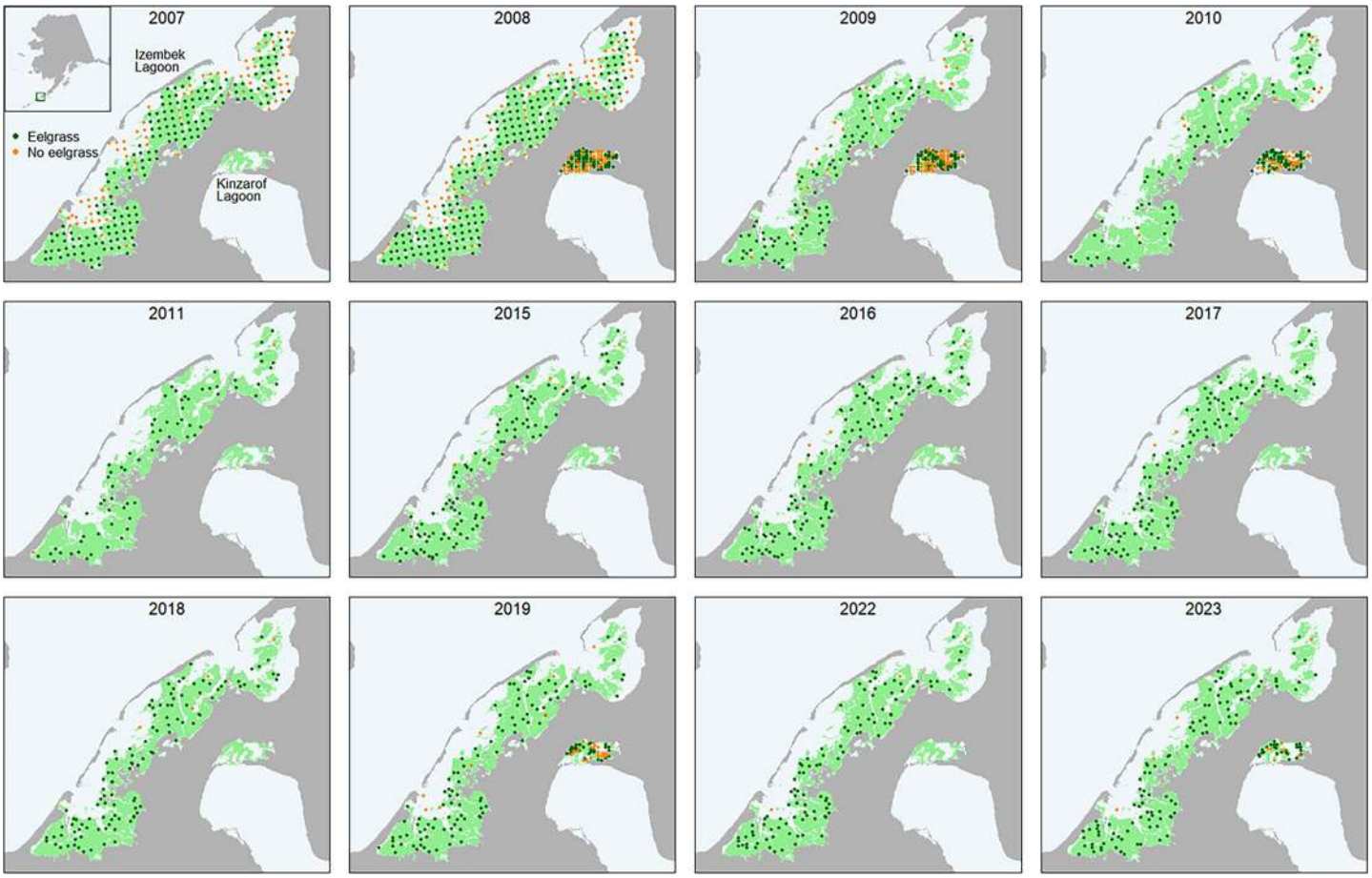

Figure 1. Map of eelgrass sampling locations in each year at Izembek and Kinzarof lagoons on the Alaska Peninsula. Kinzarof was sampled in only 5 yr. Areas with eelgrass beds as determined by remote sensing (Ward and Hogrefe 2022) are shaded in light green; our analysis included only data from within those areas. Sampling points are colored based on whether eelgrass was detected (Ward 2021).

Given these changing conditions, along with the importance of eelgrass to fish and wildlife in this system, effective long-term monitoring of eelgrass is intended to help inform management of eelgrass and species that depend upon it. No overall change was detected in eelgrass biomass in Izembek and Kinzarof lagoons from 2009 to 2018 (Ward and Amundson 2019), but eelgrass was lost at some specific locations in the lagoons between 2006 and 2020 (Douglas et al. 2024). However, no formal evaluation has taken place to evaluate the statistical power of the current monitoring efforts at Izembek and Kinzarof lagoons. Here we conducted a power analysis of the annual monitoring scheme to evaluate whether the current number of points surveyed annually is sufficient to provide an 80% chance of detecting a 25% decline in mean eelgrass biomass over 5 yr at Izembek and Kinzarof lagoons. Additionally, we assessed if eelgrass monitoring could be conducted at a less frequent interval while still allowing detection of that decline.

