SAN FRANCISCO BAY
BIRD OBSERVATORY

Phalarope Migration Surveys
June – September 2024

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Executive Summary

The South Bay Salt Pond Restoration Project (SBSPRP) is restoring over 15,000 acres of former salt evaporation ponds to a mix of tidal marsh and ponded wetland habitats. These wetlands provide habitat for many waterbirds, including migrating Red-necked Phalarope (*Phalaropus lobatus*) and Wilson's phalarope (*P. tricolor*). Sustaining baseline population goals for wildlife populations requires understanding how species are responding to restoration actions over time. While many waterbird guilds have increased in abundance from pre-restoration baselines, phalarope numbers have declined >50% below baseline numbers. The purpose of this ongoing study is to understand observed declines in phalarope numbers within the SBSPRP area and their relationship with broader population trends and phalarope movements. This report serves as a data summary and coarse-scale assessment of phalarope monitoring efforts in the South San Francisco Bay during the 2024 migration.

From June 17, 2024 to September 26, 2024, we conducted 8 rounds of phalarope migration surveys at 32 sites (18 SBSPRP managed ponds, 7 Cargill-managed salt production ponds, and 7 sites outside of these areas). The surveys found that counts of Wilson's Phalaropes peaked at 787 phalaropes on 07/16 and counts of Red-necked Phalaropes peaked at 1,082 phalaropes on 09/12. Across all sites, a total of 4,835 phalaropes were counted throughout the summer (2433 Red-necked, 1,352 Wilson's, and 1,050 that could not be identified to species). Compared to last year, this was -11,819 fewer sightings of phalaropes (-11,634 Red-necked, -286 Wilson's, and +101 that could not be identified to species).

Looking only at the 25 sites within SBSPRP area, a total of 3,161 phalaropes were counted throughout the survey season (1938 red-necked, 1,157 Wilson's, and 66 that could not be identified to species). Within only the SBSPRP and salt pond complexes from the early July through end of September surveys, the average number of sightings of phalaropes per survey was 448. This was -91.6% below the revised 2005-2007 baseline of 5,324, and counts therefore remain below the Adaptive Management Plan trigger threshold of a decline of 50% or more below baseline values.

We gathered and analyzed foraging data for the first time this year and assessed how salinity affected foraging behavior. Based on a literature review on phalarope prey ecology, we designed six new salinity bins: near-fresh, brackish, saltwater, and low, medium, and high hypersaline. The overwhelming majority of observations of Red-necked and Wilson's Phalaropes were of foraging, rather than roosting, birds. Phalaropes demonstrated a strong association with a relatively wide band of moderately high salinity (30-150 ppt), which closely aligned with the reported tolerance of *Ephydra* and *Artemia* spp. reported in the literature. Surprisingly, Wilson's Phalaropes were most commonly observed at sites with salinity matching the marine environment (30-50 ppt), despite their tight association with hypersaline lakes.

Behavioral observations showed that brackish (2-30 ppt) and marine (30-50 ppt) ponds had significantly fewer foraging attempts and successful prey captures. Outside of low hypersaline ponds (50-80 ppt) appeared to provide the best habitat, with medium hypersaline ponds (80-150 ppt) trending towards lower numbers of successful prey captures and high hypersaline ponds (>150 ppt) having significantly lower foraging attempts. Sunnyvale WPCP, the only near-freshwater (0-2 ppt) pond used by phalaropes, was an outlier site not comparable to most ponds in the South Bay because it is a sewage treatment pond with unnaturally high nutrients and had notably higher rates of foraging attempts and successful prey captures.

Continuing observations of phalaropes outside of the original set of 31 sites indicate the current spatial coverage of sampling may be insufficient. We reanalyzed historical count data from the entire 2002-2024 record of waterbird surveys conducted for the SBSPRP, and identified 17 new sites (in addition to one that was already added, E5C) that are potentially important habitat. While some of these sites, such as those that have been breached to tidal action, are likely no longer suitable, others appear likely to remain habitat. We identify six sites in particular that are high-priority for adding to the set of surveys.

Introduction

In 2003, the U.S. Fish and Wildlife Service (USFWS) and California Department of Fish and Wildlife (CDFW, formerly California Department of Fish and Game) entered into a historic agreement with Cargill Salt to acquire 15,100 acres of salt evaporator ponds in the South San Francisco Bay (South Bay). The South Bay Salt Pond Restoration Project (SBSPRP) has begun to restore the area to a mix of tidal and ponded habitats while continuing to provide flood protection and improved public access to many sites.

Salt ponds have been present in the San Francisco Bay for over 150 years (Ver Planck 1958) and have significant wildlife value (Anderson 1970, Accurso 1992, Takekawa et al. 2001, Warnock et al. 2002). Due to the loss of wetlands elsewhere, the ponds now provide important foraging and roosting areas for many waterbirds. As a major migratory and wintering location along the Pacific Flyway, the San Francisco Bay supports more than a million birds throughout the year (Page et al. 1999, Warnock et al. 2002). The SBSPRP has committed to restoring some ponds to tidal marsh, while retaining some pond habitat (as managed ponds) within the project area for waterbirds. Today, the South Bay is a mix of these former salt ponds now managed for wildlife or restored to tidal marsh, ponds still operated for salt production by Cargill Salt, and other pond and wetland habitats scattered throughout the baylands.

More than a decade of waterbird counts show that many waterbird guilds increased in abundance relative to SBSPRP baselines, which were established prior to implementation of project restoration and enhancement actions (Tarjan 2019a). However, combined counts of two phalarope species, Red-necked Phalarope (*Phalaropus lobatus*) and Wilson's Phalarope (*P. tricolor*) showed a decline of 78% from prior to the start of restoration project activities by 2017 (Tarjan 2019a). This decline has been driven by declines in both Wilson's Phalarope and Red-necked Phalarope (Fig. 1; Van Schmidt & Parsons 2024). Less frequent summer salt pond surveys during phalarope migration impeded our ability to capture phalarope use of the SBSPRP area. NEPA/CEQA significance thresholds require that a decline is attributed to restoration activities (South Bay Salt Pond Restoration Project, 2007). Understanding phalarope trends within the SBSPRP and how they relate to broader population trends requires targeted surveys during the peak phalarope season, supplemented with evaluation of external datasets that may offer insight into broader regional trends and explanatory factors such as habitat quality and SBSPRP management changes. In accordance with these recommendations, SFBBO piloted two surveys during peak phalarope migration in 2019. In 2020, due to Covid-19 safety requirements, we conducted surveys across the full range of seven dates at only a subset of sites. We conducted surveys at all sites on all dates during years 2021–2024.

This report summarizes the results of SFBBO's phalarope surveys in the South San Francisco Bay from 2019 through 2024. We applied the change model of Van Schmidt & Parsons (2023) to these data to analyze whether site-scale habitat characteristics or annual climate could explain observed changes in phalarope population sizes. New this year, we also conducted behavioral observations on foraging to help understand how prey ecology may relate to observed trends and spatial patterns of habitat use. This was coupled with a new analysis of phalarope-salinity relationships. To shed light on historical trends and large detections of phalaropes outside the core set of survey sites, we also reanalyzed data from the historical salt pond surveys.

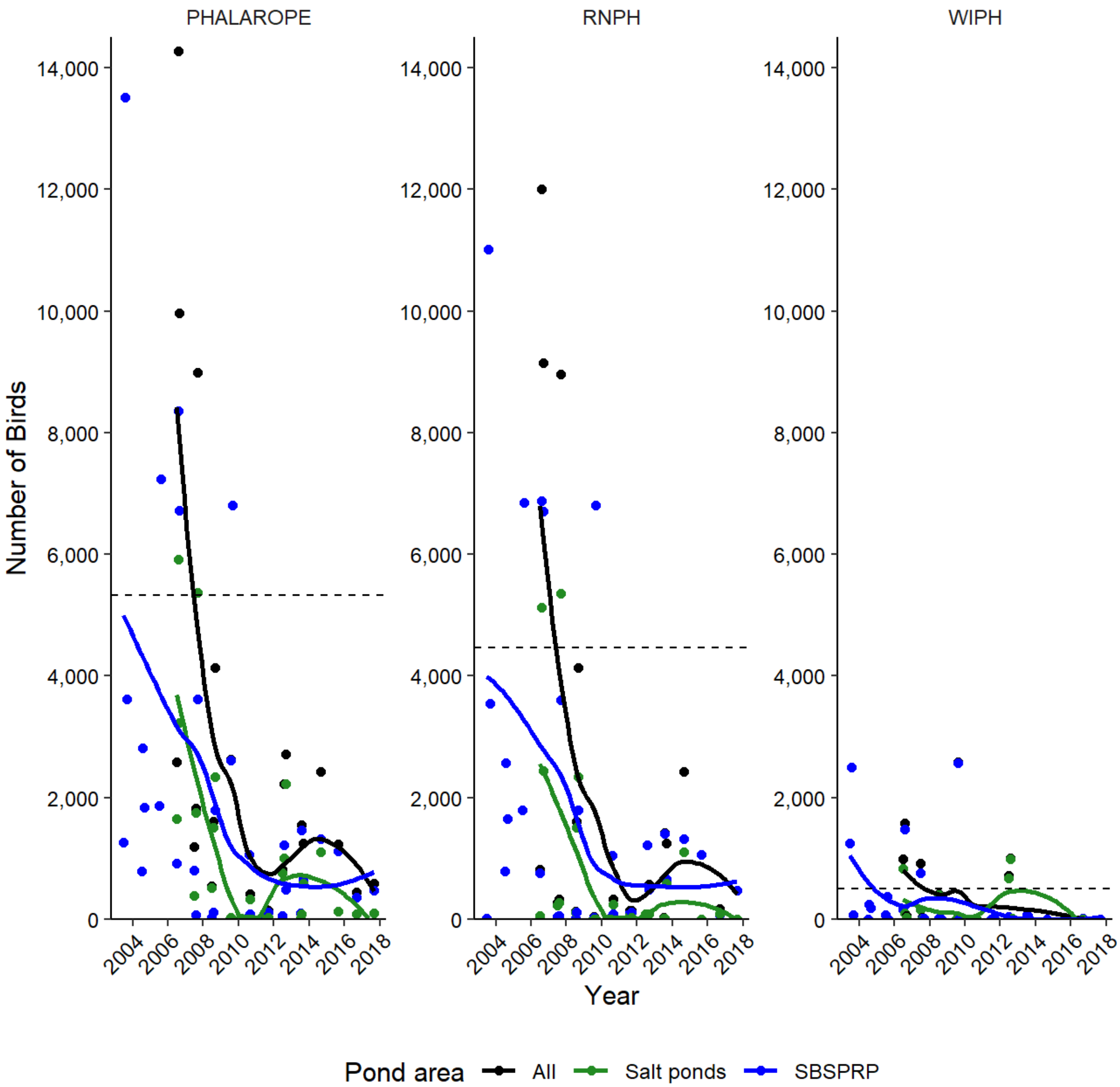


Figure 1. Counts of total phalaropes, Red-necked Phalarope, and Wilson's Phalarope during summer migration (July-September) within the South Bay Salt Pond Restoration Project (SBSPRP) and salt production ponds of south San Francisco Bay, California. Lines represent LOESS regression curves and the dashed lines denote recommended new SBSPRP targets based on recalculated baseline values (average counts from 2005–2007; Burns & Van Schmidt 2023). Republished from Van Schmidt & Parsons (2024).

Methods

Study Area

The study area for SFBBO's annual waterbird monitoring for SBSPRP includes 82 current and former salt ponds in the Santa Clara, Alameda, and San Mateo counties of California. Coyote Hills, Dumbarton, and Mowry salt ponds are owned by Don Edwards San Francisco Bay National Wildlife Refuge and managed for salt production by Cargill Salt. Alviso and Ravenswood complexes are owned and managed by Don Edwards San Francisco Bay National Wildlife Refuge. Eden Landing Ecological Reserve (Eden Landing) ponds are owned and managed by California Department of Fish and Wildlife (CDFW).

Through an analysis of the existing dataset and eBird observations (Tarjan 2019b), SFBBO identified a set of sites that covered 99% of phalarope observations within the South Bay from 2014-2018. We found that Wilson's phalarope exhibited one peak that clustered in late summer, whereas Red-necked Phalarope showed one peak in spring and were present for a more prolonged period in late summer to early fall. Both species showed similar preferences for select sites, suggesting that surveys could target specific areas. Based on these results, SFBBO made the following recommendations: 1) Capturing the peak migration for the two common phalarope species in San Francisco Bay requires conducting two sets of surveys, one set centered around July 17 for Wilson's phalarope and one set centered around August 24 for Red-necked Phalarope; and 2) Surveys of 24 SBSPRP and Cargill-managed ponds are likely to capture >95% of phalaropes on SBSPRP ponds. eBird sightings suggest adding 7 sites outside the set of traditional SBSPRP survey area to the phalarope surveys. Sites outside of the SBSPRP footprint are owned by various other entities, including City of Sunnyvale (Sunnyvale Water Pollution Control Plant ponds) and East Bay Regional Parks District (Coyote Hills Park).

Phalarope migration surveys were conducted at 32 of these ponds and additional sites outside of the SBSPRP footprint (Fig. 2). The current and former salt ponds surveyed include 4 ponds in the Alviso complex, 1 pond in the Coyote Hills complex, 3 ponds in the Dumbarton complex, 12 ponds in the Eden Landing complex, 3 ponds in the Mowry complex, and 3 ponds in the Ravenswood complex. Seven sites outside of the project were surveyed: Coyote Hills Regional Park, Crittenden Marsh, Northeast of N1, New Chicago Marsh, Spreckles Marsh, Alviso Marina, and the Sunnyvale Water Pollution Control Plant.

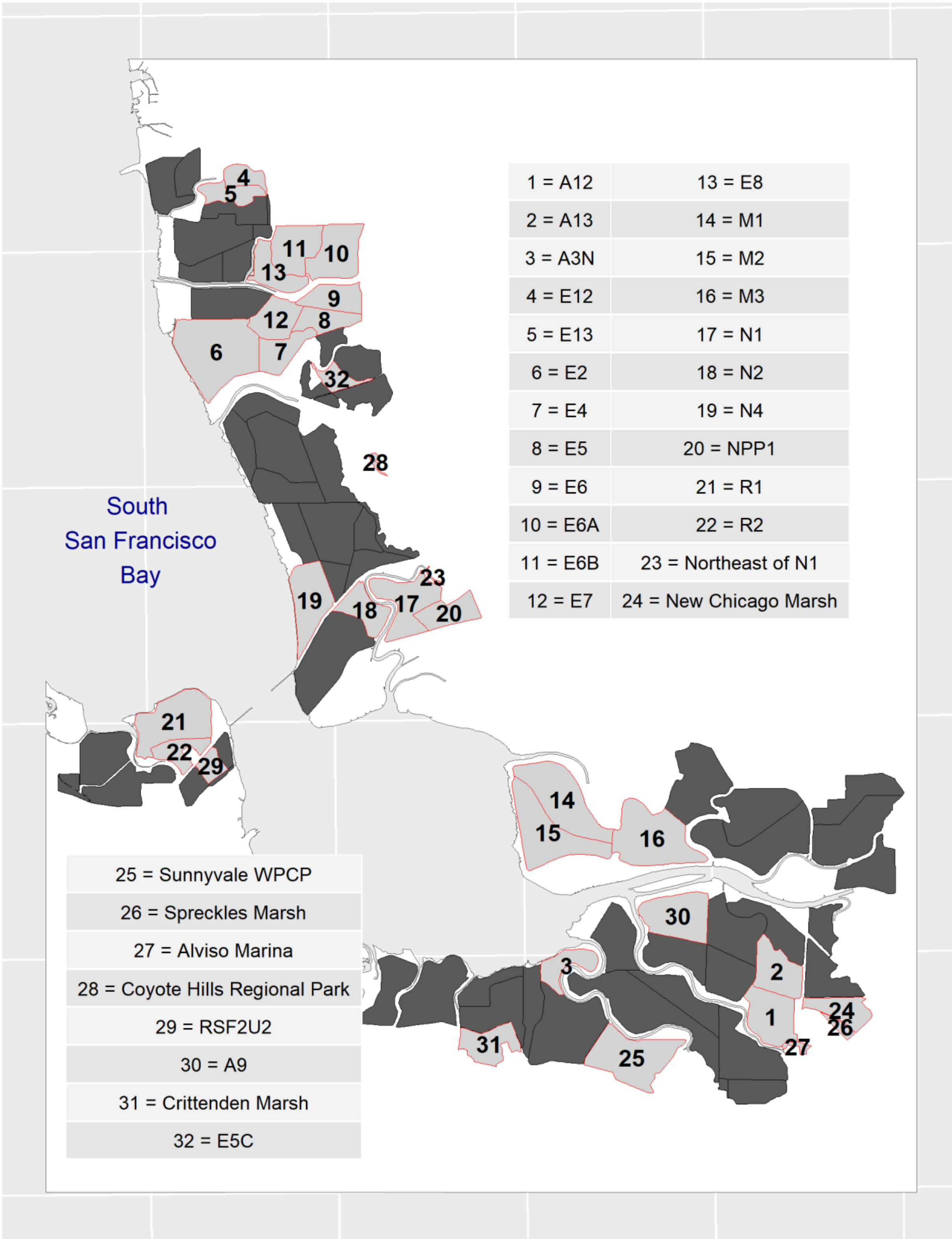


Figure 2. Map of target sites for Phalarope migration surveys.

Field Data Collection

Phalarope Migration Surveys

We conducted 8 surveys in 2024 during the season of peak phalarope migration. Surveys occurred every two weeks with the following target dates: 06/18, 07/02, 07/16, 07/30, 08/13, 08/29, 09/12, 09/24 (Table 1). Surveyors did not complete at E8 and Alviso Marina during the first survey period due to a communication error, but surveys have never detected phalaropes at these sites before the mid-July survey window, so this error is unlikely to have significantly impacted counts. Counts were collected by a combination of SFBBO staff and trained community scientist volunteers to allow all sites to be surveyed synchronously or near synchronously. Surveyors were trained to identify Red-necked Phalaropes, Wilson's Phalaropes, and Red Phalaropes (*P. fulicarius*). However, Red Phalaropes are rare in South San Francisco Bay and were not observed; they are therefore not covered by this report.

During each survey period, all sites were surveyed as close in time as possible, with most being surveyed simultaneously on the morning of the Tuesday of each observation window. Surveyors identified and counted phalaropes at each site from the nearest accessible levee using spotting scopes. Data included species counts, date, survey start time, survey end time, an estimate of the percent of the site that was visible to the observer, and an estimate of the percent of the site that was covered in water. In order to ensure we do not miss sites that are newly colonized by phalaropes as habitats change, we also encouraged surveyors to scan other nearby ponds for phalaropes and collected observations from agency staff. We report significant incidental sightings in this report, but excluded them from counts to allow for comparison with previous years and because these ponds were not surveyed in all time periods.

Foraging Observations

This year, we also began to collect foraging data. Species counts were divided into the number of birds that were foraging versus roosting. Surveyors were trained using videos and heuristics on how to estimate distances on the ponds and identify foraging behaviors of phalaropes. Foraging was defined as spinning behaviors, or pecking or probing at the pond surface (Frank & Conover 2021).

Once a flock was found at the site, behavioral observations were recorded. Surveyors picked a random phalarope <100 m from the observer; if none were present, birds up to 200 m away could be selected. The estimated distance to the bird was recorded based on visual estimation to the nearest five meters. Observations of birds that were too far to observe by species or that were estimated at >200 m were discarded from the foraging analysis. Once the bird was selected, the surveyor watched the bird for one minute and used two clickers to record (1) the total number of foraging attempts (all pecks at the open water surface) and (2) the number of successful foraging attempts (pecks followed by beak-opening as described in Rubega & Obst 1993, or a swallowing motion where the bird rapidly jerked its head back-and-forth). Birds sometimes foraged by swishing their bill from side to side for a moment, which was counted as a single peck.

At some sites, phalaropes were observed foraging too rapidly and continuously to count (>2 attempts/second), though high-speed video showed that the birds were successfully foraging at these sites. In these instances, a separate "continuous foraging" protocol was used. Rather than counting individual pecks, surveyors used a stopwatch to record the number of seconds a bird was actively foraging during the one-minute survey.

After the one-minute observation was complete for either protocol, surveyors selected another random bird and repeated the process until ten observations of each species were gathered at each site, or until there were no more phalaropes close enough to observe. In some instances, observers recorded >10 observations of a single species; in these instances, we retained these extra observations in analyses in order to include as much data on foraging rates as possible.

Historical Surveys

The current survey effort began in August 2019 as a pilot program, with only two surveys in August of that year (Table 1). The first full round of surveys was conducted in 2020, with six surveys spanning from late June/early July to the end of September, though due to COVID-19 restrictions surveys were not conducted at ponds within the Alviso and Ravenswood pond complexes in Don Edwards NWR. Beginning in 2022, an additional mid-June survey window was added to ensure surveys caught the leading edge of Wilson's Phalarope migration, which had peaked in 2021 in what was previously the first late June/early July survey window (Table 1).

We also report analyses derived from historical salt pond surveys, which are described more fully in Van Schmidt & Parsons (2025). Briefly, ponds were surveyed monthly from roughly 2002-2013. Beginning in 2014, there was a period of more intermittent surveys, with survey frequency reduced to twice per quarter and only once during the summer, with the summer survey dropped altogether beginning in 2019. There were additional intermittent gaps in some survey rounds due to funding and/or COVID 19 restrictions (see Van Schmidt & Parsons 2025 for details).

Water Quality Sampling

We sampled water quality at all survey sites (including sites outside of the SBSPRP and Cargill ponds) during summer (July 2–July 9) and in early fall (September 1–October 12). The water sampling did not necessarily occur on the same day as the phalarope counts. We followed water quality monitoring methods outlined by Murphy et al. (2007). Water quality samples were collected from the surface of the water (depth of <0.5 m). Within non-SBSPRP sites, only a single water quality sample was taken, but within SBSPRP sites multiple points were generally sampled in accordance with established sampling points (Van Schmidt & Parsons 2024). The number of sampling points per pond varied based on the size, configuration, and accessibility of the pond. Some sampling points could also temporarily not be reached due to low water levels within the pond during some surveys. Whenever possible, water quality data was collected on the day of the bird survey, but otherwise was collected as close to the date of the bird survey as possible. We recorded dissolved oxygen (mg/L), salinity (ppt), conductivity (mS), pH, temperature (°C), and barometric pressure (not used as a water quality measure) at 1-4 pre-determined sampling sites at each pond using a Hanna HI98494 Sonde (Hanna Instruments Inc, Woonsocket, RI). When salinities exceeded approximately 70 ppt (the maximum value registered by the Hanna HI98494 Sonde), we calculated salinity using a hydrometer (Ertco, West Paterson, NJ) to measure specific gravity in combination with a temperature reading from the water sample. We calibrated the LDO sensor before the start of each sampling day, and conductivity and pH sensors at the beginning of each week of sampling.

During each survey we recorded water levels by reading the water level on staff gauges (if present). In ponds with multiple staff gauges, we recorded only the master staff gauge (indicated by a circle of yellow paint on the gauge post). Observers also visually estimated the proportion of any pond substrate exposed to the air (dry pond bottom or mudflat exposed) to provide a finer-scale characterization of habitat variability.

Table 1. Schedule for phalarope migration surveys. Sites = the number of sites visited during each survey, where each site was visited once between the start and end dates. RNPH = number of Red-necked Phalaropes; WIPH = number of Wilson's phalaropes; XXPH = number of phalaropes that could not be identified to species. No red phalaropes were observed.

Survey ID	Survey period	Year	Start	End	Sites	RNPH	WIPH	XXPH	Total
1	Mid Aug	2019	08/15	08/20	21	1447	251	2	1700
2	Late Aug	2019	08/26	09/01	22	1063	114	1	1178
3	Early July	2020	07/06	07/09	15	1	548	0	549
4	Mid July	2020	07/20	07/23	17	182	767	140	1089
5	Early Aug	2020	08/04	08/06	15	758	446	5	1209
6	Mid Aug	2020	08/18	08/20	15	904	162	0	1066
7	Late Aug	2020	08/31	09/01	14	1700	110	0	1810
8	Mid Sept	2020	09/15	09/16	13	935	0	32	967
9	Late Sept	2020	09/29	09/29	14	40	1	0	41
10	Early July	2021	07/06	07/07	31	35	738	16	789
11	Mid July	2021	07/19	07/20	31	313	12	414	739
12	Early Aug	2021	08/02	08/03	31	370	240	84	694
13	Mid Aug	2021	08/16	08/17	30	1196	24	3	1223
14	Late Aug	2021	08/30	08/31	31	2992	33	792	3817
15	Mid Sept	2021	09/13	09/14	31	6767	0	0	6767
16	Late Sept	2021	09/28	09/30	30	811	0	0	811
17	Late June	2022	06/20	06/21	29	0	18	0	18
18	Early July	2022	07/05	07/06	29	0	83	2	85
19	Mid July	2022	07/18	07/19	29	186	735	0	921
20	Early Aug	2022	08/01	08/02	29	87	10	0	97
21	Mid Aug	2022	08/15	08/16	31	1557	19	30	1606
22	Late Aug	2022	08/29	08/30	31	4357	220	0	4577
23	Mid Sept	2022	09/12	09/13	31	2882	15	875	3772
24	Late Sept	2022	09/27	09/27	30	1963	0	0	1963
25	Late June	2023	06/19	06/20	31	0	0	0	0
26	Early July	2023	07/03	07/05	29	0	177	0	177
27	Mid July	2023	07/17	07/19	29	157	1035	2	1194
28	Early Aug	2023	07/31	08/03	31	774	146	59	979
29	Mid Aug	2023	08/14	08/15	31	1669	136	200	2005
30	Late Aug	2023	08/28	09/01	31	6554	100	334	6988
31	Mid Sept	2023	09/11	09/12	31	3344	43	301	3688
32	Late Sept	2023	09/25	09/26	31	1565	1	53	1619
33	Late June	2024	06/17	06/19	30	6	44	0	50
34	Early July	2024	07/01	07/03	32	8	30	0	38
35	Mid July	2024	07/15	07/19	32	11	787	16	814
36	Early Aug	2024	07/29	07/30	32	31	243	110	384
37	Mid Aug	2024	08/13	08/13	32	71	182	484	737
38	Late Aug	2024	08/29	08/30	32	680	38	238	956
39	Mid Sept	2024	09/12	09/12	32	1082	28	82	1192
40	Late Sept	2024	09/24	09/26	32	544	0	120	664

Analysis

Data Summary

We calculated the total counts and peak counts for each species, and both combined. Counts were summarized across (1) the entire study area, (2) across just the footprint of the current and former salt ponds, (3) at specific sites, and (4) in each of a set of salinity bins. Loading and initial reprocessing of habitat variables was done in R v4.1.3 (x86) to permit pulling from Microsoft Access databases, and all other steps were done in R v4.3.1 (x64; R Development Core Team 2018). We visualized phalarope trends using the ggplot2 package (Wickham 2016) in R v4.3.1.

Data from the pilot season in 2019 were excluded from all analyses of historical data unless otherwise noted. We report sum sightings as the total across all surveys, regardless of the addition of the new late June/early July survey in 2024. While this may introduce a slight bias towards underrepresenting counts in 2020 and 2021, there are only 23 phalarope sightings during this survey on average, so the impact should be minimal.

Change Model

We iterated on a model relating static pond characteristics, changes in pond hydrology, and annual weather (Table 2) to changes in waterbird abundance in order to identify drivers of observed changes in waterbird populations (Van Schmidt & Parsons 2024). Because summer monitoring efforts are no longer conducted and other recent efforts have assessed changes in breeding waterbird habitat selection in salt ponds vs. tidal marsh (Hartman et al. 2021, Schacter et al. 2023), we assessed only non-breeding (migratory and wintering) populations of waterbirds within South San Francisco Bay. Phalaropes were included in our assessment, but it was based on summer records within the long-term SBSPRP monitoring dataset rather than the new phalarope migration surveys (Van Schmidt & Parsons 2024), which will be the subject of a different report. Least Terns were excluded because they had prohibitively low abundances, which made fitting complex models difficult, and because summer surveys of this target species are now covered by other recent studies at SFBBO (Schwarz et al., 2024).

