SAN FRANCISCO BAY
BIRD OBSERVATORY

Phalarope Migration and Foraging Activity in South San Francisco Bay in 2025

Prepared By:

Nathan D. Van Schmidt, Director of Regional Strategies
Amy Parsons, Lead Waterbird Biologist
San Francisco Bay Bird Observatory
1905 Concourse Dr, San Jose, CA 95131

Prepared For:

Dave Halsing, Executive Project Manager
South Bay Salt Pond Restoration Project
State Coastal Conservancy

Rachel Tertes, Wildlife Biologist
Don Edwards San Francisco Bay National Wildlife Refuge
U.S. Fish & Wildlife Service

John Krause, Senior Environmental Scientist & Garrett Allen, Reserve Manager
Eden Landing Ecological Reserve
California Department of Fish and Wildlife

Amy Larson
California Wildlife Foundation

Laura Cholodenko
California State Coastal Conservancy

FINAL REPORT
June 29, 2026

Suggested citation: Van Schmidt, Nathan D. and A. Parsons. 2026. Phalarope Migration and Foraging Activity in South San Francisco Bay in 2025. Report prepared for the South Bay Salt Pond Restoration Project.

Table of Contents

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 2025 migration.

From 2023-2024 large sightings (>1,000 birds in some cases) were observed on ponds outside of the set of 31 sites initially believed to be sufficient to capture >95% of recent phalarope sightings. In the previous report, we therefore reanalyzed the historical data from the full 2002-2024 dataset and identified those sites that may also support significant numbers of phalaropes. From this list, this year we nearly doubled our survey effort by surveying 27 new ponds: A10, A11, A14, A15, A16, A22, A23, A3W, A5, A7, A8, E10, E11, E14, E1C, E2C, E3C, E4C, E6C, M4, M5, M6, N3, R3, R4, R5, and RSF2U3 (in addition to E5C which was added in 2024).

From June 17, 2025 to September 25, 2025, we conducted 8 rounds of phalarope migration surveys at 59 sites (41 SBSPRP managed ponds, 11 Cargill-managed salt production ponds, and 7 sites outside of these areas). Across *all* sites, a total of 9,066 phalaropes were counted throughout the summer (3,498 Red-necked, 3,583 Wilson's, and 1,985 that could not be identified to species). Compared to last year, this was 4,231 more sightings of phalaropes (+1,065 Red-necked, +2,231 Wilson's, and +935 that could not be identified to species). Counts of Wilson's Phalaropes peaked at 1,324 phalaropes on July 1st and counts of Red-necked Phalaropes peaked at 1,894 phalaropes on September 9th.

Looking at *only* the 52 sites within SBSPRP and salt pond area, a total of 6,871 phalaropes were counted throughout the survey season (3,347 Red-necked, 1,542 Wilson's, and 1,982 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 982. This was -81.6% 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.

Across all new sites, the only phalaropes sighted were 33 Red-necked Phalaropes, all of which were seen during the late August survey at just three sites: A8 (17 birds), A15 (7 birds), and E3C (9 birds). This suggests that the core set of 32 sites, though imperfect, is largely sufficient for capturing the key trends. To increase confidence in this conclusion, we recommend continuation of phalarope migration surveys at the expanded set of sites in 2026 to confirm this initial finding of a lack of phalaropes. In future years, the set of sites may be able to be winnowed down again.

The mid-June migratory period was added in 2022 because the peak count of Wilson's Phalaropes in 2021 was in early July, the previous first survey window, creating uncertainty in whether this was the true peak. This mid-June survey period has only had 68 phalarope sightings across all four years since then (0.2% of all sightings). This survey is also not conducted at any other staging sites. It therefore appears unlikely that this survey window is necessary and we recommend removing it from the protocol in order to reduce the cost of monitoring.

The recent range-wide modeling report showed a growth in phalarope peak abundances across their entire range of 5–25%, indicating that while the partial recovery of San Francisco Bay abundances may have benefited from range-wide increases, they exceeded rather than simply tracked global trends. The largest increases in abundance were all outside of the SBSPRP (M2, M1, Crittenden Marsh, N2, and N1), with the exception of A9, which had 557 more sightings this year and was the third-most abundant site. This aligns with the predictions of the change model, which predicted that shifting hydrology would drive a reduction in abundance in the SBSPRP that would be outweighed by an increase in the other sites.

The increasing amount of data and an improved model-averaging approach allowed us to uncover several new statistically significant predictors of phalarope trends. Changes in Red-necked Phalarope abundance was strongly positively related to increases in pH, and more weakly inversely related to changes in salinity and water elevation (shallow water better). Wilson's Phalarope abundance increases were strongly positively related to colder water, which may be related to receiving influxes of new water as ponds dry out and water warms, especially in the salt pond complexes. Without data on invertebrate communities within the ponds, the mechanisms underlying these relationships remain unclear. There was also evidence that both species were more abundant in the South Bay in drier, colder years, though the small number of years in the present monitoring dataset weakens the strength of this conclusion. The R^2 of the total phalaropes change model was less than half that of the other models, indicating that grouping guilds in models may be reducing statistical power if individual species are responding differently to habitat shifts.

Due to the small sample sizes of complete observations of both phalarope population sizes and water quality variables, results from this model should be interpreted very cautiously. However, the R^2 of the best Wilson's Phalarope model was 0.28, indicating over a quarter of the variance was explained by the model—a record high for waterbird surveys within the South Bay which highlights the value of both targeted and continuing monitoring.

Salinity was not a strong driver in the change model, but had clear impacts on phalarope habitat selection in the now two years of data assessing salinity associations with foraging behavior. Phalaropes are less abundant and forage less at highly hypersaline sites (>150 ppt). This statistically significant relationship offers a clear explanation for why the analysis of the historical phalarope data showed an increase in abundance at sites where salinity fell from 2002–2017. Phalaropes show a marked preference for marine to moderately hypersaline environments (30–150 ppt). E7, which had marine salinity, was the SBSPRP site with the highest numbers of attempts and foraging successes both by Wilson's and Red-necked Phalaropes. This aligns with the optimal salinity conditions of their hypothesized main food source: *Ephydra* and potentially *Artemia* spp. Studies on *Ephydra* at Mono Lake suggest that a salinity high enough to exclude fish but otherwise near the low end of this spectrum (~50 ppt) may be ideal. Preliminary data in the previous report suggested Wilson's Phalaropes preferred the less-salty marine salinity sites, contrary to expectations as this species is more specialized on saline lake habitats. However, the additional year of data indicates that both phalaropes species are selecting for the same marine to moderately hypersaline salinity range and Wilson's Phalaropes are more strictly avoiding near-fresh or brackish sites, in line with expectations for a hypersaline specialist species. The Sunnyvale WPCP pond was the only near-fresh or brackish pond that was used with 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. Brackish ponds had visibly lower attempt and success rates, but this relationship was not significant because of the small number of foraging observations at these sites. An additional year of data may enable us to confirm that this is statistically significant.

Unique amongst waterbirds monitored in the South Bay, this internationally coordinated survey protocol is now producing range-wide collaborative research on how changes in South Bay populations relate to changes in range-wide abundances and habitat conditions. In early 2026, the federal government began a formal process to review Wilson's Phalarope for listing under the Endangered Species Act. A Phalarope Core Workgroup of managers and Technical Workgroup of scientists are now being formed to address declines in this species. These surveys provide the backbone of this scientific and conservation effort.

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 2021–2025.

This report summarizes the results of SFBBO's phalarope surveys in the South San Francisco Bay from 2019 through 2025. 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, we also reanalyzed data from the historical salt pond surveys to model species-specific trendlines and thresholds using the new recommended July–September survey window (Burns & Van Schmidt 2023).

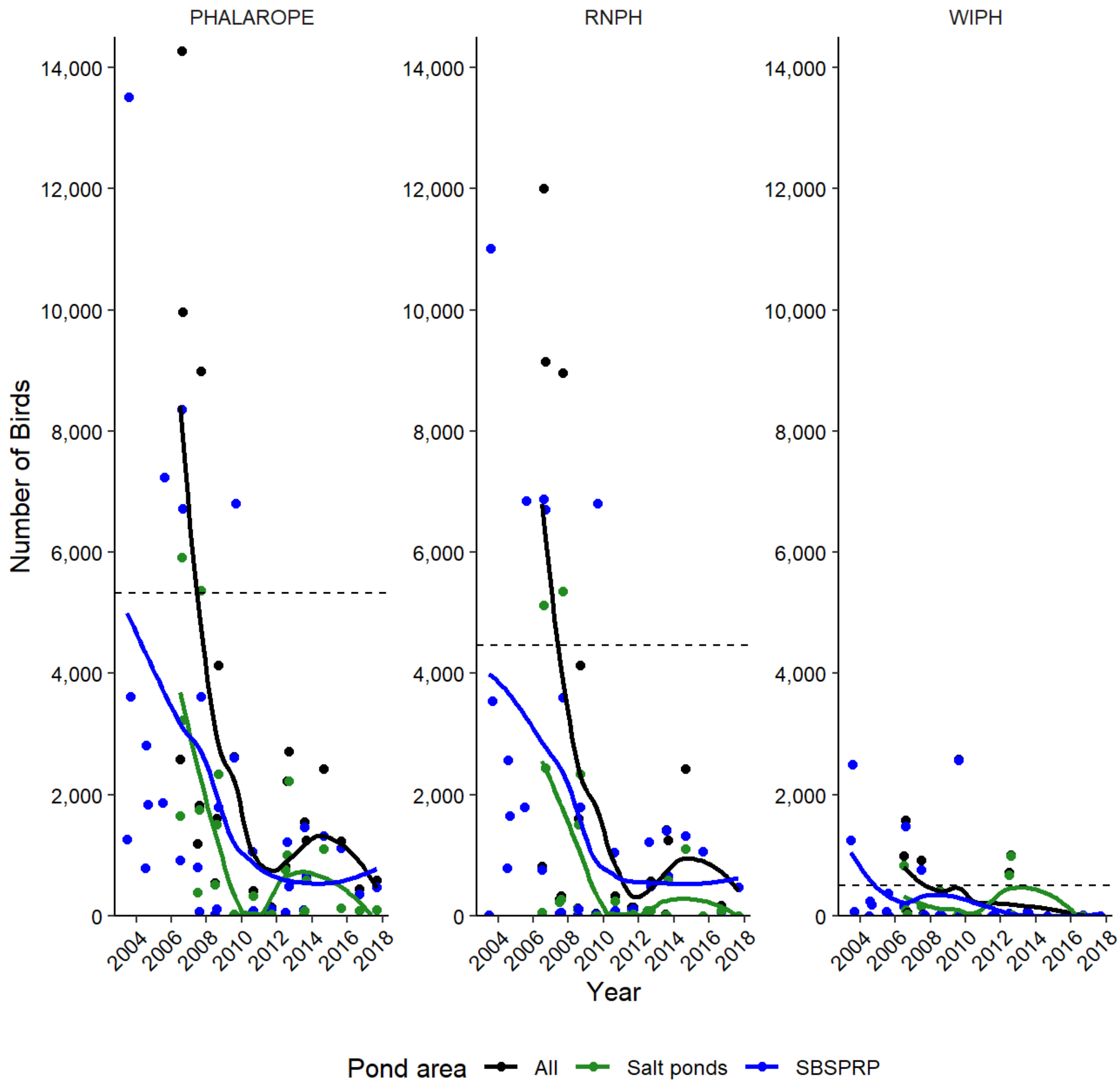


Figure 1. Counts of total phalaropes, Red-necked Phalarope, and Wilson's Phalarope during summer migration (July-September) within the South Bay Salt Pond Restoration Project (SBSPRP) and salt production ponds of south San Francisco Bay, California. Lines represent LOESS regression curves and the dashed 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. Both phalarope species showed similar preferences for select sites, suggesting that surveys could target a subset of 24 SBSPRP and Cargill-managed ponds to capture the vast majority 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 (WPCP) ponds) and East Bay Regional Parks District (Coyote Hills Park). However, from 2023-2024 large sightings (>1,000 birds in some cases) were observed on ponds outside of this set of sites. This prompted SFBBO to carry out a reanalysis of the historical data from the full 2002-2024 dataset (Van Schmidt & Parsons 2024). In 2025, in order to ensure that we were not missing phalaropes at unsurveyed sites we added 27 new ponds: A10, A11, A14, A15, A16, A22, A23, A3W, A5, A7, A8, E10, E11, E14, E1C, E2C, E3C, E4C, E6C, M4, M5, M6, N3, R3, R4, R5, and RSF2U3. Most of these ponds were selected based on the ponds ranked likely to have phalaropes reanalysis of historical phalarope records in the 2024 report (Van Schmidt & Parsons 2024), though a few low-ranked ponds were added because they were already being surveyed by other research projects or were adjacent to surveyed ponds.

In 2025, phalarope migration surveys were conducted at 59 sites (Fig. 2). Most of these sites were current salt ponds or former salt ponds within the SBSPRP footprint, but seven additional baylands sites were surveyed: Coyote Hills Regional Park, Crittenden Marsh, Northeast of N1, New Chicago Marsh, Spreckles Marsh, Alviso Marina, and the Sunnyvale WPCP.

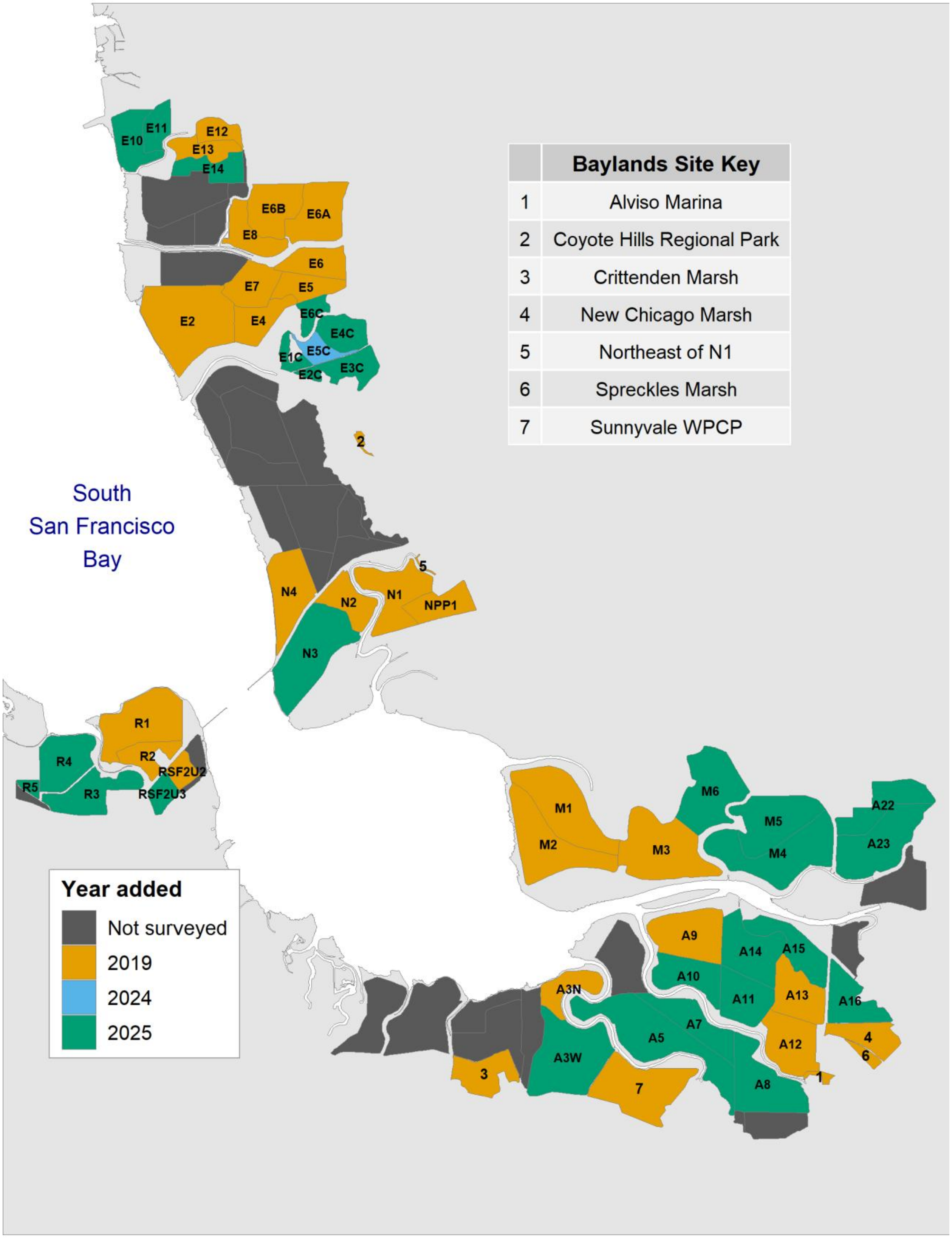


Figure 2. Map of target sites for Phalarope migration surveys, color-coded by the year they were added to the set of monitored sites.

Field Data Collection

Phalarope Migration Surveys

We conducted 8 surveys in 2025 during the season of peak phalarope migration. Surveys occurred every two weeks with the following target dates: 06/17, 07/01, 07/15, 07/29, 08/12, 08/26, 09/09, 09/23 (Table 1). Surveys were two weeks apart to ensure the peak of migration was captured and in alignment with rangewide North American staging site surveys carried out by the International Phalarope Working Group (Carle et al. 2024). 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

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 per second), though high-speed video showed that the birds were successfully foraging. 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 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).

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	20	1447	251	2	1700
2	Late Aug	2019	08/26	09/01	21	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	30	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

Survey ID	Survey period	Year	Start	End	Sites	RNPH	WIPH	XXPH	Total
29	Mid Aug	2023	08/14	08/15	31	1669	136	200	2005
30	Late Aug	2023	08/28	09/01	30	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
41	Late June	2025	06/17	06/19	56	0	0	0	0
42	Early July	2025	06/30	07/03	58	0	1324	0	1324
43	Mid July	2025	07/14	07/17	59	76	835	1	912
44	Early Aug	2025	07/28	07/31	58	162	1019	49	1230
45	Mid Aug	2025	08/11	08/14	59	73	352	0	425
46	Late Aug	2025	08/25	08/28	58	848	37	1045	1930
47	Mid Sept	2025	09/08	09/11	57	1894	10	775	2679
48	Late Sept	2025	09/22	09/25	57	445	6	115	566

Water Quality Sampling

We sampled water quality at 56 of the 59 survey sites (including Bayland sites) between September 2 and October 15. 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 SBSPRP sites multiple points were generally sampled in accordance with established sampling points (Van Schmidt & Parsons 2024). Within non-SBSPRP sites, only a single water quality sample was taken except for Sunnyvale WPCP, where one sample was taken from each pond. The number of sampling points per pond varied based on the size, configuration, and accessibility of the pond. Some sampling points or entire sites could not be reached due to low water levels within the pond. Whenever possible, water quality data were 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.

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 2022. While this may introduce a slight bias towards underrepresenting counts in 2020 and 2021, there are only 17 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 just four 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:

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

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 in a first AICc model selection procedure, followed by "Main" stage covariates. "Change model" column indicates whether the variable 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	Tides	Breached	Breached to tidal action	Binary	Annual
		Gated	Gated culvert / semi-tidal	Binary	Annual
		BreachOrGate	Breached or gated	Binary	Annual
	Hydrology	WaterElev_m_AC	Mean water depth	cm	Change
		Salinity	Mean salinity	ppt	Annual + Change
		Salinity_AC	Mean salinity	ppt	Annual + Change
		pH	Mean pH	pH	Annual + Change
		pH_AC	Mean pH	pH	Annual + Change
		LDO_mgL	Mean dissolved oxygen	mg/L	Annual + Change
		LDO_mgL_AC	Mean dissolved oxygen	mg/L	Annual + Change
		Temp_C	Mean water temperature	°C	Annual + Change
		Temp_C_AC	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, the 2019–2020 and 2020–2021 change data had zero complete cases for use in statistical modeling (Table 2). As a result, the final sample sizes were small ($n = 85$ for total phalaropes, $n = 82$ for Red-necked Phalarope, and $n = 35$ 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 8 covariates for total phalaropes and Red-necked Phalarope, and up to 4 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. 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 1. 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. We report R^2 of the best model for each guild as an estimate of goodness-of-fit.