MATERIALS AND METHODS

The eelgrass monitoring program was initiated in 2007 (Ward and Amundson 2019) because of the ecological importance of Izembek and Kinzarof lagoons, increasing threats to and declines of eelgrass along the west coast of North America (Short and Wyllie-Eciieverria 1996, Ward et al. 2003, Sherman and DeBruyckere 2018), and rapidly changing environment of the North Pacific and Bering Sea region (Wood et al. 2015). A broad-scale estimation of the spatial extent of eelgrass has been made using satellite data (Hogrefe et al. 2014, Douglas et al. 2024). Due to the infrequency of suitable satellite images (cloud-free at low tide with complete visibility of the lagoons; Ward et al. 1997, Douglas et al. 2024), this approach is not suitable for annual monitoring. Instead, the satellite-based estimate of eelgrass extent is used to establish the sampling area, and then annual field sampling is conducted as described below (Ward and Amundson 2019).

Field sampling

We used a previously published dataset (Ward 2021) to characterize biomass of eelgrass at Izembek and Kinzarof lagoons (Figure 1). The dataset included 12 yr for Izembek (2007–2011, 2015–2019, 2022–2023) and 5 yr for Kinzarof (2008–2010, 2019, 2023). In each year, a sample of points was selected across each lagoon, with the locations and numbers varying across years (averaging about 100 points per year for Izembek and 50 for Kinzarof, with the Kinzarof points being more closely spaced). Each year, 75% percent

Table 1. Number of points sampled in each year and each lagoon within the area determined by remote sensing to contain eelgrass beds. These are the points that were used in the power analysis, excluding points that were sampled outside the remotely sensed eelgrass areas. Kinzarof was not sampled in many years (—). Original data are from Ward (2021) but here we excluded the points outside the eelgrass area.

Year	Izembek	Kinzarof
2007	150	—
2008	163	71
2009	94	64
2010	60	62
2011	59	—
2015	98	—
2016	103	—
2017	105	—
2018	100	—
2019	92	35
2022	98	—
2023	101	29
Mean	102	52

of the points sampled in the previous year were retained to sample again, and the remaining 25% were replaced with new points, to balance monitoring trends over time with a cumulative evaluation of lagoon-wide eelgrass distribution. Timing of surveys coincided with peak eelgrass biomass (late July to early August).

The sampling area within each lagoon varied across the years of the study. In 2007–2008 at Izembek and 2008–2010 at Kinzarof, sampling was widespread across each lagoon (Figure 1). Later an analysis of satellite imagery was performed to delineate the spatial extent of eelgrass in each lagoon (Ward and Hogrefe 2022), and field sampling was almost entirely limited to areas with eelgrass in subsequent years. Therefore, to ensure that the power analysis was representative of the current sampling frame and informed by data that were comparable across years, we used data only from within the area delineated as containing eelgrass for each lagoon in all following analyses (Table 1).

Sampling locations (points) in the published field dataset (Ward 2021) were typically accessed by boat and sampled by snorkeling in dry suits. In a few instances when water depth dropped too low during a survey, final access to the point was by foot. Four quadrats were sampled per point, with the quadrats placed haphazardly 1–4 m from the point in each of the four cardinal directions. Percent cover of eelgrass, defined as the portion of the quadrat area that is covered by eelgrass when observed from above, was estimated in each quadrat (Ward and Amundson 2019). If eelgrass was present in a quadrat, 10 (years 2007–2009) or 5 (2010–2023) representative vegetative shoots were collected and measured for shoot length (meristem to tip of longest leaf) and width. Additional data were collected on environmental and biological conditions, but were not used in the present analysis. To calculate the relationship between observed metrics and eelgrass biomass (dry weight g per m^2), all eelgrass shoots were collected from a subset of 15–30 quadrats per year. Shoot length was measured for 10 representative fresh shoots from each calibration quadrat; then all shoots from the quadrat were dried to constant mass, and the dry mass of each quadrat was recorded. A calibration model was used to determine the relationship between eelgrass dry mass, shoot length, and percent cover for each calibration quadrat (Patil 2024). This relationship was then used to obtain estimates of eelgrass biomass in all remaining quadrats where biomass was not directly measured. When some information was missing for a quadrat, the mean value of biomass (159 g per m^2 across all quadrats in all years) was used. We used these predicted values of eelgrass biomass by quadrat to inform our power analysis.