Model Description

A detailed description of the model development process and methods is provided in Van Schmidt & Parsons (2024). Briefly, we modeled per-pond inter-annual change in abundance of water species or guilds with linear models (base package *stats*); unlike previous versions of this model, we did not include a random effect for year because, with only three years of change data, this resulted in unstable parameters estimates when coupled with annual weather variables. Because the annual weather variables were of greater importance to the goals of the modeling effort, we therefore dropped the random year effect. For each species/guild, we calculated abundance in a given season as the average count at that pond across all surveys during that season. We calculated response variable, percent change in abundance, as:

$$\%\ Change\ in\ Abundance = Ln(1 + ((Abundance[t] - Abundance[t\text{-}1]) / (Abundance[t\text{-}1] + 1)))\ (Eq.\ 1)$$

We took the natural log and added 1 to equalize the leverage of percent increases and percent decreases. Because this function is not defined for years where abundance in the first year is zero, we added an adjustment factor of one to the denominator to allow us to capture new colonizations, which could be important for capturing the dynamics of rare species. Zero percent change could indicate either a stable population, or an unoccupied site in both years (i.e., a likely unsuitable site). Because we sought explicitly to model drivers of observed changes in populations, not habitat selection, we removed all rows that were zero in both years from the dataset.

Table 2. Predictor variables used to predict change in seasonal mean waterbird abundance within current and former salt ponds of the South San Francisco Bay. "Base" stage variables were selected over in a first AICc model selection procedure, followed by "Main" stage covariates. "Change model" column indicates whether the variable varied was static across years, varied annually, or was modeled as the year-to-year change in this variable ("Change", with name suffix "_AC").

Stage	Group	Name	Description	Units	Change model
Base	Site characteristics	Area_km2	Pond area	km2	Static
		Islands_num	Number of islands	count	Static
		OpenPublic_pct	Maximum % of levees open to public across all years	%	Static
		OpenHunting_pct	Maximum % of levees open for hunting across all years	%	Static
		BayDist_km	Distance to San Francisco Bay	km	Static
		CreekDist_km	Distance to creek/slough	km	Static
		LandfillDist_km	Distance to Tri-cities Landfill	km	Static
Main	Hydrology	Breached	Breached to tidal action	Binary 1 / 0	Annual
		Gated	Gated culvert / semi-tidal	Binary 1 / 0	Annual
		WaterElev_m	Mean water depth	cm	Change
		Salinity	Mean salinity	ppt	Annual + Change
		pH	Mean pH	pH scale	Annual + Change
		LDO_mgL	Mean dissolved oxygen	mg/L	Annual + Change
		Temp_C	Mean water temperature	°C	Annual + Change
	Weather	PPT0	Cumulative annual precipitation – year-to-date	mm	Annual
		PPT1	Cumulative annual precipitation – last year's only	mm	Annual
		PPT3	Cumulative annual precipitation – last three years	mm	Annual
		TMX	Mean daily maximum temperature for the season	°C	Annual
		TMN	Mean daily minimum temperature for the season	°C	Annual

Model Selection

A full list of variables we assessed is in Table 2. We trimmed the dataset to only rows of complete cases for all variables included in the final modelset, in order to increase the validity of AICc comparisons by ensuring the sample size of each candidate model was the same. Table 3 shows the resulting number of ponds with phalarope counts, water quality samples, and complete cases in each year. Note that for a pond to have a "complete case" it needed to have detections in both the current and previous year, and a full set of water quality measurements. Because surveys only began in 2019, and water quality information was not collected in year 2020 due to the COVID-19 pandemic, only 2021 to 2022 and 2022-to-2023 had complete cases that could be included in the analysis. As a result, the final sample sizes were small **(*n* =** 63 for total phalaropes, *n* = 63 for Red-necked Phalarope, and *n* = 23 for Wilson's Phalarope). As a rule of thumb, statisticians recommend 10 observations per covariate. Our modelset therefore only included all combinations of up to 6 covariates for total phalaropes, up to 6 for Red-necked Phalarope, and up to 2 for Wilson's Phalarope.

AICc provides a way to compare the relative fit of models which penalizes the inclusion of additional covariates to avoid overfitting models. For the present analyses, we selected only the best model (lowest AICc), but note that as a rule of thumb models with AICc <2 higher than the best model are considered highly competitive (Bozdogan 1987). Due to the large number of habitat variables assessed, we used a structured model selection procedure based on AICc to create a tractable set of candidate models per guild/species modified (described in Van Schmidt & Parsons 2024).Because of the limited number of covariates we could include, we did not bundle change measures and raw measures as in Van Schmidt & Parsons (2023), but allowed them to stand-alone (e.g., we allowed "Salinity_AC" without "Salinity"). We excluded from the final set of candidate models any that contained combinations of variables that were problematically correlated (correlation coefficient >0.7; Table A1.1) from the final modelset. We first estimated a "base" model of static pond characteristics (Table 3, "base" in column Stage). We selected the top model from this assessment to carry forward as the base model. Because Wilson's Phalaropes had so few available variables and because the base covariates were not of management interest, we excluded this first stage of model selection for that species and only used an intercept-only "base" model. The model selection tables are in Appendix I. Model selection on the remaining variables was the main focus of our study. The full modelset was all permutations of column variables in the "main", appended to a single (static) "base" model that was unique to each of the three taxa groups.

Model Assessment

We assessed goodness-of-fit of the best model for each guild based on R^2. To estimate the impact of the habitat changes and weather on trends within the current year, we used an approaching of effect partitioning and counterfactual analysis. We first predicted the expected percent change for each pond included in the model (i.e., excluding ponds that had missing habitat data for some variables and ponds that were unoccupied in both years). We then predicted the percent change at each pond under several counterfactual alternatives with groups of covariates "removed": (1) no random effect for year, (2) no change in site hydrology (breached, gated, and water quality measurements set to last year's values, and "AC" water quality change variables set to 0), (3) average weather (weather variable set to their mean long-term value across all site-years), and (4) each static site characteristic set to the mean value across all site-years. Because the log percent change measure is difficult to interpret in terms of meaningful impacts at the population-scale, we transformed site-scale predictions to predicted actual change in mean abundance by applying the inverse of equation Eq. 1 with the actual value of abundance in the previous year t-1. We then summed these values for each alternative scenario across all sites included in the model. The effect of each of the four groups of covariates on change in overall abundance was then calculated as:

Effect = Sum Abundance Change - Sum Abundance Change in Alternative (Eq. 2)

It is important to note that these estimated effect sizes are based on holding other variables at their real values, and additional variation may arise from the intercept term and residual model error. Therefore, estimated effects from each group of models do not sum to the total predicted change, and should only be compared in terms of the relative magnitude and direction of effects rather than a conclusive determination of drivers of abundance change.

Interpreting the results of this model can be challenging because a positive relationship between a predictor variable and percent change in population size could be the result of that predictor driving population growth, population declines, or both, and could be the result of increases or decreases in the relevant habitat variable. Thus, for a bird that prefers low-salinity habitat, a positive relationship between changes in salinity could be the result of a population growing as salinity falls in one pond, or declining as salinity rises in another. For this report, we focus on reporting whether each term is a direct (positive) relationship, or an inverse (negative) relationship. For all variables, however, it is important to keep in mind that the observed relationships may have multiple potential explanations.

Foraging Data Analysis

Previous reports have grouped salinity into three bins: low (0-60 ppt), medium (60-120 ppt), and high (>120 ppt) bins of salinity. These bins were obtained from earlier reports on the salt ponds, but had an unclear relationship to actual phalarope habitat needs. Therefore, we first conducted a literature review on both phalarope foraging and salinity tolerances of their probable prey items. We synthesized information from these diverse sources to create a revised set of salinity bins to more directly align with the salinity tolerances of probable phalarope prey items.

We then applied these salinity bins to the entire phalarope migration survey dataset from 2021-2024, analyzing differences in Red-necked, Wilson's, and total phalarope abundance across the bins. For the foraging data, for each species we assessed differences in forgaging behavior across two measures: (1) the total number of foraging attempts, (2) the number of successful foraging attempts, and (3) the success rate (# successes / # of attempts). We graphed data with boxplots using base package *stats* and package *ggplot2*. For number of attempts and successes, we used GLMMs with a negative binomial distribution to test for significant effects of salinity bins, with a random effect for site to account for non-independence of the samples. For the success rate, we used a Gaussian regression with a random effect for site. Models were fit with package *lme4* (v1.1). We used package *DHARMa* (v0.4.7) to simulate residuals from the fitted models and assess them for overdispersion, accurate modeling of zero-inflation, and test normality of residuals and heteroskedasticity. Models showed slight violations of heteroskedasticity so we used package *lmerTest* (v3.1) to calculate robust p-values via the Satterthwaite method.

Historic Data Reanalysis

The current set of ponds was selected by Tarjan (2019b) with the criteria that they captured 95% of all phalarope observations within the last five years (from 2014-2018) based on SFBBO surveys, along with supplementary analysis of eBird records to identify other sites with significant concentrations of phalaropes in the surrounding region. We recommended in Burns & Van Schmidt (2023) that opportunistic surveys of other sites continue to ensure that, should spatial patterns of phalarope use shift again, the set of surveyed sites expand to capture any additional important stopover sites. This year, we for the first time added pond E5C after a count of 2,400 phalaropes were opportunistically detected there on August 22, 2023.

During waterbird surveys on May 7, 2024, surveyors counted 900 Red-necked Phalaropes at pond A5 and 377 at the adjoining pond A7 (Van Schmidt & Parsons 2025). These ponds were not within the set of surveyed sites because the last time more than 10 birds were detected there was in 2004; however, counts of up to 1,000 birds phalaropes were observed frequently there in 2003-2004. Examination of the salt

Table 3. Number of ponds that had phalarope counts, water quality samples, and complete cases for a change in abundance model (both phalarope counts and water quality samples in the focal year and previous year). Note that the true sample size is reduced further if phalaropes were not present in either the current or previous year.

Water Year	Waterbird Counts	Water Quality Samples	Complete Cases
2020	17	0	0
2021	31	0	0
2022	31	0	20
2023	31	31	20
2024	32	31	23

pond survey data showed that pond E11 and E10 also had sightings of >100 phalaropes since the 2018 analysis (299 in Sept 2019 and 153 in Sept 2021, respectively). In light of these ongoing observations, and the known high variability in which ponds phalaropes utilize during any given migration (Tarjan 2019b), it seemed plausible that the current set of sites was insufficient to capture the range of spatial variation in phalarope habitat use within South San Francisco Bay. Especially in light of worrying declines in Red-necked Phalaropes this year (described in *Results*), a less conservative reanalysis of the set of survey sites seemed warranted.

We therefore reassessed recommendations for the set of ponds by examining the full waterbird survey dataset from 2002-2024. Furthermore, recent years of surveys have highlighted both potentially divergent habitat use by Wilson's and Red-necked Phalaropes, with Wilson's Phalaropes more specialized on hypersaline habitats and thus faring better within the remaining salt ponds (Figure 1). Despite historical records of very high abundance within South San Francisco Bay (Jehl 1988, LaBarbera et al. 2023) this species is today relatively rare, representing only 13% of phalarope sightings from 2019-2024. Only examining the set of ponds that captures 95% of all phalarope records may then risk ignoring sites that are critical to recovery of Wilson's Phalaropes. In order to address this, we identified the set of ponds that satisfied three different criteria: capturing 95% of all (1) Red-necked Phalarope, (2) Wilson's Phalarope, and (3) total phalaropes. Preliminary examination of results showed that in the resulting set of sites, only two sites met the condition for total phalaropes but not Red-necked Phalaropes (M3 and M4) due to large unidentified phalarope counts; however, these counts were in mid-August to mid-September, meaning they were very likely Red-necked Phalaropes. Therefore, we discarded the third condition and assigned each site a rank of 2 (captured 95% of both phalarope species) or 1 (captured 95% of only one species). Given that the habitat conditions of many SBSPRP have changed since the Initial Stewardship Plan, which corresponded with the decline in phalaropes (Burns & Van Schmidt 2023), it is possible that some ponds with very high historic sightings are no longer viable habitat. To more heavily weight recent sightings, we also gave a third rank vote to ponds that had sightings of 50 or more phalaropes (~1% of the Adaptive Management Plan baseline) after 2009 (the last year in which total sightings were >10,000, and after which the decline appeared to level off; Fig. 1).

Results & Discussion

Overview of Counts and Trigger Status

Across n = 254 visits during the 8 surveys of the 32 sites in 2024 we counted a total of 1,352 Wilson's Phalarope, 2,433 Red-necked Phalarope, and 1,050 phalarope of unidentified species, for a grand total of 4,835 birds (Fig. 2). This was 11,819 fewer phalaropes than were detected last year (a -71.0% decrease; Table 1, Fig. 2) due to decreases in Red-necked Phalaropes as well as decreases in Wilson's Phalaropes.

Within only the 25 sites within the SBSPRP and salt pond complexes, the average number of sightings of phalaropes per survey during the early July through end of September surveys was 448. This was 91.6% below the revised 2005-2007 baseline of 5,324 (Burns & Van Schmidt 2023). Counts therefore remain below the Adaptive Management Plan trigger threshold of a decline of 50% or more below baseline values in a single year. The alternative trigger of a decline of 25% or more the baseline over the last three consecutive years was also triggered because counts were below this threshold in summer 2022, 2023, and 2024.

Observations of unidentified phalarope present a particular challenge when analyzing data to understand species-specific trends. In 2024, unidentified species represented **21.7%** of the total observations, versus a mean of 6.0% from 2020-2023 (range 2.6% - 8.8%). Handling of unidentified observations is a challenge for phalarope studies across their migration range and there is no agreed upon best practice for whether or how to include them in analysis (Carle et al. 2024). One estimation method is to assign all unidentified individuals to species based on which migration peak they were observed in (e.g., unidentified phalarope in June-July would be assumed to be Wilson's Phalarope, and unidentified phalaropes observed later would be assumed Red-necked Phalarope). Across all sites, counts of Wilson's Phalaropes peaked at 787 birds (803 assuming June-July unidentified phalaropes were Wilson's) during the survey centered on 07/16. Counts of Red-necked Phalaropes peaked at 1,082 birds (assuming August-September unidentified phalaropes were Red-necked did not change this peak count) during the survey centered on 09/12. These peaks indicate that the current survey schedule largely spans the period when phalaropes are present in the area (Fig. 3).

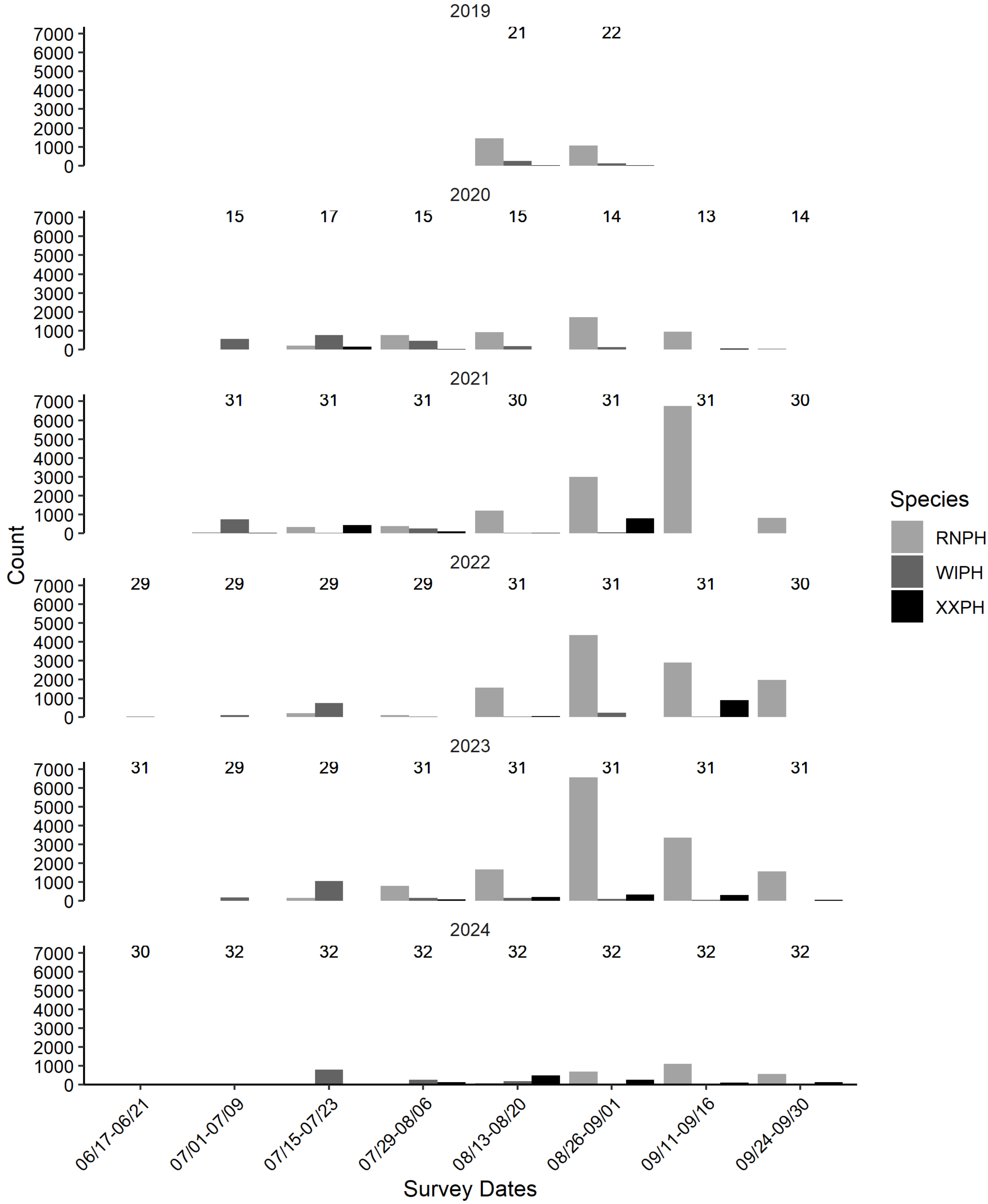


Figure 3. Counts of phalarope species observed during the phalarope migration surveys in 2019-2024. RNPH = Red-necked Phalarope; WIPH = Wilson's Phalarope; XXPH = phalaropes of unidentified species. No Red Phalaropes were observed. Counts are summed across all sites visited during each survey. Number above each date is the number of sites surveyed during that round.

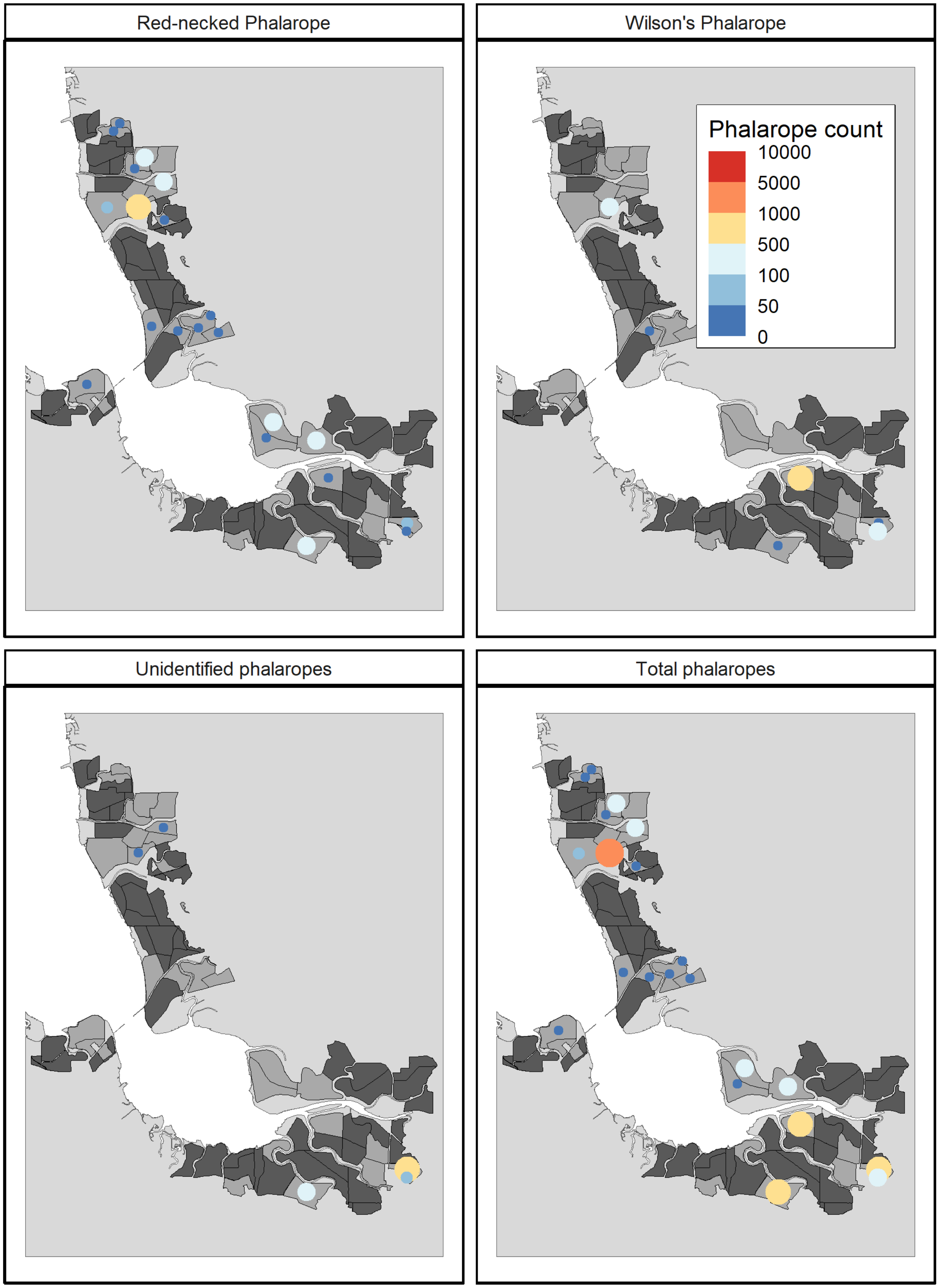


Figure 4. Counts of the total number of phalaropes observed during the phalarope migration surveys from June 17 to September 26, 2024. Species grouping is indicated at the top of each map. Dots are scaled to represent the relative number of phalaropes observed. Ponds with no dot had zero phalarope observations during 2024 surveys.

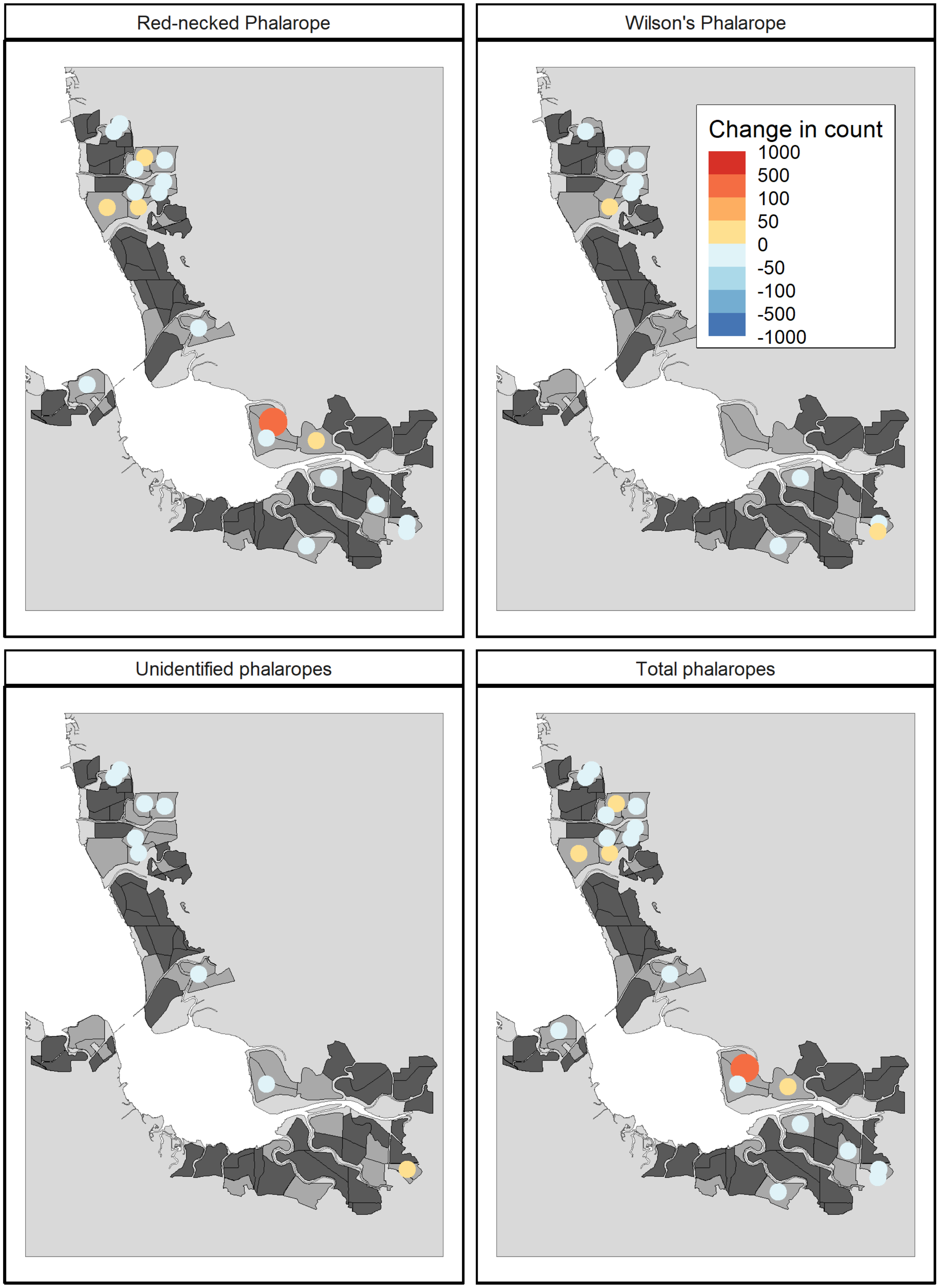


Figure 5. Change in the total number of phalaropes observed during the phalarope migration surveys from 2023 to 2024. Species grouping is indicated at the top of each map. Dots are scaled to represent the absolute change in number of phalaropes observed. Ponds with no dot had zero phalarope observations during both 2023 and 2024 surveys. Pond E5C is not plotted because it was not surveyed in 2023.