Model Averaging and Assessment

To increase model stability and leverage information from the set of competitive models, we calculated model-averaged coefficient estimates and predicted changes in abundance by calculating the coefficient or predicted value in each model, and then taking the average across all models weighted by that model's AICc weight (i.e., its relative degree of support in model selection). Averaging the full set of candidate models is inadvisable because it produces many redundant models and includes spurious covariates that do not actually improve model fit (Burnham & Anderson 2002, Grueber et al. 2011). For this reason, we applied the "nesting rule" of Richards (2005, 2008) which can improve inference and parameter accuracy by removing models from the final model set that are more complex versions of nested models with better AICc support (indicating that the additional covariates did not improve model fit; Burnham 2015). This reduced the size of the modelset from thousands of models to dozens.

For calculating coefficient estimates, there are two types of averaging: conditional, which averages over models where the parameter appears (weights are rescaled to sum to 1 among those models), and with shrinkage, which sets the coefficient equal to zero if the variable is not included in a given model (Grueber et al. 2011). Shrinkage-based estimates report the expected effect accounting for uncertainty about whether this variable belongs in the model at all, while conditional estimates report the effect of the variable assuming that it belongs in the model to begin with. Conditional estimates are therefore less conservative (Grueber et al. 2011, Nakagawa & Freckleton 2011). Shrinkage estimates can only be calculated when the number of models with and without a variable are equal. Because we used a multi-staged modelset with some covariates not able to be included together (i.e., PPT0 and PPT1 or highly correlated variables) and then applied the nesting rule, this assumption was not met, so estimates with shrinkage could not be calculated. When interpreting conditional coefficient estimates, we caution that (1) they are less conservative, though application of the nesting rule should remove completely uninformative covariates, and (2) the set of reported coefficients does not represent the behavior of any single actual model and could produce misleading inferences if predictors are correlated (e.g., if a predictor's estimated effect changes from +1 to -1 when a negatively correlated covariate is included in the model, averaging across the two models would pull the estimate towards 0 despite that not being a value in either model; Banner & Higgs 2017). This possibility could be relevant especially when the final predictor set includes closely related covariates (e.g., pH and change in pH).

For all predictions, we averaged across all models in the final modelset. To estimate the impact of the habitat changes and weather on trends within the current year we used 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 variables 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

We classified each pond into one of six salinity bins (Table 3) based on a review of likely phalarope prey salinity tolerances from the published literature (Van Schmidt & Parsons 2025). Salinity was averaged within each pond for each year (first across sampling points within a survey date, then across summer-early fall surveys within each year), so a pond's salinity category could differ between years. Bins were adapted from Takekawa et al. (2006). 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). *Ephydra* and *Artemia* spp. are most common between 30-80 ppt but tolerate a wider range from 25-150 ppt (Brown et al. 2010). *E. millbrae* can live in tide pools in the Bay Area with salinities up to 42 ppt, while *E. cinerea* prefers more hypersaline environments (up to 130 ppt) and is more commonly found in the salt ponds (Collins 1980). *A. franciscana* (the locally dominant brine shrimp species) are most abundant between 90-150 ppt but have 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). In relatively lower salinity parts of Great Salt

Lake, phalaropes forage on chironomid larvae (midges), corixids (water boatmen), and *Daphnia* spp. (water fleas), indicating some flexibility in prey item selection (Frank & Conover 2019, Frank & Conover 2021).

We analyzed differences in Red-necked, Wilson's, and total phalarope abundance and foraging behavior across these six bins. For the foraging data, for each species we assessed differences in foraging behavior across three primary measures: (1) the total number of foraging attempts, (2) the number of successful foraging attempts, and (3) the success rate (# successes / # of attempts). At the small subset of sites where continuous foraging was observed, we assessed the length of time foraging as a separate measure.

For abundance, number of attempts, and number of successes, we tested for statistically significant differences across bins using generalized linear mixed models (GLMMs) with a negative binomial (NB2) distribution and a log link, implemented in package *glmmTMB* (v1.1). For the success rate and continuous foraging time, we used Gaussian linear mixed models fit with package *lmerTest* (v3.2) to use Satterthwaite p-values that are more robust for small sample sizes. All models included salinity category and random effects for both pond (to account for non-independence of samples) and observer (to account for individual variation in counting).We attempted to fit species-salinity interactions but found that data were deficient for Wilson's Phalaropes for many salinity bins, so we excluded these from the final analysis. 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.

We used package *DHARMa* (v0.4.7) to simulate residuals from the fitted models and assess them for overdispersion, normality of residuals, and heteroskedasticity. These tests showed heteroskedasticity for the number of attempts, number of successes, and foraging time models. We recalculated the attempts model in *glmmTMB* with salinity bin added as a parameter on dispersion to correct for this. We attempted the same procedure for number of successes and foraging time, but they did not converge; we report data as-is and caution that these models should be interpreted as preliminary. Data were graphed with boxplots using base package *stats* and package *ggplot2*.

During data proofing we identified several observer-visits where the number of recorded successes exceeded the number of attempts, indicating the observer likely recorded attempts and successes as independent counts rather than successes as a subset of attempts. We corrected all data from each of these observer-days by summing the recorded attempts and successes to obtain the full attempt count.

Table 3. Salinity bins for analysis of phalarope abundance and foraging ecology in South San Francisco Bay, California. All salinity values are in ppt (g/L).

Range	Name	Notes
0–2 ppt	Near-fresh	Freshwater plus unique near-freshwater conditions at Sunnyvale WPCP
2–30 ppt	Brackish	Typical range observed in estuarine systems
30–50 ppt	Marine	Lower optimal limit for *Ephydra* and *Artemia*; 42 ppt is the upper limit for *E. Millbrae*
50–80 ppt	Low hypersaline	Upper optimal limit for *Ephydra* and *Artemia*
80–150 ppt	Mid hypersaline	Fish generally absent; high range may be upper limit for *Ephydra* and *Lipochaeta slossonae*
>150 ppt	High hypersaline	Increasingly only *Artemia* survive, and even they reach tolerances by 200 ppt

Table 4. 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.

Season ID	Season	Water Year	Phalarope Counts	Water Quality Samples	Complete Cases
2	Summer	2020	17	0	0
3	Summer	2021	31	72	0
4	Summer	2022	31	74	20
5	Summer	2023	31	74	20
6	Summer	2024	32	83	23
7	Summer	2025	59	73	22

Results

Overview of Counts and Trigger Status

Across *n* = 462 visits during the 8 surveys of the 59 sites in 2025 we counted a total of 3,583 Wilson's Phalarope, 3,498 Red-necked Phalarope, and 1,985 phalarope of unidentified species, for a grand total of 9,066 birds (Fig. 3). The majority of observed birds were foraging, not roosting (Fig. 4), though relatively more birds were observed roosting this year than in 2024 (Van Schmidt & Parsons 2025). This was 4,231 more phalaropes than were detected last year (an 87.5% increase; Table 1, Fig. 2) due to increases in Red-necked Phalaropes as well as increases in Wilson's Phalaropes.

Within only the 52 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 982. This was 81.6% 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 below the baseline over the last three consecutive years was also triggered because counts were below this threshold from 2023–2025.

Observations of unidentified phalarope present a particular challenge when analyzing data to understand species-specific trends. In 2025, unidentified species represented 21.9% of the total observations, versus a mean of 9.2% from 2020-2024 (range 2.6% - 21.7%). 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 1,324 birds (no change assuming June-July unidentified phalaropes were Wilson's) during the survey centered on 7/01. Counts of Red-necked Phalaropes peaked at 1,894 birds (no change assuming August-September unidentified phalaropes were Red-necked) during the survey centered on 09/09. These peaks indicate that the current survey schedule largely spans the period when phalaropes are present in the area (Fig. 3).

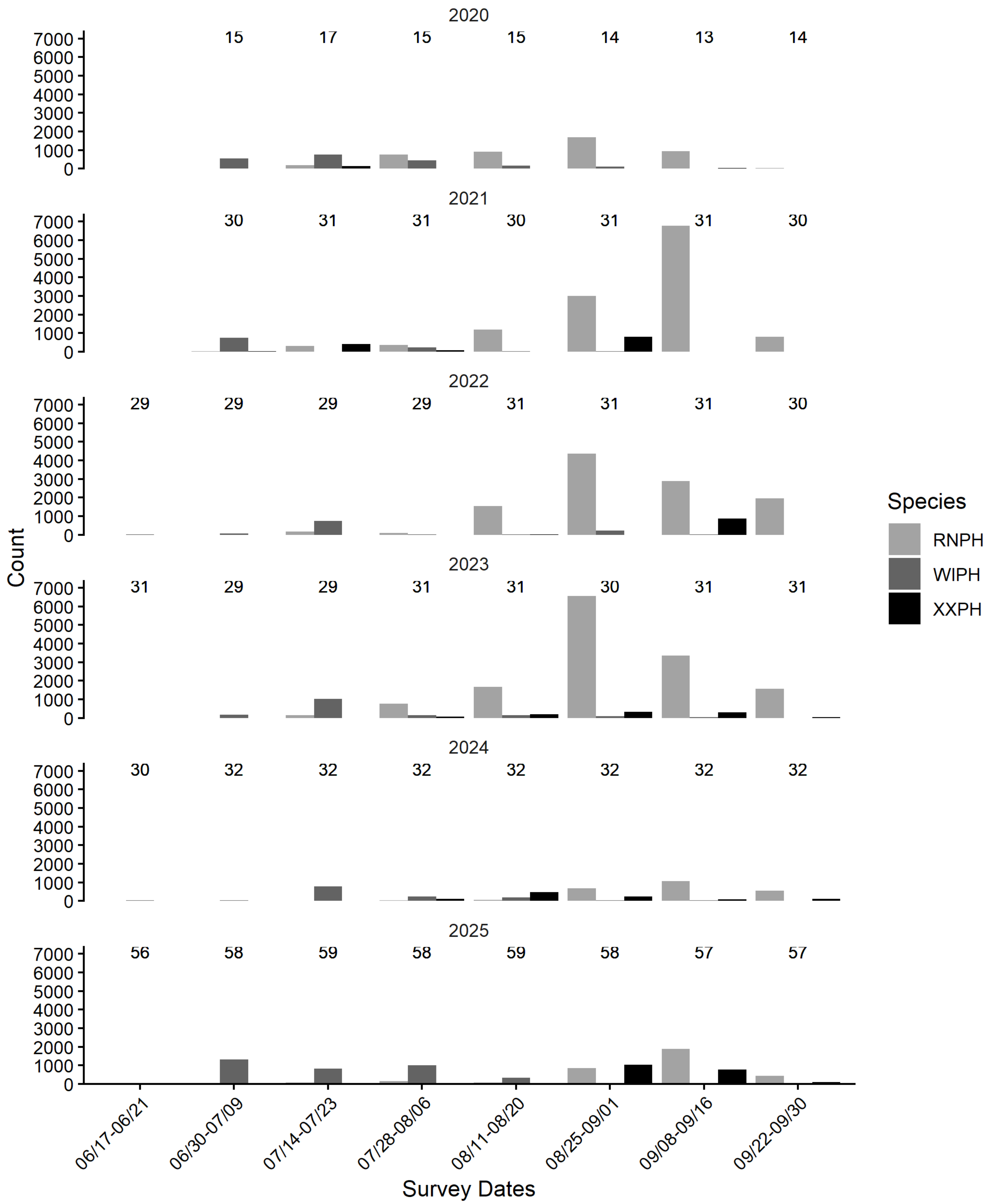


Figure 3. Counts of phalarope species observed during the phalarope migration surveys in 2020-2025 (two partial surveys in 2019 not shown). RNPH = Red-necked Phalarope; WIPH = Wilson's Phalarope; XXPH = phalaropes of unidentified species. No Red Phalaropes were observed. Counts are summed across all sites visited during each survey. Number above each date is the number of sites surveyed during that round.

Figure 4. Total counts of roosting and foraging Red-necked (RNPH), Wilson's (WIPH), and unidentified (XXPH) Phalaropes within survey sites in South San Francisco Bay, California, 2025. Sites are ordered by descending salinity; sites labeled with a * did not have salinity measurements.

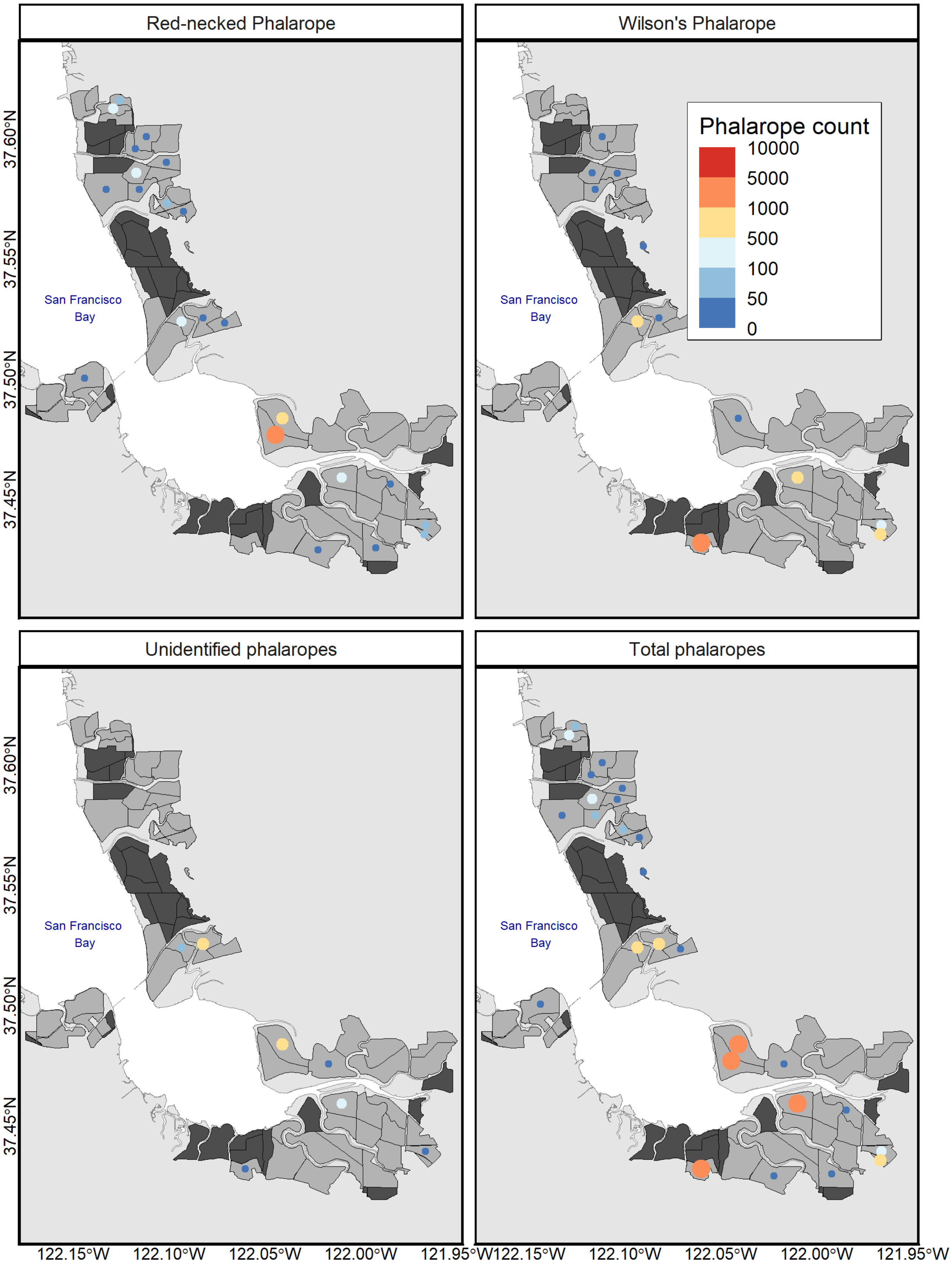


Figure 5. Counts of the total number of phalaropes observed during the phalarope migration surveys from June 17 to September 25, 2025. Species grouping is indicated at the top of each map. Dots are scaled to represent the relative number of phalaropes observed. Ponds with no dot had zero phalarope observations during 2025 surveys. Dark gray ponds were not surveyed.

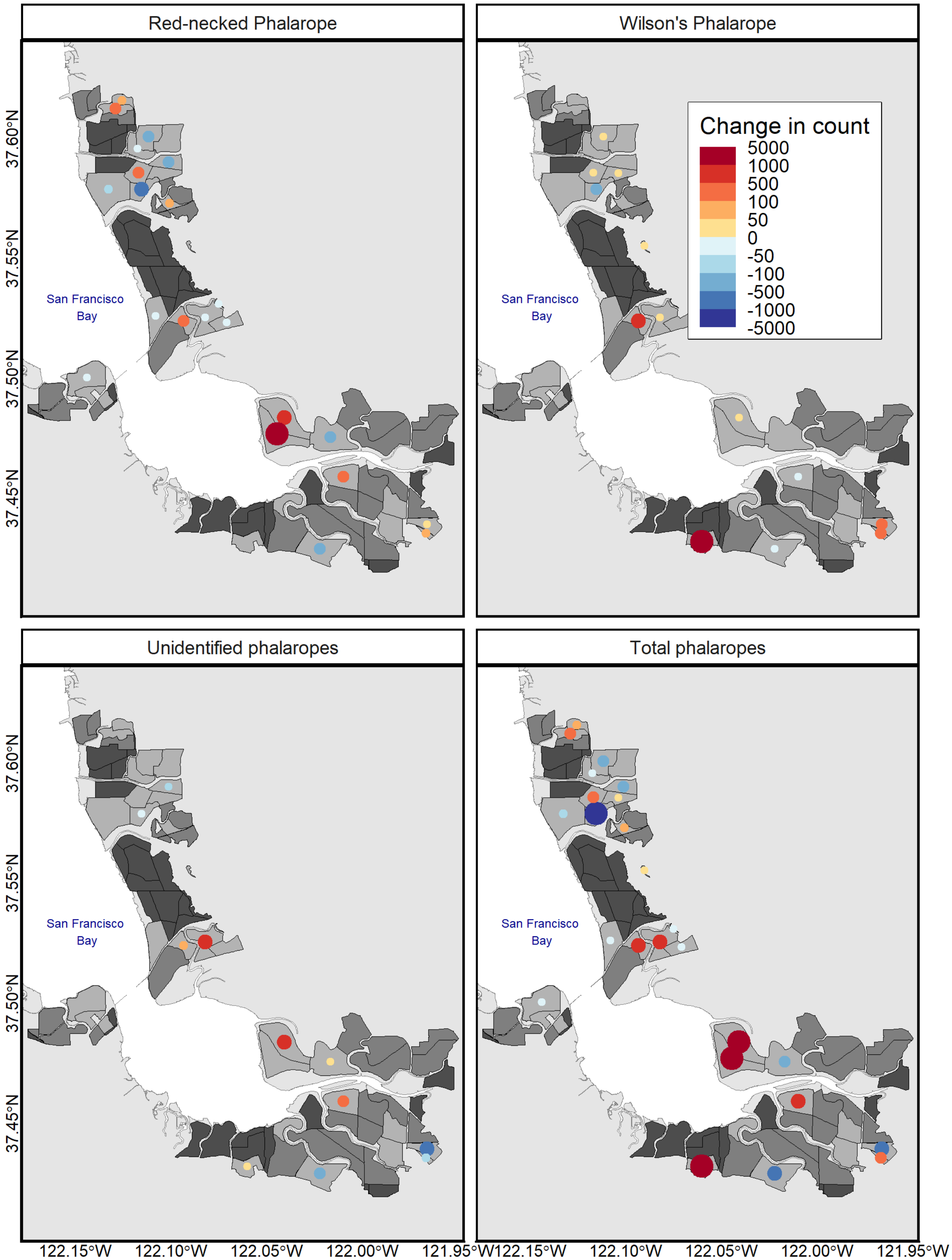


Figure 6. Change in the total number of phalaropes observed during the phalarope migration surveys from 2024 to 2025. Species grouping is indicated at the top of each map. Dot size is scaled to represent the change in number of phalaropes observed. Ponds with no dot had zero phalaropes in both years. Dark gray ponds were not surveyed in either year; slightly dark ponds were surveyed in only one year.