Analysis

As detailed below, we first evaluated any annual trend in eelgrass biomass in the field dataset to provide context for our power analysis. We then conducted a power analysis to determine if the current sampling effort could detect a 25% decline over 5 yr in eelgrass biomass at Izembek and Kinzarof lagoons by 1) estimating biological parameters from the existing dataset to inform the power analysis, 2) identifying a set of candidate sampling designs to evaluate (scenarios), 3) simulating data following those sampling designs and the biological parameters, and 4) evaluating the proportion of simulations in which a trend in eelgrass biomass was successfully detected under each scenario.

Modeling framework. To the field dataset, we applied a log gamma hurdle model in a Bayesian framework following a previous analysis (Ward and Amundson 2019, Patil 2024) with a response variable of eelgrass biomass in dry grams per square meter. The model included a presence-absence hurdle process, which described whether each quadrat was occupied by eelgrass (i.e., occupancy) using logistic regression, as well as a gamma regression process to define eelgrass biomass on the log scale at occupied quadrats ("biomass-when-present"). Together, the two processes determined gross biomass (occupancy $\times$ biomass-when-present). The model accounted for the hierarchical nature of the sampling design with three elements: 1) random effects of point on quadrat-level occupancy and biomass-when-present (i.e., quadrats were nested within points), 2) categorical effects of lagoon (Izembek as the baseline) on occupancy and biomass-when-present, and 3) a categorical effect of year on biomass-when-present. With the inclusion of the year effect on biomass-when-present, we could not also include a categorical effect of year on occupancy, as the two effects would have been unidentifiable. Given the current design of sampling only within the remotely sensed eelgrass area, we expected little change in occupancy, so we considered effects of year only on biomass-when-present throughout this study.

Our model differed from the previous version (Patil 2024) in that we did not include other covariates. Values of most environmental covariates would be difficult or impossible to predict in the datasets we simulated for the power analysis (below), so in the simulated data, any underlying covariates are treated as unexplained variation or noise in the data. For consistency with that framework, when estimating parameters from the field dataset, we likewise ignored other covariates (treating them as unexplained noise). However, when a real dataset is analyzed in the future, including other covariates could help explain variation and improve statistical power to detect a trend.

Table 2. Parameters estimated from the field dataset (directly or from our model) as input values to inform the data simulation used in the power analysis for detecting trends in eelgrass biomass (dry weight g per m^2) at Izembek and Kinzarof lagoons in Alaska. Values were calculated using only samples taken within the area of each lagoon that had been deemed by remote sensing to include eelgrass (Ward and Hogrefe 2022). We used Izembek estimates of interannual variation for Kinzarof (—) because the field dataset included only 5 yr for Kinzarof, thus potentially incompletely representing interannual variation. Some other values are identical between lagoons where the model was structured such that only a global value was estimated, rather than lagoon-specific values.

Metric	Parameter	Description	Scale	Izembek	Kinzarof
Occupancy	Quadrat mean	Mean occupancy probability per quadrat[1]	Natural	0.9993	0.9785
	Interannual SD	Variation across years[2]	Natural	0.3719	—
	Interpoint SD	Variation across points (within years)[1]	Natural	0.0004	0.0122
	Interpoint SD	Variation across points (within years)[1]	Logit	4.5049	4.5049
	SD in interpoint SD	Variation across replicates in the interpoint SD[1]	Logit	0.4536	0.4536
Biomass-when-present	Interannual SD	Variation across years[1]	Log	0.4844	—
	SD in interannual SD	Variation across replicates in the interannual SD[1]	Log	0.0124	—
	Interpoint SD	Variation across points (within years)[1]	Log	1.0437	1.0437
	SD in interpoint SD	Variation across replicates in the interpoint SD[1]	Log	0.0365	0.0365
	Interquadrat SD	Variation across quadrats (within points)[2]	Log	0.2179	0.3508
Gross biomass	Mean	Overall mean biomass across all points and years[1]	Log	5.0224	5.0869
	Interannual SD	Variation across replicates in mean gross biomass[1]	Log	0.3853	—
Calibration uncertainty	CV across samples	Variation in the relationship between field measurements (length, percent cover) and empirical estimates of eelgrass biomass[3]	Proportion	0.1400	0.1400

[1]Estimated with the model.
[2]Estimated directly from the raw data (Ward 2021).
[3]Estimated by Patil (2024).

The model was previously applied in JAGS (Plummer 2003), but we implemented it in Nimble (de Valpine et al. 2017) to reduce runtime. We discarded 15,000 iterations as burn-in, ran another 30,000 iterations, and thinned by 10 to reduce memory requirements, thus retaining 3,000 iterations. This was sufficient to achieve good mixing of chains based on visual assessment of trace plots as well as convergence based on Gelman-Rubin statistics ($\hat{R}$) < 1.1 (Gelman and Rubin 1992), except the overdispersion term was unidentifiable. Uncertainty in the value of the overdispersion term due to nonconvergence could have slightly increased the standard errors of the other parameter estimates but would not bias their values.