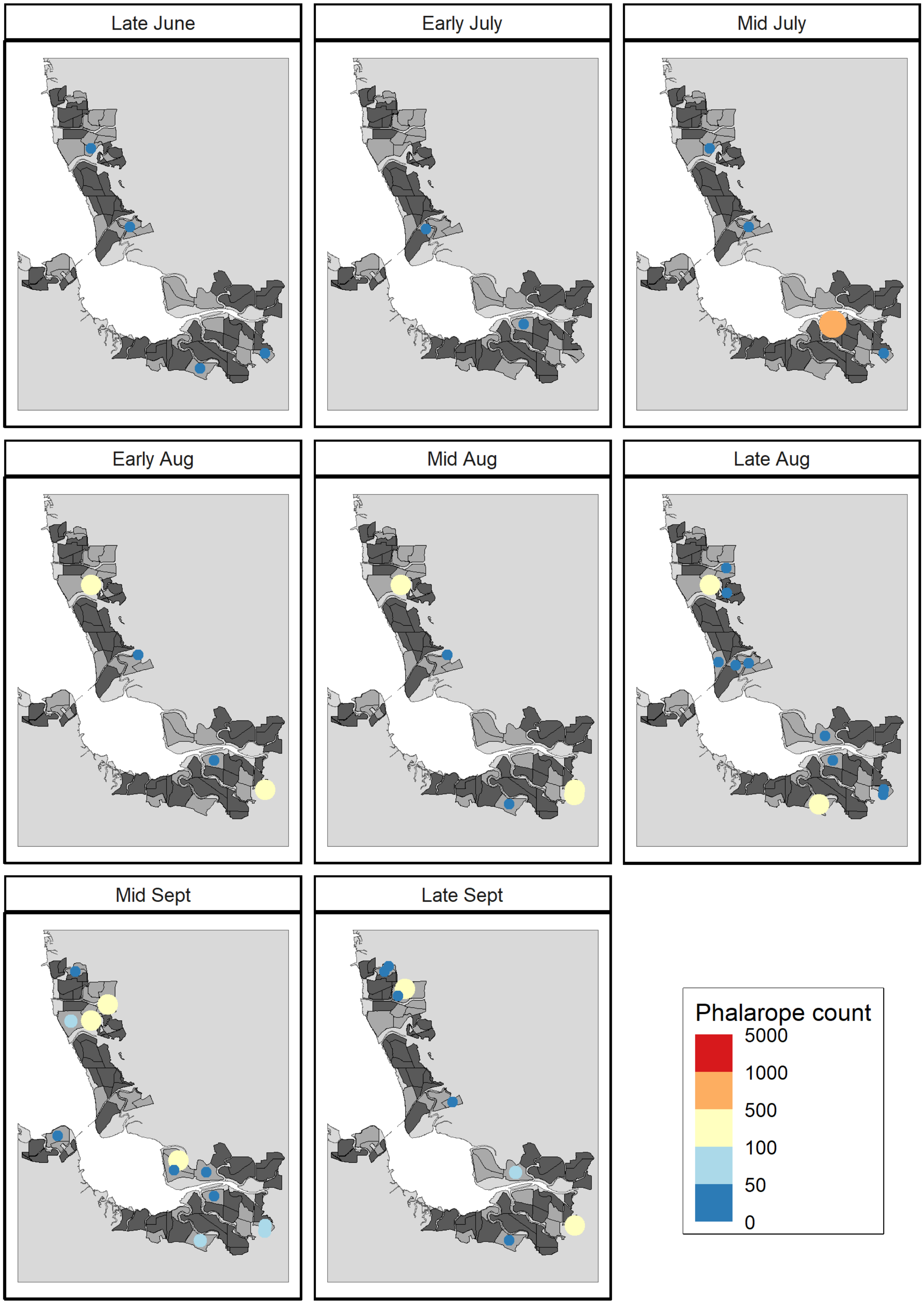


Figure 6. Counts of the total number of phalaropes observed during each phalarope migration survey block from June 17 to September 26, 2024. Survey block is indicated at the top of each map. Dots are scaled to represent the relative number of phalaropes observed. Ponds with no dot had zero phalarope observations during this period.

Red-necked Phalaropes

Trends

Across all complexes, there were 2,433 total sightings of Red-necked Phalaropes (sum sightings during the entire survey period). Compared to the previous year, this was a decrease of 11,630 total sightings (-82.7%; Fig. 4). Excluding the seven sites outside of the SBSPRP and salt pond areas, we observed a total of 1,938 Red-necked Phalaropes (79.6% of all counts), a decrease of -9,683 total sightings (-83.3%) from last year. At the pond level, E4 had the highest abundance (730 sum sightings), followed by E6B (441) and Sunnyvale WPCP (407; Fig. 4). If considering only the SBSPRP and salt pond footprints, the top three most abundant ponds also included E6 (300 sum sightings).

The site with the largest increase in Red-necked Phalaropes from last year was E4 (+466), while the site with the largest decrease was A9 (-5,972; Fig. 5). Total sightings of Red-necked Phalaropes at A13 (which has had its water level drawn down for levee construction), E13, New Chicago Marsh (also adjacent to the levee construction project), Spreckles Marsh, and Sunnyvale WPCP were lower than in any previous phalarope migration survey season. At E6B, M1, M3, and N2, sum sightings of Red-necked Phalaropes were higher than ever previously recorded there during phalarope migration surveys (Fig. 4).

Change Model

The best change model for Red-necked Phalaropes found that years with lower minimum summer temperatures had greater declines in phalarope abundance (Fig. 7). We also found a significant relationship supporting that as a site's pH increased, so too did phalarope abundance at that pond (and vice versa). Change in pH was inversely correlated the change in salinity (R^2 = -0.56). The relationship with changes in pH was supported in most of the top models, while there was also alternative support for models that included impacts of precipitation and maximum temperatures (Table A1.4). This relationship with pH changes was not found during analysis of the 2003–2018 salt pond waterbird surveys, but positive relationships with salinity and negative relationships with increasing salinity, increasing water depth, and previous year's precipitation were (Van Schmidt & Parsons 2025). Model power was low due to the small sample sizes, with only **3** years of turnover able to be modeled due to missing water quality samples (Table 3). However, unlike in the 2023 report (Van Schmidt & Parsons 2024), both relationships were significant in this year's report, highlighting the increasing statistical power of the models as additional years of data are added to the dataset.

We report predictions made by the change model within 23 ponds, excluding sites that were unoccupied by Red-necked Phalaropes in summer of both 2024 and 2023 (A12, A3N, Alviso Marina, Coyote Hills Regional Park, Crittenden Marsh, E5C, R2 and RSF2U2) and sites with missing water quality measurements (Northeast of N1, where surveyors could not sample water quality in 2023)**.** Across this set of sites, the actual change in abundance of Red-necked Phalaropes in summer was -11,638. In comparison, the model predicted a total change of -9,911, with this decrease driven by effects of weather. The proportion of variance in percent change in counts explained by the best model was low (R^2 = 0.15).

The change model estimated that changes in site hydrology could explain a 3,390 increase in abundance of Red-necked Phalaropes from last year due to increasing pH. The sites predicted to experience the largest increase in Red-necked Phalaropes abundance due to hydrology changes were Sunnyvale WPCP, N1, and A9, while the sites predicted to experience the largest increase in suitability (i.e., predicted percent change irrespective of actual abundance last year) were N1, Sunnyvale WPCP, and M2 (Table 4). The sites predicted to experience the largest negative impacts from changes in hydrology in summer 2024 were E5, N4, and A13 (based on predicted decreases in abundance) and E5, E12, and N4 (based on predicted decrease in suitability). Static site characteristics were not included in the best model.

The change model predicted that deviations from mean weather could drive a 39,322 decrease in abundance due to high minimum temperature (Table 4). This unrealistically large "predicted increase" is due to the fact that hydrology changes (increasing pH) would have otherwise been predicted to cause a large increase in several highly abundant sites (i.e., this difference in abundance was calculated by comparing to a hypothetical year that had the increase in pH but not the higher minimum temperatures, not comparing to last year). While this does suggest the current weather played a role in the unusual decline, this extreme fluctuation is likely due to the limited number of data points; with only three years of turnover (2021–2022, 2022–2023, and 2023–2024) the estimated weather effect is likely unstable.

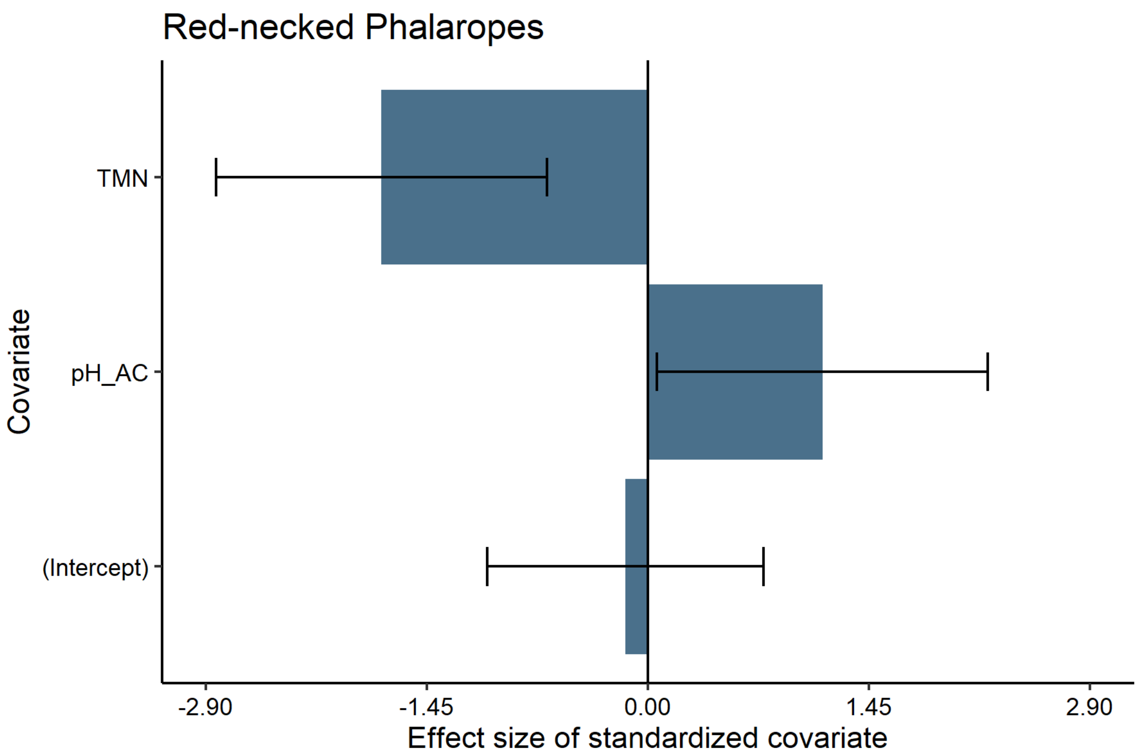


Figure 7. Fixed effects for the top model for interannual change in abundance of Red-necked Phalaropes during summer in current and former salt ponds in South San Francisco Bay, California, water year 2003–2024 (n = 63). Dark bars show coefficients and error bars show 95% confidence intervals. The year random effect was not included in this model due to sample size limits.

Table 4. Actual and predicted changes in mean abundance per survey of Red-necked Phalaropes from summer 2023 to summer 2024 (sorted greatest decrease to greatest increase). Predicted effect sizes of covariate groups are shown for hydrology (pH; compared to no change from last year), weather (minimum temperature; compared to mean annual weather), and a year random effect. Change values were calculated by back-transforming the predictions on the ln(% change + 1) scale (shown in parentheses and interpretable as change in relative habitat suitability) and then multiplying by the actual previous year's abundance.

Pond	Previous abun.	Current abun.	Actual change	Predicted change	Hydrology effect	Weather effect
A9	5,988	16	-5,972 (-5.86)	-5,421 (-2.35)	+354 (+0.97)	-9,118 (-2.84)
E13	1,240	6	-1,234 (-5.18)	-1,017 (-1.71)	+175 (+1.51)	-3,205 (-2.73)
E6	1,423	300	-1,123 (-1.55)	-1,176 (-1.75)	+91 (+0.46)	-1,129 (-1.71)
E5	1,078	0	-1,078 (-6.98)	-1,016 (-2.84)	-49 (-0.57)	-309 (-1.77)
Sunnyvale WPCP	1,432	407	-1,025 (-1.26)	+832 (+0.46)	+2,175 (+3.23)	-19,909 (-2.28)
New Chicago Marsh	754	74	-680 (-2.31)	-699 (-2.60)	+39 (+1.19)	-1,461 (-3.30)
E12	659	8	-651 (-4.30)	-632 (-3.15)	+2 (+0.07)	-404 (-2.73)
E6A	284	0	-284 (-5.65)	-222 (-1.51)	+32 (+0.69)	-285 (-1.71)
Spreckles Marsh	256	10	-246 (-3.15)	-242 (-2.84)	+11 (+1.24)	-525 (-3.59)
E8	190	3	-187 (-3.87)	-157 (-1.73)	+18 (+0.76)	-215 (-1.99)
E7	93	0	-93 (-4.54)	-76 (-1.64)	+10 (+0.85)	-116 (-1.99)
N1	89	13	-76 (-1.86)	+356 (+1.60)	+434 (+3.61)	-1,593 (-1.52)
R1	68	3	-65 (-2.85)	-45 (-1.04)	+16 (+1.12)	-106 (-1.68)
M2	64	2	-62 (-3.08)	-36 (-0.79)	+26 (+2.08)	-288 (-2.38)
A13	1	0	-1 (-0.69)	-2 (-2.57)	+0 (+0.80)	-3 (-2.88)
NPP1	0	4	+4 (+1.61)	-0 (-0.01)	+1 (+1.92)	-3 (-1.44)

Pond	Previous abun.	Current abun.	Actual change	Predicted change	Hydrology effect	Weather effect
N2	0	25	+25 (+3.26)	-1 (-0.96)	+0 (+1.02)	-1 (-1.49)
E2	65	100	+35 (+0.43)	-54 (-1.72)	+6 (+0.76)	-75 (-1.99)
N4	0	41	+41 (+3.74)	-1 (-2.04)	+0 (+0.38)	-1 (-1.93)
M3	65	127	+62 (+0.66)	-56 (-1.89)	+7 (+1.33)	-143 (-2.73)
M1	1	115	+114 (+4.06)	-1 (-0.70)	+1 (+1.82)	-7 (-2.03)
E6B	49	441	+392 (+2.18)	-38 (-1.47)	+8 (+1.06)	-77 (-2.03)
E4	264	730	+466 (+1.01)	-210 (-1.57)	+33 (+0.92)	-350 (-1.99)

Wilson’s Phalaropes

Trends

Across all complexes, there were 1,352 total sightings of Wilson’s Phalaropes (sum sightings during the entire survey period). Compared to the previous year, this was a decrease of 286 total sightings (-17.5%; Fig. 4). Excluding the seven sites outside of the SBSPRP and salt pond areas, we observed a total of 1,157 Wilson’s Phalaropes (85.6% of all counts), a decrease of -158 total sightings (-12.0% from last year. At the pond level, A9 had the highest abundance (805 sum sightings), followed by E4 (337) and Spreckles Marsh (157; Fig. 4). If considering only the SBSPRP and salt pond footprints, the top three most abundant ponds also included N2 (15 sum sightings).

The site with the largest increase in Wilson’s Phalaropes from last year was E4 (+297), while the site with the largest decrease was A9 (-455; Fig. 5). Total sightings of Wilson’s Phalaropes at Sunnyvale WPCP were lower than in any previous phalarope migration survey season. At E4 and N2, sum sightings of Wilson’s Phalaropes were higher than ever previously recorded there during phalarope migration surveys (Fig. 4).

Change Model

The best change model for Wilson’s Phalaropes found that rainier years had significantly higher abundance of phalaropes (Fig. 8). The only models that had stronger support than an intercept-only model without covariates were those that included only weather covariates (Table A1.5). However, inference was limited because there were only 23 observations in the dataset, precluding fitting more than 2 covariates in a single model. Because of this, we also did not fit base model covariates for this species in order to retain as model power for fitting those covariates of management interest (Table A1.9).

We report predictions made by the change model within 11 ponds, excluding sites that were unoccupied by Wilson’s Phalaropes in summer of both 2024 and 2023 (A12, A13, A3N, Alviso Marina, Coyote Hills Regional Park, Crittenden Marsh, E12, E2, E5C, E7, E8, M1, M2, M3, N1, N4, Northeast of N1, NPP1, R1, R2 and RSF2U2). Across this set of sites, the actual change in abundance of Wilson’s Phalaropes in summer was -286. In comparison, the model predicted a total change of -718, with this decrease driven by effects of weather. The proportion of variance in percent change in counts explained by the best model was low (R^2 = 0.31).

The change model predicted that deviations from mean weather (i.e., a drier than average year) could drive a 1,153 decrease in abundance due to low year-to-date precipitation (Table 5). The change model did not find a relationship with changes in site hydrology or static site characteristics.

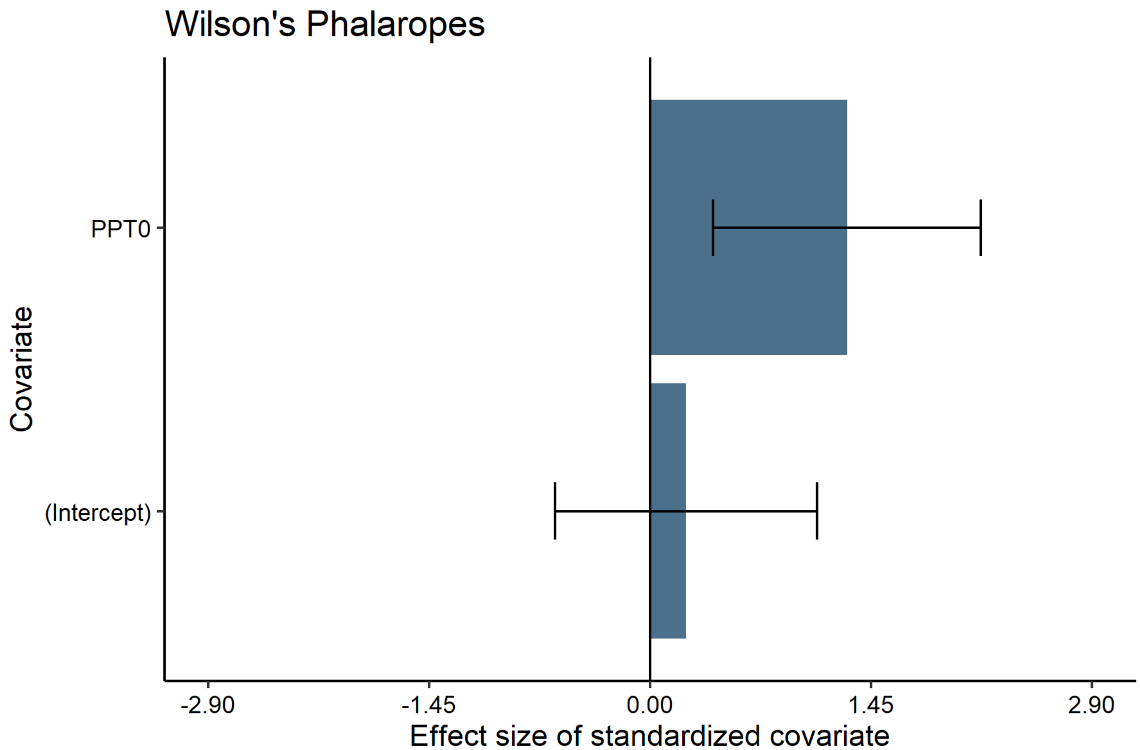


Figure 8. Fixed effects for the top model for interannual change in abundance of Wilson's Phalaropes during summer in current and former salt ponds in South San Francisco Bay, California, water year 2003–2024 (n = 23). Dark bars show coefficients and error bars show 95% confidence intervals. The year random effect was not included in this model due to sample size limits.

Table 5. Actual and predicted changes in mean abundance per survey of Wilson's Phalaropes from summer 2023 to summer 2024 (sorted greatest decrease to greatest increase). Predicted effect sizes of covariate groups are shown for weather (year-to-date precipitation; compared to mean annual weather) and a year random effect. Change values were calculated by back-transforming the predictions on the ln(% change + 1) scale (shown in parentheses and interpretable as change in relative habitat suitability) and then multiplying by the actual previous year's abundance.

Pond	Previous abun.	Current abun.	Actual change	Predicted change	Weather effect
A9	1,260	805	-455	-557	-889
			(-0.45)	(-0.58)	(-0.82)
New Chicago Marsh	189	16	-173	-80	-130
			(-2.41)	(-0.55)	(-0.78)
Sunnyvale WPCP	38	22	-16	-19	-29
			(-0.53)	(-0.65)	(-0.88)
E5	8	0	-8	-3	-5
			(-2.20)	(-0.40)	(-0.64)
E6A	4	0	-4	-2	-3
			(-1.61)	(-0.42)	(-0.66)
E13	1	0	-1	-0	-1
			(-0.69)	(-0.27)	(-0.50)
E6	1	0	-1	-1	-1
			(-0.69)	(-0.42)	(-0.66)
E6B	1	0	-1	-1	-1
			(-0.69)	(-0.32)	(-0.56)
N2	0	15	+15	-0	-1
			(+2.77)	(-0.47)	(-0.70)
Spreckles Marsh	96	157	+61	-45	-71
			(+0.49)	(-0.63)	(-0.86)
E4	40	337	+297	-11	-22
			(+2.11)	(-0.32)	(-0.55)

Total Phalaropes

Trends

Across all complexes, there were 4,835 total sightings of total phalaropes (sum sightings during the entire survey period). Compared to the previous year, this was a decrease of 11,815 total sightings (-71.0%; Fig. 4). Excluding the seven sites outside of the SBSPRP and salt pond areas, we observed a total of 3,161 total phalaropes (65.4% of all counts), a decrease of -10,662 total sightings (-77.1% from last year. At the pond level, E4 had the highest abundance (1,083 sum sightings), followed by A9 (821) and New Chicago Marsh (789; Fig. 4). If considering only the SBSPRP and salt pond footprints, the top three most abundant ponds also included E6B (441 sum sightings).

The site with the largest increase in total phalaropes from last year was E4 (+392), while the site with the largest decrease was A9 (-6,427; Fig. 5).

Change Model

The best change model for total phalaropes was very similar to that of Red-necked Phalaropes, with inverse relationships with minimum summer temperatures and positive relationships with pH (Fig. 9; Table A1.6). This is unsurprising because Red-necked Phalaropes are currently much more abundant within the South San Francisco Bay (Table 1), and therefore the results for total phalaropes are primarily describing their trends and habitat preferences.

We report predictions made by the change model within 23 ponds, excluding sites that were unoccupied by phalaropes in summer of both 2024 and 2023 (A12, A3N, Alviso Marina, Coyote Hills Regional Park, Crittenden Marsh, E5C, R2 and RSF2U2) and sites with missing water quality measurements (Northeast of N1). Across this set of sites, the actual change in abundance of phalaropes in summer was -11,823. In comparison, the model predicted a total change of -9,234, with this decrease driven by effects of weather. The proportion of variance in percent change in counts explained by the best model was NA (R^2c = NA, R^2m = 0.14).

The change model estimated that changes in site hydrology could explain a 6,244 increase in abundance of phalaropes from last year due to increasing pH. The sites predicted to experience the largest increase in phalaropes abundance due to hydrology changes were Sunnyvale WPCP, N1, and A9, while the sites predicted to experience the largest increase in suitability (i.e., predicted percent change irrespective of actual abundance last year) were N1, Sunnyvale WPCP, and M2 (Table 6). The sites predicted to experience the largest negative impacts from changes in hydrology in summer 2024 were E5, N4, and A13 (based on predicted decreases in abundance) and E5, E12, and N4 (based on predicted decrease in suitability). Static site characteristics were not supported in the best model.

The change model predicted that deviations from mean weather could drive a 54,842 decrease in abundance due to high minimum temperature (Table 6). As in the Red-necked Phalarope model, this unrealistic change is due to a large predicted swing between what site conditions would have predicted would be a high-abundance year based on increasing pH at many sites with high abundance last year.

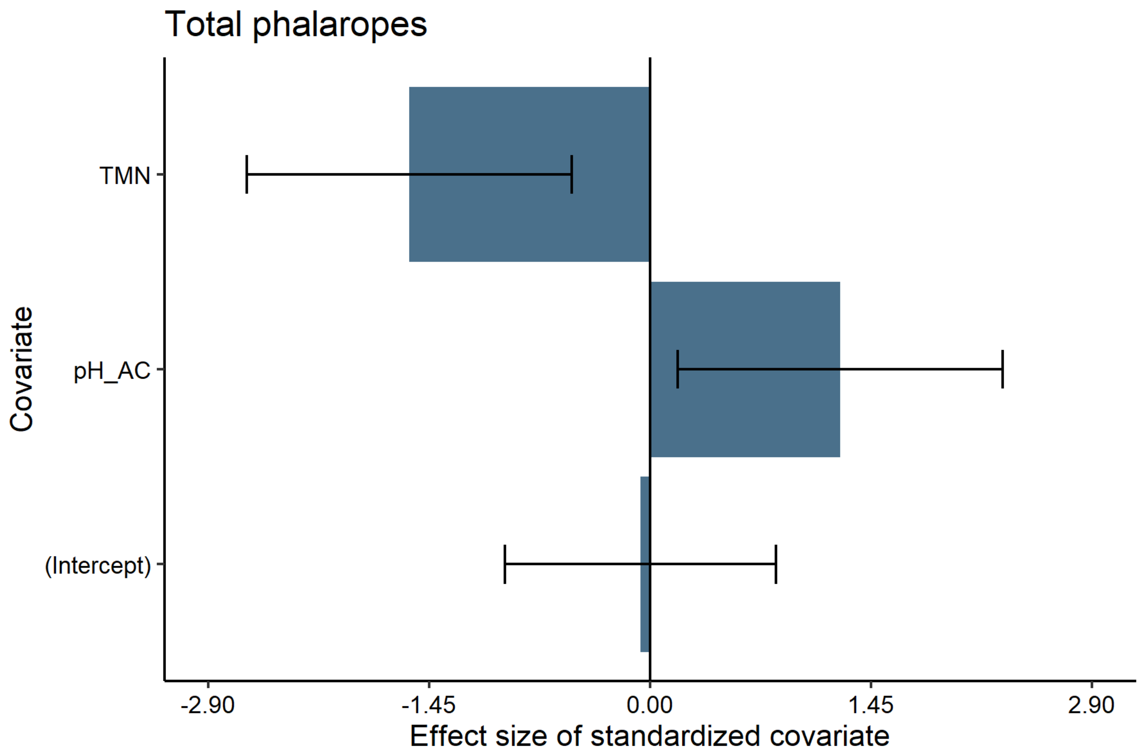


Figure 9. Fixed effects for the top model for interannual change in abundance of phalaropes during summer in current and former salt ponds in South San Francisco Bay, California, water year 2003–2024 (n = 63). Dark bars show coefficients and error bars show 95% confidence intervals. The year random effect was not included in this model due to sample size limits.

Table 6. Actual and predicted changes in mean abundance per survey of phalaropes from summer 2023 to summer 2024 (sorted greatest decrease to greatest increase). Predicted effect sizes of covariate groups are shown for hydrology (pH; compared to no change from last year), weather (minimum temperature; compared to mean annual weather), and a year random effect. Change values were calculated by back-transforming the predictions on the ln(% change + 1) scale (shown in parentheses and interpretable as change in relative habitat suitability) and then multiplying by the actual previous year's abundance.

Pond	Previous abun.	Current abun.	Actual change	Predicted change	Hydrology effect	Weather effect
A9	7,248	821	-6,427	-6,210	+679	-12,462
			(-2.18)	(-1.94)	(+1.06)	(-2.56)
E13	1,279	6	-1,273	-917	+293	-3,914
			(-5.21)	(-1.26)	(+1.65)	(-2.47)
E5	1,086	0	-1,086	-1,011	-65	-300
			(-6.99)	(-2.67)	(-0.62)	(-1.60)
E6	1,424	350	-1,074	-1,104	+126	-1,192
			(-1.40)	(-1.49)	(+0.50)	(-1.55)
Sunnyvale WPCP	1,470	617	-853	+2,619	+3,969	-28,107
			(-0.87)	(+1.02)	(+3.53)	(-2.06)
E12	853	8	-845	-804	+4	-543
			(-4.55)	(-2.83)	(+0.08)	(-2.47)
E6A	396	0	-396	-281	+62	-429
			(-5.98)	(-1.23)	(+0.76)	(-1.54)
E7	219	0	-219	-161	+36	-299
			(-5.39)	(-1.32)	(+0.93)	(-1.80)
New Chicago Marsh	1,005	789	-216	-885	+88	-2,254
			(-0.24)	(-2.12)	(+1.30)	(-2.98)
E8	190	3	-187	-145	+26	-235
			(-3.87)	(-1.42)	(+0.83)	(-1.80)
Spreckles Marsh	352	264	-88	-319	+25	-841
			(-0.29)	(-2.33)	(+1.35)	(-3.24)
M2	87	2	-85	-24	+57	-484
			(-3.38)	(-0.32)	(+2.27)	(-2.15)
N1	91	13	-78	+680	+757	-2,280
			(-1.88)	(+2.13)	(+3.94)	(-1.38)
R1	68	3	-65	-36	+24	-118
			(-2.85)	(-0.73)	(+1.23)	(-1.52)
A13	1	0	-1	-2	+0	-3
			(-0.69)	(-2.17)	(+0.88)	(-2.61)
NPP1	0	4	+4	+0	+1	-4

Pond	Previous abun.	Current abun.	Actual change	Predicted change	Hydrology effect	Weather effect
			(+1.61)	(+0.35)	(+2.09)	(-1.30)
E2	65	100	+35	-50	+9	-82
			(+0.43)	(-1.41)	(+0.83)	(-1.80)
N2	0	40	+40	-0	+0	-1
			(+3.71)	(-0.67)	(+1.11)	(-1.35)
N4	0	41	+41	-1	+0	-1
			(+3.74)	(-1.77)	(+0.42)	(-1.75)
M3	65	127	+62	-51	+12	-166
			(+0.66)	(-1.46)	(+1.45)	(-2.47)
M1	1	115	+114	-1	+1	-8
			(+4.06)	(-0.29)	(+1.99)	(-1.84)
E6B	59	441	+382	-40	+13	-103
			(+2.00)	(-1.12)	(+1.16)	(-1.84)
E4	691	1,083	+392	-492	+127	-1,014
			(+0.45)	(-1.24)	(+1.00)	(-1.80)

Salinity and Foraging Ecology

Literature Review

The most likely prey items are brine flies (*Ephydra* spp.), because brine shrimp (*Artemia* spp.) have been found to be too nutritionally poor to provide a viable diet for Red-necked Phalaropes (Rubega & Inouye 1994). However, it is unclear if the larger Wilson's Phalarope, which has a longer gut, may be able to extract sufficient nutrients from *Artemia* spp. At Great Salt Lake, phalaropes have also been observed foraging on chironomid larvae (midges), corixids (water boatmen), and *Daphnia* spp. (water fleas) in relatively lower salinity areas (Frank & Conover 2019, Frank & Conover 2021). Thus, they likely have some degree of ability to switch prey items.