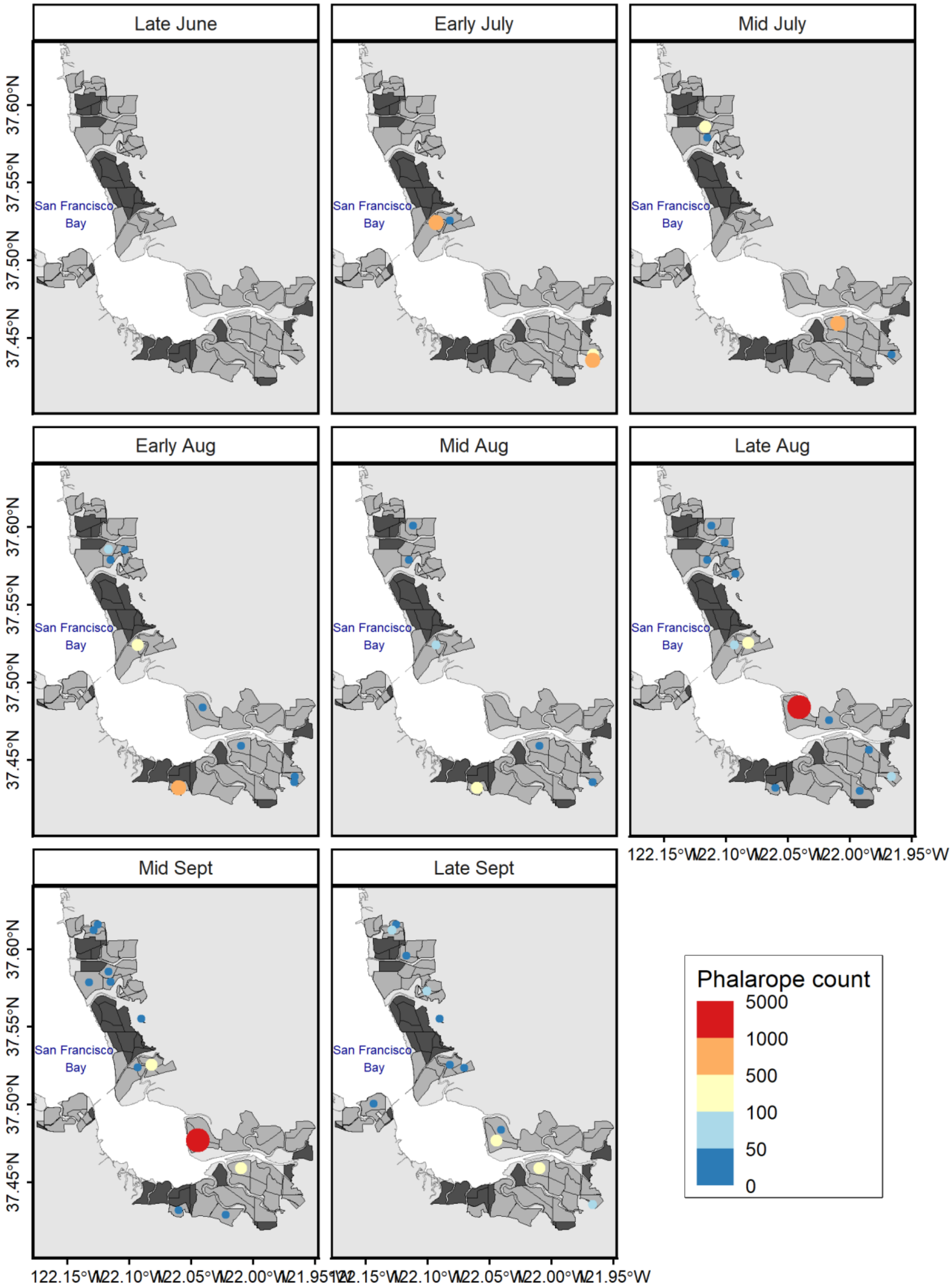


Figure 7. Counts of the total number of phalaropes observed during each phalarope migration survey block from June 17 to September 25, 2025. Survey block is indicated at the top of each map. Dots are scaled to represent the relative number of phalaropes observed. Ponds with no dot had zero phalarope observations during this period. Dark gray ponds were not surveyed.

Red-necked Phalaropes

Trends

Across all sites, there were 3,498 total sightings of Red-necked Phalaropes (sum sightings during the entire survey period). Compared to the previous year, this was an increase of 1,065 total sightings (43.8%; Fig. 5). Excluding the seven sites outside of the SBSPRP and salt pond areas, we observed a total of 3,347 Red-necked Phalaropes (95.7% of all counts), an increase of 1,409 total sightings (72.7%) from last year. At the pond level, M2 had the highest abundance (1,833 sum sightings), followed by M1 (648) and A9 (218; Fig. 5).

The site with the largest increase in Red-necked Phalaropes from last year was M2 (+1,831), while the site with the largest decrease was E4 (-704; Fig. 6). Total sightings of Red-necked Phalaropes at E2, M3, and Sunnyvale WPCP were lower than in any previous phalarope migration survey season. At E5C, M1, M2, and N2, sum sightings of Red-necked Phalaropes were higher than ever previously recorded there during phalarope migration surveys (Fig. 5).

Change Model

Model-averaged coefficient estimates found a strong positive association between changes in Red-necked Phalarope abundance and changes in pH (0.74 importance, 95% confidence intervals not overlapping zero; Figure 8). There was a weaker inverse relationship with changes in salinity, which is correlated with pH in the historical survey dataset (Van Schmidt & Parsons 2025). Decreasing water elevation also was related to increasing abundance, likely reflecting a preference for shallow foraging habitats where phalaropes' unique circular foraging may be able to more effectively bring up benthic invertebrates to the surface. Precipitation showed contraindicated effects: rainfall in the previous year, and more weakly in the current year, was related to decreasing abundance, though rainfall in the past three years was related to increasing abundance. With only five years of data, however, this last measure may be unreliable. There was also support for negative impacts of higher maximum and especially minimum air temperatures. These results indicate that Red-necked Phalaropes are more abundant in South Bay in drier, colder years.

We report predictions made by the change model within 19 ponds, excluding sites that were unoccupied by Red-necked Phalaropes in summer of both 2025 and 2024 (A10–A16, A22, A23, A3N, A3W, A5, A7, A8, Alviso Marina, Coyote Hills Regional Park, Crittenden Marsh, E10, E11, E14, E1C–E4C, E5, E6A, E6C, M4–M6, N3, R2–R5, RSF2U2 and RSF2U3) and sites with missing water quality measurements (N1, Northeast of N1 and NPP1).

Across this set of sites, the actual change in abundance of Red-necked Phalaropes in summer was +1,049 (-942 birds [-79 birds per km^2] in the SBSPRP and +1,991 [+177 per km^2] in the salt ponds and baylands). In comparison, the model predicted a total change of -1,106 (-798 [-67 per km^2] in the SBSPRP and -309 [-27 per km^2] in the salt ponds and baylands), with this predicted decrease driven by effects of changes in site hydrology, as well as effects of weather. The proportion of variance in percent change in counts explained by the best model was low ($R^2 = 0.13$).

The change model estimated that changes in site hydrology could explain a 415 decrease (-420 [-35 per km^2] in the SBSPRP and +5 [+0 per km^2] in the salt ponds and baylands) in abundance of Red-necked Phalaropes from last year due to increasing salinity, higher salinity, decreasing pH, lower pH, and/or increasing depth. Within the SBSPRP footprint, the site predicted to experience the largest increase in abundance of Red-necked Phalaropes due to hydrology changes in summer 2025 was E6, while E7, E6, and E8 were predicted to increase most in habitat suitability (predicted percent change irrespective of actual abundance last year; Table 5). All of these ponds saw increasing pH, which was likely the biggest modeled driver. Salinity decreased markedly at E6/E7 and increased at E8. The overall the effect was to move towards 50 ppt salinity at all three sites, which is close to the ideal value for phalarope prey at Mono Lake (see *Salinity Assocations and Foraging)*. The SBSPRP sites predicted to have the largest decreases in abundance due to hydrology changes were E2, E6B, and E4 and the sites with the largest predicted decreases in habitat suitability were R1, E4, and E6B. In the salt ponds and baylands, the largest predicted increases in abundance were in M3, Spreckles Marsh, and M1 and the largest predicted increases in habitat suitability were in Spreckles Marsh, M3, and M2, while the sites with predicted largest decreases were N2, N4, and Sunnyvale WPCP and N2, N4, and Sunnyvale WPCP for abundance and habitat suitability, respectively (Table 5).

The change model predicted that deviations from mean weather could drive a 175 decrease (+5 [+0 per km^2] in the SBSPRP and -179 [-16 per km^2] in the salt ponds and baylands) in abundance due to high year-to-date precipitation (only weakly supported), high precipitation in the last year, low precipitation in the last three years (indicating complex lag effects), high minimum temperature, and/or high maximum temperature (Table 5).

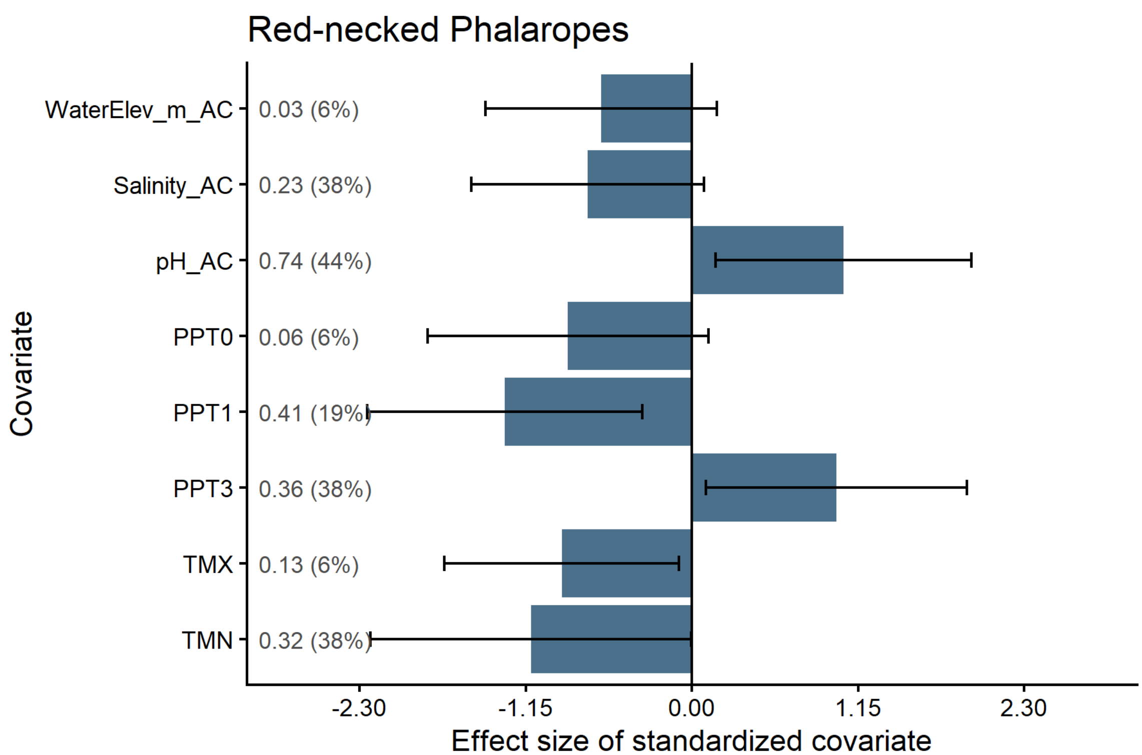


Figure 8. Fixed effects for the top model for interannual change in abundance of Red-necked Phalaropes during summer in South San Francisco Bay, California, water year 2003–2025 (n = 82). Dark bars show coefficients and error bars show 95% confidence intervals. The year random effect was not included in this model due to sample size limits. Covariates are labeled with their importance (cumulative AICc weight) in model selection and the percent of models they were in within the final model set.

Table 5. Actual and predicted changes in mean abundance per survey of Red-necked Phalaropes from summer 2024 to summer 2025 (sorted greatest decrease to greatest increase). Predicted effect sizes of covariate groups are shown for hydrology (salinity, pH, and depth; compared to no change from last year), weather (precipitation last year, precipitation in the last three years, minimum temperature, year-to-date precipitation, and maximum 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. Predicted changes were model-averaged from the 16 candidate models that remained after filtering out uncompetitive nested models.

Pond	Previous abun.	Current abun.	Actual change	Predicted change	Hydrology effect	Weather effect
E4	730	26	-704 (-3.30)	-538 (-1.33)	-398 (-1.12)	+13 (+0.07)
E6B	441	15	-426 (-3.32)	-346 (-1.52)	-219 (-1.19)	-5 (-0.06)
Sunnyvale WPCP	407	1	-406 (-5.32)	-303 (-1.36)	-154 (-0.90)	-20 (-0.17)
E6	300	8	-292 (-3.51)	+153 (+0.41)	+227 (+0.69)	-1 (-0.00)
M3	127	0	-127 (-4.85)	+51 (+0.34)	+110 (+0.95)	-70 (-0.33)
E2	100	1	-99 (-3.92)	-48 (-0.64)	-27 (-0.41)	+3 (+0.06)
N4	41	0	-41 (-3.74)	-12 (-0.32)	-6 (-0.17)	+4 (+0.13)
E8	3	1	-2 (-0.69)	-0 (-0.06)	+1 (+0.27)	-0 (-0.04)
R1	3	1	-2 (-0.69)	-3 (-1.00)	-2 (-0.96)	+0 (+0.24)
New Chicago Marsh	74	78	+4 (+0.05)	-36 (-0.64)	+12 (+0.38)	-43 (-0.74)
E12	8	61	+53 (+1.93)	-4 (-0.57)	-1 (-0.12)	-1 (-0.17)
Spreckles Marsh	10	72	+62 (+1.89)	+23 (+1.14)	+31 (+2.17)	-38 (-0.74)
E5C	4	74	+70 (+2.71)	-0 (-0.09)	+0 (+0.03)	+1 (+0.16)
E13	6	108	+102 (+2.75)	-4 (-0.77)	-1 (-0.35)	-0 (-0.14)
N2	25	160	+135 (+1.82)	-6 (-0.25)	-4 (-0.16)	+3 (+0.19)
E7	0	156	+156 (+5.06)	+1 (+0.81)	+1 (+1.09)	+0 (+0.00)
A9	16	218	+202 (+2.56)	-10 (-0.83)	-1 (-0.07)	-4 (-0.47)
M1	115	648	+533 (+1.72)	-28 (-0.27)	+14 (+0.17)	-15 (-0.16)
M2	2	1,833	+1,831 (+6.42)	+0 (+0.04)	+1 (+0.48)	-1 (-0.16)

Wilson's Phalaropes

Trends

Across all sites, there were 3,583 total sightings of Wilson's Phalaropes (sum sightings during the entire survey period). Compared to the previous year, this was an increase of 2,231 total sightings (165.0%; Fig. 5). Excluding the seven sites outside of the SBSPRP and salt pond areas, we observed a total of 1,542 Wilson's Phalaropes (43.0% of all counts), an increase of 385 total sightings (33.3%) from last year. At the pond level, Crittenden Marsh had the highest abundance (1,293 sum sightings), followed by A9 (776) and N2 (687; Fig. 5). If considering only the SBSPRP and salt pond footprints, the top three most abundant ponds also included E7 (33).

The site with the largest increase in Wilson's Phalaropes from last year was Crittenden Marsh (+1,293), while the site with the largest decrease was E4 (-306; Fig. 6). Total sightings of Wilson's Phalaropes at Sunnyvale WPCP were lower than in any previous phalarope migration survey season. At Crittenden Marsh, E7, N2, and Spreckles Marsh, sum sightings of Wilson's Phalaropes were higher than ever previously recorded there during phalarope migration surveys (Fig. 5).

Change Model

Model-averaged parameters showed that lower (and to a lesser degree, decreasing) water temperature was the strongest predictor of increases in Wilson's Phalarope population (or vice versa) over the past half-decade (Figure 9). Changes in water temperature was by far the dominant driver (0.92 importance). Negative impacts of high rainfall over the past three years and high minimum temperature were also supported, albeit more weakly. Thus like Red-necked Phalaropes, Wilson's Phalaropes are also more abundant in South Bay in drier, colder years.

We report predictions made by the change model within 12 ponds, excluding sites that were unoccupied by Wilson's Phalaropes in summer of both 2025 and 2024 (A10–A16, A22, A23, A3N, A3W, A5, A7, A8, Alviso Marina, E10–E14, E1C–E5C, E6, E6A, E6C, E8, M2–M6, N3, N4, Northeast of N1, NPP1, R1–R5, RSF2U2 and RSF2U3) and sites with missing water quality measurements (N1).

Across this set of sites, the actual change in abundance of Wilson's Phalaropes in summer was +2,230 (-294 birds [-59 birds per km^2] in the SBSPRP and +2,524 [+381 per km^2] in the salt ponds and baylands). In comparison, the model predicted a total change of +4,220 (+1,554 [+311 per km^2] in the SBSPRP and +2,666 [+402 per km^2] in the salt ponds and baylands), with this predicted increase driven by effects of changes in site hydrology. The proportion of variance in percent change in counts explained by the best model was moderate ($R^2 = 0.28$).

The change model estimated that changes in site hydrology could explain a 5,198 increase (+2,401 [+480 per km^2] in the SBSPRP and +2,797 [+422 per km^2] in the salt ponds and baylands) in abundance of Wilson's Phalaropes from last year due to lower water temperature, decreasing water temperature, and/or decreasing water temperature. Within the SBSPRP footprint, the sites predicted to experience the largest increase in abundance of Wilson's Phalaropes due to hydrology changes in summer 2025 were E4, A9, and E5, while E4, E5, and A9 were predicted to increase most in habitat suitability (predicted percent change irrespective of actual abundance last year; Table 6). No SBSPRP sites were predicted to decreases in abundance due to hydrology changes. In the salt ponds and baylands, the largest predicted increases in abundance were in Spreckles Marsh, New Chicago Marsh, and Crittenden Marsh and the largest predicted increases in habitat suitability were in New Chicago Marsh, Spreckles Marsh, and Crittenden Marsh, while the sites with predicted largest decreases were Coyote Hills Regional Park and Coyote Hills Regional Park for abundance and habitat suitability, respectively (Table 6).

The change model predicted that deviations from mean weather could drive a 593 decrease (-596 [-119 per km^2] in the SBSPRP and +3 [+0 per km^2] in the salt ponds and baylands) in abundance due to high precipitation in the last three years and/or high minimum temperature (Table 6).