Previous trend. We first applied the modeling framework to the field dataset to evaluate any previous trend in eelgrass biomass. This evaluation provided context for our goal of detecting a 25% decline over 5 yr. We used the estimated intercept (which differed between lagoons) and the categorical year effect (which did not differ between lagoons) to calculate an annual estimate of eelgrass biomass (dry g per m^2) for each lagoon. We then applied a linear regression to the annual estimates to estimate the observed trend over time for each lagoon. We considered a trend to be significantly detected if $p < 0.05$ for the slope of the linear regression, and divided the slope by the estimated mean biomass from the first year to calculate the observed percent change per year (i.e., to compare to our goal of detecting 25% decline over 5 yr, which equates to –5.6% per year).

Power analysis. The second component of our analysis was a simulation-based power analysis. We informed the power analysis with parameter estimates from the model as applied to the field dataset described above (Table 2). Where possible, we incorporated model uncertainty by using the standard error (SE) of the model estimate as the standard deviation (SD) of the simulated values. For three additional parameters needed for the power analysis, we calculated values directly from the field data, as these parameters were not estimated by the model: the SD across quadrats (within points) in biomass-when-present, the interannual SD in occupancy, and the uncertainty in converting field measurements to biomass from the calibration model (expressed as a coefficient of variation [CV]).

We simulated a suite of scenarios where sampling parameters varied. While our goal was to detect a 25% decline in 5 yr, monitoring programs often have low power to detect change over short time series (Thogmartin et al. 2007, Urquhart 2012), so we also evaluated scenarios with the same annual rate of decline (–5.6% per year) but where monitoring occurred over 10 or 15 yr. Our scenarios also included cases where sampling was conducted every 1, 2, or 3 yr and where 50, 100, or 150 points were sampled each year within the eelgrass polygon. In all cases, we assumed four quadrats were sampled per point.

For each lagoon separately, we simulated data under the same hierarchical structure as the real dataset: four quadrats within N points in each of Y years in each lagoon, using the summary statistics from the corresponding lagoon (Table 1) and the trend of interest (–25% every 5 yr). Here we treated year as a continuous variable, rather than categorical as above, to represent a trend over time. Interannual variation was also applied to biomass-when-present, such that the trend was not perfectly linear over time but rather showed random noise as observed in the field dataset. Trend was applied lagoon-wide, not point by point, but points were assigned mean offsets from the lagoon mean to represent random spatial variation that was constant over time. The annual mean for each point was drawn from a distribution with a mean = lagoon mean + point-specific offset. Finally, the biomass value for each quadrat was drawn from a distribution defined by the annual point mean and the interquadrat SD estimated from the field dataset. We also simulated occupancy with interannual

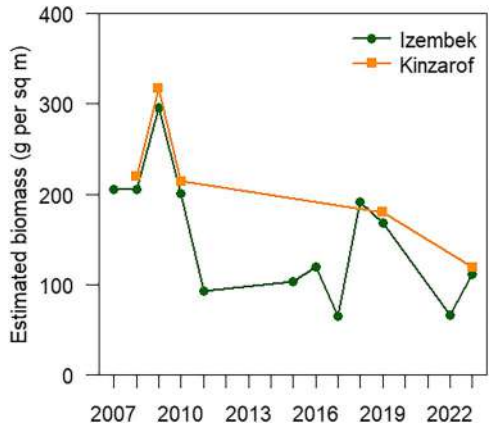

Figure 2. Annual estimates of eelgrass biomass in Izembek and Kinzarof lagoons, Alaska, for each year in the field dataset. Estimates are shown only for years in which each lagoon was sampled. Estimates between lagoons were very similar because the model allowed the lagoons to differ only in their intercepts, not in the categorical year effect on biomass, due to limited data from Kinzarof Lagoon.

variation and spatial variation, but without an overall trend over time.

To each simulated dataset under each scenario, we then applied the same model used on the field dataset, but with year as continuous variable (rather than categorical) such that the year effect represented the trend over time. Here we modeled each lagoon separately to best evaluate whether a trend could be detected in each lagoon. The model for the power analysis otherwise matched the model described above. We repeated the data simulation and model fitting for 500 replicates of each scenario. If the 95% Bayesian credible interval (BCI) of the year effect did not overlap zero and the mean was < 0, we considered the replicate to have successfully detected the decline. We calculated the power of each scenario as the proportion of replicates in which the trend was detected, and compared that power to the target of an 80% chance of detecting the decline to determine which sampling designs would successfully meet the monitoring goal (detecting a decline of –5.6% per year). Our scripts to implement the power analysis, including simulating data, applying the model, and calculating power, are publicly available (Weiser 2025).

RESULTS AND DISCUSSION

Our analysis of the previous trend showed a significant decline in eelgrass biomass over 2007–2023 in Izembek Lagoon (slope = –7.91, 95% CI = –14.80, –1.02; $P = 0.03$) and a nonsignificant decline in Kinzarof Lagoon (slope = –8.36, 95% CI = –17.56, 0.84; $P = 0.09$; Figure 2). The estimated slope for Izembek Lagoon equated to a –3.9% per year decline relative to the first year in the dataset (2007). The estimated decline at Kinzarof Lagoon was also –3.9% per year, but was not significantly detected with only 5 yr of sampling over the 17-yr period.