Takekawa et al. (2006) found communities in North Bay salt ponds could be grouped into freshwater (0 - 0.5 ppt), mixohaline (0.5 - 30 ppt), low (31-80 ppt), mid (81-150 ppt), and high (>150 ppt) categories in alignment with observed trends in zooplankton and avian abundance. Most generally, *Ephydra* and *Artemia* spp. are most common between 30-80 ppt, corresponding to the Takekawa's "low" category, but tolerate a wider range from 25-150 ppt (Brown 2010). A review by the Goals Project (2000) found that *E. millbrae* can live in tide pools in the Bay Area with salinities up to 42 ppt. *E. cinerea* prefers more hypersaline environments and is more commonly found in the salt ponds; we could not find specific salinity tolerances published for this species within San Francisco Bay, however in Great Salt Lake this species can be found in water as salty as 130 ppt (Collins 1980). In contrast, the highest densities for *A. franciscana*, the dominant brine shrimp species within San Francisco Bay, show highest densities between 90-150 ppt but a broader tolerance of 70-200 ppt. The absence of fish, generally when salinity exceeds 80 ppt, can also significantly alter biotic communities (Carpelan 1957).

Salinity and Abundance

Based on this review, we created the classification system described in Table 7. Very few ponds were actually low salinity. Sunnyvale WPCP was the only pond that was near-fresh (1.1 mean ppt), so we slightly expanded this category to capture this distinct pond separately. This unique site is a treatment pond at a Water Pollution Control Plant and thus presumably very high in nutrients, and unlike other ponds within the study area or other freshwater ponds. Thus, it is appropriate that it was in its own category. The only pond in the brackish bin was E13, which was itself nearly in the saltwater bin (29.1 mean ppt).

Phalaropes demonstrated a strong association with a relatively wind band of moderately high salinity (30-150 ppt, encompassing the marine to medium hypersaline categories; Fig. 10), which closely aligned with the reported tolerance of *Ephydra* and *Artemia* spp. (Brown 2010). This trend was highly statistically significant (Table 8); we used sites with the marine salinity levels as the reference level in our regression models because these ponds were the most commonly used by phalaropes. Despite Wilson's Phalaropes more tightly associating with saline lakes while Red-necked Phalaropes spend significant time in the marine environment (Brown et al. 2010, Hunnewell et al. 2016, Colwell & Jehl 2020), we found that Wilson's Phalarope were relatively more common within the marine salinity band, while most phalarope observations at very high salinity sites were Red-necked Phalarope (Fig. 11).

Table 7. Salinity bins for analysis of phalarope abundance and foraging ecology in South San Francisco Bay, California. All salinity values are in ppt (g/L).

Lower (ppt)	Upper (ppt)	Name	Notes
0	2	Near-fresh	Freshwater up to the range of irrigation water
2	30	Brackish	Typical range observed in estuarine systems
30	50	Marine	Lower optimal limit for *Ephydra* and *Artemsia*; 42 ppt is the upper limit for *E. millbrae*
50	80	Low hypersaline	Upper optimal limit for *Ephydra* and *Artemsia*
80	150	Mid hypersaline	Fish generally absent; high range may be upper limit for *Ephydra* and *Lipochaeta slossonae*
150	+	High hypersaline	Increasingly only *Artemia* suvive, which reach tolerances by 200 ppt

Table 8. Regression coefficients of a negative binomial model (size parameter = 0.17) with a random effect for site (variance = 0.3244) to test for significant effects of water salinity on phalarope abundance (sum sightings over the summer and early fall) within South San Francisco Bay, California, 2024 (n = 110 surveys of 32 sites).

Parameter	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	6.4896913	1.517810	4.2756947	0.0000191
Salinity_Category6Brackish	-4.0662611	1.889964	-2.1515017	0.0314366
Salinity_Category6Marine	-0.8758886	1.607504	-0.5448749	0.5858396
Salinity_Category6Low hypersaline	-0.2635644	1.618900	-0.1628047	0.8706722
Salinity_Category6Mid hypersaline	-0.8668680	1.657946	-0.5228567	0.6010740
Salinity_Category6High hypersaline	-4.3351800	1.846041	-2.3483656	0.0188560

Foraging Observations

In 2024, the overwhelming majority of observations of Red-necked and Wilson's Phalaropes were of foraging, rather than roosting, birds (Fig. 11). Observations of phalarope foraging behavior showed patterns that were similar but diverged in important ways. Sunnyvale WPCP, the only near-freshwater (0-2 ppt) pond used by phalaropes, was a clear outlier site, with significantly more foraging attempts and successful prey captures per minute than any other salinity category (Table 9, 10; Fig. 12, 13). Excluding this site, brackish (2-30 ppt) and marine (30-50 ppt) ponds had significantly less foraging attempts and successful prey captures. Low hypersaline ponds (50-80 ppt) otherwise appeared to provide the best quality habitat, with medium hypersaline ponds (80-150 ppt) trending towards lower numbers of successful prey captures and high hypersaline ponds (>150 ppt) having significantly lower foraging attempts (Table 9, 10; Fig. 12, 13). The foraging efficiency (rate of successful captures per attempt) appeared to trend higher at hypersaline sites (Fig. 14), but this relationship was not significant (Table 11).

At three sites, phalaropes were observed probing into the water so rapidly that individual pecks could not be counted. One surveyor recorded high-speed video and confirmed that the birds did appear to be successfully capturing and swallowing prey very rapidly. These sites were Sunnyvale WPCP, A9, and E4, and their salinity was near-freshwater (1.1 ppt), marine (35.0 ppt), and low hypersaline (58.3 ppt), respectively. Birds at Sunnyvale WPCP and A9 spent 12% and 15% of the measured minutes foraging continuously in this manner, while birds at E4 spent 58% of each minute doing so. Both Wilson's and Red-necked Phalaropes were observed engaging in the behavior. These observations were not entered in quantitative analysis due to their rarity, but appear to indicate sites with very high prey density.

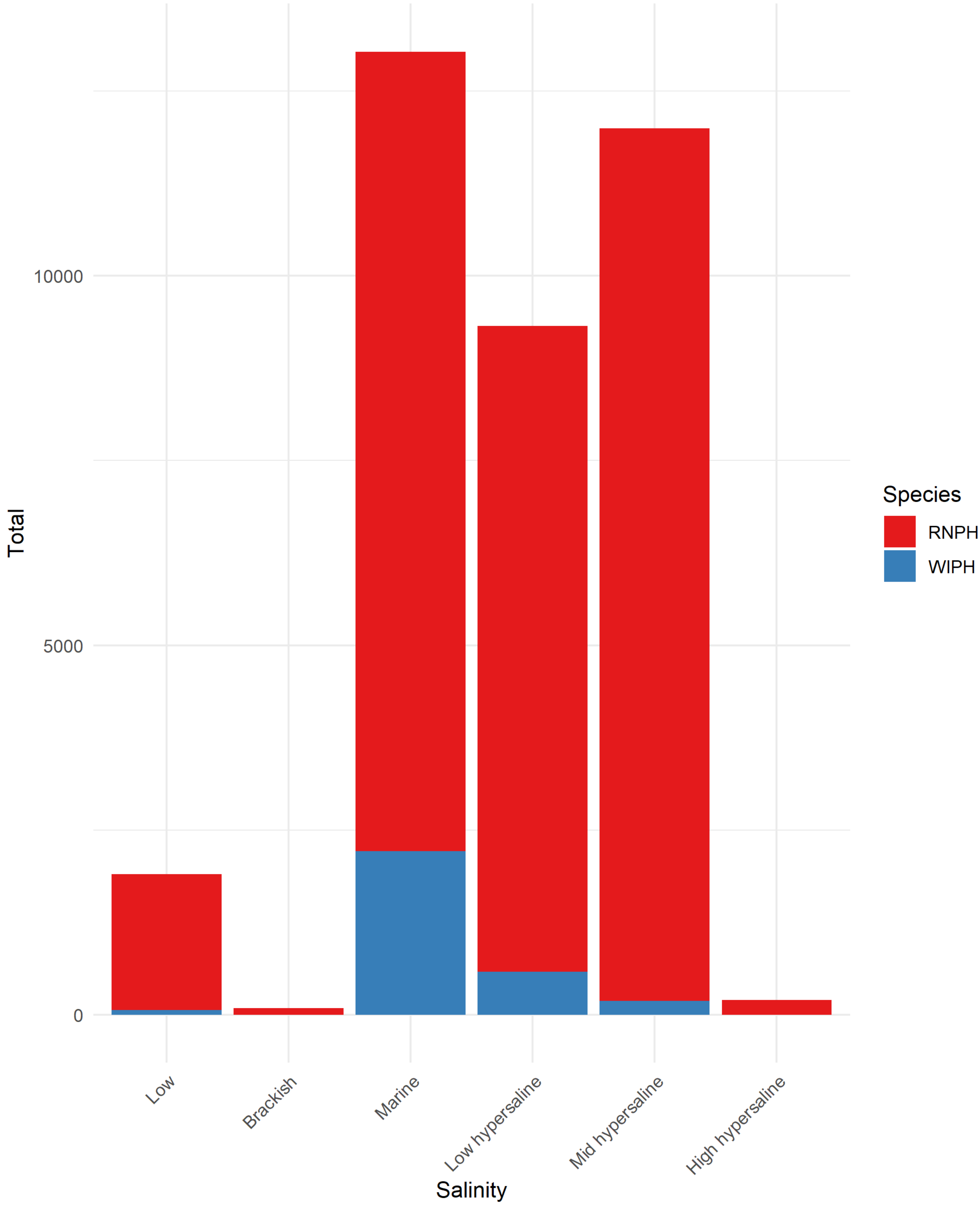


Figure 10. Total sightings of Wilson's (WIPH) and Red-necked (RNPH) Phalarope in phalarope migration surveys from 2021-2024 across ponds grouped into six bands of salinity.

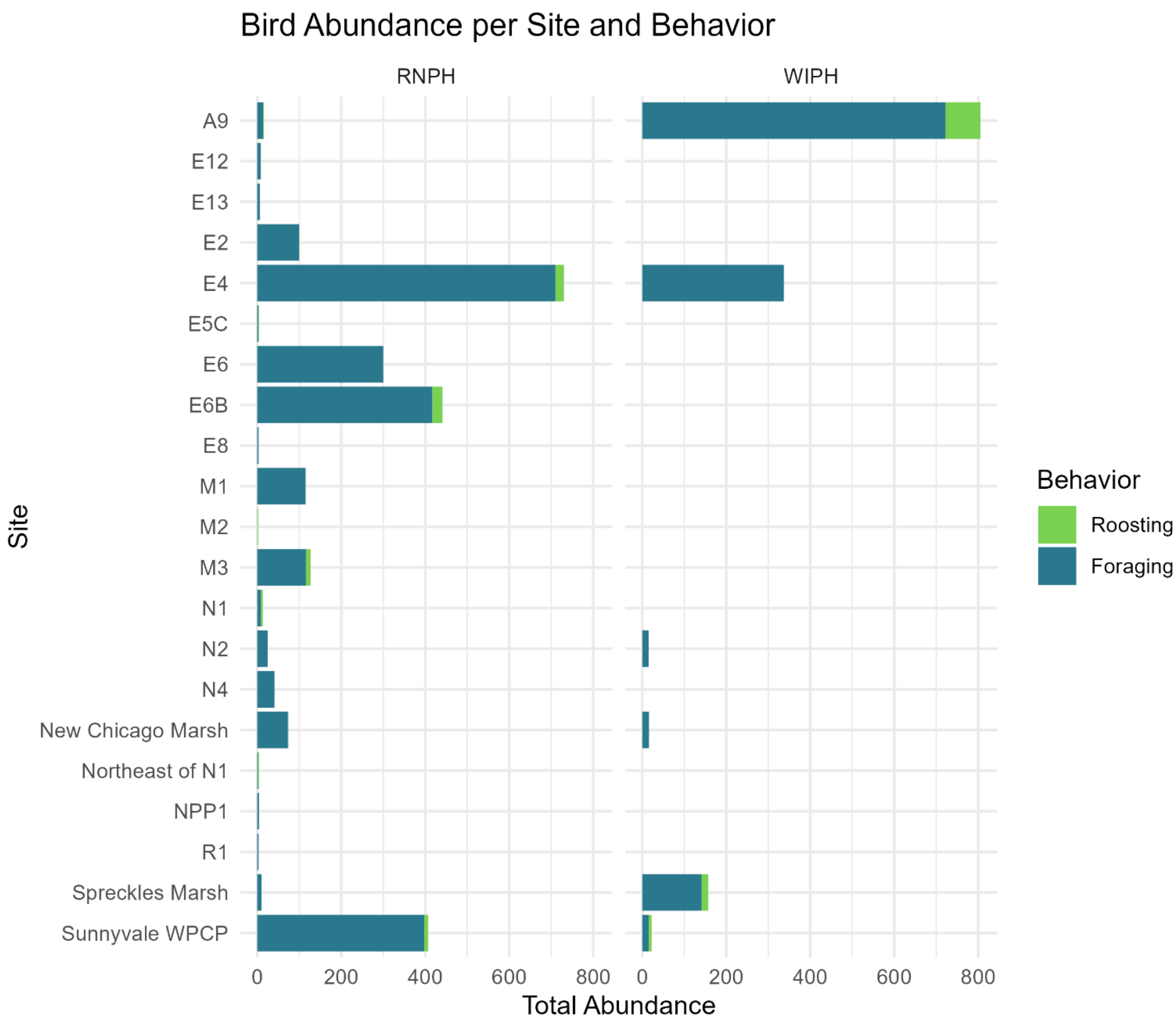


Figure 11. Total counts of the number of roosting and foraging Red-necked (RNPH) and Wilson's (WIPH) Phalaropes within 32 sites in South San Francisco Bay, California, June-September 2024. The remaining 11 sites not listed had zero phalaropes detected at them during the survey period.

Table 9. Regression coefficients a negative binomial model (size parameter = 2.7) with a random effect for site (variance = 0.0712) to test for significant effects of water salinity on the number of foraging successes for phalaropes within South San Francisco Bay, California, 2024 (*n* = 196 observations from 16 sites).

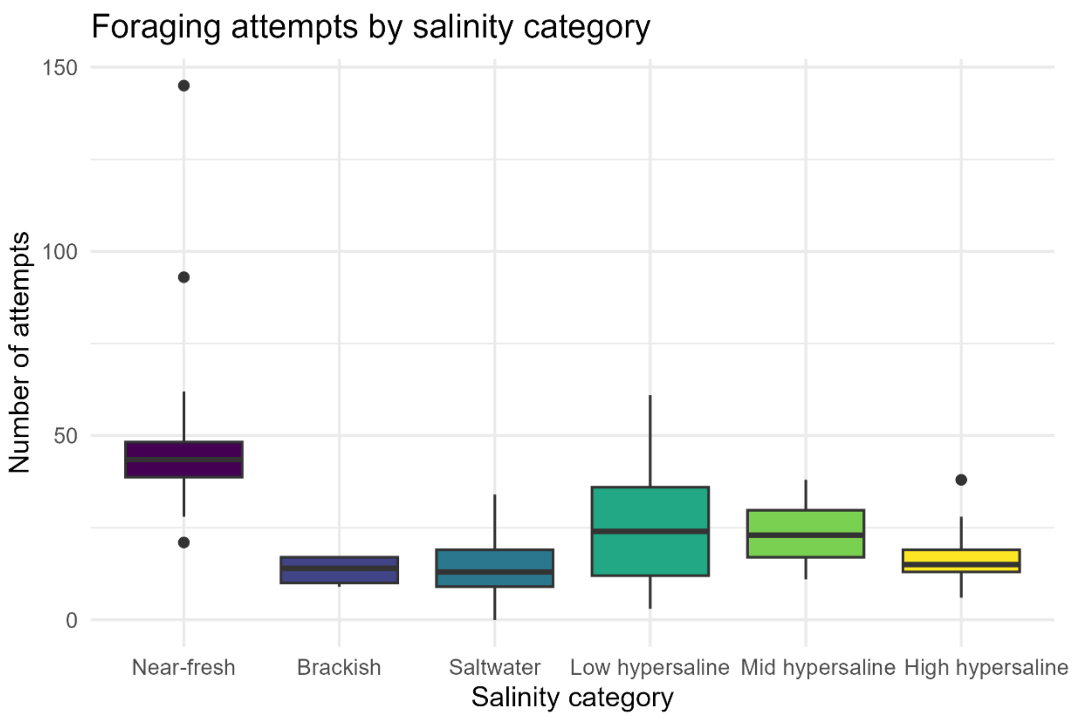
Parameter	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	2.5868823	0.1674325	15.4502957	0.0000000
Salinity_Category6Near-fresh	1.3444494	0.3493287	3.8486654	0.0001188
Salinity_Category6Brackish	0.0049327	0.4280449	0.0115239	0.9908055
Salinity_Category6Low hypersaline	0.5656081	0.2143170	2.6391196	0.0083122
Salinity_Category6Mid hypersaline	0.5784596	0.4427700	1.3064560	0.1913975
Salinity_Category6High hypersaline	0.1622866	0.3015955	0.5380934	0.5905126

Table 10. Regression coefficients a negative binomial model (size parameter = 2.7) with a random effect for site (variance = 0.0296) to test for significant effects of water salinity on the number of foraging attempts for phalaropes within South San Francisco Bay, California, 2024 (*n* = 187 observations from 16 sites).

Parameter	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	1.4070076	0.1548154	9.0882895	0.0000000
Salinity_Category6Near-fresh	1.2435841	0.3060135	4.0638215	0.0000483
Salinity_Category6Brackish	-0.6254750	0.4674626	-1.3380215	0.1808894
Salinity_Category6Low hypersaline	0.9067210	0.1847317	4.9083133	0.0000009
Salinity_Category6Mid hypersaline	-0.0257474	0.4567776	-0.0563674	0.9550491
Salinity_Category6High hypersaline	0.8959938	0.3048926	2.9387196	0.0032957

Table 11. Regression coefficients a normal (Gaussian) model with a random effect (variance = 0.3244) for site to test for significant effects of water salinity on the foraging success ratio (# of successful prey captures / # of foraging attempts) for phalaropes within South San Francisco Bay, California, 2024 (n = 171 observations from 16 sites).

Parameter	Estimate	Std. Error	df	t value	Pr(>|t|)
(Intercept)	0.2922287	0.0732601	10.744744	3.9889209	0.0022258
Salinity_Category6Near-fresh	0.0620655	0.1628413	8.458152	0.3811412	0.7124948
Salinity_Category6Brackish	-0.0921914	0.1761560	11.578809	-0.5233506	0.6106000
Salinity_Category6Low hypersaline	0.1607069	0.0963366	8.753926	1.6681804	0.1305737
Salinity_Category6Mid hypersaline	-0.1227488	0.1819333	13.158276	-0.6746915	0.5115528
Salinity_Category6High hypersaline	0.1387772	0.1241614	13.345894	1.1177161	0.2834054



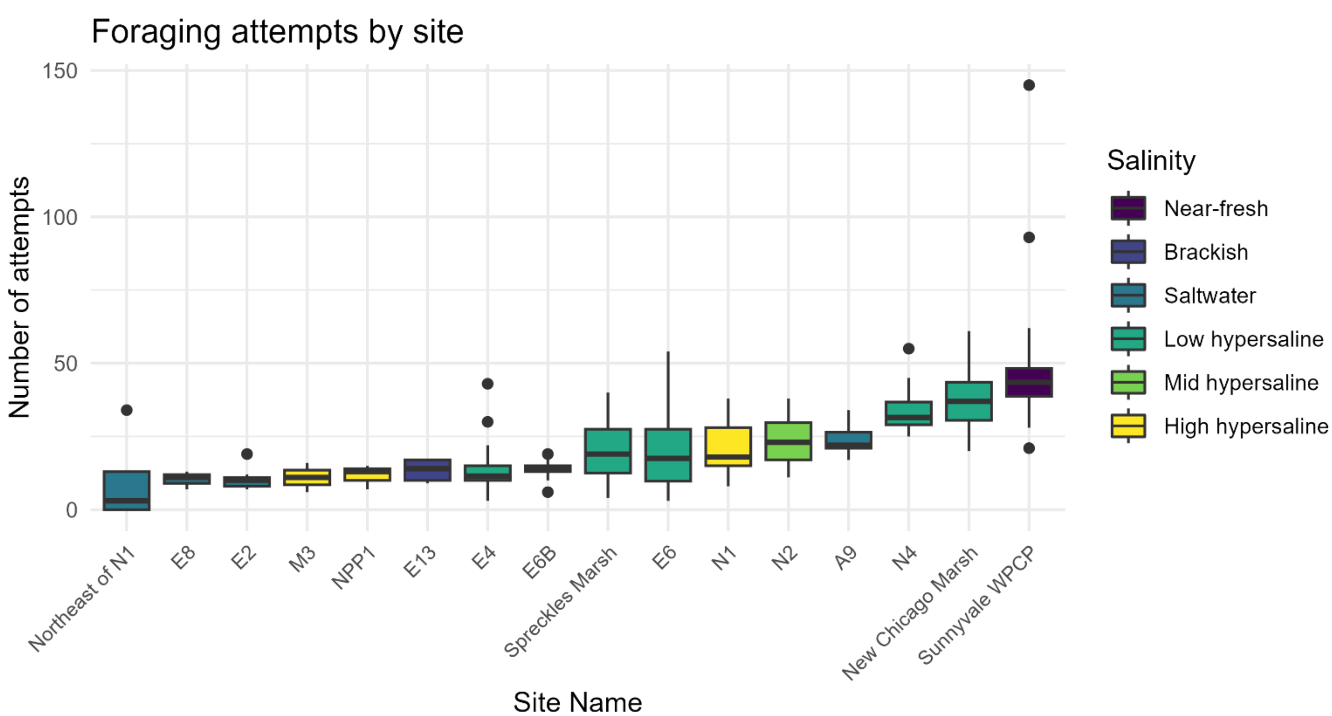
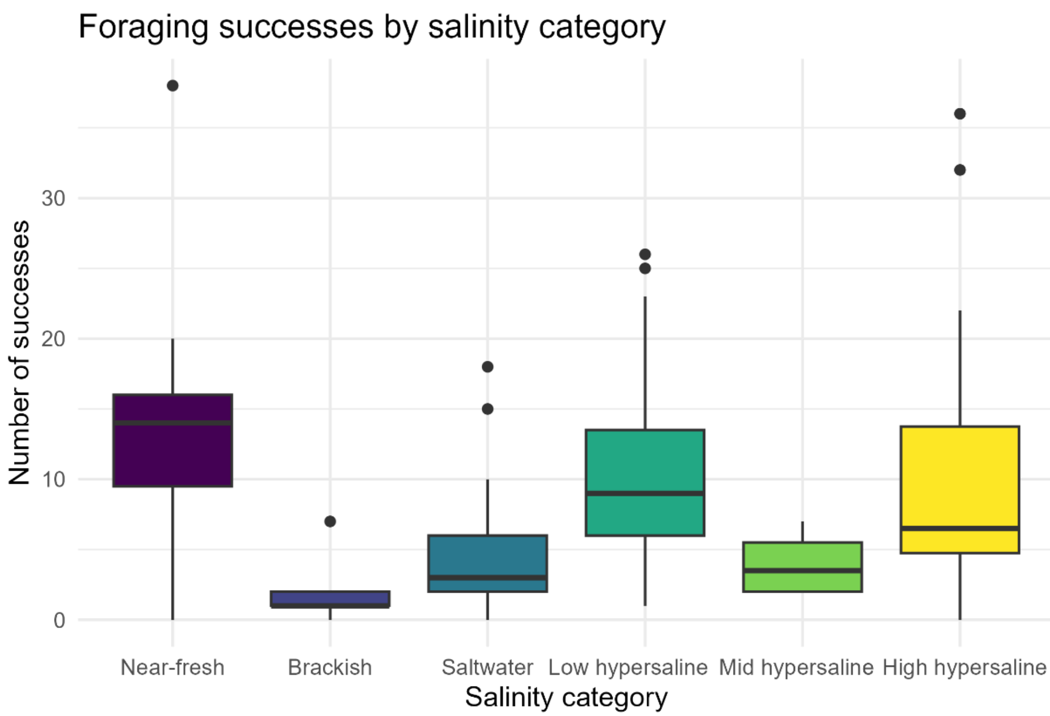


Figure 12. Boxplots of the number of foraging attempts (i.e., pecks) per minute by phalaropes within South San Francisco Bay, divided by salinity category (top) and site shaded by salinity category (bottom). Dots represent outlier observations.



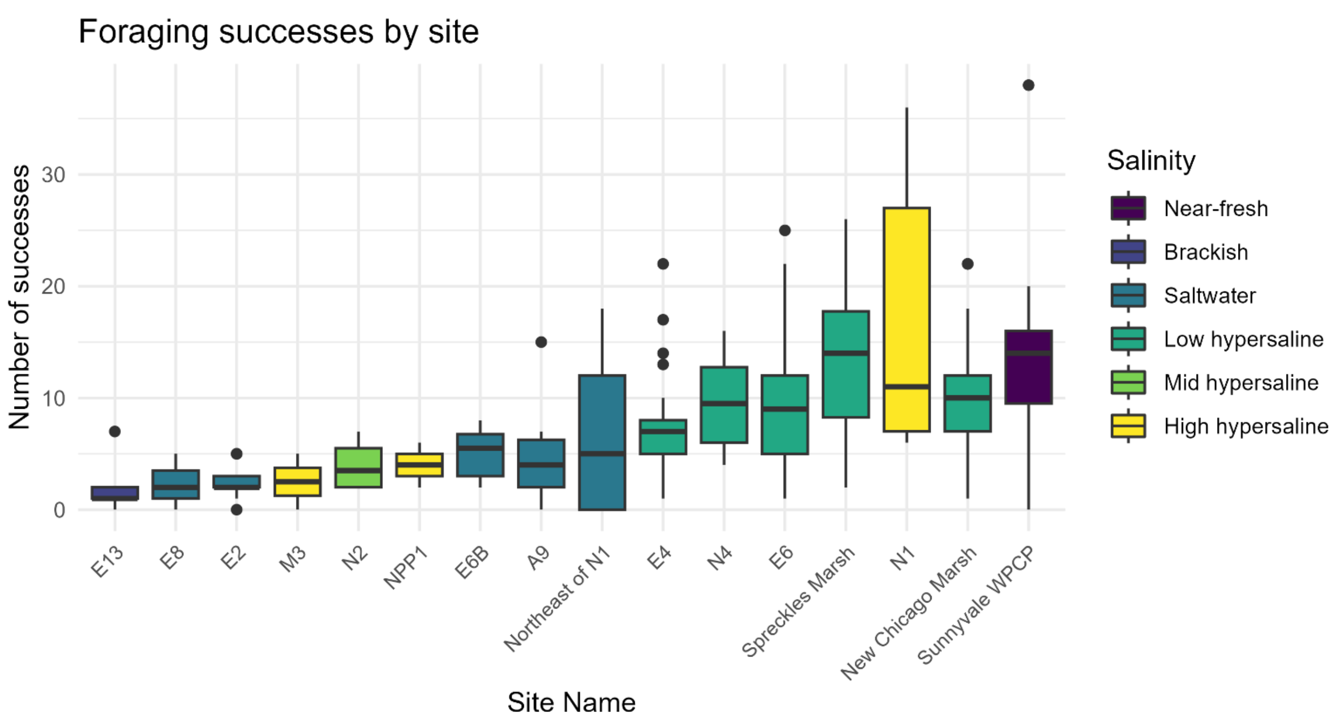


Figure 13. Boxplots of the number of successful foraging attempts (i.e., pecks followed by swallows) per minute by phalaropes within South San Francisco Bay, divided by salinity category (top) and site shaded by salinity category (bottom). Dots represent outlier observations.