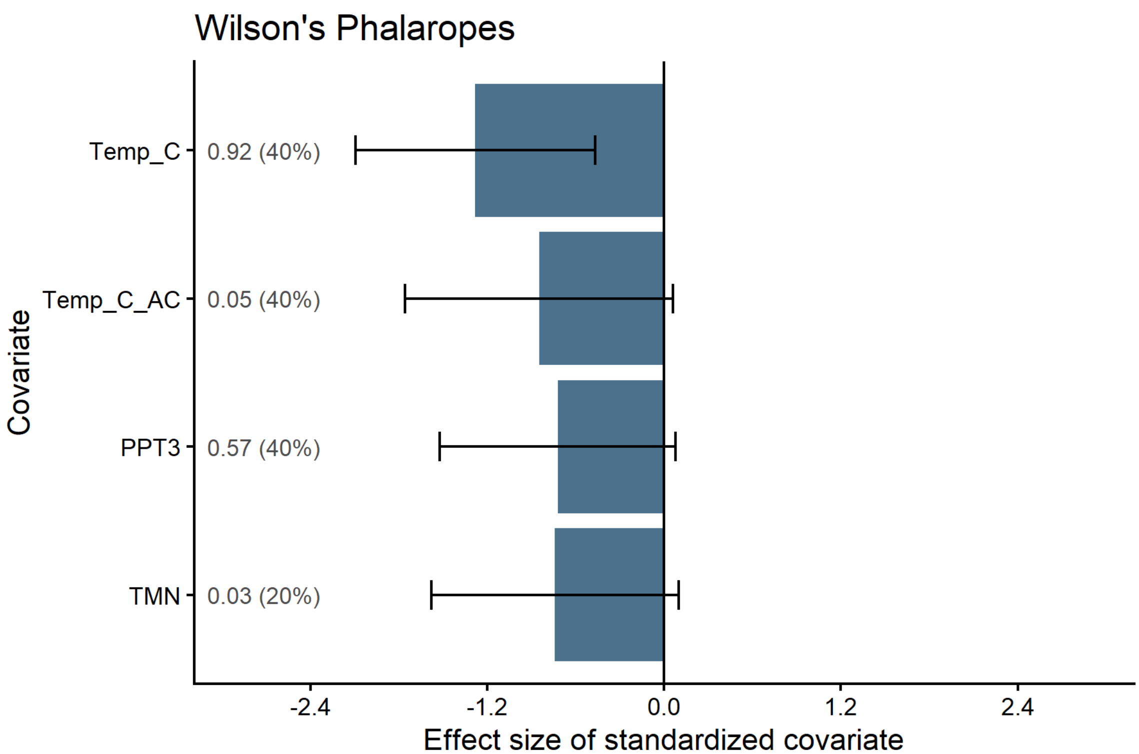


Figure 9. Fixed effects for the top model for interannual change in abundance of Wilson's Phalaropes during summer in South San Francisco Bay, California, water year 2003–2025 (n = 35). Dark bars show coefficients and error bars show 95% confidence intervals. The year random effect was not included in this model due to sample size limits. Covariates are labeled with their importance (cumulative AICc weight) in model selection and the percent of models they were in within the final model set.

Table 6. Actual and predicted changes in mean abundance per survey of Wilson's Phalaropes from summer 2024 to summer 2025 (sorted greatest decrease to greatest increase). Predicted effect sizes of covariate groups are shown for hydrology (dissolved oxygen and water temperature; compared to no change from last year), weather (maximum temperature, precipitation in the last three years, and 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. Predicted changes were model-averaged from the 9 candidate models that remained after filtering out uncompetitive nested models.

Pond	Previous abun.	Current abun.	Actual change	Predicted change	Hydrology effect	Weather effect
E4	337	31	-306 (-2.36)	+1,961 (+1.92)	+2,173 (+2.91)	-581 (-0.23)
A9	805	776	-29 (-0.04)	-411 (-0.71)	+223 (+0.83)	-13 (-0.03)
Sunnyvale WPCP	22	0	-22 (-3.14)	-4 (-0.21)	+14 (+1.38)	+0 (+0.01)
E5	0	4	+4 (+1.61)	+3 (+1.48)	+4 (+2.73)	-1 (-0.23)
E6B	0	4	+4 (+1.61)	+1 (+0.63)	+0 (+0.27)	-1 (-0.26)
M1	0	6	+6 (+1.95)	+1 (+0.87)	+1 (+0.77)	-0 (-0.05)
Coyote Hills Regional Park	0	7	+7 (+2.08)	+0 (+0.20)	-0 (-0.29)	-0 (-0.18)
E7	0	33	+33 (+3.53)	+0 (+0.08)	+0 (+0.48)	-0 (-0.24)
New Chicago Marsh	16	137	+121 (+2.09)	+213 (+2.61)	+227 (+4.43)	+0 (+0.00)
Spreckles Marsh	157	604	+447 (+1.34)	+2,265 (+2.73)	+2,387 (+4.21)	+7 (+0.00)
N2	15	687	+672 (+3.76)	+21 (+0.84)	+0 (+0.00)	-5 (-0.13)
Crittenden Marsh	0	1,293	+1,293 (+7.17)	+169 (+5.14)	+167 (+4.18)	+2 (+0.01)

Total Phalaropes

Trends

Across all sites, there were 9,066 total sightings of total phalaropes (sum sightings during the entire survey period). Compared to the previous year, this was an increase of 4,231 total sightings (87.5%; Fig. 5). Excluding the seven sites outside of the SBSPRP and salt pond areas, we observed a total of 6,871 total phalaropes (75.8% of all counts), an increase of 3,710 total sightings (117.4%) from last year. At the pond level, M2 had the highest abundance (1,833), followed by M1 (1,579) and A9 (1,378; Fig. 5).

The site with the largest increase in total phalaropes from last year was M2 (+1,831), while the site with the largest decrease was E4 (-1,026; Fig. 6).

Change Model

The best change model for total phalaropes was a mixture of strongest results for Red-necked and Wilson's Phalaropes, with both diluted, and several others covariates dropping out (Figure 10). Salinity was not important, unlike in Red-necked Phalaropes, and the contradictory positive and negative effects of past three years' precipitation on Wilson's and Red-necked Phalaropes, respectively, resulted in an overall positive effect because Red-necked Phalaropes are much more abundant within South Bay. The other parameters were generally the same, albeit weaker, reflecting the potential pitfall of combining multiple species with potentially divergent habitat needs and responses in a single model.

We report predictions made by the change model within 22 ponds, excluding sites that were unoccupied by phalaropes in summer of both 2025 and 2024 (A10–A16, A22, A23, A3N, A3W, A5, A7, A8, Alviso Marina, E10, E11, E14, E1C–E4C, E6A, E6C, M4–M6, N3, R2–R5, RSF2U2 and RSF2U3) and sites with missing water quality measurements (N1, Northeast of N1 and NPP1).

Across this set of sites, the actual change in abundance of phalaropes in summer was +3,602 (-918 birds [-73 birds per km2] in the SBSPRP and +4,520 [+370 per km2] in the salt ponds and baylands). In comparison, the model predicted a total change of -728 (-411 [-33 per km2] in the SBSPRP and -317 [-26 per km2] in the salt ponds and baylands), with this predicted decrease driven by effects of weather. The proportion of variance in percent change in counts explained by the best model was very low ($R^2 = 0.05$).

The change model estimated that changes in site hydrology could explain a 331 increase (+54 [+4 per km2] in the SBSPRP and +277 [+23 per km2] in the salt ponds and baylands) in abundance of phalaropes from last year due to lower water temperature, increasing pH, and/or increasing pH. Within the SBSPRP footprint, the sites predicted to experience the largest increase in abundance of phalaropes due to hydrology changes in summer 2025 were E6, A9, and E4, while E7, E5, and E6 were predicted to increase most in habitat suitability (predicted percent change irrespective of actual abundance last year; Table 7). The SBSPRP sites predicted to have the largest decreases in abundance due to hydrology changes were E13, R1, and E6B and the sites with the largest predicted decreases in habitat suitability were E13, E6B, and R1. In the salt ponds and baylands, the largest predicted increases in abundance were in New Chicago Marsh, Spreckles Marsh, and M3 and the largest predicted increases in habitat suitability were in Spreckles Marsh, New Chicago Marsh, and M3, while the sites with predicted largest decreases were N4, N2, and Sunnyvale WPCP and Sunnyvale WPCP, Crittenden Marsh, and Coyote Hills Regional Park for abundance and habitat suitability, respectively (Table 7).

The change model predicted that deviations from mean weather could drive a 881 decrease (-292 [-23 per km2] in the SBSPRP and -589 [-48 per km2] in the salt ponds and baylands) in abundance due to high precipitation last year, low precipitation in the last three years, high minimum temperature, and/or high year-to-date precipitation (Table 7).

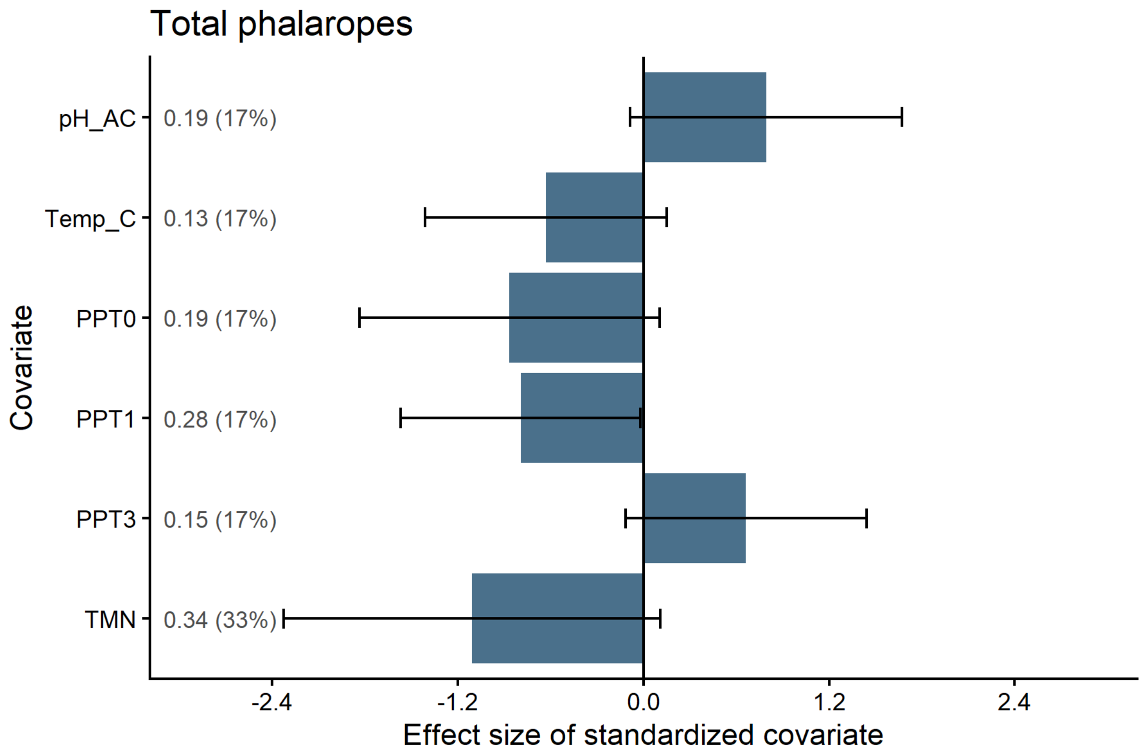


Figure 10. Fixed effects for the top model for interannual change in abundance of phalaropes during summer in South San Francisco Bay, California, water year 2003–2025 (n = 85). Dark bars show coefficients and error bars show 95% confidence intervals. The year random effect was not included in this model due to sample size limits. Covariates are labeled with their importance (cumulative AICc weight) in model selection and the percent of models they were in within the final model set.

Table 7. Actual and predicted changes in mean abundance per survey of phalaropes from summer 2024 to summer 2025 (sorted greatest decrease to greatest increase). Predicted effect sizes of covariate groups are shown for hydrology (water temperature, dissolved oxygen, and pH; compared to no change from last year), weather (precipitation last year, precipitation in the last three years, minimum temperature, year-to-date precipitation, and maximum 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. Predicted changes were model-averaged from the 15 candidate models that remained after filtering out uncompetitive nested models.

Pond	Previous abun.	Current abun.	Actual change	Predicted change	Hydrology effect	Weather effect
E4	1,083	57	-1,026 (-2.93)	-42 (-0.04)	+26 (+0.03)	+13 (+0.01)
Sunnyvale WPCP	617	1	-616 (-5.73)	-98 (-0.17)	-42 (-0.08)	+19 (+0.04)
New Chicago Marsh	789	216	-573 (-1.29)	-222 (-0.33)	+173 (+0.36)	-410 (-0.54)
E6B	441	19	-422 (-3.10)	-115 (-0.30)	-70 (-0.19)	-44 (-0.13)
E6	350	8	-342 (-3.66)	+36 (+0.10)	+62 (+0.17)	-25 (-0.06)
M3	127	7	-120 (-2.77)	-8 (-0.06)	+20 (+0.18)	-28 (-0.21)
E2	100	1	-99 (-3.92)	+0 (+0.00)	+7 (+0.08)	-3 (-0.03)
N4	41	0	-41 (-3.74)	+7 (+0.15)	-1 (-0.03)	+6 (+0.14)
E8	3	1	-2 (-0.69)	-1 (-0.16)	-0 (-0.11)	-0 (-0.12)
R1	3	1	-2 (-0.69)	+0 (+0.07)	-1 (-0.26)	+1 (+0.29)
E5	0	4	+4 (+1.61)	+0 (+0.12)	+0 (+0.22)	-0 (-0.01)
Coyote Hills Regional Park	0	7	+7 (+2.08)	-0 (-0.06)	-0 (-0.30)	+0 (+0.22)
E12	8	61	+53 (+1.93)	-2 (-0.31)	-1 (-0.09)	-2 (-0.25)

Pond	Previous abun.	Current abun.	Actual change	Predicted change	Hydrology effect	Weather effect
E5C	4	74	+70	+1	+1	+1
			(+2.71)	(+0.16)	(+0.11)	(+0.12)
E13	6	108	+102	-2	-1	-1
			(+2.75)	(-0.36)	(-0.15)	(-0.23)
E7	0	189	+189	+0	+0	-0
			(+5.25)	(+0.13)	(+0.23)	(-0.06)
Spreckles Marsh	264	676	+412	-7	+121	-182
			(+0.94)	(-0.03)	(+0.63)	(-0.53)
A9	821	1,378	+557	-286	+31	-231
			(+0.52)	(-0.43)	(+0.06)	(-0.36)
N2	40	901	+861	+11	-2	+11
			(+3.09)	(+0.24)	(-0.03)	(+0.23)
Crittenden Marsh	0	1,295	+1,295	+0	-0	+0
			(+7.17)	(+0.11)	(-0.20)	(+0.27)
M1	115	1,579	+1,464	+1	+8	-6
			(+2.61)	(+0.01)	(+0.07)	(-0.05)
M2	2	1,833	+1,831	+0	+0	-0
			(+6.42)	(+0.01)	(+0.08)	(-0.05)

Salinity Associations and Foraging

Summing across all years of full survey rounds (2021-2025), phalaropes demonstrated a strong association with a relatively wide band of moderately high salinity (30-150 ppt, encompassing the marine to medium hypersaline categories; Fig. 11). This closely aligned with the reported tolerance of *Ephydra* and *Artemia* spp. (Brown 2010). This trend was highly statistically significant (Table 8). 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 both species used all three of these bins. Ponds outside of these limits were generally avoided. The one exception was freshwater ponds, where 77% of sightings were at Sunnyvale Water Pollution Control Plant (WPCP) whose waste ponds are presumably uniquely very high in nutrients and therefore unlike other ponds within the study area or other freshwater ponds. The remaining sightings in near-fresh were in N1.

Table 8. Regression coefficients of a negative binomial model (dispersion parameter = 0.0489) with random effects for observer (variance = 1.3085) and pond (variance = 6.7479) to test for significant effects of water salinity on phalarope abundance (sum sightings over summer and early fall) within South San Francisco Bay, California, 2021-2025 (n = 1224 pond-surveys). Marine salinity is the reference level.

	Estimate	**Std. Error**	**z value**	**Pr(>|z|)**
(Intercept)	1.7282	0.7420	2.3291	0.0199
Near-fresh	2.1376	1.9790	1.0801	0.2801
Brackish	-4.3109	1.1338	-3.8021	0.0001
Low hypersaline	0.7997	0.7800	1.0253	0.3052
Mid hypersaline	-0.6800	0.8134	-0.8361	0.4031
High hypersaline	-4.8378	1.3614	-3.5536	0.0004

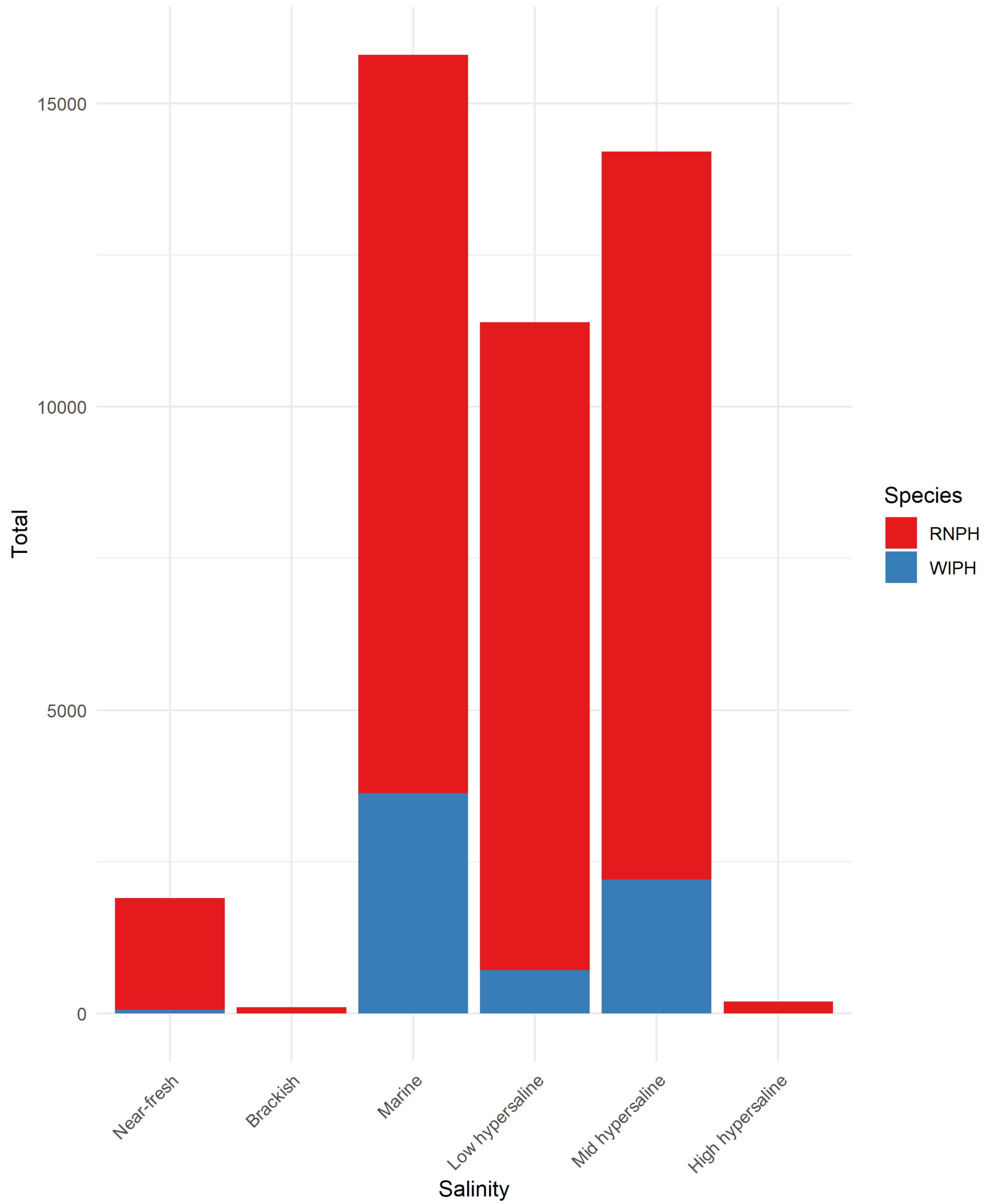


Figure 11. Total sightings of Wilson's (WIPH) and Red-necked (RNPH) Phalarope in phalarope migration surveys from 2021-2025 across ponds grouped into six bands of salinity.