Our simulation-based power analysis showed that the current study design (~100 points per year at Izembek and 50 at Kinzarof within the eelgrass polygon for each lagoon) did not meet our goal of an 80% chance of detecting a 25% decline in eelgrass biomass-when-present over 5 yr (Figures 3a and 3d). However, we found a ≥80% chance of detecting a decline of the same magnitude (–5.6% per year) after 10 yr at either lagoon if 100 points were monitored

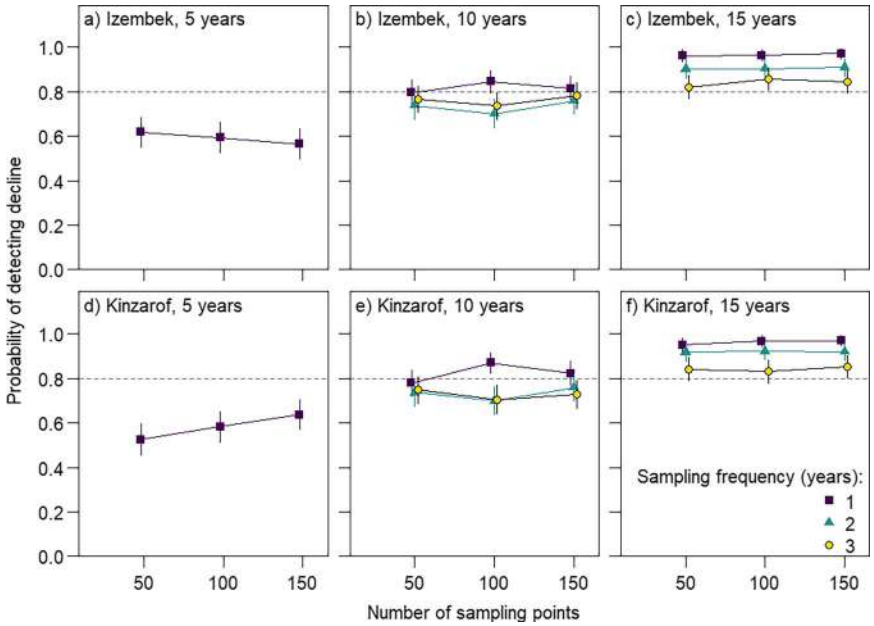

Figure 3. Predicted probability of detecting a decline in eelgrass biomass for each lagoon (Izembek: a–c; Kinzarof: d–f) under various sampling schemes: 50, 100, or 150 points sampled per year (horizontal axis) with sampling occurring every 1, 2, or 3 yr (lines) over 5, 10, or 15 yr (panels). In every case, the simulated decline was –5.6% per year (–25% every 5 yr) and a dashed horizontal reference line is shown at the power target of 0.80 (80% chance of detecting the decline). The left-hand column (a, d) shows only one line per lagoon because sampling at a frequency of less than annually would not provide robust estimates of a trend over 5 yr, so we did not test those scenarios.

per lagoon per year within the eelgrass polygon (Figures 3b and 3c), or after 15 yr with monitoring conducted at least every third year (Figures 3c and 3f). The number of years was much more influential than the number of points, agreeing with previous work showing that the power to detect a trend over time typically depends strongly on the number of years in the sample (Thogmartin et al. 2007, Urquhart 2012). The fundamental sampling unit in a trend analysis is the year rather than samples within a year, and thus interannual variation has a strong influence, with higher variation introducing more noise that hinders detection of a trend. In our dataset, the standard deviation across years was equivalent to 11.6% of the mean value, whereas the trend we modeled was only –5.6% per year. Thus, the noise introduced by interannual variation often masked the trend in our simulations, especially when fewer years were included.

The previous field dataset included ∼100 points sampled in 12 of 17 yr at Izembek, which is intermediate between the 1- and 2-yr sampling frequency that we simulated in our power analysis (Figure 3c). Our model of the previous trend detected a statistically significant decline of –3.9% per year at Izembek, in agreement with the power analysis that a stronger trend (–5.6% per year) could be easily detected with a similar dataset over 17 yr. In contrast, the previous dataset for Kinzarof sampled ∼50 points about every 3 yr over 17 yr, which is roughly equivalent to the 3-yr sampling frequency in Figure 3f. As shown in the figure, our power analysis indicated that a –5.6% trend would usually be detected with that sampling design, but it is unsurprising that a weaker trend (–3.9% in the field dataset) would not be significantly detected.