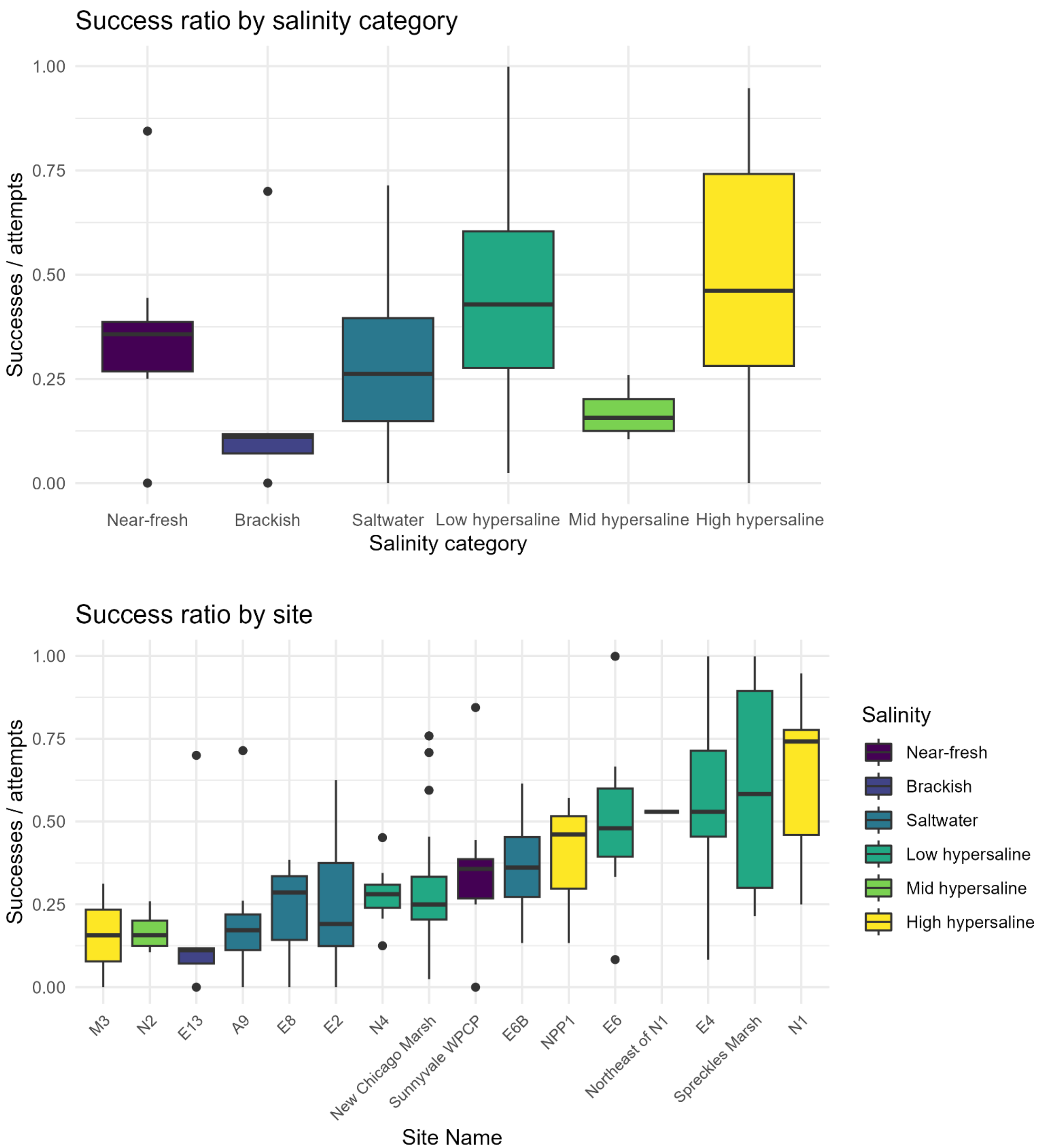


Figure 14. Boxplots of the foraging success ratio (number of successful foraging attempts / total number of foraging attempts in a minute) for phalaropes within South San Francisco Bay, divided by salinity category (top) and site shaded by salinity category (bottom). Dots represent outlier observations.

Water Quality

We here report trends in water quality, averaged across two samples near the beginning of the survey period (July 2 to July 9) and near the end of the survey period (September 4 to October 8). Survey-specific water quality data are reported in Appendix 2. Because of the small number of surveyed ponds within some pond complexes, we report results grouped by the main land managing agency for each set of ponds (plus all other surveyed sites within the baylands).

Don Edwards

Don Edwards NWR contains two pond complexes: the southern Alviso complex, which was characterized by sites with a mix of different salinities, and the northern Ravenswood complex, which was characterized by high salinity ponds to the north and lower salinity ponds to the south in the RSF2 group (with the exception of RSF2U3). Average salinity ranged from 28.39 ppt at A3N to 329.67 ppt at A12. Salinity changes from last year to this year ranged from -29.03 ppt at A3N to +121 ppt at R2 (Fig. 20). The mean interannual change in salinity during this survey round across all sites was +16.71 ppt.

Average dissolved oxygen concentrations in the Don Edwards complex ranged from a low of 3.46 mg/L at A13 to a high of 20.25 mg/L at RSF2U2. Changes in dissolved oxygen since last year ranged from -6.55 mg/L at R2 to +13.18 mg/L at RSF2U2 (Fig. 21). The mean interannual change in dissolved oxygen across all sites was +1.31 mg/L.

Average pH values ranged from a low of 6.99 in R2 to a high of 9.03 in A3N. Changes in pH from last year ranged from -0.41 at A3N to +3.27 at RSF2U2 (Fig. 22). The mean interannual change in pH across all sites was +0.54.

Water temperature at the surface ranged from 22.57 ˚C at A3N to 35.02 ˚C at A12. Compared to last year, water temperature changes in ponds ranged from -3.88 ˚C at RSF2U2 to +9.98 ˚C at A12 (Fig. 23). The mean interannual change in temperature across all sites was +2.22 ˚C.

Staff gauges are present and functional on all ponds. For ponds where readings could be taken, staff gauge levels ranged in the Don Edwards complex from 1.4 ft in A9 to 5.9 ft in RSF2U2. For ponds where staff gauge measurements could be obtained in both years, changes in seasonal water depth from last year ranged from -0.2 ft in RSF2U2 to +0.1 ft in RSF2U2 (Fig. 24). The mean interannual change in water level across all sites was -0.02 ft.

Some water quality information was unavailable for certain surveys at ponds A13 and R2 because they were surveyed on days when they were dry or when water quality equipment was malfunctioning.

Eden Landing

The Eden Landing complex was characterized by mostly marine to low hypersaline salinity, with the exception of E6C which was high hypersaline. Average salinity ranged from 32.42 ppt at E13 to 156.67 ppt at E5. Salinity changes from last year to this year ranged from -11.38 ppt at E5C to +83.28 ppt at E5 (Fig. 20). The mean interannual change in salinity during this survey round across all sites was +12.42 ppt.

Average dissolved oxygen concentrations in the Eden Landing complex ranged from a low of 1.72 mg/L at E5 to a high of 14.32 mg/L at E6A. Changes in dissolved oxygen since last year ranged from -12.8 mg/L at E6A to +9.19 mg/L at E13 (Fig. 21). The mean interannual change in dissolved oxygen across all sites was +0.02 mg/L.

Average pH values ranged from a low of 7.64 in E5 to a high of 9.28 in E6A. Changes in pH from last year ranged from -0.42 at E12 to +1.05 at E2 (Fig. 22). The mean interannual change in pH across all sites was +0.29.

Water temperature at the surface ranged from 19.11 °C at E8 to 31.27 °C at E5. Compared to last year, water temperature changes in ponds ranged from -7.8 °C at E8 to +11 °C at E7 (Fig. 23). The mean interannual change in temperature across all sites was +3.26 °C.

Staff gauges are present and functional on all ponds. For ponds where readings could be taken, staff gauge levels ranged in the Eden Landing complex from 1.7 ft in E6B to 7.2 ft in E12. For ponds where staff gauge measurements could be obtained in both years, changes in seasonal water depth from last year ranged from -1.3 ft in E13 to +0.7 ft in E6A (Fig. 24). The mean interannual change in water level across all sites was -0.18 ft.

Salt Ponds

Within the three active salt production pond complexes, the Coyote Hills complex is generally the least saline (though increasingly saline in the southern ponds), the Dumbarton complex is moderately saline (with salinity increasing as water moves towards the eastern ponds), and the Mowry complex is the most saline (with salinity also increasing as water moves east towards the eastern ponds). Average salinity ranged from 44.89 ppt at M1 to 207.25 ppt at M3. Salinity changes from last year to this year ranged from -101.74 ppt at M1 to +55.1 ppt at M3 (Fig. 20). The mean interannual change in salinity during this survey round across all sites was -0.52 ppt.

Average dissolved oxygen concentrations in the Salt Ponds complex ranged from a low of 2.34 mg/L at N1 to a high of 7.54 mg/L at M1. Changes in dissolved oxygen since last year ranged from -15.55 mg/L at NPP1 to +0.81 mg/L at M1 (Fig. 21). The mean interannual change in dissolved oxygen across all sites was -4.24 mg/L.

Average pH values ranged from a low of 7.9 in M3 to a high of 8.65 in N4. Changes in pH from last year ranged from +0.02 at N4 to +2.88 at N4 (Fig. 22). The mean interannual change in pH across all sites was +0.8.

Water temperature at the surface ranged from 19.53 °C at N4 to 28.17 °C at N4. Compared to last year, water temperature changes in ponds ranged from -4.92 °C at N4 to +4.67 °C at M3 (Fig. 23). The mean interannual change in temperature across all sites was +1.45 °C.

Staff gauges are present and functional on all ponds except M4, M6, N4AB, N4B, and NPP1. For ponds where readings could be taken, staff gauge levels ranged in the Salt Ponds complex from 0.3 ft in N1 to 3.2 ft in M2. For ponds where staff gauge measurements could be obtained in both years, changes in seasonal water depth from last year ranged from -1.9 ft in M1 to +1.4 ft in M3 (Fig. 24). The mean interannual change in water level across all sites was -0.51 ft.

Some water quality information was unavailable for certain surveys at ponds M1, M2, M3, and M3 because they were surveyed on days when they were dry or when water quality equipment was malfunctioning.

Other Baylands

The remaining sites outside of the SBSPRP and salt pond footprints had heterogeneous hydrology and environmental conditions. Staff gauges were not present on any of these sites.

Average salinity ranged from 0.6 ppt at Sunnyvale WPCP to 145 ppt at Alviso Marina. Salinity changes from last year to this year ranged from -40 ppt at Alviso Marina to +132.77 ppt at Alviso Marina (Fig. 20). The mean interannual change in salinity during this survey round across all sites was +22.85 ppt.

Average dissolved oxygen concentrations in the Baylands complex ranged from a low of 4.43 mg/L at New Chicago Marsh to a high of 15.45 mg/L at Sunnyvale WPCP. Changes in dissolved oxygen since last year ranged from -13.59 mg/L at Coyote Hills Regional Park to +11.82 mg/L at New Chicago Marsh (Fig. 21). The mean interannual change in dissolved oxygen across all sites was +0.72 mg/L.

Average pH values ranged from a low of 8.12 in Alviso Marina to a high of 9.54 in Sunnyvale WPCP. Changes in pH from last year ranged from -0.6 at Coyote Hills Regional Park to +1.63 at Sunnyvale WPCP (Fig. 22). The mean interannual change in pH across all sites was +0.42.

Water temperature at the surface ranged from 22.1 ˚C at Coyote Hills Regional Park to 35.9 ˚C at Alviso Marina. Compared to last year, water temperature changes in ponds ranged from -6.68 ˚C at Coyote Hills Regional Park to +14.45 ˚C at New Chicago Marsh (Fig. 23). The mean interannual change in temperature across all sites was +8.21 ˚C.

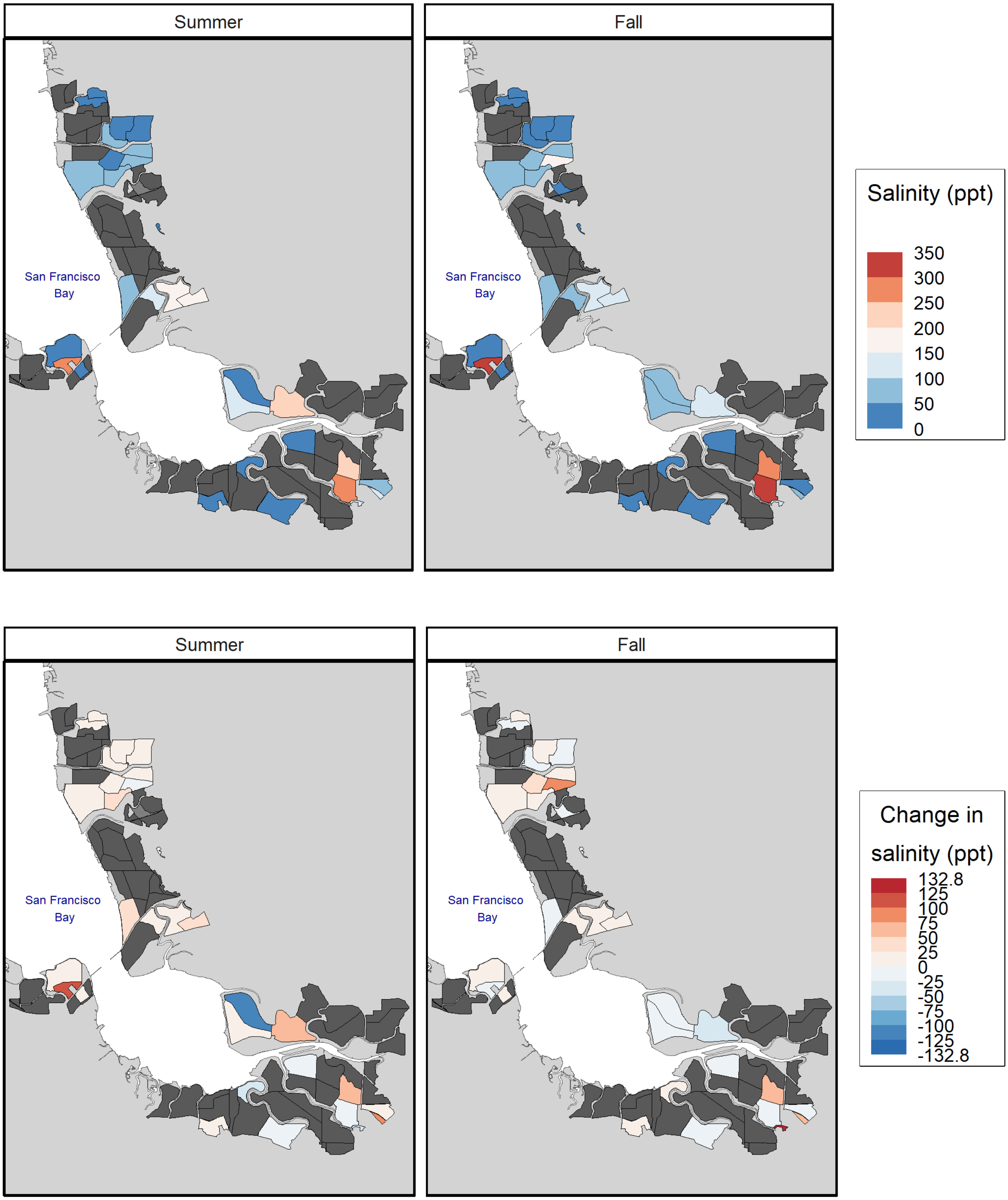


Figure 20. Seasonal salinity (ppt) during July 2 to October 8, 2024 (top panels) and change in salinity (ppt) between 2024 and 2023 (bottom panels) in pond habitats of South San Francisco Bay, California. Dark grey ponds were not measured for this pond characteristic in one or both seasons.

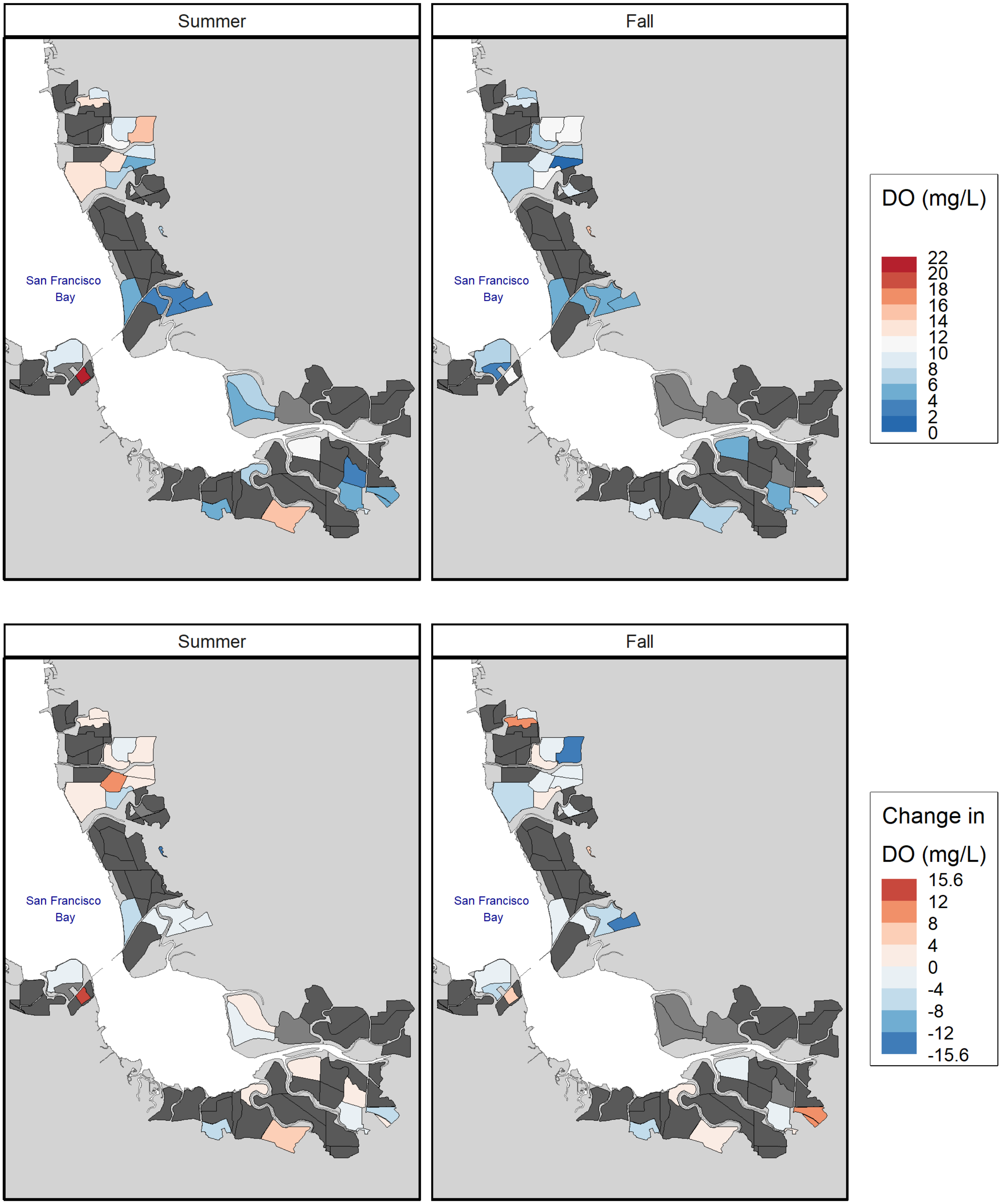


Figure 21. Seasonal dissolved oxygen (mg/L) during July 2 to October 8, 2024 (top panels) and change in dissolved oxygen (mg/L) between 2024 and 2023 (bottom panels) in pond habitats of South San Francisco Bay, California. Dark grey ponds were not measured for this pond characteristic in one or both seasons.

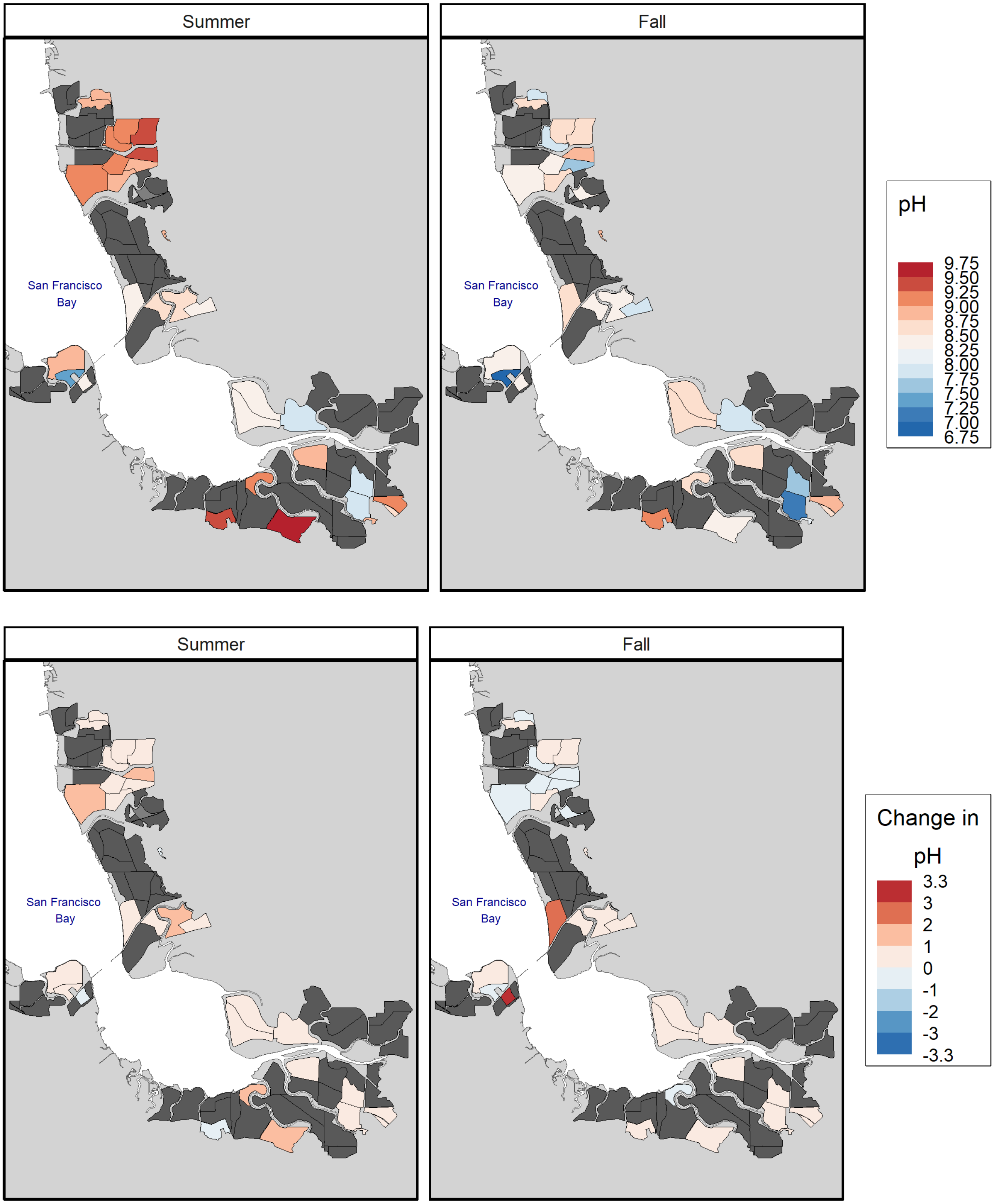


Figure 22. Seasonal pH during July 2 to October 8, 2024 (top panels) and change in pH between 2024 and 2023 (bottom panels) in pond habitats of South San Francisco Bay, California. Dark grey ponds were not measured for this pond characteristic in one or both seasons.

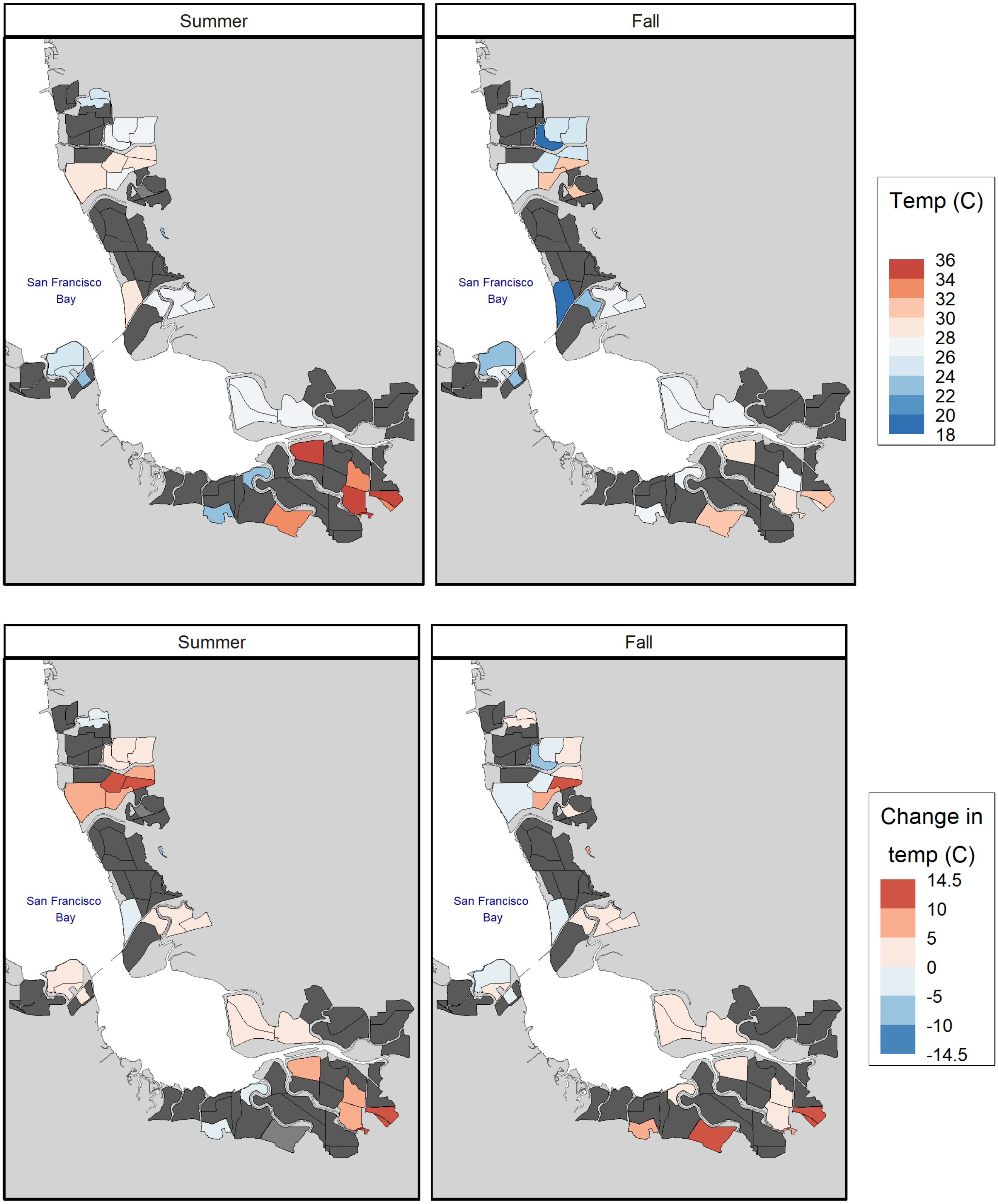


Figure 23. Seasonal temperature (C) during July 2 to October 8, 2024 (top panels) and change in temperature (C) between 2024 and 2023 (bottom panels) in pond habitats of South San Francisco Bay, California. Dark grey ponds were not measured for this pond characteristic in one or both seasons.