Foraging Attempts

A total of 416 one-minute foraging observations were recorded across 24 sites by 26 observers. At one visit to Sunnyvale WPCP in early 2024, Wilson's Phalaropes were observed engaging in the continuous foraging behavior for the first time and recorded as a mean of 78 attempts/minute (no success data was recorded as pecks were too rapid to count). This outlier observation of the continuous foraging behavior under the discrete foraging protocol was removed from the dataset. It was otherwise the only observation of Wilson's Phalaropes foraging at ponds below the marine salinity level (<30 ppt).

The number of foraging attempts per minute varied across the salinity gradient and was generally highest in the low-hypersaline category and decreased as salinities lowered below or raised above this range (Fig. 12). High hypersaline ponds (>150 ppt) had significantly fewer foraging attempts relative to marine, and brackish sites trended lower as well (Table 9). An exception was the near-fresh Sunnyvale WPCP, which is unusually nutrient-rich and not representative of normal freshwater sites; both Red-necked and Wilson's Phalaropes showed elevated foraging attempt rates here, though this effect was particularly pronounced for Wilson's Phalarope (Fig. 13). Observations of phalarope foraging behavior showed patterns that were broadly similar between species, though Red-necked Phalaropes were observed using more freshwater habitats.

Table 9. Regression coefficients of a negative binomial model (dispersion modeled as a function of salinity category) with random effects for observer (variance = 0.1236) and pond (variance = 0.0156) for foraging attempts within South San Francisco Bay, California, 2024-2025 (n = 386 observations). Marine salinity is the reference level.

Parameter	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	3.1133	0.1173	26.5423	0.0000
Near-fresh	0.0868	0.2662	0.3261	0.7443
Brackish	-0.4964	0.4082	-1.2162	0.2239
Low hypersaline	-0.0600	0.1193	-0.5031	0.6149
Mid hypersaline	-0.0551	0.1187	-0.4644	0.6423
High hypersaline	-0.5397	0.2590	-2.0841	0.0372
Dispersion: (Intercept)	1.9784	0.1849	10.7004	0.0000
Dispersion: Near-fresh	1.7051	1.0098	1.6886	0.0913
Dispersion: Brackish	0.7884	1.0834	0.7277	0.4668
Dispersion: Low hypersaline	0.2952	0.2723	1.0840	0.2784
Dispersion: Mid hypersaline	-0.1659	0.2625	-0.6323	0.5272
Dispersion: High hypersaline	0.6143	1.4901	0.4123	0.6801

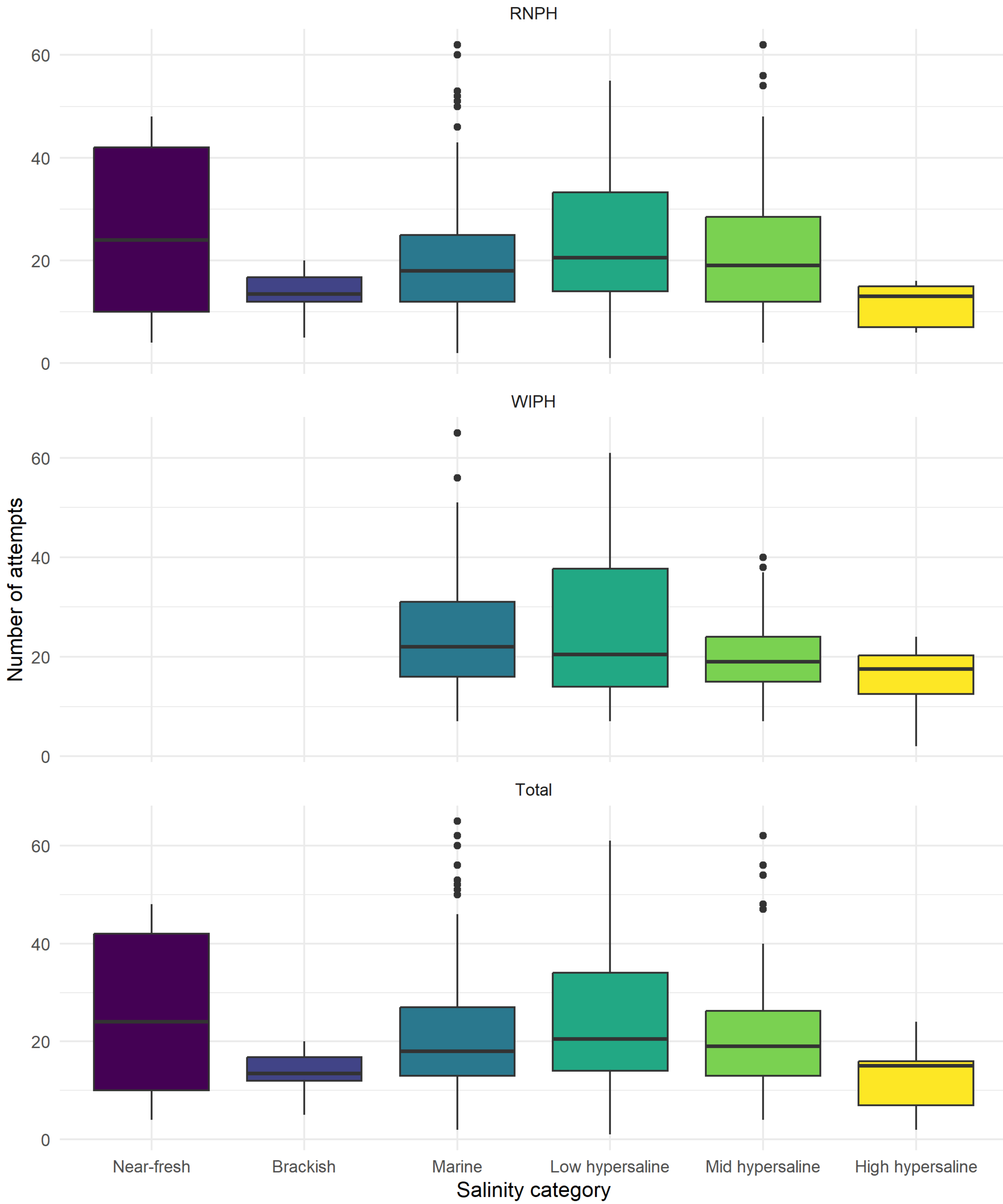


Figure 12. Boxplots of the number of foraging attempts (pecks) per minute by phalaropes within South San Francisco Bay, 2024-2025, divided by salinity category. Panels show Red-necked Phalaropes (RNPH, top), Wilson's Phalaropes (WIPH, middle), and total observations (bottom).

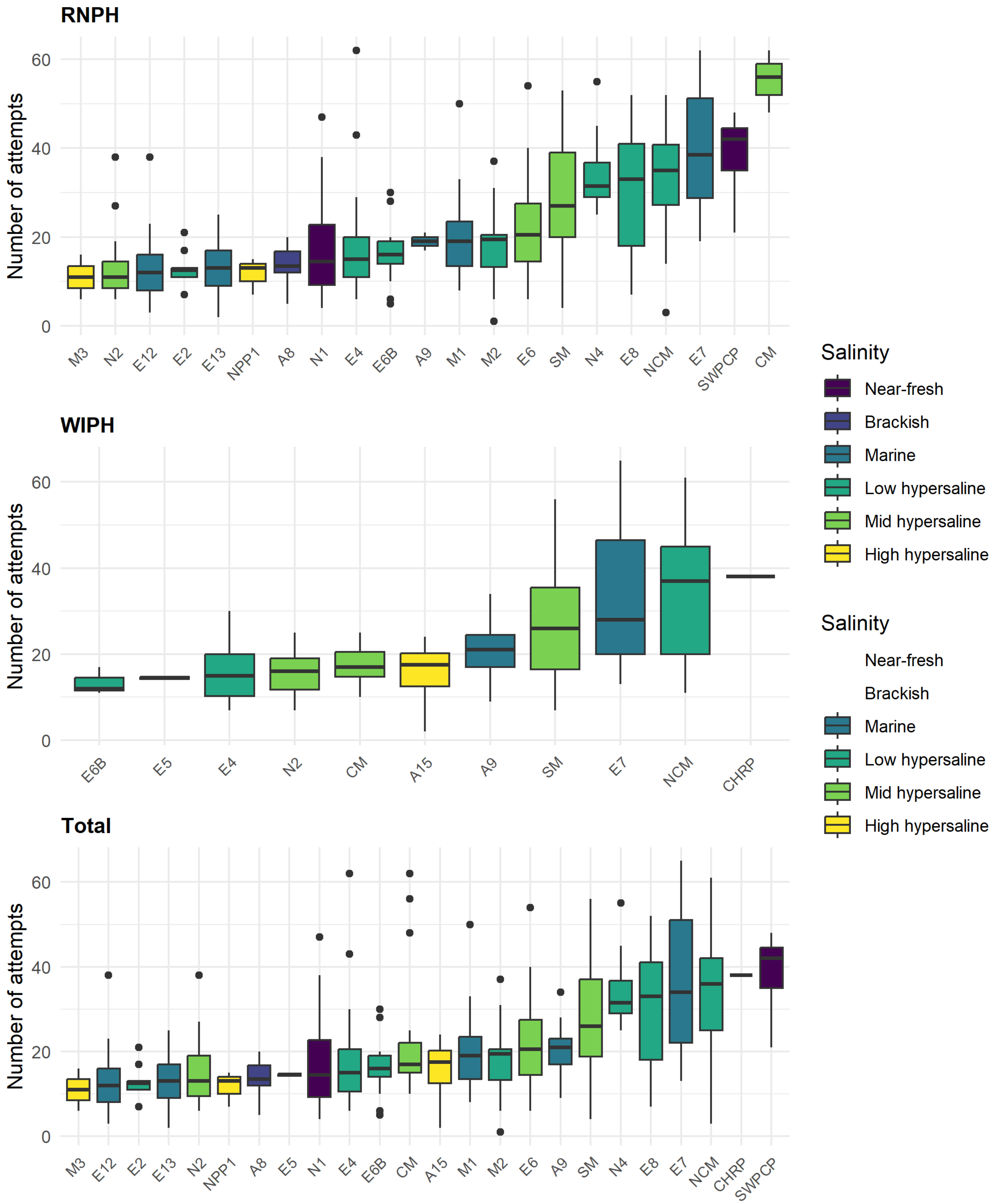


Figure 13. Boxplots of foraging attempts per minute by site, shaded by modal salinity category. Sites are ordered by median attempts within each panel. Panels show Red-necked Phalaropes (RNPH, top), Wilson's Phalaropes (WIPH, middle), and total observations (bottom).

Foraging Successes

Patterns in successful prey captures broadly paralleled those of total attempts, with brackish and high hypersaline sites having fewer successful foraging attempts on average (Fig. 14). However, neither category was significantly different from marine (Table 10). As Wilson's Phalaropes generally did not forage at near-fresh or brackish sites, they also did not succeed at foraging there, strongly suggesting ponds with salinity <30 ppt have little habitat value for Wilson's Phalaropes. Based on successful foraging attempts, the sites that appeared to offer the greatest foraging habitat value were E7, Spreckles Marsh, Sunnyvale WPCP, New Chicago Marsh, and N4 (Fig. 15; excluding Coyote Hills Regional Park where only a single bird was observed foraging).

Table 10. Regression coefficients of a negative binomial model (dispersion parameter = 4.8854) with random effects for observer (variance = 0.3987) and pond (variance = 0.0895) for foraging successes within South San Francisco Bay, California, 2024-2025 (n = 378 observations). Marine salinity is the reference level. This model showed signs of heteroskedasticity, so p-values may be unreliable.

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	2.1954	0.1930	11.3743	0.0000
Near-fresh	0.1208	0.4349	0.2777	0.7813
Brackish	-0.8630	0.7568	-1.1403	0.2542
Low hypersaline	-0.3770	0.2262	-1.6667	0.0956
Mid hypersaline	-0.1151	0.1915	-0.6010	0.5479
High hypersaline	-0.6178	0.4973	-1.2423	0.2141

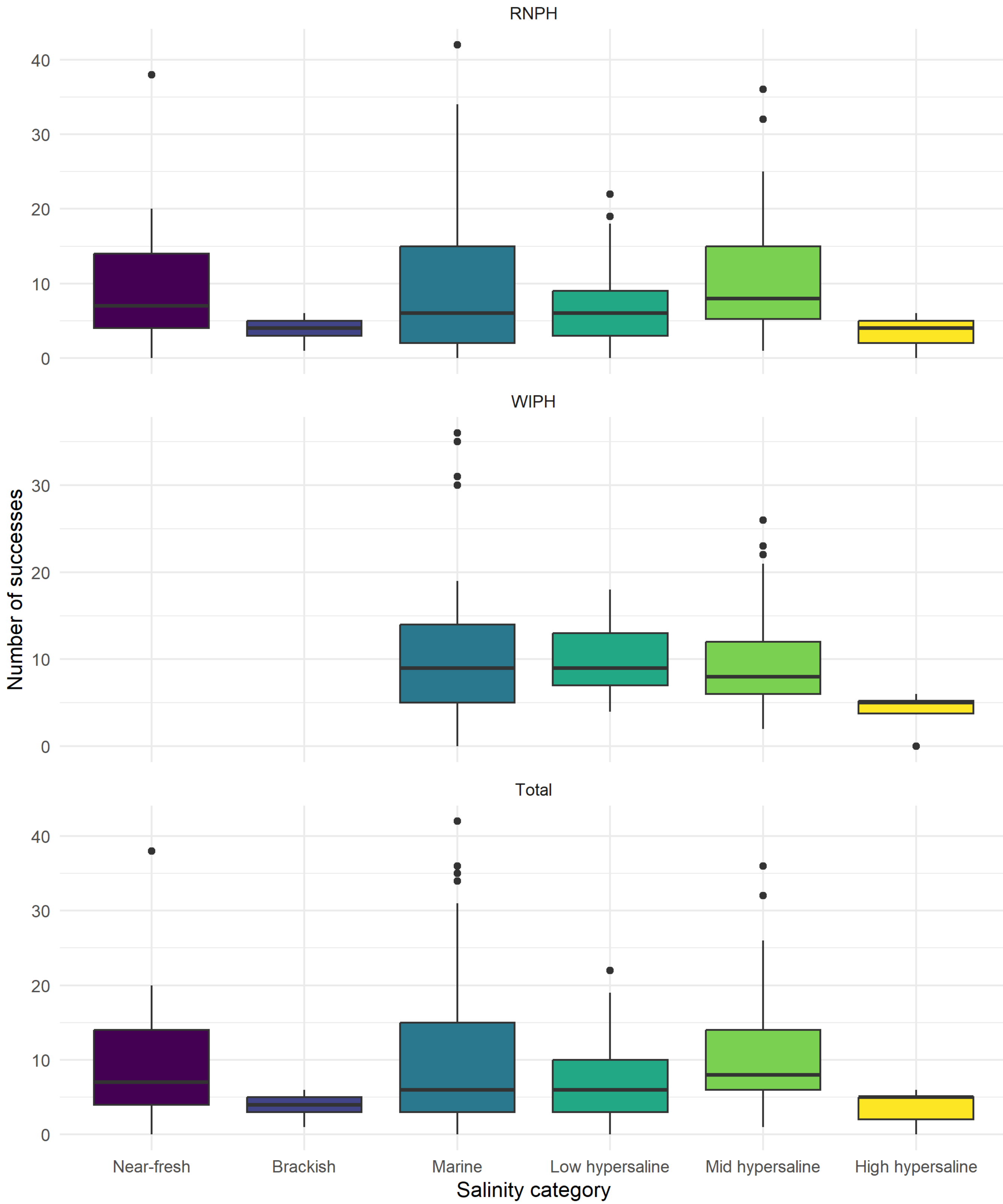


Figure 14. Boxplots of the number of successful foraging attempts (pecks followed by swallows) per minute by phalaropes within South San Francisco Bay, 2024-2025, divided by salinity category. Panels show Red-necked Phalaropes (RNPH, top), Wilson's Phalaropes (WIPH, middle), and total observations (bottom).

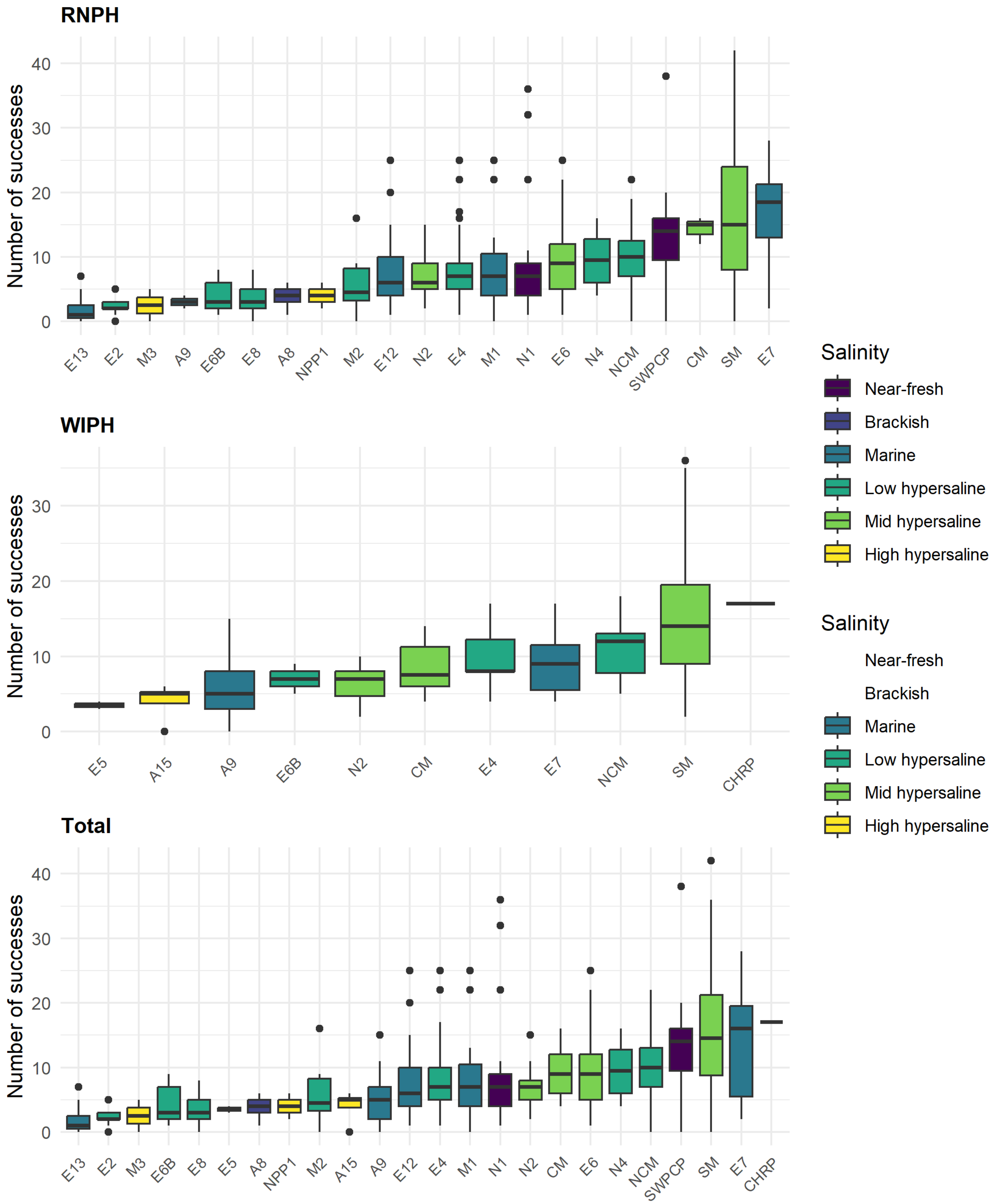


Figure 15. Boxplots of foraging successes per minute by site, shaded by modal salinity category. Sites are ordered by median successes within each panel. Panels show Red-necked Phalaropes (RNPH, top), Wilson's Phalaropes (WIPH, middle), and total observations (bottom).