Our results were similar between the two lagoons (Figure 3), but this similarity could be attributed to our use of Izembek's interannual variation for both lagoons due to the limited time-series of field data available for Kinzarof. If further monitoring reveals that Kinzarof has lower or higher interannual variation than Izembek, power to detect a trend may be higher or lower (respectively) than shown here. The two lagoons are in close proximity but are not part of the same oceanic system (Izembek opens to the Bering Sea, while Kinzarof opens to the Gulf of Alaska/Pacific Ocean), so differing ocean-influenced conditions such as water temperature or salinity could result in different magnitudes of interannual variation between the two lagoons. Local factors such as ice scouring or grazing by herbivores could also potentially manifest differently in the two lagoons, thus differently affecting their rates of interannual variation. After a few more years monitoring at Kinzarof, recalculating interannual variation would be useful to evaluate whether the results of this power analysis will be accurate for Kinzarof. However, based on our results, it would still be unlikely that a decline of –5.6% per year could be detected at Kinzarof in less than 10 yr.

The number of points sampled had little influence on the power to detect a trend in this study, with 50 points generally performing nearly as well as 150 points (Figure 3). However, this analysis evaluated only the goal of detecting a decline in biomass-when-present, and thus did not consider any changes in occupancy (i.e., where eelgrass is present). If local (point-scale) occupancy is of interest, such as to evaluate localized loss of eelgrass due to ice scouring, continuing to monitor the current sample size of ∼100 points in Izembek and ∼50 in Kinzarof could be useful. In addition to tracking point-scale occupancy, monitoring at least 100 and 50 points (respectively) could be useful to evaluate lagoon-scale changes in occupancy if satellite imagery is not consistently available. In other parts of the world, both point-scale occupancy and the extent of eelgrass meadows can change substantially from year to year (Balsby et al. 2017, Bartenfelder et al. 2022, Munsch et al. 2023), suggesting that monitoring occupancy would be useful in this system and could justify continuing to sample extensively across both lagoons.

In our power analysis, we assumed that monitoring was conducted throughout the extent of the eelgrass meadows in each lagoon, and that no sampling was conducted in areas where eelgrass was absent. If this sampling frame is accurately designated, it provides an efficient way to monitor each lagoon's total eelgrass biomass without wasting effort on sampling points where eelgrass is unlikely to occur. However, if the eelgrass extent changes in the future, adjusting the sampling frame would be necessary to continue ensuring that the lagoons are adequately and efficiently sampled. When new satellite images become available in future years, the remote-sensing process (Douglas et al. 2024) could be repeated to update the estimated area in which eelgrass is present. The area for field sampling could then be updated accordingly. After the sampling frame changes, an evaluation of trend across years with different sampling extents could be conducted by using the sum total of eelgrass biomass present in each lagoon (biomass in g per m^2 × area of the current eelgrass extent in m^2) in each year. This sum total, and any trend therein, would be relevant to management of other species of interest because it would indicate the total amount of eelgrass available to those species.

While eelgrass is not currently directly managed in Izembek or Kinzarof lagoons, it is of management interest due to its role in sustaining populations of Pacific black brant and other waterfowl (Ward et al. 2005). The monitoring program is therefore intended to detect any declines in eelgrass abundance and distribution that could be caused by changing conditions. Eelgrass occurrence and biomass in these lagoons can be affected by ice scouring, storms (wave action), water temperature, light intensity, and water depth (McRoy 1966, 1970, Ward and Amundson 2019). Future human development or other habitat change could also potentially affect eelgrass extent or biomass in the lagoons. Numbers of brant overwintering in both lagoons have also increased in recent years, as ice cover in the lagoons has decreased, potentially changing herbivory pressure on eelgrass over winter (Ward et al. 2009, Stillman et al. 2021, U.S. Fish and Wildlife Service 2022). While grazing can stimulate growth of eelgrass (Shaughnessy et al. 2021), it is unclear how grazing may affect eelgrass biomass at high latitudes in winter, when eelgrass is not growing, thus providing another potential avenue for changing eelgrass biomass. The waterfowl that use Izembek and Kinzarof lagoons are harvested, and thus any documented decline in eelgrass at their primary fall staging site could inform harvest limits

and other management decisions for brant or other waterfowl (Olson 2019). Additionally, if the monitoring program reveals declines in eelgrass that are of management concern, restoration of eelgrass beds could be pursued, perhaps through methods employed at lower latitudes (e.g., transplantation and seeding; Tanner et al. 2010, Ward and Beheshti 2023).

In conclusion, our power analysis indicates that the current monitoring scheme at Izembek and Kinzarof lagoons is unlikely to meet our goal of an 80% chance of detecting a 25% decline in eelgrass biomass over 5 yr. However, a longer time series would likely detect the same rate of decline (–5.6% per year). Our results suggest that additional sampling per year would not help to meet the goal of monitoring trends, but findings elsewhere of variable eelgrass occupancy and extent would support continuing to monitor the current samples of ∼100 (Izembek) and ∼50 (Kinzarof) points. The improvement in power with a longer time series highlights the utility of long-term monitoring of natural resources, particularly when interannual variation is large. Continued monitoring of eelgrass, brant populations (e.g., Ward et al. 2009), and factors that contributed to healthy eelgrass ecosystems (e.g., water temperature) in this area would help inform management of eelgrass itself or of other species of interest, particularly with any continuing environmental or ecological changes and increasing numbers of overwintering black brant.