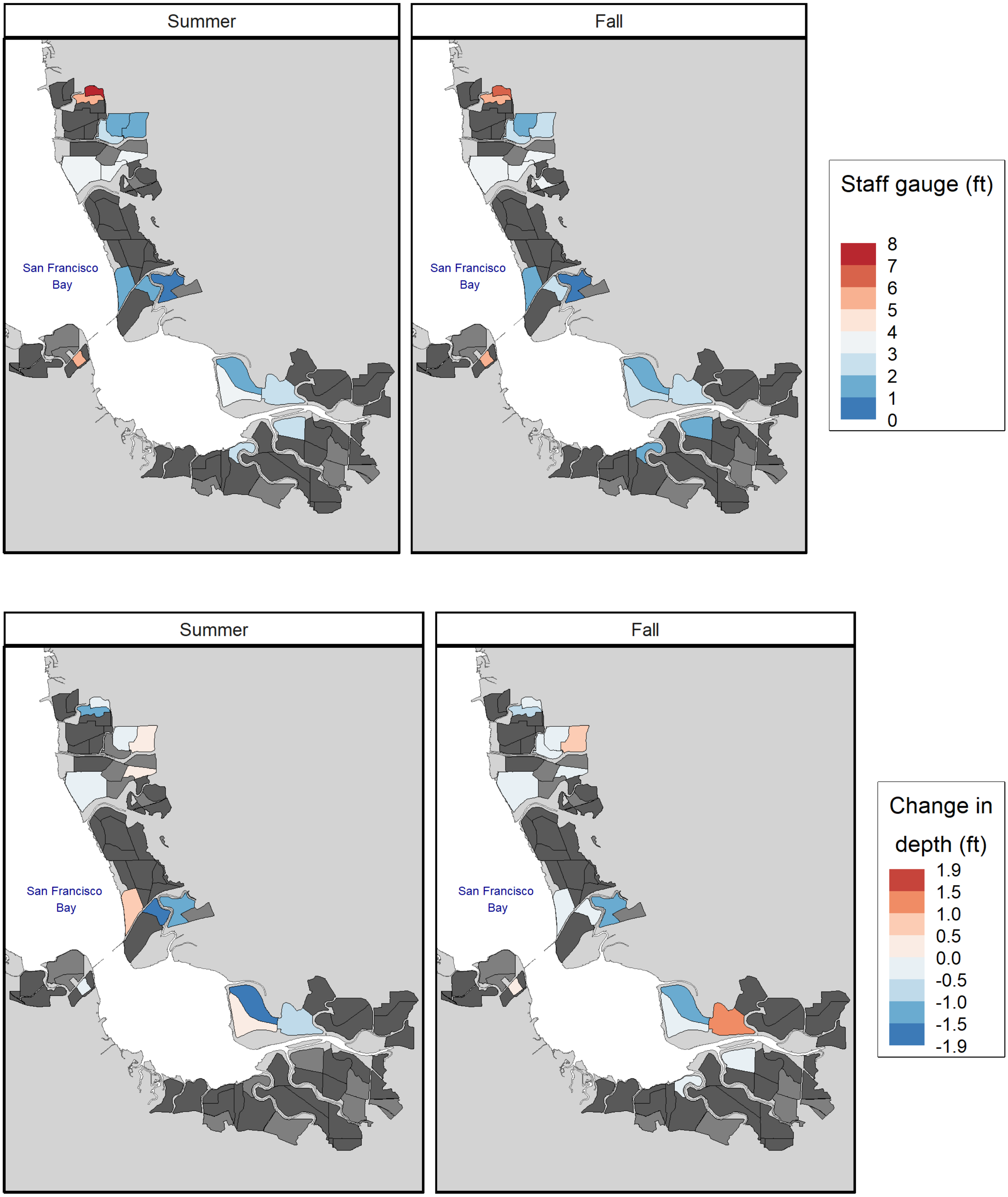


Figure 24. Seasonal staff gauge reading (ft) during July 2 to October 8, 2024 (top panels) and change in staff gauge reading (ft) between 2024 and 2023 (bottom panels) in pond habitats of South San Francisco Bay, California. Dark grey ponds were not measured for this pond characteristic in one or both seasons.

Reanalysis of Phalarope Data

Reanalysis of the set of sites that would target 95% of all historic 2002-2024 counts for all three groupings (Wilson's, Red-necked, and total phalaropes) identified 17 new potential survey sites (Table 8). Taken together, new sites covered 40% of the set of candidate SBSPRP or salt pond sites, accounted for 34% of total sum phalarope sightings, and represented 40% of the interannual peak single-day counts (an alternative indicator of habitat potential; Table 4). The highest priority of these ponds were A7, A15, A16, and N3. Of the three ponds with >100 counts in the past several years that spurred this analysis, A5 and E11 were only rank 2, because they lacked a sufficient number of Wilson's and Red-necked Phalarope sightings, respectively. E10 was not ranked at all, despite the sighting of 153 Red-necked Phalarope in September 2021.

Two currently surveyed sites (A3N and RSF2U2) did not meet any of the four criteria within the historical waterbird survey dataset. During phalarope migration surveys, 1,760 phalaropes sighted at A3N in 2022, meriting its continued inclusion in the survey. However, to date no phalaropes have been observed at RSF2U2 during phalarope migration surveys, with the last sighting of >50 birds within the SBSPRP dataset in 2005. Nevertheless, as a high salinity pond it is in theory phalarope habitat.

Table 12. Ranking of ponds that capture 95% of either Red-necked Phalarope or Wilson's Phalarope sightings from general waterbird surveys within 82 current and former salt ponds of South San Francisco Bay, California, between 2002-2024. "Current" indicates the pond is already included in the set of 31 ponds originally selected for surveys (excluding the 7 sites outside of the pond complexes; Tarjan 2019b), plus E5C (added this year). "Interannual Peak" describes the peak number of sightings during a single survey in any year. [B] indicates a pond that has since been breached to restore tidal action.

Pond	Current	Interannual Peak	Rank	Last 50+ Count	95% of RNPH	95% of WIPH
A7	0	6141	3	2024-05-07	1	1
A13	1	5401	3	2021-10-08	1	1
E5C	1	3738	3	2022-09-20	1	1
R1	1	2490	3	2022-09-06	1	1
N3	0	2253	3	2013-08-21	1	1
E5	1	1952	3	2023-09-12	1	1
N1	1	1656	3	2016-07-18	1	1
A12	1	1653	3	2013-08-08	1	1
M1	1	1390	3	2024-09-23	1	1
A15	0	1165	3	2012-08-08	1	1
E6A	1	1124	3	2024-09-09	1	1
A16	0	744	3	2013-08-09	1	1
E6	1	697	3	2023-09-12	1	1
A5	0	6121	2	2024-05-07	1	0
N4	1	4596	2	2014-09-11	1	0
E6B	1	3334	2	2024-10-16	1	0
E4	1	2611	2	2024-09-05	1	0
M2	1	2521	2	2023-09-12	1	0
N2	1	2120	2	2022-05-19	1	0
A8	0	1694	2	2009-09-04	1	1
E13	1	1121	2	2024-10-08	1	0
E2	1	1056	2	2024-09-17	1	0
M3	1	998	2	2024-09-23	0	1
E7	1	860	2	2024-09-16	1	0
A9	1	678	2	2023-09-11	1	0
E9 [B]	0	674	2	2009-09-01	1	1
E8AE [B]	0	630	2	2010-08-02	1	0
NPP1	1	413	2	2022-05-19	1	0
E11	0	299	2	2019-09-17	0	1
E6C	0	205	2	2013-05-01	0	1
E8	1	191	2	2015-09-08	1	0
E4C	0	2595	1	2009-09-02	1	0
M4	0	1448	1	2012-09-04	0	0
E8X [B]	0	1429	1	2005-08-09	1	0
E3C	0	1116	1	2008-09-04	1	0
E12	1	410	1	2023-09-26	0	0
N9	0	393	1	2008-07-03	0	1
A14	0	155	1	2010-05-13	0	0
E14	0	142	1	2006-05-10	0	1
R2	1	130	1	2019-04-19	0	0
RSF2U2	1	96	0	2005-04-27	0	0
A3N	1	34	0	N/A	0	0

Discussion

Trends

The total counts for phalaropes were markedly lower this year than in any other year of complete surveys (2021-2023), and even lower than the counts in 2020 when less than half as many sites were surveyed. This decline was largely driven by dramatic loss of Red-necked Phalaropes, which declined by 82.7%. Counts of Wilson's Phalarope also declined, but more modestly. Coordinated monitoring of phalarope staging sites in western North America began in 2019 (Carle et al. 2024). Between 2019 and 2022 there was an overall decrease in the range-wide peak counts for both Wilson's and to a lesser extent Red-Necked Phalaropes, but 2023 saw increasing phalarope counts (Carle et al. 2024). A 2024 rangewide report has not yet been drafted, but rangewide trends this year show an additional increase (R. Carle, pers comm.) indicating that the severe decline within South San Francisco Bay was unusual.

Counts of wildlife almost always underestimate the true population size because surveyors either miss birds or birds are not present on the day of the survey. Therefore, the key factor in discussing trends is to determine whether the counts are declining over time, even when accounting for differences in survey effort. While the current report has highlighted that we are missing some phalaropes at some currently unsurveyed sites and the severe decline from last year is therefore likely an overestimate of the true degree of decline, Burns & Van Schmidt (2023) showed that a >50% decline had still occurred even when comparing only abundance within the 24 current and former salt pond sites identified by Tarjan (2019b) to baseline abundance at those same 24 sites. While the current spatial protocol may miss birds at unsurveyed sites, the baseline data also did not include any sites in the Coyote Hills, Dumbarton, and Mowry complexes in the first year (Burns & Van Schmidt 2023; see p. 14-16 of that report for a more detailed discussion of the caveats of these comparisons). Furthermore, the addition of more frequent surveys reduce the uncertainty in whether the birds will be present on the day the site is surveyed. Therefore, we believe the finding of a >50% decline remains robust.

Expansion of Survey Sites

As noted in Burns & Van Schmidt (2023), surveying only the set of 31 sites that had high counts between 2014-2018 years (Tarjan 2019b) risks missing critical counts if other ponds that had substantial phalarope counts in the more distant past remain viable habitat. That report specifically highlighted ponds E5C and the A5-A8 pond cluster as having very high historical counts that were nevertheless not covered by the current set of surveys. In the two years since then, opportunistic observations have demonstrated that this set of 31 sites is likely insufficient to predict which sites phalaropes will use as migratory stopover each summer. This issue is particularly pressing given the apparently marked 82.7% decrease in Red-necked Phalarope sightings this year, to only 2,429 total sightings. To put the potential impact of the missing sites in perspective, the 1,277 Red-necked Phalaropes sighted in May at A5 and A7 represented 53% of this year's total count during migration surveys, while the opportunistic sighting in 2023 of 2,400 Red-necked Phalaropes at E5C would represent nearly the entirety of the year's Red-necked Phalarope total in a single day. Without a more comprehensive picture of ponds that are plausibly occupied but unsurveyed, it is extremely difficult to determine the accuracy of the observed decline. Given these sightings, current estimates of decline are almost certainly overestimates. Accurately accounting for all birds on the landscape is key to both identifying when Adaptive Management Targets for recovery have been met, and improve the quality of data feeding into the models that otherwise may take many years to gain enough statistical samples to accurately disentangle the degree to which weather or management is driving trends.

The dramatic declines and apparent two missed counts of >1,000 individuals strongly suggests a need to expand the set of survey sites. Along with E5C newly added this year, surveying the more comprehensive set of sites identified in this report would bring the total number to 49 sites (a 53% increase in effort). This is likely not feasible without expanded funding, assistance from refuge staff, or a relaxation in the assumption of same-day surveys. Three ranked ponds (E8X, E8AE, and E9) have been breached to restore tidal action and are presumably no longer habitat, and can presumably still be skipped in any expanded survey effort.

Nevertheless, adding a portion of these sites could be more feasible. For example, adding the 4 identified as rank 3, plus A5 and E11 (6 total sites), would only require a more modest 19% increase in survey effort. A8, which was also ranked 2, could be surveyed along with A5 and A7 as a set by a single surveyor on a bike. Alternatively, 1-2 years of intensive efforts to survey the entire set of 42 current and candidate sites would help identify which sites are currently truly habitat, allowing a reduced set of sites to be surveyed again in subsequent years.

At a minimum, we recommendation continuation of phalarope migration surveys at current sites in 2025 and in future years. The current interval of at least every two weeks during the peak migration is unlikely to be able to be relaxed without sacrificing data quality. For guilds that migrate through the area over a short timespan, such as phalaropes, frequent sampling is required during phalarope migration to understand their use of the SBSPRP area and surrounding areas in South San Francisco Bay. The roughly two-week period between each survey for phalaropes was determined by the the Internationl Phalarope Working Group to be the best balance between feasibility and likelihood of avoiding missing birds (Carle et al. 2021). The distribution of count data continues to indicate that current survey schedule largely spans the period when phalaropes are present in the area, but with notable variability in the timing of peak migration across years (Table 1). Annual surveys allow for estimation of year-to-year changes in phalarope abundance that can be compared to changes in other regions to test whether declines are driven by broad-scale annual climatic conditions or local-scale habitat conditions (Coates et al. 2021). Comparisons could also be made between rates within SBSPRP-managed wildlife and tidally breached ponds and rates within remaining active salt ponds and other bayland sites, which have presumably had fewer changes in management. This can provide a robust approach to determining whether management actions within the SBSPRP are a significant factor in the observed ongoing declines, but requires an annual estimate of lambda to account for the high inter-annual variability in population counts. The statistical power of this method increases when exceptional declines are observed in multiple consecutive years. We now have 4 complete years of data and one partial year (2020), which has enabled preliminary models to be developed. While their statistical power remains low due to the high variability of migration, each year of continued surveys will further improve our ability to disentangle significant drivers.

Habitat Associations in the Change Model

The results from this year found statistically significant relationship indicating that as pond pH increased, so too did phalarope abundance (and vice versa), a relationship which was suggested by trends last year as well (Van Schmidt & Parsons 2024). Changes in pH in the long-term 2003–2018 waterbird survey dataset for phalaropes were strongly and inversely correlated with changes in salinity ($R^2 = -0.56$), and therefore this may also indicate an inverse relationship with changes in salinity, as seen in that dataset (Van Schmidt & Parsons 2025). Thus, it may be helpful for managers to consider how pH and salinity change together when altering salinity management of ponds. However, the inverse pH relationship was not observed in the model developed from the historical salt pond survey dataset, while negative relationships with salinity, dissolved oxygen, water depth, and minimum air temperature were. This relationship was also observed in Wilson's Phalaropes, with the third-ranked competitive model (dAICc = 1.90) showing positive a relationship with pH (Table A1.6). Unlike last year, none of the three final selected models showed that phalaropes were responding positively to changes in the percent of the pond that was flooded with open water increased. However, our statistical power to detect relationships was weak, especially for Wilson's Phalaropes which had only 23 total cases of changes.

We found a direct relationship between greater precipitation in the focal water year and increases in Wilson's Phalaropes; however, this was opposite of our findings from last year, which suggested populations declining in response to greater precipitation in the previous three years (Van Schmidt & Parsons 2024). While this could be interpreted to suggest that current population trends are primarily driven by precipitation, we again caution that this model is very preliminary and the high annual variability in both Wilson's Phalarope trends and precipitation in the Bay Area may be spurious. Many covariates could not be included together in the same model at all due to high collinearity (Table A1.1).

We conducted a preliminary model of the relationship between site-scale inter-annual population trends and habitat and climate variables. However, it is important to note that due to the very small sample sizes of complete observations of both phalarope population sizes and water quality variables, results from this model should be interpreted very cautiously. Statisticians recommended sample sizes have at least 10 observations per covariate assessed, so we do not yet have adequate sample sizes to conduct a full model. It is critically important to maintain annual surveys because a single missed year of either water quality sampling or waterbird counts renders not only the data in that year unusable for modeling drivers of changes in abundance, but also data in the previous and subsequent year.

Assuming we continue this concerted effort to conduct water quality sampling at all sites going forward, we would expect at least 20 additional usable observations of population change each year. As predicted last year, this year we were able to fit a model properly assessing all six of the "change in water quality" variables for total phalaropes and Red-necked Phalaropes, though Wilson's Phalaropes rarity meant they did not add as many sample sizes. In three more years, a model should be able to also assess the five climate variables. However, a model assessing these and the raw water quality parameters (i.e., "Salinity" instead of "change in salinity") and their interactions will require over a decade of surveying. Surveying additional ponds described above would improve the rate of annual sample collection to hasten this process.

Improved statistical power could be achieved more quickly by developing a data fusion model that includes both historic waterbird counts (Van Schmidt & Parsons 2023) and contemporary migration surveys. Due to differences in survey protocols, this would require a more complex Bayesian hierarchical model that could directly estimate the underlying abundances (the "process model") that underlie the raw counts (the "observation model"). SFBBO has obtained funds to partner with Oikonos to develop a Bayesian model for rangewide surveys over the course of 2025, which could form the basis of a local model for South San Francisco Bay.

Foraging Ecology

As migratory stopover habitat, the factors that influence phalarope site preferences within the salt pond project area are likely driven not directly by site conditions but by their effect on prey availability. Understanding these relationships is key in order to inform management decisions and develop plans to recover these species. Management actions in comparable ponds in North San Francisco Bay have been shown to significantly shift invertebrate biomass with direct links to changing waterbird habitat use (Athearn et al. 2009). Results from our population change model indicate that phalarope increases have occurred where salinity has fallen (and vice versa), contrary to expectations for this saline habitat specialist (Van Schmidt and Parsons 2023). These observations have significant management implications and merit further investigation. Prey abundance might more directly correlate with phalarope site selection than salinity, dissolved oxygen, pH, water temperature, and water depth, which are likely underlying factors that drive phalarope population trends because they support prey abundance.

In this study, we reassessed and managed to reconcile these confounding patterns to some degree. While phalaropes are selecting for hypersaline habitat (an exception being Sunnyvale WPCP), they are not selecting for high hypersaline habitats (>150 ppt). Rather, they seem to show a marked preference for marine to moderately hypersaline environments (30-150 ppt). E4 was the most-used site and one of the few sites used highly both by Wilson's and Red-necked Phalaropes. Its salinity value was 70.66 ppt (Table A2.1), in the "low hypersaline" range that showed the higher foraging attempts and success rates, though it was on the lower side compared to other low hypersaline sites. It notably had the third-highest average success ratio. These patterns align well with observations of phalaropes at other sites and knowledge about the optimal salinity conditions of their hypothesized main food source—*Ephydra* and potentially *Artemia* spp. It remains surprising that Wilson's Phalaropes appear to be selecting for a lower optimal salinity than Red-necked Phalaropes; they may be foraging on *E. millbrae* in the marine-salinity ponds, which disappear above 42 ppt (though which in turn may be replaced by *E. cinerea* at these higher salinities. The Sunnyvale WPCP pond was the only non-hypersaline pond that was used that had high foraging rates, but the invertebrate community at this site is likely unique to the high nutrients at this wastewater site and therefore not representative of general habitat needs. However, the bottom two salinity bins only had a single pond in each with behavioral observations in this study, limiting the robustness of our statistical conclusions and necessitating statistical corrections to p-value calculations due to this unbalanced sample. A second year of data is likely to significantly strengthen the robustness of these findings.

We also conducted pilot sampling of six ponds in a paired design of salinity bins and historical phalarope abundance: RSF2U2 (low salinity / low phalarope abundance), A9 (low salinity / high abundance), N2 (moderate salinity / low abundance), N1 (moderate salinity / high abundance), R2 (high salinity / low abundance), and M3 (high salinity / high abundance). The samples have not yet been processed, but the pilot effort confirmed the feasibility of collecting and processing macroinvertebrates from the water column following the methods of Keith et al. (2022), preserving samples in the lab, and counting and identifying them as a first step toward assessing their abundance and diversity. In total, preparation, fieldwork, and lab work took an average of 3 hours per site.

Movement Ecology

Additional study would be useful to investigate the stopover time of phalaropes in the Bay Area during migration. This would make it possible to estimate the likelihood of counting the same individuals across multiple surveys, enabling the instantaneous counts produced by the surveys to be better converted to true population size estimates. There is some evidence that suggests birds may switch from using one site to another during years when conditions are less favorable (Carle et al. 2022), but research has yet to confirm the frequency or scope of this behavior. Wilson's Phalaropes fitted with satellite tags by researchers in Tule Lake in summer 2023 were detected migrating through South San Francisco Bay, though it is unclear whether they used it as a stopover point (Tarr & Carle 2023).

One avenue of investigation would be to analyze data from Motus towers that have recently been or are currently being installed around the Bay Area. Motus tags have been successfully deployed by researchers on phalaropes for the past two summers (Tarr & Carle 2023). Their success suggests that boat captures and spotlighting at night may be effective. These birds continued on to Baja California, with a distinctly different migratory pathway than other phalaropes tagged in Tule Lake that continued inland via Mono Lake to central Mexico. SFBBO has installed one Motus tower near the Alviso ponds and is currently working to install two other towers in the South Bay. If some of the tagged birds migrate through the Bay Area and are detected by one of the new Motus towers, it will give us insight into their stopover time. Because of the now low number of Wilson's Phalarope in San Francisco Bay compared to Mono Lake, relying on birds tagged elsewhere to elucidate movement between these two potentials paths may be an unreliable plan.

Tagging phalaropes in San Francisco Bay will likely be much more effective at determining how (and how frequently) phalaropes move between San Francisco Bay and other stopover sites (e.g. Mono Lake) within and between migrations. However, at this time we recommend prioritizing expanding monitoring sites and studies on invertebrate prey to better understand local habitat use dynamics. It will be more fruitful to attempt capture, tagging, and tracking once phalarope tagging methods have been better developed on their main staging grounds and the Motus network has been expanded.

Management Recommendations

After controlling for protocol differences between these surveys and those used to compute the NEPA/CEQA baseline, the 2024 counts are 91.6% below the baseline, which is below the trigger value of 50% below baseline (LaBarbera et al. 2023). A link between SBSPRP management actions and ongoing declines has not been scientifically proven, but the precautionary principle suggests conservative management to prevent further declines until such a statistical test has been conducted. Significant work has been completed in recent years to contextualize the declines within South San Francisco Bay with historical trends throughout California (Tarjan 2019a, LaBarbera et al. 2023), develop a model that can relate the observed decline in the early years of the SBSPRP to habitat and climate covariates (Van Schmidt & Parsons 2023), and we have here extended these models to be species-specific and assess the phalarope migration survey data. These efforts have begun to identify optimal conditions for the phalaropes that remain within South Bay, though the cause of their decline remains somewhat unclear given many appear to be selecting for a wide range of marine to moderately hypersaline habitat that still exist.

In order for the South Bay to retain its current bird numbers, we recommend that the SBSPRP's Project Management Team, the Refuge, and Eden Landing Ecological Reserve consider managing ponds to support use by phalaropes to address declines at SBSPRP sites. Based on the current report, we recommend managers maintain ponds used by phalaropes at a moderate level of salinity (30-120 ppt), with the low hypersaline range (50-80 ppt) potentially the safest bet given it is in the middle of this broader range. Until these relationships can be thoroughly interrogated with additional analyses (i.e., predictive models or decomposing the relative contribution of each variable to observed trends), we still recommend being cautious with changes to phalarope habitats.

Special consideration should be given to ponds that have hosted large numbers of phalaropes in recent years (e.g., E4, E6, and A13) when making management decisions such as seasonal adjustments to water level. Due to the substantial uncertainty in phalarope habitat selection, we suggest avoiding changes to these ponds whenever possible, and generally managing for conditions in each pond that were similar to conditions in the years where its count was highest. Leaving sufficient suitable habitat for phalaropes should also be a consideration when making plans around which sites will be maintained as managed ponds or converted for tidal marsh restoration. In some cases, it may not be practical to alter management to support phalaropes due to the complex system of water intake structures that link the ponds (e.g., E6 and E5 depend on circulation from E1 and E2) and the slate of tidal breach developments (e.g., A13 is already undergoing construction). When continuing with the design of subsequent phases of the SBSPRP, it could be useful to identify opportunities to alter hydrological management structures to increase the flexibility of water level and salinity management across the different ponds. This could improve the responsiveness of management to species' needs.

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Appendix 1

This appendix presents tables summarizing the model selection procedure and results for the regression models relating inter-annual changes in total phalarope sightings to changes in habitats and weather.

Table A1.1. List of variables that were too highly correlated (correlation coefficient >0.7) to be included in the same models for a given phalarope species/guild within South San Francisco Bay, California, summer 2021-2024. Variables 1 and 2 are listed; any models within the set that included both were dropped from all analysis for that species/guild.

Species/Guild	Var1	Var2
PHAL	LDO_mgL	PPT3
PHAL	LandfillDist_km	OpenHunting_pct
PHAL	LDO_mgL_AC	PPT0
PHAL	PPT0	TMX
PHAL	PPT1	PPT3
PHAL	PPT1	TMN
RNPH	LDO_mgL	PPT3
RNPH	LandfillDist_km	OpenHunting_pct
RNPH	LDO_mgL_AC	PPT0
RNPH	PPT0	TMX
RNPH	PPT1	PPT3
RNPH	PPT1	TMN
WIPH	Salinity_AC	Temp_C
WIPH	LDO_mgL	PPT3
WIPH	LandfillDist_km	OpenHunting_pct
WIPH	LDO_mgL_AC	PPT0
WIPH	LDO_mgL_AC	TMX
WIPH	Salinity_AC	Temp_C_AC
WIPH	PPT0	TMX
WIPH	PPT1	PPT3
WIPH	PPT1	TMN

Table A1.2. Model selection table for base model of pond characteristics related to change in abundance of phalaropes during summer in current and former salt ponds in South San Francisco Bay, California, summer 2021-2024. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only the top 20 models are shown.

Formula	K	dAICc	AICc	-2LL	w
Abun_PC ~ 1	2	0.00	348.41	-172.11	0.15
Abun_PC ~ LandfillDist_km	3	2.14	350.55	-172.07	0.05
Abun_PC ~ OpenHunting_pct	3	2.15	350.56	-172.08	0.05
Abun_PC ~ OpenPublic_pct	3	2.15	350.56	-172.08	0.05
Abun_PC ~ Area_km2	3	2.20	350.61	-172.10	0.05
Abun_PC ~ BayDist_km	3	2.20	350.61	-172.10	0.05
Abun_PC ~ CreekDist_km	3	2.20	350.61	-172.10	0.05
Abun_PC ~ Islands_num	3	2.21	350.62	-172.11	0.05
Abun_PC ~ Area_km2 + LandfillDist_km	4	4.38	352.79	-172.05	0.02
Abun_PC ~ Area_km2 + OpenHunting_pct	4	4.40	352.81	-172.06	0.02
Abun_PC ~ BayDist_km + OpenPublic_pct	4	4.40	352.82	-172.06	0.02
Abun_PC ~ LandfillDist_km + OpenPublic_pct	4	4.41	352.82	-172.06	0.02
Abun_PC ~ CreekDist_km + LandfillDist_km	4	4.41	352.82	-172.06	0.02
Abun_PC ~ CreekDist_km + OpenHunting_pct	4	4.41	352.82	-172.07	0.02
Abun_PC ~ CreekDist_km + OpenPublic_pct	4	4.41	352.83	-172.07	0.02
Abun_PC ~ OpenHunting_pct + OpenPublic_pct	4	4.42	352.83	-172.07	0.02
Abun_PC ~ BayDist_km + LandfillDist_km	4	4.42	352.83	-172.07	0.02
Abun_PC ~ Islands_num + LandfillDist_km	4	4.42	352.83	-172.07	0.02
Abun_PC ~ BayDist_km + OpenHunting_pct	4	4.42	352.84	-172.07	0.02
Abun_PC ~ Islands_num + OpenHunting_pct	4	4.43	352.84	-172.07	0.02

Table A1.3. Model selection table for base model of pond characteristics related to change in abundance of Red-necked Phalaropes during summer in current and former salt ponds in South San Francisco Bay, California, summer 2021-2024. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only the top 20 models are shown.