Success Rate

The median foraging success rate (proportion of attempts that resulted in successful prey capture) was highest at moderately hypersaline sites for Red-necked Phalaropes and at low hypersaline sites for Wilson's Phalaropes, and notably lower for Wilson's Phalarope at highly hypersaline sites (Fig. 16). However, these patterns were not statistically significant (Table 11). The consistency of the success ratio across salinity categories, in contrast to the variation in absolute attempt and success counts, suggests that differences in foraging activity across the gradient are driven primarily by differences in foraging effort (attempts) rather than differences in per-attempt capture efficiency.

Table 11. Regression coefficients of a linear Gaussian model with random effects for observer (variance = 0.0158) and pond (variance = 0.0055) for foraging success rate within South San Francisco Bay, California, 2024-2025 (n = 378 observations). Marine salinity is the reference level.

	Estimate	Std. Error	df	t value	Pr(>|t|)
(Intercept)	0.4077	0.0447	24.2135	9.1226	0.0000
Near-fresh	0.0251	0.1115	17.0408	0.2251	0.8246
Brackish	-0.0942	0.1627	20.2539	-0.5789	0.5690
Low hypersaline	-0.0692	0.0558	18.2994	-1.2406	0.2304
Mid hypersaline	0.0380	0.0504	45.2526	0.7539	0.4548
High hypersaline	-0.0535	0.1208	36.8668	-0.4427	0.6605

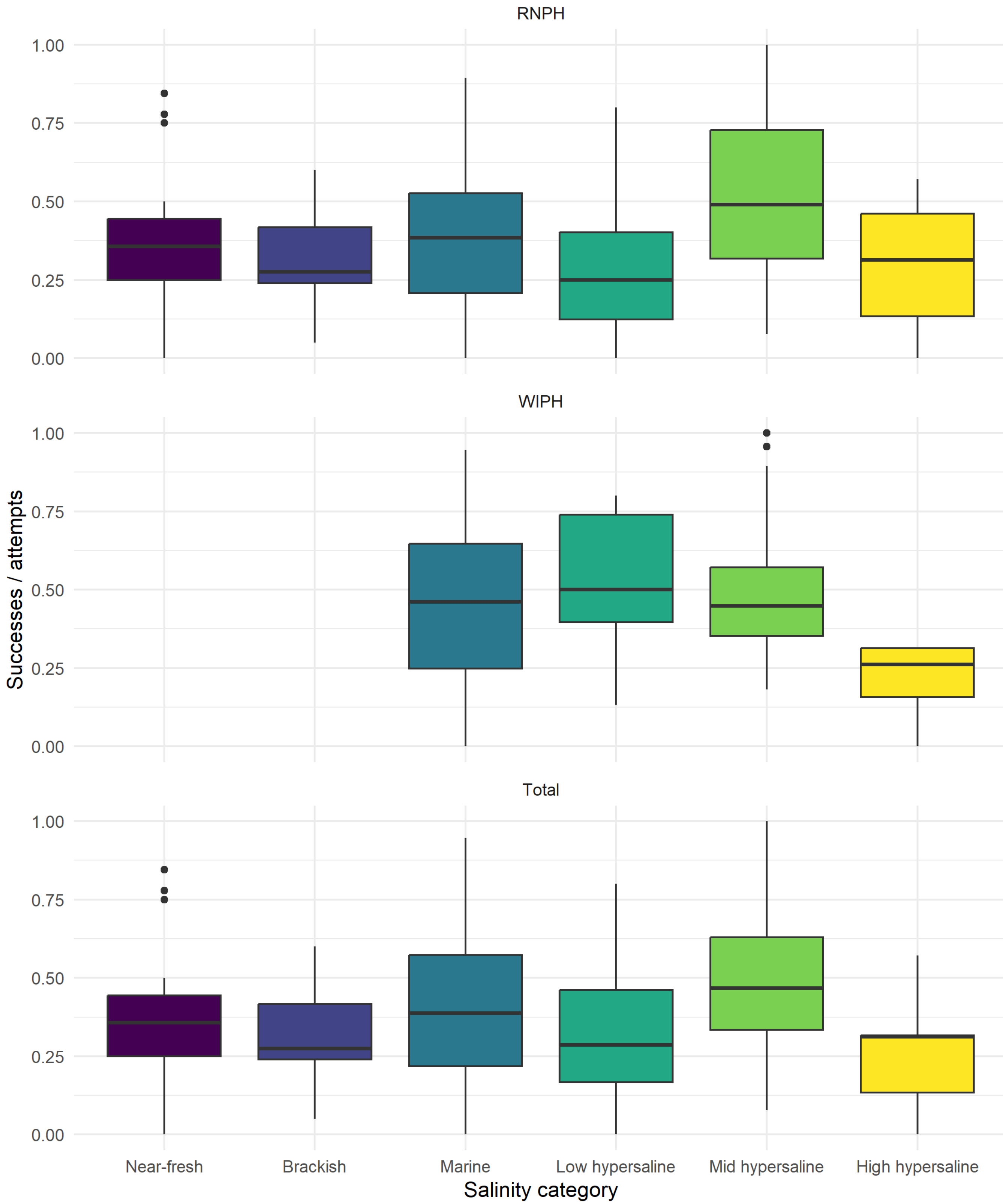


Figure 16. Boxplots of the foraging success ratio (successes / attempts in one minute) for phalaropes within South San Francisco Bay, 2024-2025, divided by salinity category. Panels show Red-necked Phalaropes (RNPH, top), Wilson's Phalaropes (WIPH, middle), and total observations (bottom).

Continuous Foraging Time

At 5 sites (A9, E4, Sunnyvale WPCP, Spreckles Marsh, and New Chicago Marsh), 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. For these observations (n = 74), surveyors recorded the number of seconds spent continuously foraging within the one-minute observation window rather than counting individual pecks. These sites spanned three salinity categories (near-fresh, marine, and low hypersaline). Despite the small sample size, phalaropes at the near-fresh Sunnyvale WPCP site spent significantly more time in continuous foraging behavior compared to the marine reference (Table 12), averaging approximately 14 additional seconds per minute of continuous foraging. Birds at low hypersaline sites trended toward 8 seconds less foraging per minute than marine, but this was not significant. These observations, while preliminary, appear to indicate sites with very high prey density where rapid continuous foraging is more efficient than individual strike attempts.

Table 12. Regression coefficients of a linear Gaussian model with random effects for pond (variance = 0.1778) for continuous foraging time (seconds per minute) within South San Francisco Bay, California, 2024-2025 (n = 74 observations from 5 sites). Marine salinity is the reference level. This model showed signs of heteroskedasticity, so p-values may be unreliable.

	Estimate	**Std. Error**	**df**	**t value**	**Pr(>|t|)**
(Intercept)	38.3655	3.6604	6.8533	10.4813	0.0000
Near-fresh	14.2709	6.0410	9.0052	2.3623	0.0424
Low hypersaline	-8.8409	4.3850	4.5254	-2.0162	0.1058

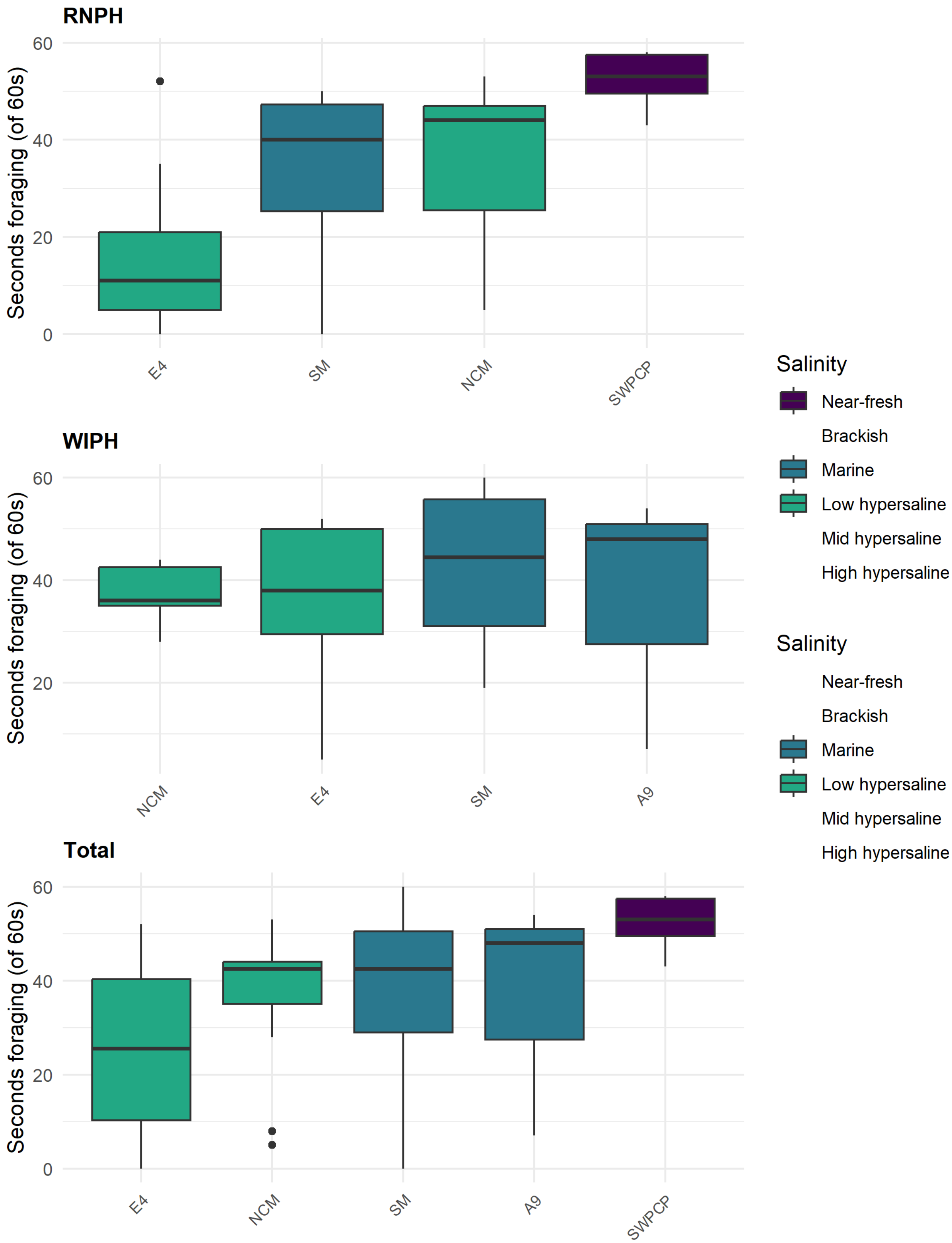


Figure 17. Boxplots of continuous foraging time (seconds actively foraging out of 60 seconds) by site, shaded by modal salinity category. Sites are ordered by median foraging time within each panel. Panels show Red-necked Phalaropes (RNPH, top), Wilson's Phalaropes (WIPH, middle), and total observations (bottom).

Water Quality

We here report trends in water quality from samples taken between September 2 and October 15. 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 17.75 ppt at A8 to 334 ppt at R2. Salinity changes from last year to this year ranged from -6.52 ppt at R1 to +27.5 ppt at R2 (Fig. 18). The mean interannual change in salinity during this survey round across all sites was +8.18 ppt.

Average dissolved oxygen concentrations in the Don Edwards complex ranged from a low of 2.66 mg/L at A15 to a high of 15.36 mg/L at A5. Changes in dissolved oxygen since last year ranged from -4.71 mg/L at A3N to +3.31 mg/L at RSF2U2 (Fig. 19). The mean interannual change in dissolved oxygen across all sites was -1.4 mg/L.

Average pH values ranged from a low of 6.83 in A12 to a high of 9.08 in A14. Changes in pH from last year ranged from -0.35 at A12 to +1.19 at A14 (Fig. 20). The mean interannual change in pH across all sites was +0.07.

Water temperature at the surface ranged from 21.33 ˚C at A3N to 32.13 ˚C at R2. Compared to last year, water temperature changes in ponds ranged from -6.64 ˚C at A3N to +6.58 ˚C at A14 (Fig. 21). The mean interannual change in temperature across all sites was +0.53 ˚C.

For ponds where readings could be taken, staff gauge levels ranged in the Don Edwards complex from 2 ft in A3W to 5.66 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 -0.3 ft in A16 to +3.2 ft in A3W (Fig. 22). The mean interannual change in water level across all sites was +1.45 ft.

Some water quality information was unavailable for certain surveys at ponds A12, A15, A22, A23, and RSF2U3 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 32.31 ppt at E2C to 228.5 ppt at E6C. Salinity changes from last year to this year ranged from -51.15 ppt at E6 to +105.36 ppt at E1C (Fig. 18). The mean interannual change in salinity during this survey round across all sites was +7.01 ppt.

Average dissolved oxygen concentrations in the Eden Landing complex ranged from a low of 2.05 mg/L at E4 to a high of 19.07 mg/L at E7. Changes in dissolved oxygen since last year ranged from -9.67 mg/L at E4 to +9.79 mg/L at E7 (Fig. 19). The mean interannual change in dissolved oxygen across all sites was +0.39 mg/L.

Average pH values ranged from a low of 7.76 in E6C to a high of 9.5 in E7. Changes in pH from last year ranged from -0.67 at E10 to +1.01 at E7 (Fig. 20). The mean interannual change in pH across all sites was +0.13.

Water temperature at the surface ranged from 19.08 °C at E10 to 30.86 °C at E8. Compared to last year, water temperature changes in ponds ranged from -9.92 °C at E4 to +11.75 °C at E8 (Fig. 21). The mean interannual change in temperature across all sites was -1.89 °C.

For ponds where readings could be taken, staff gauge levels ranged in the Eden Landing complex from 3.6 ft in E6 to 6.29 ft in E2C. For ponds where staff gauge measurements could be obtained in both years, changes in seasonal water depth from last year ranged from -0.5 ft in E10 to +2.59 ft in E2C (Fig. 22). The mean interannual change in water level across all sites was +1.03 ft.

Some water quality information was unavailable for certain surveys at pond E11 because it was surveyed on days when it was dry or when water quality equipment was malfunctioning.

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 0.56 ppt at N1 to 238.67 ppt at M6. Salinity changes from last year to this year ranged from -114.77 ppt at N1 to +26 ppt at N4 (Fig. 18). The mean interannual change in salinity during this survey round across all sites was -30.36 ppt.

Average dissolved oxygen concentrations in the Salt Ponds complex ranged from a low of 2.04 mg/L at M6 to a high of 9.58 mg/L at M1. Changes in dissolved oxygen since last year ranged from -1.16 mg/L at N4 to +1.35 mg/L at N3 (Fig. 19). The mean interannual change in dissolved oxygen across all sites was +0.01 mg/L.

Average pH values ranged from a low of 7.58 in M6 to a high of 8.69 in M2. Changes in pH from last year ranged from -0.37 at NPP1 to +0.42 at M3 (Fig. 20). The mean interannual change in pH across all sites was +0.01.

Water temperature at the surface ranged from 22.6 °C at N3 to 26.39 °C at M6. Compared to last year, water temperature changes in ponds ranged from -6.46 °C at M4 to +3.62 °C at N4 (Fig. 21). The mean interannual change in temperature across all sites was -1.51 °C.

For ponds where readings could be taken, staff gauge levels ranged in the Salt Ponds complex from 0.6 ft in N1 to 8.06 ft in M3. For ponds where staff gauge measurements could be obtained in both years, changes in seasonal water depth from last year ranged from -0.9 ft in N2 to +5.16 ft in M3 (Fig. 22). The mean interannual change in water level across all sites was +0.85 ft.

Some water quality information was unavailable for certain surveys at ponds N1 and NPP1 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. Crittenden Marsh underwent a dramatic shift in conditions from last year as water levels fell, increasing markedly in salinity.

Average salinity ranged from 1.94 ppt at Sunnyvale WPCP to 138 ppt at Crittenden Marsh. Salinity changes from last year to this year ranged from -49.2 ppt at Spreckles Marsh to +113.7 ppt at Crittenden Marsh (Fig. 18). The mean interannual change in salinity during this survey round across all sites was +19.9 ppt.

Average dissolved oxygen concentrations in the Baylands complex ranged from a low of 4.87 mg/L at Northeast of N1 to a high of 65.65 mg/L at Sunnyvale WPCP. Changes in dissolved oxygen since last year ranged from -7.11 mg/L at Coyote Hills Regional Park to +59.01 mg/L at Sunnyvale WPCP (Fig. 19). The mean interannual change in dissolved oxygen across all sites was +8.29 mg/L. The extreme extreme dissolved oxygen values measured at Sunnyvale WPCP may indicate an algal bloom at this unique site.

Average pH values ranged from a low of 7.74 in Crittenden Marsh to a high of 9.69 in Spreckles Marsh. Changes in pH from last year ranged from -1.39 at Crittenden Marsh to +1.1 at Spreckles Marsh (Fig. 20). The mean interannual change in pH across all sites was -0.07.

Water temperature at the surface ranged from 12.45 ˚C at Crittenden Marsh to 27.97 ˚C at Sunnyvale WPCP. Compared to last year, water temperature changes in ponds ranged from -13.81 ˚C at Crittenden Marsh to -2.32 ˚C at Coyote Hills Regional Park (Fig. 21). The mean interannual change in temperature across all sites was -8.17 ˚C.