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LITERATURE CITED

Balsby TJS, Clausen P, Krause-Jensen D, Carstensen J, Madsen J. 2017. Long-term patterns of eelgrass (*Zostera marina*) occurrence and associated herbivorous waterbirds in a Danish coastal inlet. *Front. Mar. Sci.* 3:285. https://doi.org/10.3389/fmars.2016.00285.

Bartenfelder A, Kenworthy WJ, Puckett B, Deaton C, Jarvis JC. 2022. The abundance and persistence of temperate and tropical seagrasses at their edge-of-range in the western Atlantic Ocean. Front. Mar. Sci. 9.

Christiaen B, Gaeckle J, Ferrier L. 2018. Eelgrass abundance and depth distribution in East Kitsap. Final report to the Suquamish Tribe, DNR IAA 15-17 Amendment 1. https://www.dnr.wa.gov/publications/aqr__nrsh__east__kitsap__report__2018.pdf. Accessed November 21, 2024.

de Valpine P, Turek D, Paciorek CJ, Anderson-Bergman C, Lang DT, Bodik R. 2017. Programming with models: Writing statistical algorithms for general model structures with NIMBLE. J. Comput. Graph. Stat. 26:403–413.

Douglas DC, Fleming MD, Patil V, Ward DH. 2024. Mapping eelgrass cover and biomass at Izembek Lagoon, Alaska, using in-situ field data and Sentinel-2 satellite imagery. USGS Open File Report, Anchorage, Alaska. 35 pp.

Gelman A, Rubin DB. 1992. Inference from iterative simulation using multiple sequences. Stat. Sci. 7:457–472.

Green EP, Short FT. 2003. World atlas of seagrasses. University of California Press, Berkeley. 310 pp.

Harris PM, Neff AD, Johnson SW. 2012. Changes in eelgrass habitat and faunal assemblages associated with coastal development in Juneau, Alaska. U.S. Department of Commerce, NOAA Tech. Memo. NMFS-AFSC-240. 47 p.

Hogrefe KR, Ward DH, Donnelly TF, Dau N. 2014. Establishing a baseline for regional scale monitoring of eelgrass (*Zostera marina*) habitat on the lower Alaska Peninsula. Remote Sens. 6:12447–12477.

Holland GJ, Alexander JSA, Johnson P, Arnold AH, Halley M, Bennett AF. 2012. Conservation cornerstones: Capitalising on the endeavours of long-term monitoring projects. Biol. Conserv. 145:95–101.

Johnson SW, Thedinga JF. 2005. Fish use and size of eelgrass meadows in southeastern Alaska: A baseline for long-term assessment of biotic change. NW Sci. 79:141–155.

McRoy CP. 1966. Standing stocks and ecology of eelgrass (*Zostera marina*) at Izembek Lagoon, Alaska. M.S. thesis. University of Washington, Seattle.

McRoy CP. 1970. On the biology of eelgrass, *Zostera marina*, Izembek Lagoon, Alaska. Ph.D. dissertation. University of Alaska, Fairbanks.

Munsch SH, Beaty FL, Beheshti KM, Chesney WB, Endris CA, Gerwing TG, Hessing-Lewis M, Kiffney PM, O'Leary JK, Reshitnyk L, Sanderson BL, Walter RK. 2023. Northeast Pacific eelgrass dynamics: Interannual expansion distances and meadow area variation over time. Mar. Ecol. Prog. Series 705:61–75.

Murray DL, Krebs CJ. 2024. Editorial: Long-term monitoring in ecology and evolution: Establishing a sound baseline to help inform our future. Front. Ecol. Evol. 12.

Olson SM. 2019. Pacific Flyway data book, 2019. U.S. Department of Interior, Fish and Wildlife Service, Division of Migratory Bird Management, Vancouver, Washington.

Patil VP. 2024. Eelgrass biomass model (ver 1.0.1, May 2024). U.S. Geological Survey software release. https://doi.org/10.5066/P13EG9KS.

Plummer M. 2003. JAGS: A program for analysis of Bayesian graphical models using Gibbs sampling. pp. 1–10. In: K. Hornik, F. Leisch, and A. Zeileis (eds.). Proceedings of the 3rd International Workshop on Distributed Statistical Computing (DSC 2003). R Foundation for Statistical Computing, Vienna, Austria.

Plummer ML, Harvey CJ, Anderson LE, Guerry AD, Ruckelshaus MH. 2013. The role of eelgrass in marine community interactions and ecosystem services: Results from ecosystem-scale food web models. Ecosystems 16:237–251.