Formula	K	dAICc	AICc	-2LL	w
Abun_PC ~ 1	2	0.00	351.54	-173.67	0.12
Abun_PC ~ OpenPublic_pct	3	1.35	352.89	-173.24	0.06
Abun_PC ~ LandfillDist_km	3	1.89	353.43	-173.51	0.05
Abun_PC ~ OpenHunting_pct	3	1.93	353.47	-173.53	0.05
Abun_PC ~ BayDist_km	3	2.01	353.56	-173.58	0.04
Abun_PC ~ Islands_num	3	2.07	353.61	-173.60	0.04
Abun_PC ~ Area_km2	3	2.12	353.66	-173.63	0.04
Abun_PC ~ CreekDist_km	3	2.18	353.72	-173.66	0.04
Abun_PC ~ CreekDist_km + OpenPublic_pct	4	3.37	354.92	-173.11	0.02
Abun_PC ~ LandfillDist_km + OpenPublic_pct	4	3.61	355.15	-173.23	0.02
Abun_PC ~ Area_km2 + OpenPublic_pct	4	3.61	355.15	-173.23	0.02
Abun_PC ~ Islands_num + OpenPublic_pct	4	3.63	355.17	-173.24	0.02
Abun_PC ~ BayDist_km + OpenPublic_pct	4	3.63	355.17	-173.24	0.02
Abun_PC ~ OpenHunting_pct + OpenPublic_pct	4	3.63	355.17	-173.24	0.02
Abun_PC ~ Area_km2 + LandfillDist_km	4	3.88	355.42	-173.37	0.02
Abun_PC ~ Islands_num + LandfillDist_km	4	3.91	355.45	-173.38	0.02
Abun_PC ~ Islands_num + OpenHunting_pct	4	3.95	355.49	-173.40	0.02
Abun_PC ~ BayDist_km + LandfillDist_km	4	3.96	355.50	-173.41	0.02
Abun_PC ~ Area_km2 + OpenHunting_pct	4	3.97	355.52	-173.41	0.02
Abun_PC ~ BayDist_km + OpenHunting_pct	4	4.03	355.57	-173.44	0.02

Table A1.4. Model selection table for interannual change in abundance of phalaropes during summer in current and former salt ponds in South San Francisco Bay, California, summer 2021-2024. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only the top 20 models are shown. Site characteristics were not subject to model selection (see text for details), but were instead obtained from De La Cruz et al. (2018).

Formula	K	dAICc	AICc	-2LL	w
Abun_PC ~ pH_AC + TMN	4	0.00	343.58	-167.45	0.01
Abun_PC ~ pH_AC + PPT1	4	0.01	343.59	-167.45	0.01
Abun_PC ~ pH_AC + PPT1 + TMX	5	0.91	344.50	-166.72	0.01
Abun_PC ~ pH_AC + PPT1 + Salinity_AC	5	1.31	344.89	-166.92	0.01
Abun_PC ~ pH_AC + Salinity_AC + TMN	5	1.43	345.02	-166.98	0.01
Abun_PC ~ pH_AC + PPT3 + TMN	5	1.44	345.02	-166.98	0.01
Abun_PC ~ pH_AC + PPT0 + TMN	5	1.82	345.40	-167.17	0.01
Abun_PC ~ LDO_mgL_AC + pH_AC + TMN	5	1.89	345.47	-167.21	0.01
Abun_PC ~ LDO_mgL + pH_AC + TMN	5	1.99	345.57	-167.26	0.01
Abun_PC ~ pH_AC + Temp_C + TMN	5	2.18	345.76	-167.36	0.00
Abun_PC ~ LDO_mgL + pH_AC + PPT1	5	2.21	345.79	-167.37	0.00
Abun_PC ~ pH_AC + TMN + WaterElev_m_AC	5	2.25	345.83	-167.39	0.00
Abun_PC ~ LDO_mgL_AC + pH_AC + PPT1	5	2.25	345.83	-167.39	0.00
Abun_PC ~ pH_AC + Salinity + TMN	5	2.26	345.84	-167.39	0.00
Abun_PC ~ BreachOrGate + pH_AC + PPT1	5	2.28	345.86	-167.41	0.00
Abun_PC ~ pH_AC + Temp_C_AC + TMN	5	2.30	345.88	-167.41	0.00
Abun_PC ~ BreachOrGate + pH_AC + TMN	5	2.32	345.90	-167.42	0.00
Abun_PC ~ pH + pH_AC + TMN	5	2.32	345.90	-167.42	0.00
Abun_PC ~ pH_AC + PPT1 + Salinity	5	2.33	345.92	-167.43	0.00
Abun_PC ~ pH_AC + PPT1 + Temp_C	5	2.34	345.92	-167.43	0.00

Table A1.6. Model selection table for interannual change in abundance of Red-necked Phalaropes during summer in current and former salt ponds in South San Francisco Bay, California, summer 2021-2024. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only the top 20 models are shown. Site characteristics were not subject to model selection (see text for details), but were instead obtained from De La Cruz et al. (2018).

Formula	K	dAICc	AICc	-2LL	w
Abun_PC ~ pH_AC + TMN	4	0.00	345.81	-168.56	0.01
Abun_PC ~ pH_AC + PPT3 + TMN	5	0.96	346.78	-167.86	0.01
Abun_PC ~ pH_AC + PPT1	4	0.97	346.79	-169.05	0.01
Abun_PC ~ pH_AC + PPT0 + TMN	5	1.18	346.99	-167.97	0.01
Abun_PC ~ pH_AC + PPT3 + TMX	5	1.24	347.06	-168.00	0.01
Abun_PC ~ pH_AC + Salinity_AC + TMN	5	1.26	347.08	-168.01	0.01
Abun_PC ~ LDO_mgL_AC + pH_AC + TMN	5	1.38	347.20	-168.07	0.01
Abun_PC ~ pH_AC + PPT1 + TMX	5	1.44	347.25	-168.10	0.01
Abun_PC ~ LDO_mgL + pH_AC + TMN	5	1.60	347.41	-168.18	0.00
Abun_PC ~ pH_AC + PPT1 + Salinity_AC	5	2.00	347.82	-168.38	0.00
Abun_PC ~ pH_AC + TMN + TMX	5	2.14	347.95	-168.45	0.00
Abun_PC ~ Salinity_AC + TMN	4	2.15	347.96	-169.64	0.00
Abun_PC ~ pH_AC + Salinity + TMN	5	2.19	348.00	-168.47	0.00
Abun_PC ~ TMN	3	2.22	348.03	-170.81	0.00
Abun_PC ~ pH_AC + Salinity + Salinity_AC + TMN	6	2.25	348.07	-167.28	0.00
Abun_PC ~ pH + pH_AC + TMN	5	2.30	348.11	-168.53	0.00
Abun_PC ~ BreachOrGate + pH_AC + TMN	5	2.31	348.12	-168.53	0.00
Abun_PC ~ pH_AC + TMN + WaterElev_m_AC	5	2.32	348.14	-168.54	0.00
Abun_PC ~ pH_AC + Temp_C_AC + TMN	5	2.34	348.16	-168.55	0.00
Abun_PC ~ PPT3 + TMX	4	2.36	348.17	-169.74	0.00

Table A1.7. Model selection table for interannual change in abundance of Wilson's Phalaropes during summer in current and former salt ponds in South San Francisco Bay, California, summer 2021-2024. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only the top 20 models are shown. Site characteristics were not subject to model selection (see text for details), but were instead obtained from De La Cruz et al. (2018).

Formula	K	dAICc	AICc	-2LL	w
Abun_PC ~ PPT0	3	0.00	101.93	-47.33	0.15
Abun_PC ~ PPT3 + TMN	4	0.88	102.81	-46.29	0.10
Abun_PC ~ pH_AC + PPT0	4	1.90	103.83	-46.80	0.06
Abun_PC ~ PPT0 + Salinity_AC	4	2.86	104.79	-47.28	0.04
Abun_PC ~ PPT0 + Temp_C	4	2.89	104.82	-47.30	0.04
Abun_PC ~ PPT0 + WaterElev_m_AC	4	2.90	104.83	-47.30	0.04
Abun_PC ~ PPT0 + Salinity	4	2.92	104.84	-47.31	0.04
Abun_PC ~ BreachOrGate + PPT0	4	2.93	104.86	-47.32	0.03
Abun_PC ~ LDO_mgL + PPT0	4	2.94	104.86	-47.32	0.03
Abun_PC ~ PPT0 + TMN	4	2.94	104.87	-47.32	0.03
Abun_PC ~ pH + PPT0	4	2.96	104.88	-47.33	0.03
Abun_PC ~ PPT0 + Temp_C_AC	4	2.96	104.88	-47.33	0.03
Abun_PC ~ LDO_mgL_AC	3	3.14	105.07	-48.90	0.03
Abun_PC ~ TMX	3	4.15	106.08	-49.41	0.02
Abun_PC ~ PPT3 + TMX	4	4.87	106.80	-48.29	0.01
Abun_PC ~ PPT3 + Temp_C	4	5.20	107.12	-48.45	0.01
Abun_PC ~ pH_AC + TMX	4	5.32	107.24	-48.51	0.01
Abun_PC ~ LDO_mgL_AC + PPT3	4	5.41	107.34	-48.56	0.01
Abun_PC ~ LDO_mgL_AC + pH_AC	4	5.58	107.51	-48.64	0.01
Abun_PC ~ TMN	3	5.75	107.68	-50.21	0.01

Appendix 2

This appendix presents a table and line graphs (one per set of sites) of the mean water quality measures during summer (July 2 to July 9) and early fall (September 4 to October 8) of 2024.

Table A2.1. Annual and seasonal water quality and depth measures (mean across surveys), and change in those measures compared to the same time period last year, in current and former salt ponds in South San Francisco Bay, California, 2024.

Location	Pond	Season	Salinity (ppt)	DO (mg/L)	pH	Temp (C)	Staff gage (ft)	ΔSalinity (ppt)	ΔDO (mg/L)	ΔpH	ΔTemp (C)	ΔStaff gauge (ft)
Baylands	Alviso Marina	Summer	112.00	8.63	9.00	35.90		-40.00	-2.39	0.79	13.41	
	Alviso Marina	Fall	145.00	6.25	8.12	29.73		132.77	0.16	0.05	8.09	
	Coyote Hills Regional Park	Summer	4.87	7.38	8.91	22.10		-2.91	-13.59	-0.60	-6.68	
	Coyote Hills Regional Park	Fall	23.28	15.06	8.83	28.42		22.47	7.57	0.76	8.38	
	Crittenden M.	Summer	12.98	5.22	9.39	22.62		2.16	-4.40	-0.35	-1.78	
	Crittenden M.	Fall	24.30	9.86	9.13	26.26		7.72	-5.46	0.45	5.66	
	New Chi. Marsh	Summer	79.80	4.43	9.14	34.10		15.65	-6.45	0.45	14.45	
	New Chi. Marsh	Fall	34.30	12.83	8.95	31.14		-3.97	11.82	0.38	12.88	
	Northeast of N1	Summer										
	Northeast of N1	Fall										
	Spreckles Marsh	Summer	133.00	4.94	8.54	33.75		90.05	1.82	0.83	14.25	
	Spreckles Marsh	Fall	94.40	8.55	8.59	29.43		51.91	8.24	0.18	10.61	
	Sunnyvale WPCP	Summer	1.11	15.45	9.54	33.18		-1.12	7.83	1.63		
	Sunnyvale WPCP	Fall	0.60	6.64	8.33	31.30		-0.49	3.49	0.53	11.04	
Don Edwards	A12	Summer	288.50	4.14	7.85	35.02		-7.50	-0.31	0.91	9.98	
	A12	Fall	329.67	4.69	7.18	29.82		-3.33	-1.52	0.75	1.62	
	A13	Summer	233.00	3.46	7.87	33.35		75.00	1.21	0.40	9.39	
	A13	Fall	288.00		7.58	27.51		59.00		0.19	1.14	
	A3N	Summer	28.39	6.47	9.03	22.57	2.2	-29.03	2.27	1.02	-0.70	
	A3N	Fall	47.55	10.89	8.62	27.97	1.8	8.03	3.56	-0.41	3.30	0.00
	A9	Summer	33.58	10.42	8.95	34.06	2.5	-5.49	1.39	0.16	6.87	
	A9	Fall	41.18	5.47	8.54	29.29	1.4	-0.42	-0.19	0.32	1.42	0.00
	R1	Summer	32.38	9.98	8.90	25.53		7.39	-1.49	0.51	0.89	
	R1	Fall	38.71	7.08	8.41	23.64		7.03	-0.81	0.37	-0.67	
	R2	Summer	292.00		7.31	25.74		121.00		0.31	0.13	
	R2	Fall	306.50	3.47	6.99	26.78		-6.83	-6.55	-0.13	0.67	
	RSF2U2	Summer	28.63	20.25	8.27	23.47	5.9	5.86	13.18	-0.14	0.88	-0.20
	RSF2U2	Fall	32.23	11.27	8.30	22.64	5.9	3.18	5.01	3.27	-3.88	0.10

Location	Pond	Season	Salinity (ppt)	DO (mg/L)	pH	Temp (C)	Staff gage (ft)	ΔSalinity (ppt)	ΔDO (mg/L)	ΔpH	ΔTemp (C)	ΔStaff gauge (ft)
Eden	E12	Summer	49.02	9.22	8.79	24.34	7.2	12.79	0.02	0.48	-0.31	-0.30
Landing	E12	Fall	42.28	6.80	7.88	25.05	6.7	5.04	-1.59	-0.42	1.55	-0.30
	E13	Summer	32.42	12.14	8.78	24.92	5.5	6.68	0.48	0.36	-2.06	-1.30
	E13	Fall	35.67	9.51	8.64	25.63	5.7	-1.77	9.19	0.79	2.13	-0.90
	E2	Summer	51.06	12.68	9.13	29.64	3.2	18.77	3.26	1.05	7.84	0.00
	E2	Fall	52.13	7.50	8.48	26.14	3.3	1.71	-4.15	-0.25	-0.01	-0.10
	E4	Summer	70.66	6.28	8.88	27.91	3.2	37.17	-4.09	0.56	7.83	
	E4	Fall	70.66	11.72	8.75	31.06	3.1	20.77	3.35	0.23	8.37	
	E5	Summer	91.20	5.25	8.96	29.22	3.7	-9.46	0.39	0.50	10.63	0.50
	E5	Fall	156.67	1.72	7.64	31.27	3.5	83.28	-3.16	-0.42	10.19	-0.10
	E5C	Summer										
	E5C	Fall	41.24	8.87	8.40	30.13	3.8	-11.38	-0.82	-0.15	4.11	
	E6	Summer	69.76	8.10	9.27	28.21		23.42	3.28	1.01	9.41	
	E6	Fall	97.07	6.30	8.88	25.33		0.30	-3.20	-0.22	0.23	
	E6A	Summer	36.24	14.32	9.28	27.23	1.9	3.04	2.70	0.51	1.53	0.20
	E6A	Fall	38.56	10.32	8.62	24.42	2.1	-0.80	-12.80	0.05	4.25	0.70
	E6B	Summer	44.70	8.82	9.11	26.73	1.7	7.02	-0.19	0.65	3.85	0.00
	E6B	Fall	43.45	10.08	8.66	24.08	1.9	8.69	-1.83	0.19	-2.81	-0.40
	E7	Summer	49.08	12.60	9.04	29.82		2.47	8.34	0.85	11.00	
	E7	Fall	78.50	9.28	8.49	24.78		35.56	-1.81	-0.21	-3.56	
	E8	Summer	50.95	10.01	9.12	27.28	2.6	18.15	2.54	0.96	2.12	
	E8	Fall	35.12	7.92	7.92	19.11	2.9	-0.60	0.41	-0.37	-7.80	-0.30
Salt	M1	Summer	44.89	7.54	8.45	26.88	1.2	-101.74	0.81	0.84	3.87	-1.90
Ponds	M1	Fall	67.97		8.59	26.73	1.7	-16.70		0.58	1.65	-1.30
	M2	Summer	148.00	4.08	8.37	27.31	3.2	1.70	-3.30	0.77	2.54	0.20
	M2	Fall	68.97		8.63	27.02	2.8	-14.43		0.83	1.89	-0.10
	M3	Summer	207.25		7.90	27.65	2.1	55.10		0.52	1.87	-0.75
	M3	Fall	132.25		7.96	27.90	2.9	-47.25		0.53	4.67	1.40
	N1	Summer	166.00	2.34	8.60	27.39	0.5	13.75	-3.24	1.51	3.35	-1.20
	N1	Fall	115.33	4.70	8.37	26.09	0.3	11.33	-6.93	0.54	0.09	-1.00
	N2	Summer	136.50	3.79	8.62	26.48	1.7	6.09	-1.78	0.40	1.18	-1.80
	N2	Fall	93.70	5.86	8.39	22.29	2.7	16.17	-0.56	0.42	0.90	-0.30
	N4	Summer	91.73	5.49	8.33	28.17	1.9	33.99	-7.51	0.02	-0.02	0.90
	N4	Fall	51.84	5.38	8.65	19.53	1.7	-1.30	-2.32	2.88	-4.92	-0.30
	NPP1	Summer	179.00	2.69	8.44	27.38		26.00	-2.05	0.99	3.16	
	NPP1	Fall	132.50	4.53	7.98	26.79		10.00	-15.55	0.32	0.13	

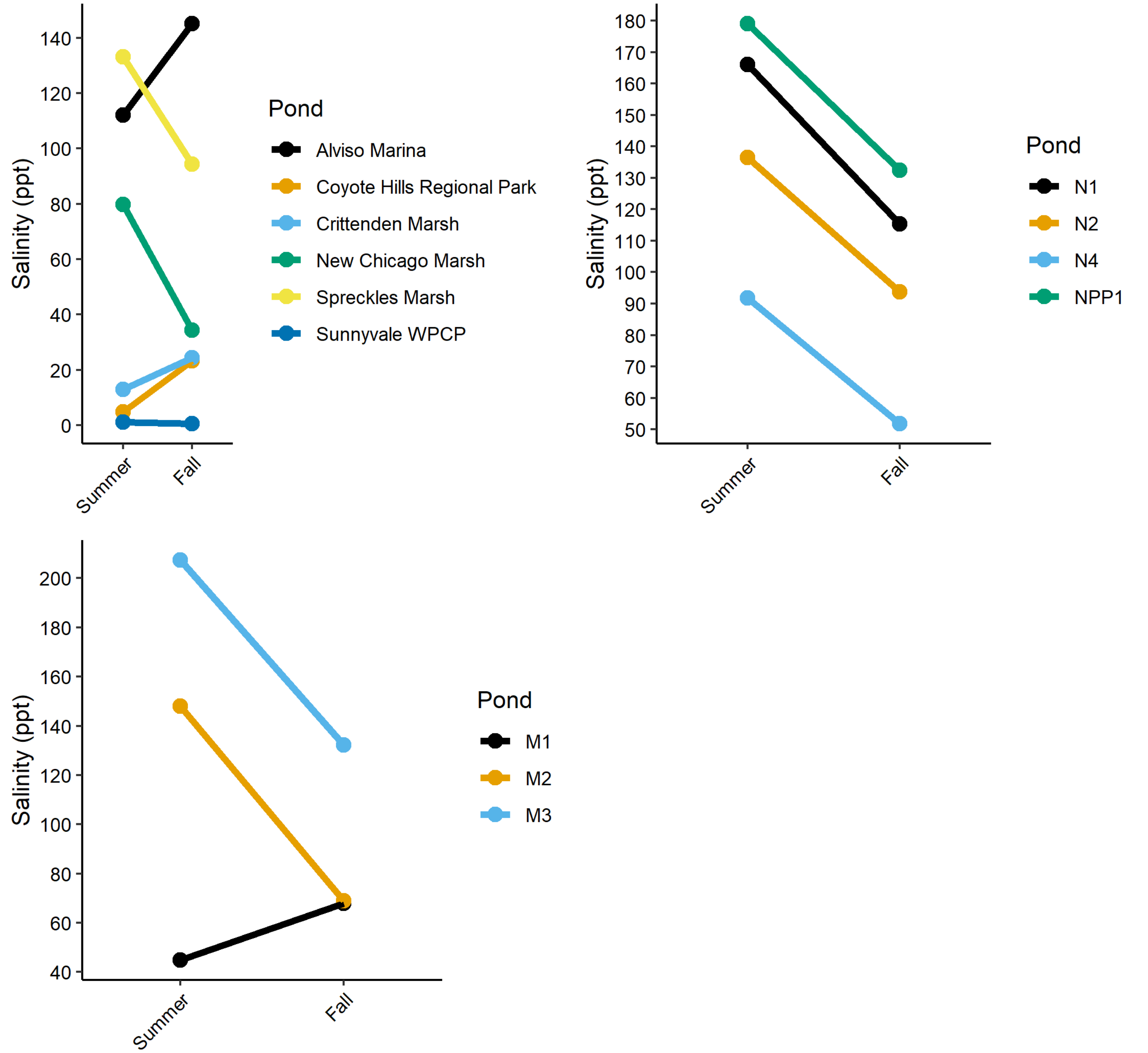

Figure A2.1. Average Salinity (ppt) at the non-SBSPRP sites within South San Francisco Bay, California; July – October 2024. Scale differs between graphs.

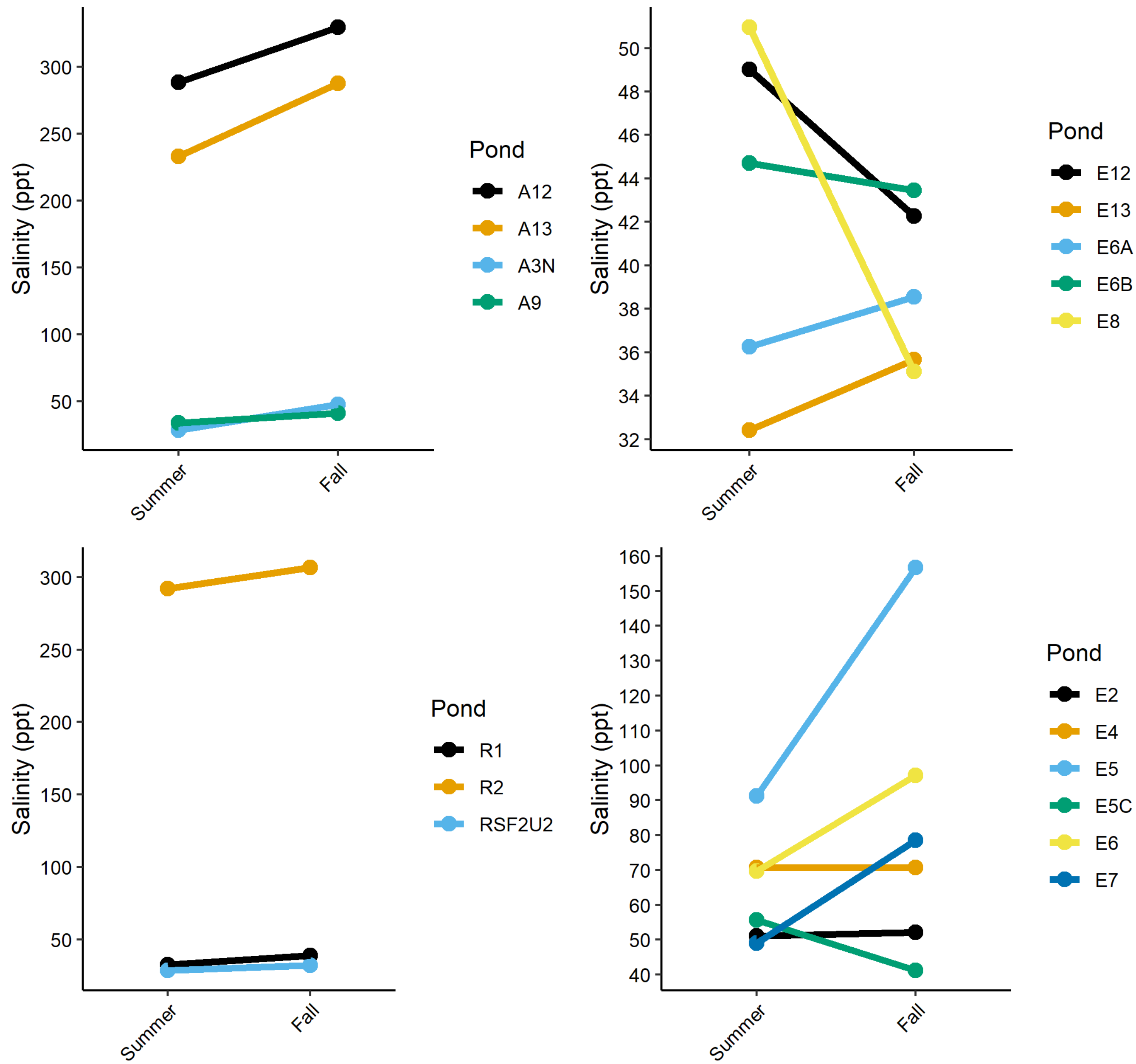

Figure A2.2. Average Salinity (ppt) at the SBSPRP sites within South San Francisco Bay, California; July – October 2024. Scale differs between graphs.

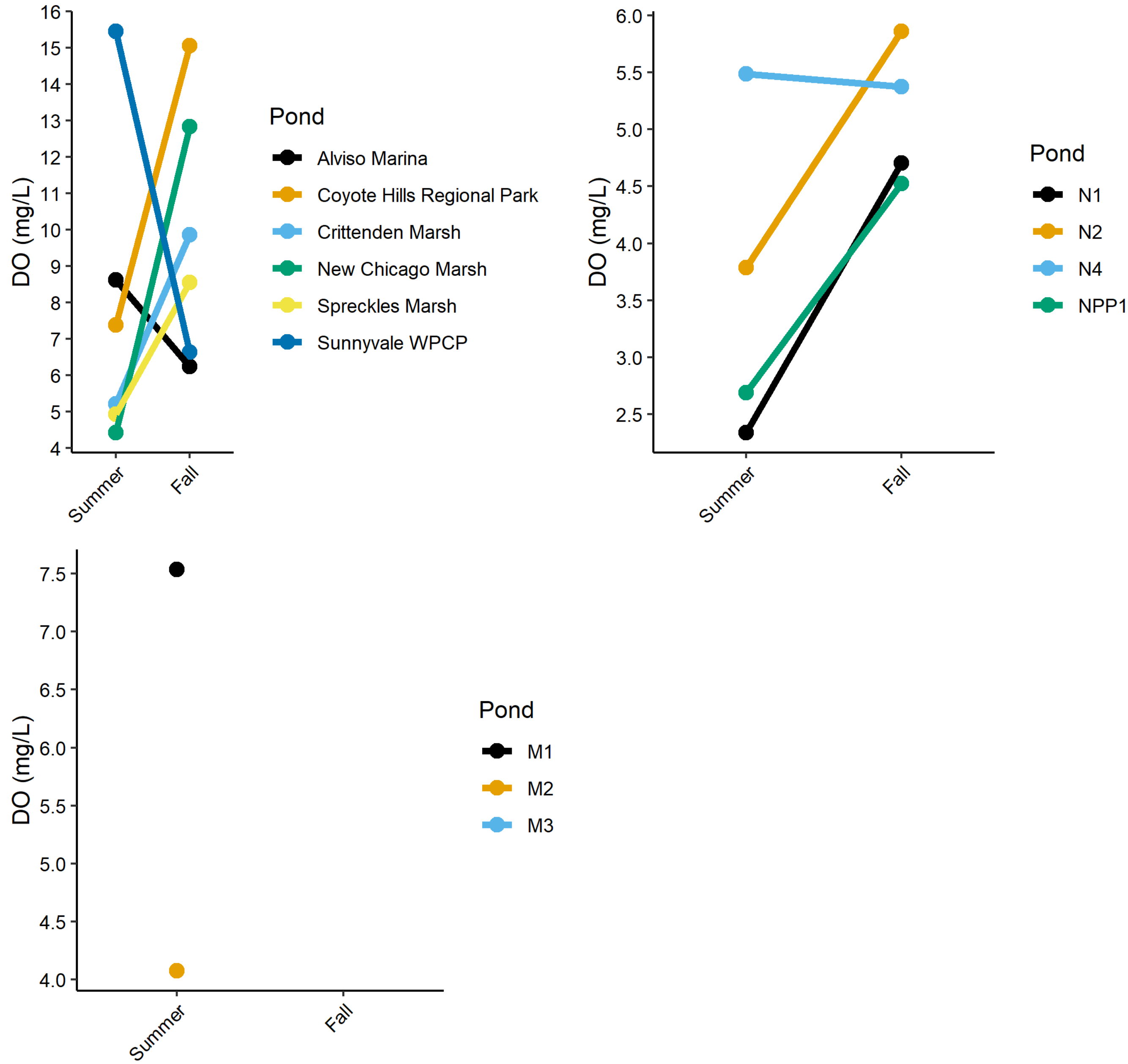

Figure A2.3. Average DO (mg/L) at the non-SBSPRP sites within South San Francisco Bay, California; July – October 2024. Scale differs between graphs.