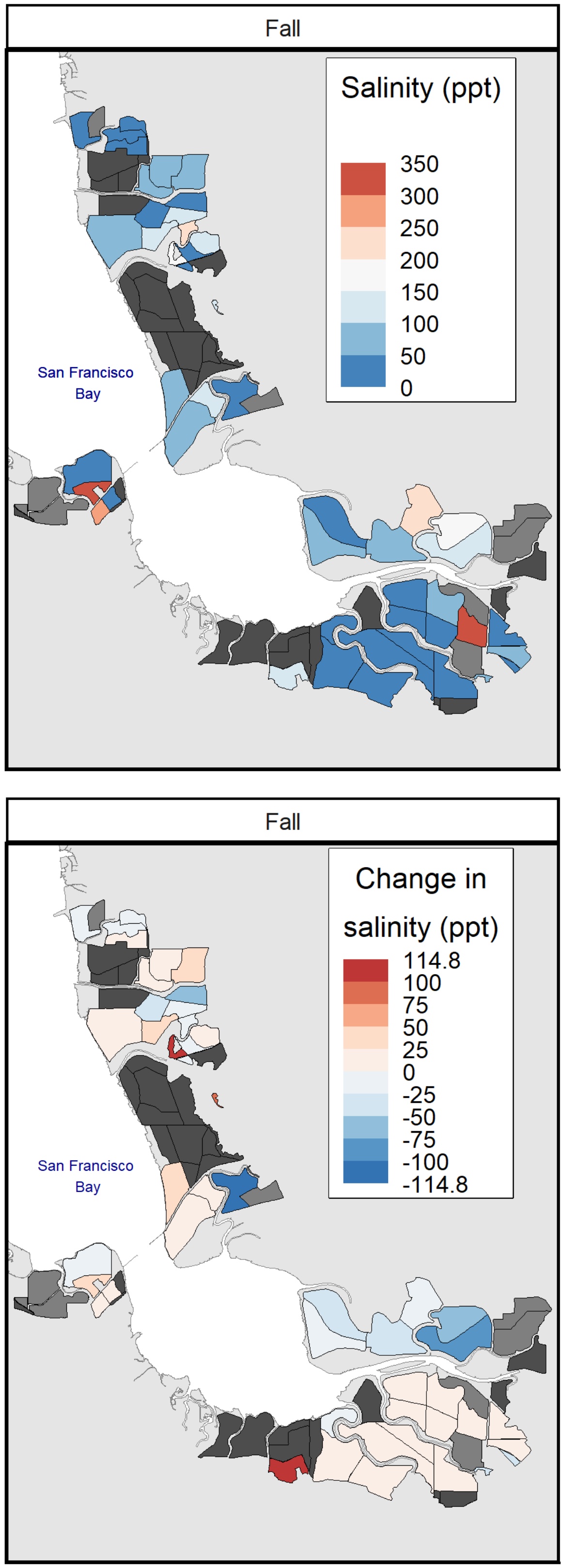


Figure 18. Seasonal salinity (ppt) during September 02 to October 15 2025 (top panels) and change in salinity (ppt) between 2025 and 2024 (bottom panels) in pond habitats of South San Francisco Bay, California. Dark grey ponds were not measured for this pond characteristic in one or both seasons.

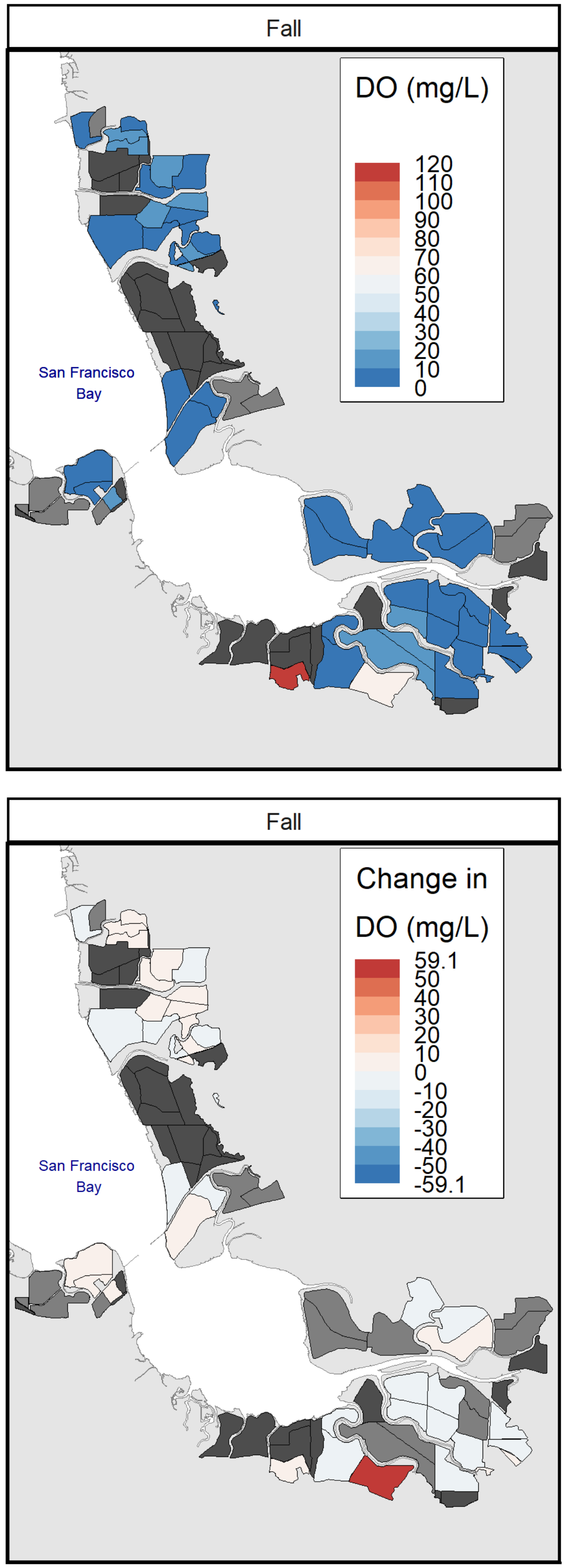


Figure 19. Seasonal dissolved oxygen (mg/L) during September 02 to October 15 2025 (top panels) and change in dissolved oxygen (mg/L) between 2025 and 2024 (bottom panels) in pond habitats of South San Francisco Bay, California. Dark grey ponds were not measured for this pond characteristic in one or both seasons.

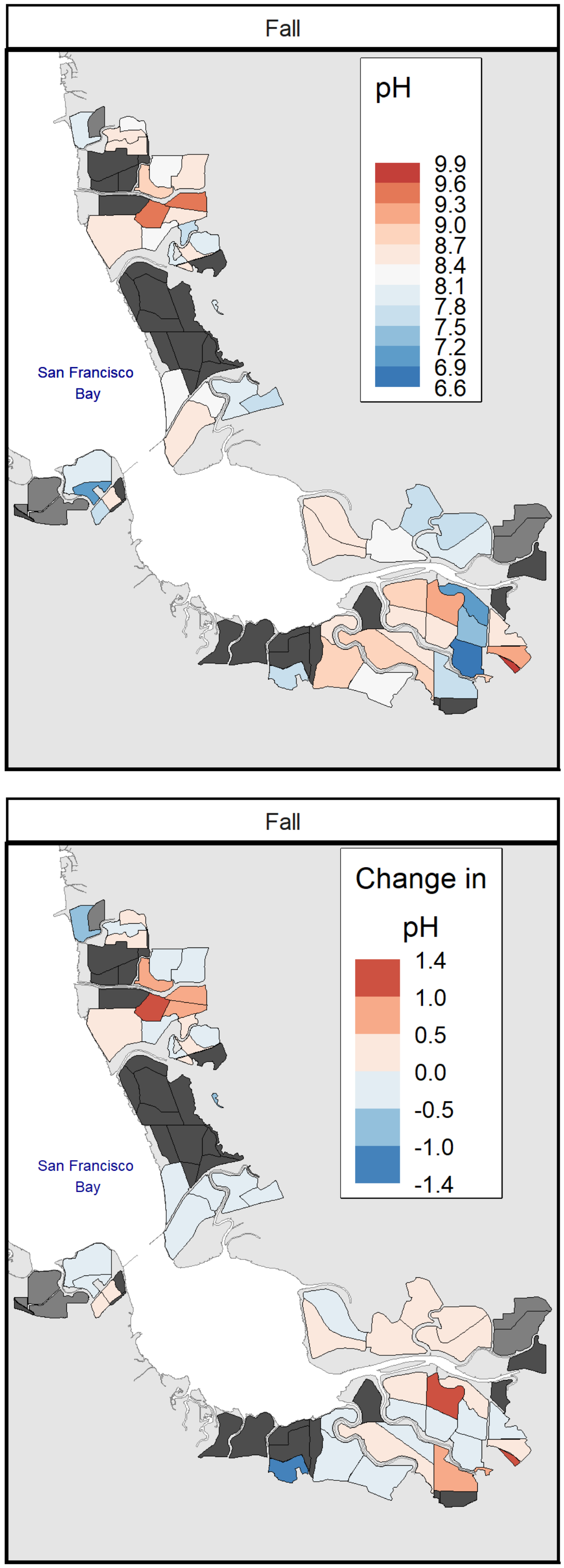
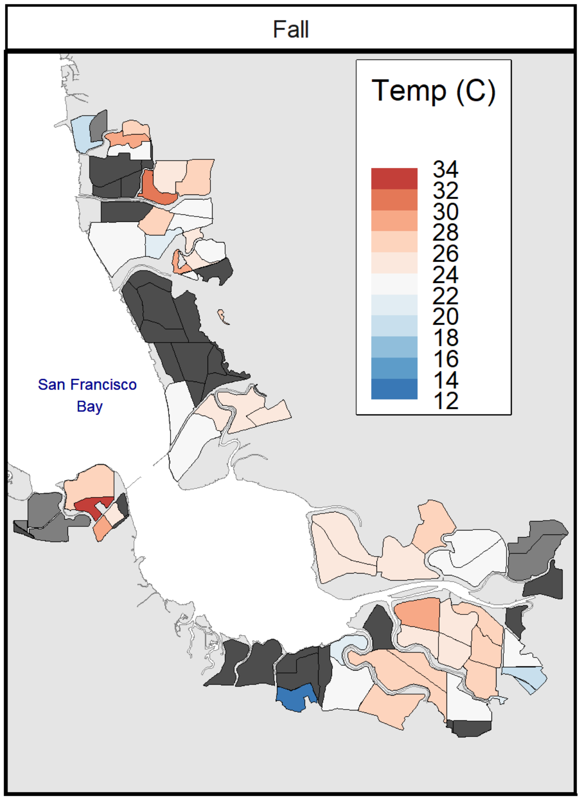


Figure 20. Seasonal pH during September 02 to October 15 2025 (top panels) and change in pH between 2025 and 2024 (bottom panels) in pond habitats of South San Francisco Bay, California. Dark grey ponds were not measured for this pond characteristic in one or both seasons.



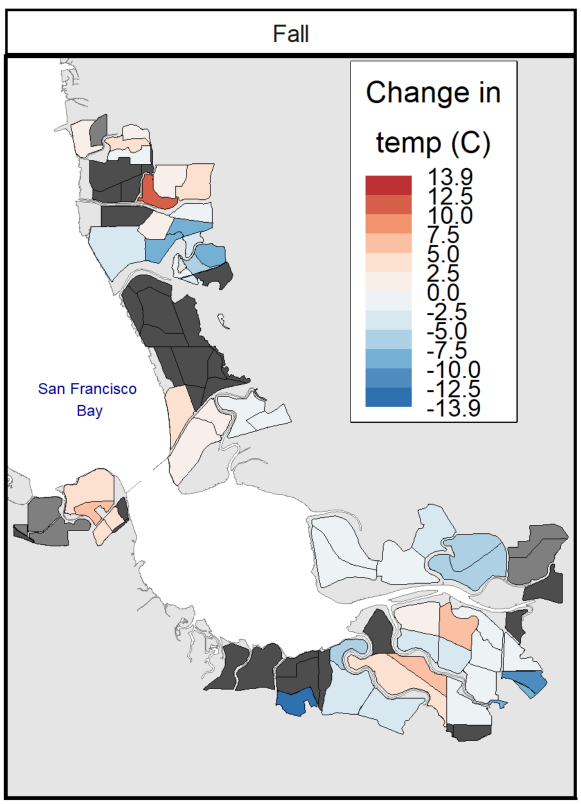


Figure 21. Seasonal temperature (C) during September 02 to October 15 2025 (top panels) and change in temperature (C) between 2025 and 2024 (bottom panels) in pond habitats of South San Francisco Bay, California. Dark grey ponds were not measured for this pond characteristic in one or both seasons.

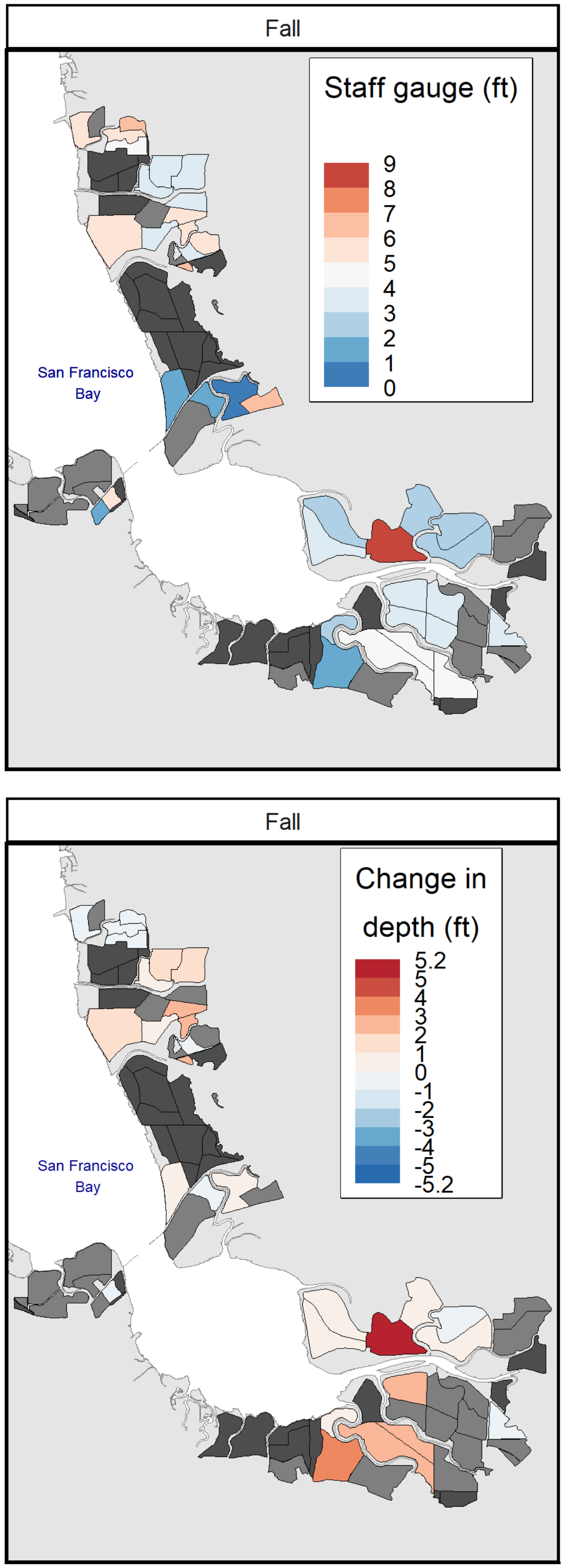


Figure 22. Seasonal staff gauge reading (ft) during September 02 to October 15 2025 (top panels) and change in staff gauge reading (ft) between 2025 and 2024 (bottom panels) in pond habitats of South San Francisco Bay, California. Dark grey ponds were not measured for this pond characteristic in one or both seasons.

Discussion

Trends

The total counts for phalaropes was nearly double that of the record low observed in 2024 (9,066 total sightings up from 4,835; +88%). However, this total is still markedly below the 13,000-16,700 total sightings observed 2021-2023. The modeled overall growth in range-wide phalarope peak abundances from 2024 to 2025 increased around 5–25% (Van Schmidt et al. 2026), indicating that while the partial recovery of San Francisco Bay abundances may have benefited from range-wide increases, the Bay Area's abundances were not simply tracking range-wide trends.

The largest increases in abundance were all outside of the SBSPRP (M2, M1, Crittenden Marsh, N2, and N1), with the exception of A9, which had 557 more sightings this year and was the third-most abundant site. This aligns with the predictions of the change model, which predicted that shifting hydrology would drive a reduction in abundance in the SBSPRP that would be outweighed by an increase in the other sites.

The 2024 report highlighted that we may be missing some phalaropes at some currently unsurveyed sites (Van Schmidt & Parsons 2025). 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. To address this concern, this year we nearly doubled the survey effort by adding 27 new sites (in addition to the 32 surveyed last year). This expansion was done at minimal additional cost by leveraging volunteers and ongoing plover surveys at most new sites. At these new sites, the only phalaropes sighted were 33 Red-necked Phalaropes, all of which were seen during the late August survey at just three sites: A8 (17 birds), A15 (7 birds), and E3C (9 birds). This suggests that the core set of 32 sites, if not perfect, is largely sufficient for capturing the key trends. Both this new data and the fact that 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 indicate that the finding of a long-term decline is robust. While it is always possible to 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) and the more frequent surveys in the modern protocol further 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.

Monitoring Program

Unique amongst waterbirds monitored in the South Bay, this two-week survey protocol is coordinated internationally by the International Phalarope Working Group (Carle et al. 2021). This international coordination is allowing us to model how changes in South Bay populations relate to changes in range-wide abundances (Van Schmidt et al. 2026) and habitat conditions (Miller et al. in prep). The phalarope monitoring data are critically important not only for assessing whether the SBSPRP has met phalarope targets set by the Adaptive Management Plan but also for informing this broader effort. In early 2026, the federal government began a formal process to review Wilson's Phalarope for listing under the Endangered Species Act. A Phalarope Core Workgroup of managers and Technical Workgroup of scientists are now being formed to reverse declines in this species. Locally, 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). Especially since we are able to widely use volunteer and plover surveyors to minimize expenses, the scientific and practical value of this key dataset is outsized in comparison to the monitoring cost.

The two-week period was determined to be the best balance between feasibility and likelihood of avoiding missing birds (Carle et al. 2021) and cannot be relaxed without sacrificing data quality and the ability to jointly model rangewide abundances and distributions. 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 South Bay, as seen in the massive sighting of Red-necked Phalaropes at M2 this year during a single visit. 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 5 complete years of data and one partial year (2020), which has enabled preliminary models to be developed. The statistical power of the Wilson's Phalarope model is markedly improving, now explaining more than 25% of the variance in counts. This highlights how each year of continued surveys is improving our ability to disentangle significant drivers.

We therefore recommended continuation of phalarope migration surveys at the expanded set of sites in 2026 to confirm whether the initial finding of few phalaropes at the new sites is robust, in order to increase confidence in this conclusion. In future years, the set of sites may be able to be winnowed down again.

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). The mid-June migratory period was added in 2022 because the peak count of Wilson's Phalaropes in 2021 was in early July, the previous first survey window, creating uncertainty in whether this was the true peak. This mid-June survey period has only had 68 phalarope sightings across all four years since then (0.2% of all sightings). This survey is also not conducted at any other staging sites. It therefore appears that this survey window is unnecessary and we recommend removing it from the protocol in order to reduce the cost of monitoring.

Habitat Associations in the Change Model

The increasing amount of data and improved model-averaging approach allowed us to uncover several new statistically significant predictors of phalarope trends. Changes in Red-necked Phalarope abundance were strongly positively related to increases in pH (which is inversely correlated with salinity; $R^2 = -0.56$ in the 2003–2018 waterbird survey dataset for phalaropes) and more weakly inversely related to changes in salinity (falling saltiness better) and water elevation (falling water levels better). Wilson's Phalarope abundance increases were strongly positively related to colder water. This could be a proxy for influxes of new, cold water into drying sites (i.e., as new batches of water are moved through the commercial salt pond complexes), matching the inverse relationship with water elevation in Red-necked Phalaropes (though neither species had both predictors in their final model set). Both species also showed non-significant inverse relationships with both maximum and minimum air temperatures. Without data on invertebrate communities within the ponds, the mechanisms underlying these relationships remain unclear.

Salinity was not a strong driver in the change model, despite having clear impacts on phalarope habitat selection. The most probable explanation for this is that the site's salinity categories in the past five years have remained fairly stable, unlike in the historical surveys (Van Schmidt & Parsons 2023), and therefore changes in salinity are not a key driver of population changes in the recent period.

The R^2 of the total phalaropes change model was less than half that of the Red-necked Phalarope model and not even a quarter that of the Wilson's Phalarope model, and results for this model showed little more than a weaker mish-mash of the stronger effects seen in each species-specific model. This indicates that guild-grouped modeling may be reducing statistical power if individual species are responding differently to habitat shifts. If true, this has significant implications for the ability to increase the statistical power of the general waterbird models (Van Schmidt & Parsons 2023) by analyzing species-specific trends.