Shaughnessy FJ, Ferson SL, Frimodig AJ, Barton DC, Hurst M, Black JM. 2021. Growth and flowering responses of eelgrass to simulated grazing and fecal addition by Brant Geese. Ecosphere 12.

Sherman K, DeBruyckere LA. 2018. Eelgrass habitats on the U.S. West Coast. State of the knowledge of eelgrass ecosystem services and eelgrass extent. Pacific Marine and Estuarine Fish Habitat Partnership for The Nature Conservancy. 67 pp.

Short FT, Wyllie-Eciieverria S. 1996. Natural and human-induced disturbance of seagrasses. Environ. Conserv. 23:17–27.

Stillman RA, Rivers EM, Gilkerson W, Wood KA, Nolet BA, Clausen P, Wilson HM, Ward DH. 2021. Predicting impacts of food competition, climate, and disturbance on a long-distance migratory herbivore. Ecosphere 12.

Tanner C, Hunter S, Reel J, Parham T, Naylor M, Karrh L, Busch K, Golden RR, Lewandowski M, Rybicki N, Schenk E. 2010. Evaluating a large-scale eelgrass restoration project in the Chesapeake Bay. Restor. Ecol. 18:538–548.

Thogmartin WE, Gray BR, Gallagher M, Young N, Rohweder JJ, Knutson MG. 2007. Power to detect trend in short-term time series of bird abundance. The Condor 109:943–948.

Urquhart NS. 2012. The role of monitoring design in detecting trend in long-term ecological monitoring studies, pp. 151–173. In: R. A. Gitzen, J. J. Millspaugh, A. B. Cooper, and D. S. Licht (eds.). Design and analysis

of long-term ecological monitoring studies. Cambridge University Press, Cambridge, UK.
U.S. Fish and Wildlife Service. 2022. Waterfowl population status, 2022. U.S. Department of the Interior, Washington, DC. https://www.fws.gov/library/collections/waterfowl-population-status-reports. Accessed November 21, 2024.
Ward DH. 2021. Point sampling data for eelgrass (*Zostera marina*) and seaweed distribution and abundance in bays adjacent to the Izembek National Wildlife Refuge, Alaska (ver 4.0, June 2024). U.S. Geological Survey data release. https://doi.org/10.5066/P9ZUDIOH.
Ward DH, Amundson CL. 2019. Monitoring annual trends in abundance of eelgrass (*Zostera marina*) at Izembek National Wildlife Refuge, Alaska, 2018. U.S. Geological Survey Open-File Report 2019-1042. 8 p. https://doi.org/10.3133/ofr20191042.
Ward DH, Dau CP, Lee T, Sedinger JS, Anderson BA, Hines JE. 2009. Change in abundance of Pacific brant wintering in Alaska: Evidence of a climate warming effect? Arctic 62:301–311.
Ward DH, Hogrefe KR. 2022. Mapping data of eelgrass (*Zostera marina*) distribution, Alaska and Baja California, Mexico. U.S. Geological Survey data release. https://doi.org/10.5066/P9WEK4JI.
Ward DH, Hogrefe KR, Swaim KR, Donnelly TF, Fairchild LL. 2022. Eelgrass (*Zostera marina*) and seaweed abundance along the Coast of Togiak National Wildlife Refuge, Alaska, 2008–10. U.S. Geological Survey Open-File Report 2020–1114. 14 p. https://doi.org/10.3133/ofr20201114.
Ward DH, Markon CJ, Douglas DC. 1997. Distribution and stability of eelgrass beds at Izembek Lagoon, Alaska. Aquat. Bot. 58:229–240.
Ward DH, Morton A, Tibbitts TL, Douglas DC, Carrera-González E. 2003. Long-term change in eelgrass distribution at Bahia San Quintin, Baja California, Mexico, using satellite imagery. Estuaries 26:1529–1539.
Ward DH, Reed A, Sedinger JS, Black JM, Derksen DV, Castelli PM. 2005. North American Brant: Effects of changes in habitat and climate on population dynamics. Global Change Biol. 11:869–880.
Ward M, Beheshti K. 2023. Lessons learned from over thirty years of eelgrass restoration on the US West Coast. Ecosphere 14.
Weiser EL. 2025. Scripts to evaluate trend and conduct a power analysis for monitoring eelgrass at Izembek and Kinzarof lagoons, Alaska. U.S. Geological Survey software release. https://doi.org/10.5066/P1PHZSE.
Wood KR, Bond NA, Danielson SL, Overland JE, Salo SA, Stabeno PJ, Whitefield J. 2015. A decade of environmental change in the Pacific Arctic region. Prog. Oceanogr. 136:12–31.
Wyllie-Echeverria S, Ackerman JD. 2003. The seagrasses of the Pacific coast of North America, pp. 199–206. In: E. Green and F. Short (eds.). World atlas of seagrasses. 2nd ed. UNEP World Conservation Monitoring Centre, University of California Press, Berkeley.