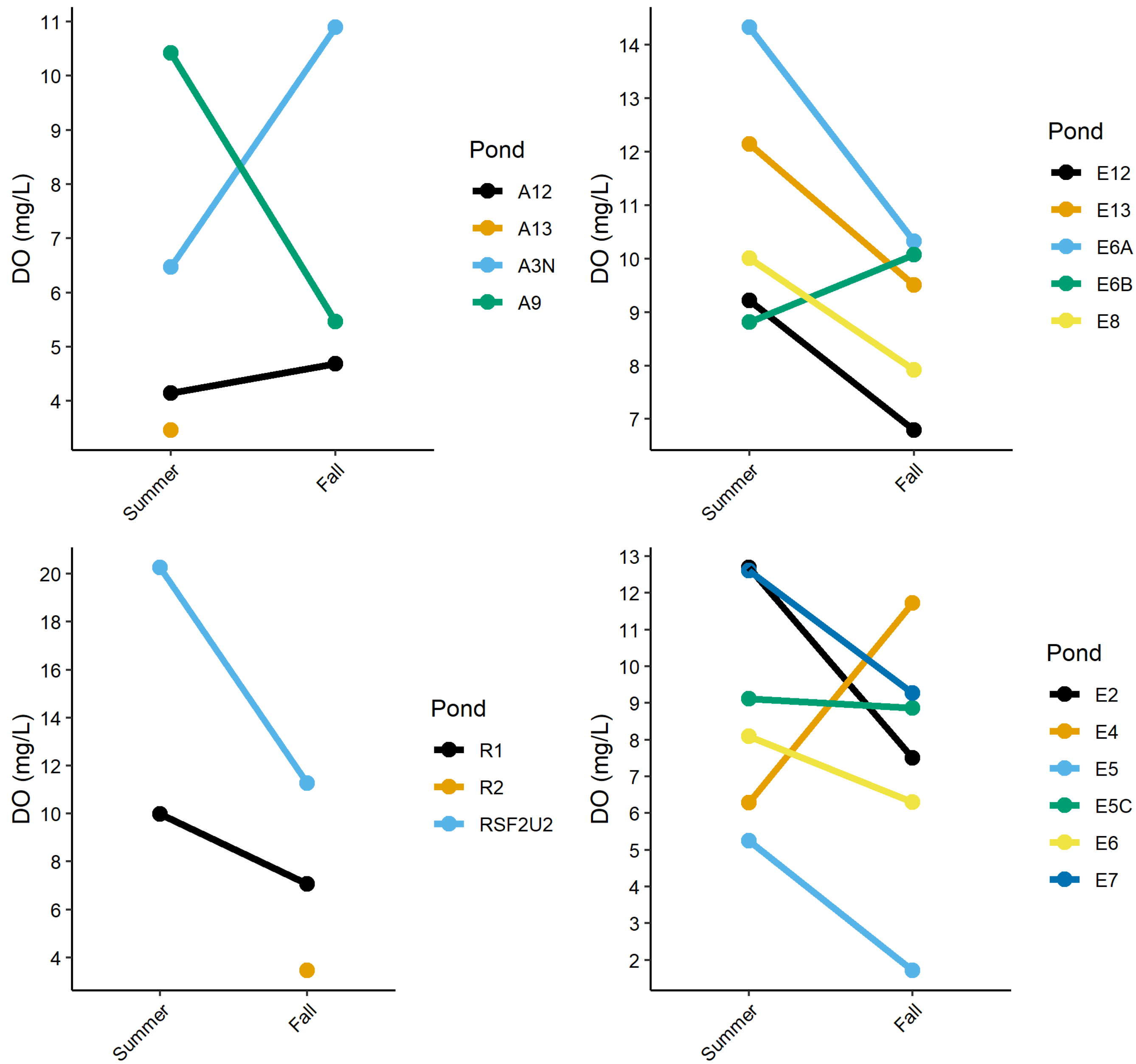

Figure A2.4. Average DO (mg/L) at the SBSPRP sites within South San Francisco Bay, California; July – October 2024. Scale differs between graphs.

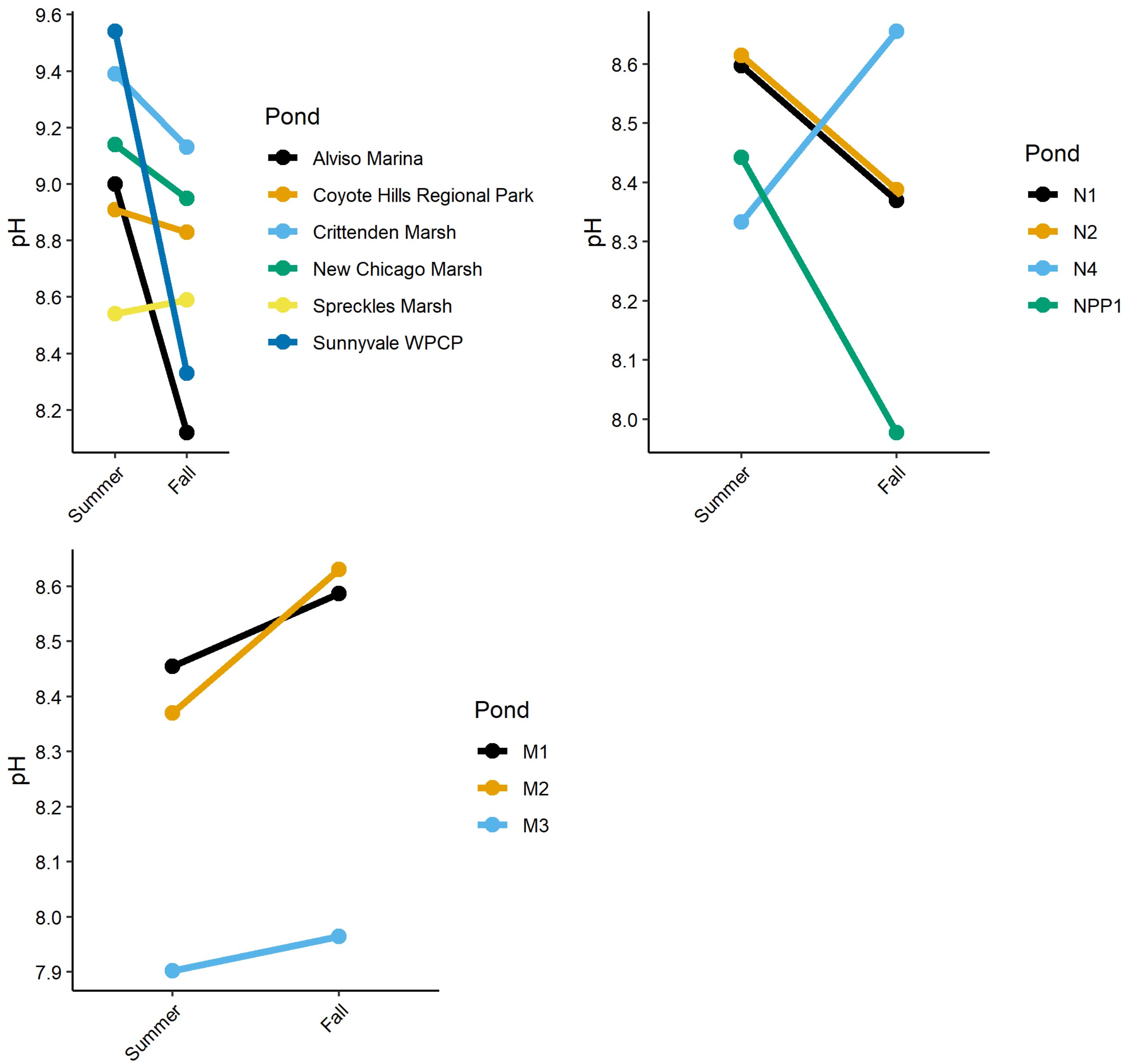

Figure A2.5. Average pH at the non-SBSPRP sites within South San Francisco Bay, California; July – October 2024. Scale differs between graphs.

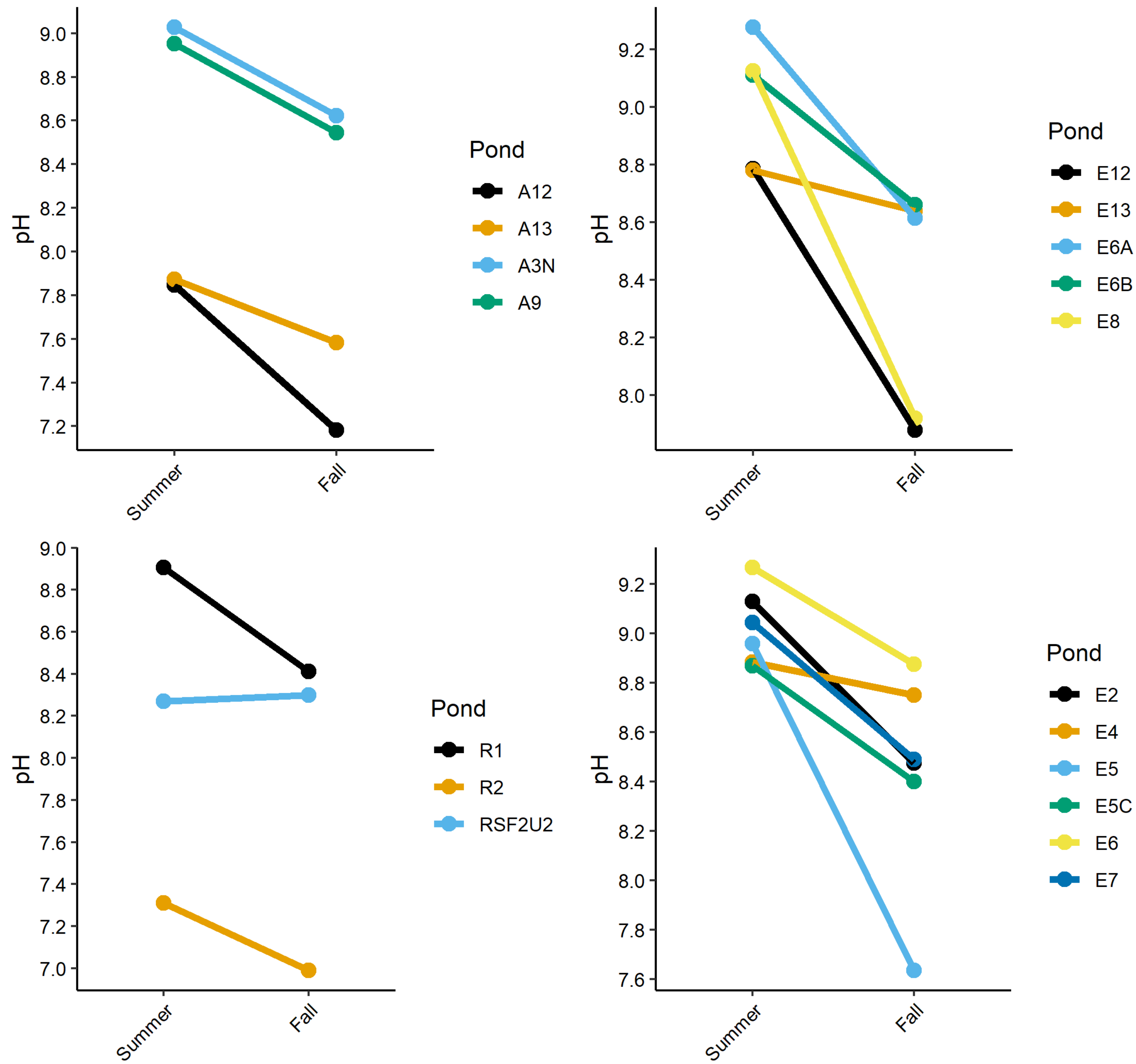

Figure A2.6. Average pH at the SBSPRP sites within South San Francisco Bay, California; July – October 2024. Scale differs between graphs.

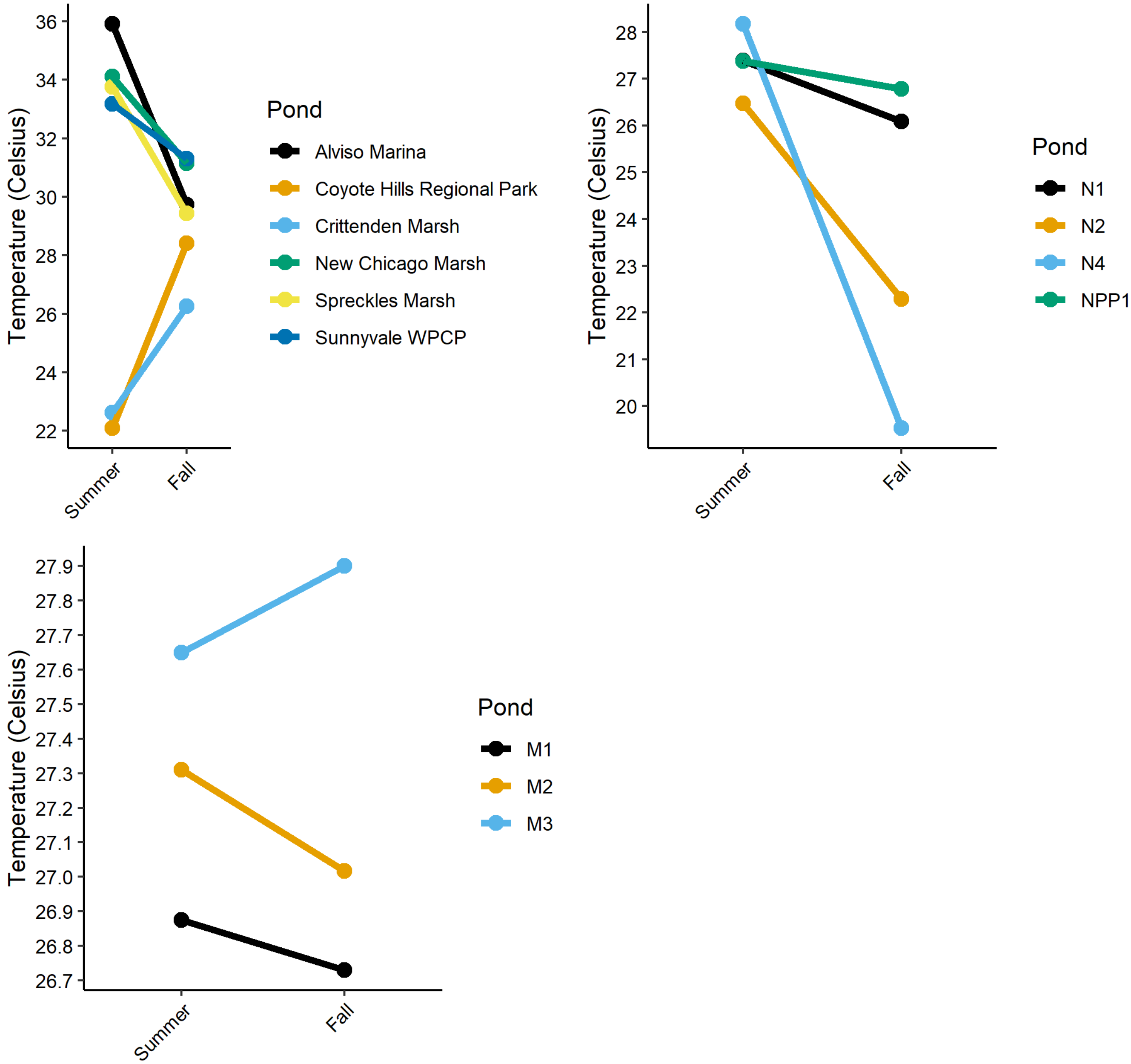

Figure A2.7. Average Temperature (Celsius) at the non-SBSPRP sites within South San Francisco Bay, California; July – October 2024. Scale differs between graphs.

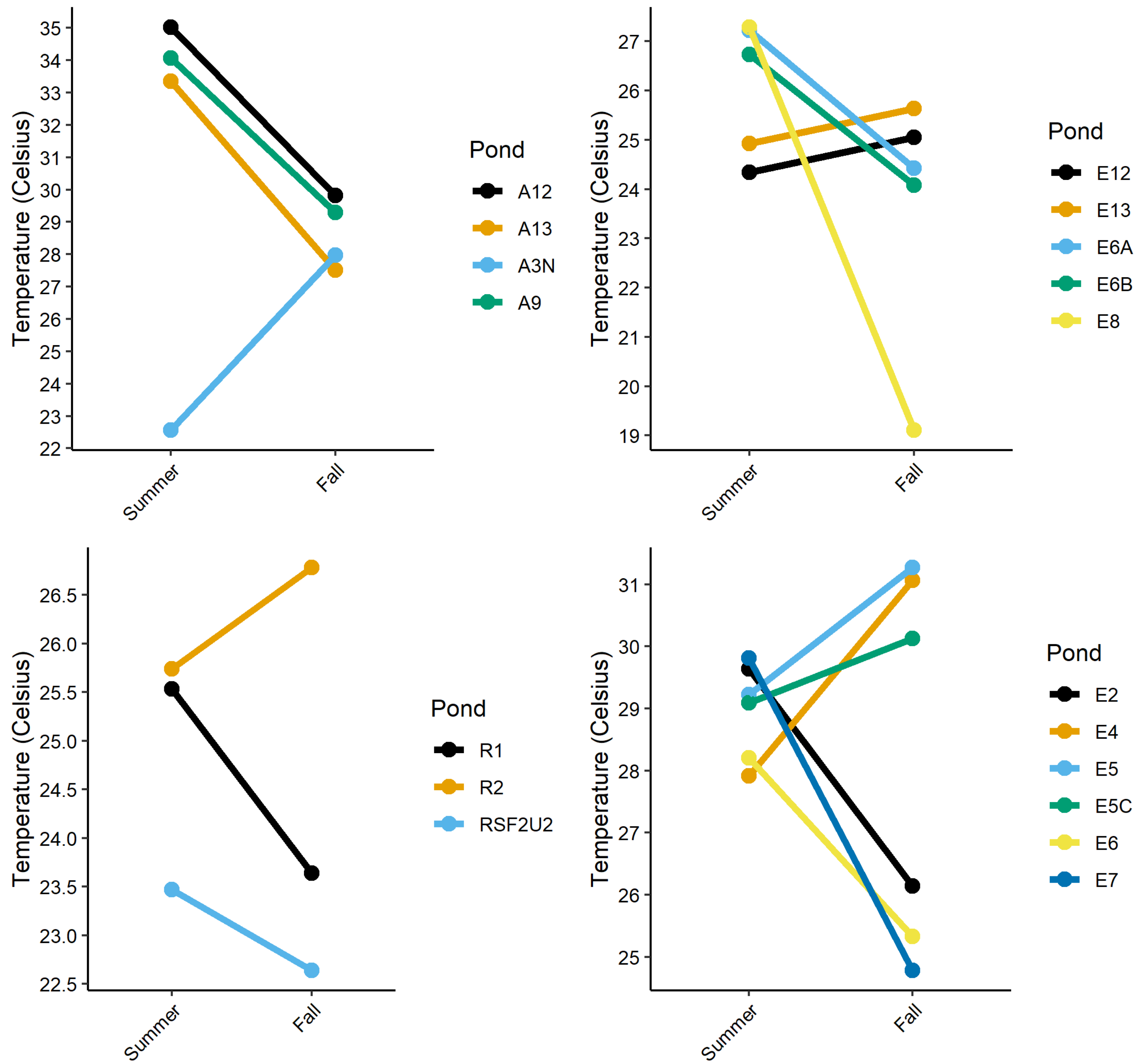

Figure A2.8. Average Temperature (Celsius) at the SBSPRP sites within South San Francisco Bay, California; July – October 2024. Scale differs between graphs.

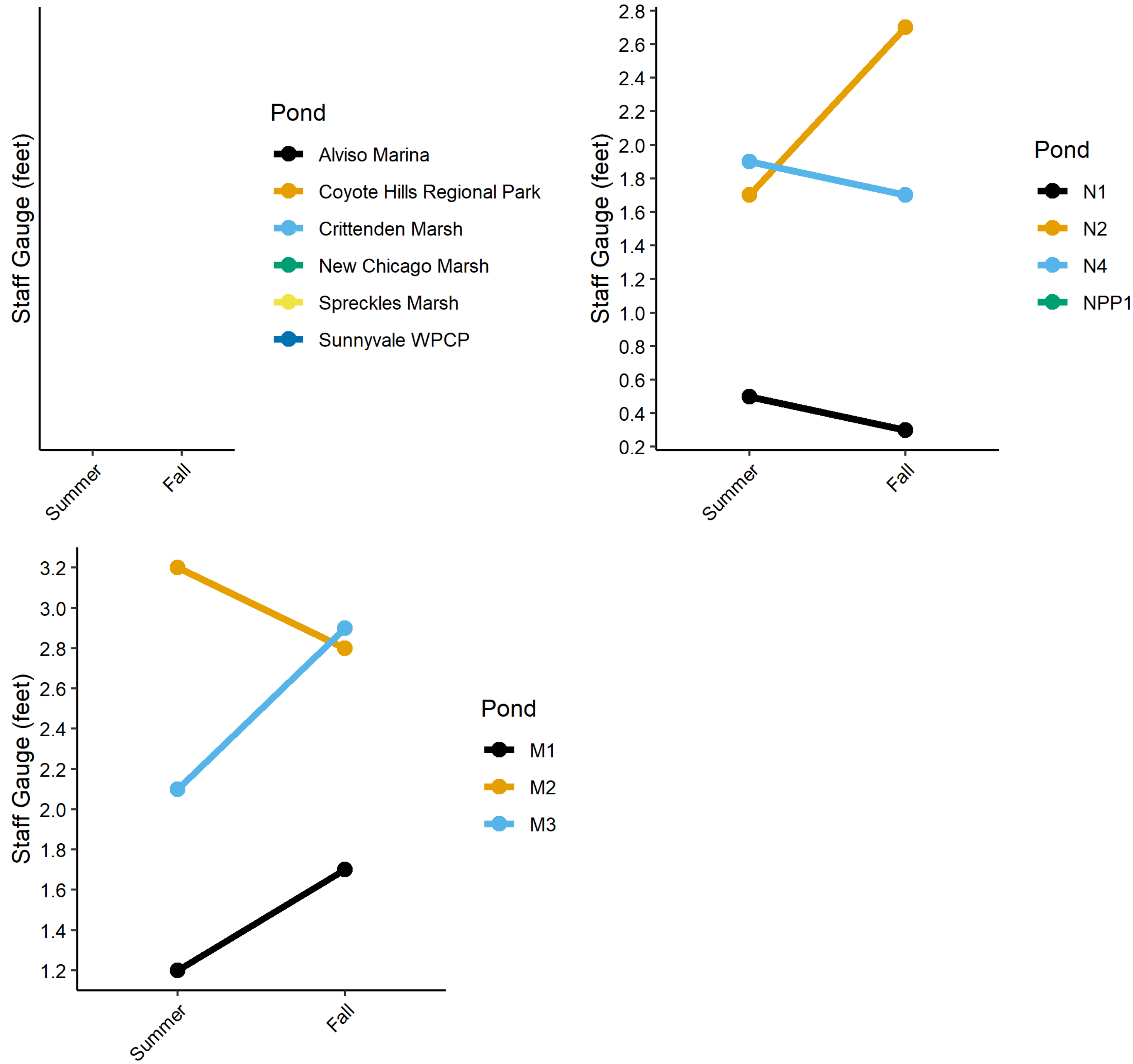

Figure A2.9. Average Staff Gauge (feet) at the non-SBSPRP sites within South San Francisco Bay, California; July – October 2024. Scale differs between graphs.

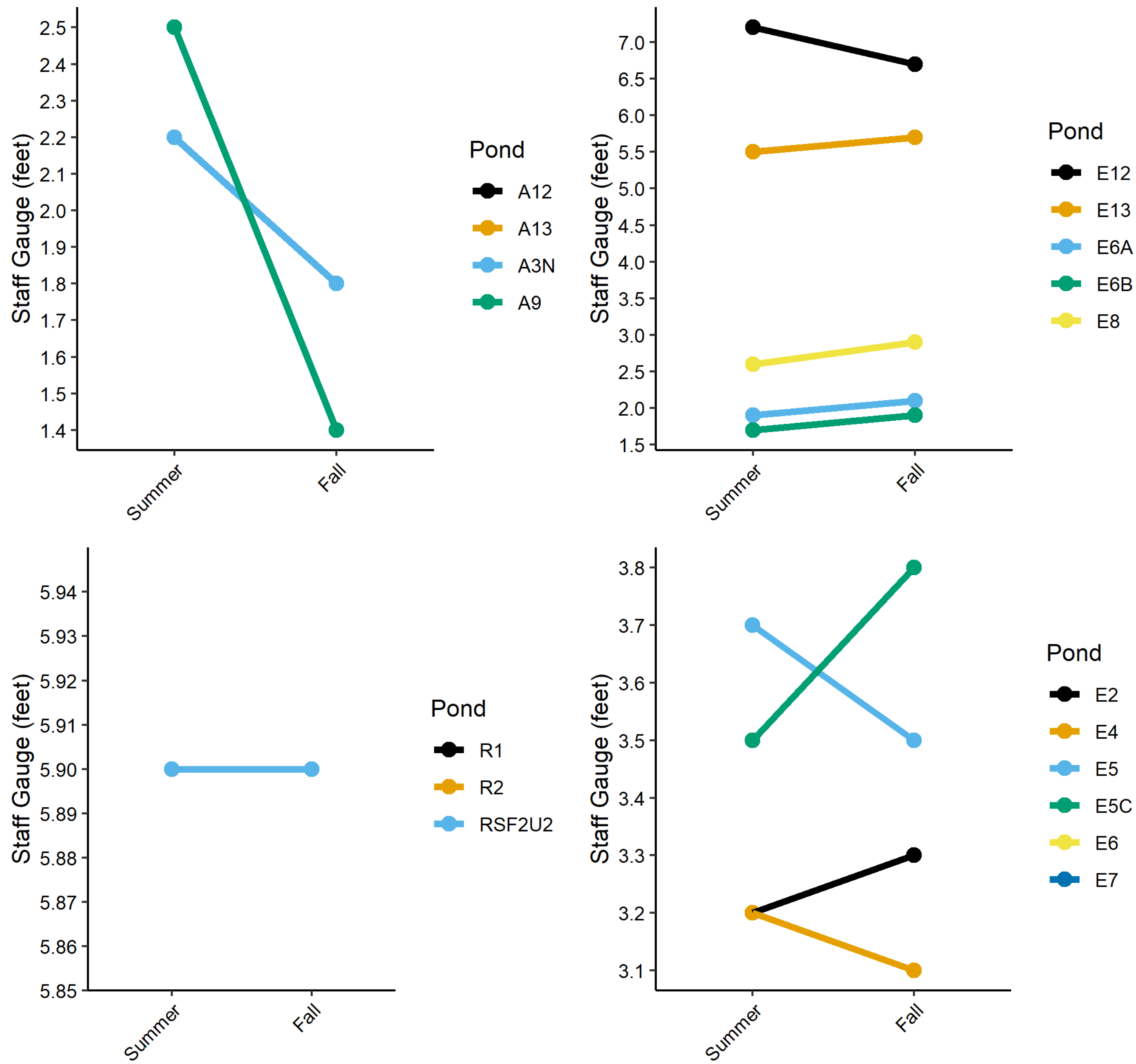

Figure A2.10. Average Staff Gauge (feet) at the SBSPRP sites within South San Francisco Bay, California; July – October 2024. Scale differs between graphs.