There was evidence that rainier years drove decreases in abundance of both species (though cumulative three-year precipitation showed a divergent pattern for Red-necked Phalaropes). In drought years, saline lakes like Lake Abert and Great Salt Lake have dried and led to dramatic reductions there, and then experienced correspondingly large increases in abundance in wet years when the lakes expand again (Van Schmidt et al. 2026). Preliminary results from other research suggests greater monthly water extents at the six saline lake staging areas correlate to greater abundance from 2019–2025 (Miller et al., in prep). Annual precipitation in South Bay is significantly and strongly correlated with annual precipitation at Mono Lake, Great Salt Lake, and Lake Owens (Pearson $r \geq 0.85$, all $p < 0.05$), and moderately correlated with Lake Abert (Pearson $r = 0.66$; sum total precipitation September–August water year 2019–2024 analysis of ERA5-Land data; Muñoz Sabater, 2019). The rainfall effects seen in this study could support the hypothesis that the South Bay may act as a refugia for phalaropes during drought and be less important in rainy years. An alternative explanation could be that rainier years raise water elevations in ponds, which correlated with declining abundance of Red-necked Phalaropes. However, rainfall in the present year had the weakest relationship, with lag effects better-supported, favoring the conclusion that poor conditions at other sites in recent years may drive phalaropes to adjust their migration route in subsequent years.

Due to the small sample sizes of complete observations of both phalarope population sizes and water quality variables, results from this model should be interpreted very cautiously. We still do not yet have adequate sample sizes to conduct a full model (recommended 10 per covariate assessed), and only fit models with up to 3 (Wilson's) to 8 (Red-necked and total phalaropes) predictors per model. The model averaging approach allowed us to work around this constraint by reporting conditional coefficient estimates across the full set of alternative models, but this is still not a test of all possible predictor interactions. A model assessing these and the raw water quality parameters (i.e., "Salinity" instead of "change in salinity") and their interactions will likely require over a decade of surveying. 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.

Given the timeline of Restoration Project Phase 2, it is unlikely that management decisions can wait a decade. Because almost none of the newly surveyed ponds had phalaropes present, surveying additional ponds appears unlikely improve the rate of annual sample collection to hasten this process. Therefore, improving statistical power more quickly by cross-walking the historic waterbird counts (Van Schmidt & Parsons 2023) and contemporary migration surveys or by developing a data fusion model appears the best option. A Bayesian hierarchical model that could directly estimate the underlying abundances (the "process model") that underlie the raw counts (the "observation model"). The recently developed Bayesian model of rangewide surveys (Van Schmidt et al. 2026) could form a foundation for such a model.

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; Van Schmidt & Parsons 2023), which likely relates to moving ponds from extreme high hypersaline conditions (>150 ppt) to marine to moderately hypersaline conditions. The two years of data assessing salinity associations and foraging now shows that phalaropes are less abundant and forage less at highly hypersaline sites (>150 ppt). This statistically significant relationship offers a clear explanation for why the analysis of the historical phalarope data showed an increase in abundance at sites where salinity fell from 2002-2017 (Van Schmidt & Parsons 2023). Phalaropes show a marked preference for marine to moderately hypersaline environments (30-150 ppt), with an aversion to sites that are near-freshwater or brackish (except for the unusual Sunnyvale WPCP). E7, which had marine salinity, was the SBSPRP site with the highest numbers of attempts and foraging successes both by Wilson's and Red-necked Phalaropes; in 2024, E4 had been most abundant (Van Schmidt & Parsons 2025).

This aligns with the optimal salinity conditions of their hypothesized main food source—*Ephydra* and potentially *Artemia* spp. Both experimental and field studies at saline lakes have shown that the best salinity optimal for productivity of alkali flies (*Ephydra* spp.) is 25–100 ppt, with 50 ppt (the cutoff between our "marine" and "low hypersaline" bins) ideal for productivity of this insect for waterbird foraging (Herbst 2023). As salinity increases, the phytoplankton food chain base decreases, *Ephydra* growth slows and osmotic stress increases, size and pupal emergence success decreases; these changes lead to a corresponding decrease in both shorebird prey abundance and nutritional value (Herbst & Bradley 1989, Herbst & Blinn 1998, Herbst 2023).

Below 50 ppt—and especially below 30 ppt—biotic interactions may be more limiting of *Ephydra* spp. abundance as other less salt-tolerant species become more abundant (Herbst 1988). Fish are more abundant in lower-salinity ponds and may offer a predatory control on invertebrate biomass. Takekawa et al. (2006) showed that fish biomass declined rapidly across a gradient of North Bay ponds, approaching near-zero by 40–50 ppt, though Takekawa et al. (2004) found salt-tolerant species (most commonly longjaw mudsucker; *Gillichthys mirabilis*) present in ponds up to 62 ppt. Above 80 ppt, salinity should be lethal for all fish (Takekawa et al., 2006).

Preliminary data in the previous report suggested Wilson's Phalaropes preferred the less-salty marine salinity sites (Van Schmidt & Parsons 2025), contrary to expectations as this species is more specialized on saline lake habitats (Jehl 1988). However, the additional year of data now indicates that both Wilson's Phalaropes and Red-necked Phalaropes are selecting for the same marine to moderately hypersaline salinity range and Wilson's Phalaropes are more strictly avoiding near-fresh or brackish sites (Figs. 12, 17; except for Sunnyvale WPCP). Thus the additional monitoring data is reinforcing confidence in the biological underpinnings of our findings.

The Sunnyvale WPCP pond was the only near-fresh or brackish pond that was used with 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. Brackish ponds had visibly lower attempt and success rates, but this relationship was not significant because of the small number of foraging observations at these sites. An additional year of data may enable us to confirm that this is statistically significant.

Additional limiting factors for alkali fly production are the depth and substrate of ponds. At Mono Lake, abundance of alkali fly larvae and pupae decreases beyond 1 m depth (Herbst & Bradley 1993). Ponds in South Bay generally have <1 m depth, so this factor is unlikely to be the limiting factor for alkali fly production in most ponds. However, it may offer a management lever for some ponds, and is in line with the optimal depths for most guilds when foraging which are generally <1 m (exceptions being foraging Eared Grebes and gulls, which prefer ~1.5 m depth). Fly larvae and pupae are also generally more abundant on hard, rocky substrates compared to mud, sand, or gravel (Herbst 1990, Herbst & Bradley

1993). Substrate may therefore be a relevant constraint in the South Bay's soft-bottomed ponds, although the effect of substrate on alkali fly production has not been measured in these ponds.

Movement Ecology

There is evidence that suggests birds may switch from using one staging site to another during years when conditions are less favorable (Van Schmidt et al. 2026, Carle et al. 2021), 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 (Tarr & Carle 2023). Motus tags have been successfully deployed by researchers on phalaropes (Tarr & Carle 2023) and SFBBO has installed two of the four Motus towers in the South Bay, while other organizations have expanded the Motus network across North and South America. The resulting Motus tracks for Wilson's Phalaropes and Red-necked Phalaropes show variable migratory pathways, with some continuing on to Baja California while others continued inland via Mono Lake to central Mexico. One Wilson's Phalarope Motus-tagged in Tule Lake was detected by a tower in Napa-Sonoma Marsh in North Bay. If more tagged birds migrate through the Bay Area and are detected by one of the new Motus towers, it may give us insight into their stopover time. However, 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 stopover times here may be an unreliable plan.

Tagging phalaropes in San Francisco Bay would 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. This would also allow us to investigate the stopover time of phalaropes in the Bay Area during migration, which would make it possible to estimate the likelihood of counting the same individuals across multiple surveys and then crosswalk survey counts to true population size estimates. However, at this time we recommend prioritizing core monitoring and studies on invertebrate prey to better understand local habitat use dynamics. Initial success of trapping efforts at Mono Lake suggests that boat captures and spotlighting at night may be effective, but it is unclear whether these approaches can be translated to the shallow salt ponds of San Francisco Bay. 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 2025 counts are -81.6% 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. 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 (30–150 ppt), with the low hypersaline range (50–80 ppt) likely the safest bet given in the middle of this broader range. Mono Lake studies indicate that 50 ppt may be optimal for alkali flies, and low fish biomass were observed in North Bay ponds ranging 40–50 ppt (Herbst 2023, Takekawa et al. 2006). Given the alkali flies are more productive in shallow water with rocky substrates, maintaining any shallow and/or rocky ponds near the discharge limit (44 ppt) where possible may improve phalarope habitat without reducing habitat quality for most other waterbird guilds. However, at a landscape scale, this should be balanced with the needs of other guilds. De La Cruz et al. (2018) found piscivores, terns, waders, dabbling ducks, and diving ducks all had optimal salinities for foraging habitats ranging 1–17 ppt (brackish), while shorebirds showed no significant salinity preferences. In contrast, Takekawa et al. (2006)'s studies of the North Bay foraging biomass and diversity of waterbirds peaked in the mid-hypersaline range (80–150 ppt). Such tradeoffs appear most likely for piscivorous guilds (fisheaters, waders, and terns).

Given that alkali flies larvae and pupae are more productive on rocky substrates, a possible habitat enhancement action could be deploying rocky structures such as concrete blocks into shallow (<1 m) ponds in the appropriate salinity level, mimicking tufa substrates at Mono Lake. Artificial concrete tufa are already being deployed at Mono Lake as part of alkali fly research there (Di Paolo 2024). This may be able to enhance forage productivity without needing to alter difficult-to-manage water transfers or compromise habitats for other organisms. If deployed in a pilot-program before-and-after study, it would provide a relatively low-cost test of whether added substrate measurably increases prey availability.

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 now to increase the rigor of these models by implementing model averaging. Furthermore, foraging studies have highlighted a clear band of suitable salinities. These efforts have begun to identify optimal conditions for the phalaropes that remain within South Bay, such as ponds A9 and E7. 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 future years these models can be applied to predict alternative historical scenarios of how phalarope populations would have changed with and without the changes in ponds within the SBSPRP. This can help determine whether SBSPRP management actions caused the observed declines.

Special consideration should be given to ponds that have hosted large numbers of phalaropes in recent years (e.g., A9, E7, E4, and E6) when making management decisions such as seasonal adjustments to water level. All but E6 are likely to need to be breached to transition into tidal marsh eventually, emphasizing the need to identify ways to adjust conditions in other ponds to mimic these successful habitats. In the meantime, we suggest avoiding changes to these ponds and managing for conditions in each pond that were similar to conditions in the years when 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 many 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). 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.

Acknowledgements

Thank you to the California Wildlife Foundation, the U.S. Fish & Wildlife Service, the California State Coastal Conservancy, and the Santa Clara Valley Water District (Valley Water) for coordinating and funding this work. We would like to thank Rachel Tertes at the Refuge for help with coordinating site access. We would like to acknowledge the South Bay Salt Pond Restoration's Pond Management Working Group that has recommended and implemented many management changes over the years as they relate to specific pond systems. Thank you to Dave Halsing of the South Bay Salt Pond Restoration Project and John Krause with the CDFW Eden Landing Ecological Reserve. Our gratitude to Susan De La Cruz and her colleagues at the U.S. Geological Survey, who gathered historical salt pond data summarized in this report. Lastly, we could not have done this work without our dedicated field crew of SFBBO staff (Cole Jower, Daniel Solorzano-Jones, Emily Chan, Fen Conway, Jeremy Reinhard, Jesse

Wentworth, Kaili Hovind, and Maddy Schwarz) and volunteers. Thank you also to current and former SFBBO staff Katie LaBarbera, Gabbie Burns, Max Tarjan, Chris Overington, and Zabet Garber for their logistical support with surveys and their work on earlier drafts of this report.

Literature Cited

Accurso, L. M. 1992. Distribution and abundance of wintering waterfowl on San Francisco Bay 1988- 1990. Master's Thesis. Humboldt State University, Arcata, CA.

Anderson, W. 1970. A preliminary study of the relationship of salt ponds and wildlife – South San Francisco Bay. California Fish and Game 56: 240–252.

Athearn ND, Takekawa JY, Shinn JM. 2009. Avian response to early tidal salt marsh restoration at former commercial salt evaporation ponds in San Francisco Bay, California, USA. Nat Resour Environ Issues 15:Article 14.

Banner, K. M., and M. D. Higgs. 2017. Considerations for assessing model averaging of regression coefficients. Ecological Applications 27:78–93.

Burns, G., and N.D. Van Schmidt. 2023. Bridging the gap between disparate phalarope survey methodologies to evaluate population status. San Francisco Bay Bird Observatory. Report prepared for the South Bay Salt Pond Restoration Project.

Burnham, K. P. 2015. Multimodel inference: understanding AIC relative variable importance values. Colorado State University, Fort Collins, CO, USA. https://sites.warnercnr.colostate.edu/kenburnham/wp-content/uploads/sites/25/2016/08/VARIMP.pdf

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. 2nd edition. Springer, New York, New York, USA.

Carle, R.D., G. Burns, S.Hecocks, R. Larson, A. Lewis, A.E. McKellar, J. Neill, M. Prather, J. Reuland, M. Rubega, and M. Tarjan. 2021. Coordinated phalarope surveys at western North American staging sites, 2019-2020. Unpublished report of the International Phalarope Working Group.

Carle, R.D., K. Tarr., J. Neill., D. House, R. Larson., A.E. McKellar, J. Ng, M. Prather, J. Reuland, M. Rubega, H. Tobiason, & N.D. Van Schmidt. 2024. Coordinated phalarope surveys at western North American staging sites 2019–2023. Unpublished report of the International Phalarope Working Group.

Coates, P.S., B.G. Prochazka, M.S. O'Donnell, C.L. Aldridge, D.R. Edmunds, A.P. Monroe, A.P., M.A. Ricca, G.T. Wann, S.E. Hanser, L.A. Wiechman, and M.P. Chenaille. 2021. Range-wide greater sage-grouse hierarchical monitoring framework—Implications for defining population boundaries, trend estimation, and a targeted annual warning system: U.S. Geological Survey Open-File Report 2020–1154, 243 p., https://doi.org/10.3133/ofr20201154.

Colwell, M.A., and J.R. Jehl Jr. 2020. Wilson's Phalarope (*Phalaropus tricolor*), version 1.0. In: Poole AF, Gill FB, editors. *Birds of the World*. Cornell Lab of Ornithology, Ithaca, New York. https://doi.org/10.2173/bow.wilpha.01

De La Cruz, S. E. W., L. M. Smith, S. M. Moskal, C. Strong, J. Krause, Y. Wang, and J. Y. Takekawa. 2018. Trends and habitat associations of waterbirds using the South Bay Salt Pond Restoration Project, San Francisco Bay, California. Page 148. Report, Reston, VA.

Frank, M. G., and M.R. Conover. 2019. Threatened habitat at Great Salt Lake: importance of shallow-water and brackish habitats to Wilson's and Red-necked Phalaropes. Condor 121:duz005. https://doi.org/10.1093/condor/duz005

Frank, M. G., and M. R. Conover. 2021. Foraging behavior of Red-necked (Phalaropus lobatus) and Wilson's (P. tricolor) phalaropes on Great Salt Lake, Utah. The Wilson Journal of Ornithology 133(4): 538–551. doi: https://doi.org/10.1676/20-00069.

Grueber, C. E., S. Nakagawa, R. J. Laws, and I. G. Jamieson. 2011. Multimodel inference in ecology and evolution: challenges and solutions. Journal of Evolutionary Biology 24:699–711.

Hartman, C.A., J.T. Ackerman, C. Schacter, M.P. Herzog, M. Tarjan, Y. Wang, C. Strong, R. Tertes, and N. Warnock. 2021. Breeding waterbird populations have declined in San Francisco Bay: an assessment over two decades. San Franc Estuary Watershed Sci. 19(3):Article 4. https://doi.org/10.15447/sfews.2021v19iss3art4

Herbst, D. B. 1988. Comparative population ecology of Ephydra hians Say (Diptera: Ephydridae) at Mono Lake (California) and Abert Lake (Oregon). Hydrobiologia, 158(1), 145-166.

Herbst, D. B. 1990. Distribution and abundance of the alkali fly (Ephydra hians) Say at Mono Lake, California (USA) in relation to physical habitat. Hydrobiologia, 197(1), 193-205.

Herbst, D. B. 2023. Developmental and reproductive costs of osmoregulation to an aquatic insect that is a key food resource to shorebirds at salt lakes threatened by rising salinity and desiccation. Frontiers in Ecology and Evolution, 11, 1136966.

Herbst, D. B., and D. W. Blinn, D. W. 1998. Experimental mesocosm studies of salinity effects on the benthic algal community of a saline lake. Journal of Phycology, 34(5), 772-778.

Herbst, D. B., and T. J. Bradley. 1989. Salinity And Nutrient Limitations On Growth Of Benthic Algae From Two Alkaline Salt Lakes Of The Western Great Basin (Usa) 1. Journal of Phycology, 25(4), 673-678.

Herbst, D. B., and T. J. Bradley. 1993. A population model for the alkali fly at Mono Lake: depth distribution and changing habitat availability. Hydrobiologia, 267(1), 191-201.

Hunnewell R.W., A.W. Diamond, and S.C. Brown. 2016. Estimating the migratory stopover abundance of phalaropes in the outer Bay of Fundy, Canada. Avian Conservation and Ecology 11(2):11. https://doi.org/10.5751/ACE-00926-110211

Jehl, J. R. 1988. Biology of the Eared Grebe and Wilson's Phalarope in the nonbreeding season: a study of adaptations to saline lakes. Studies in Avian Biology 12.

LaBarbera, K., Van Schmidt, N. D., and G. Burns. 2023. Bridging the gap between disparate phalarope survey methodologies to evaluate population status. San Francisco Bay Bird Observatory. Report prepared for the South Bay Salt Pond Restoration Project.

Miller, S. M., Reudink, M. W., Carle, R. D., Hoque, E., Donnelly, P., Rubega, M., House, D., Larson, R., Neill, J., Prather, M., Reuland, J., Van Schmidt, N., Tarr, K., and McKellar, A. E. In prep. Stable saline habitats support migrating phalaropes at North American staging sites.

Di Paolo, R. 2024. Alkali fly researcher resumes at Mono Lake. Mono Lake Newsletter (Summer 2024 (Volume XLVI, No. 1). Mono Lake Committee, Lee Vining, CA.

Murphy, A., C. Strong, D. Le Fer, and S. Hudson. 2007. Interim Cargill salt pond report. Unpublished report. San Francisco Bay Bird Observatory, Milpitas, CA.

Muñoz Sabater, J. 2019. ERA5-Land hourly data from 1950 to present. Copernicus Climate Change Service (C3S) Climate Data Store (CDS). https://doi.org/10.24381/cds.e2161bac

Nakagawa, S., and R. P. Freckleton. 2011. Model averaging, missing data and multiple imputation: a case study for behavioural ecology. Behavioral Ecology and Sociobiology 65:103–116.

R Development Core Team. 2018. R: A language and environment for statistical computing. version 3.5.3. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from http://www.r-project.org

Richards, S. A. 2005. Testing ecological theory using the information-theoretic approach: examples and cautionary results. Ecology 86:2805–2814.

Richards, S. A. 2008. Dealing with overdispersed count data in applied ecology. Journal of Applied Ecology 45:218–227.

Rubega, M. A., and B. S. Obst. 1993. Surface-tension feeding in phalaropes: discovery of a novel feeding mechanism. The Auk, 110(2):169-178.

Schacter, C.R., C.A. Hartman, M.P. Herzog, S.H. Peterson, L.M. Tarjan, Y. Wang, C. Strong, R. Tertes, N. Warnock, and J.T. Ackerman. 2023. Habitat use by breeding waterbirds in relation to tidal marsh restoration in the San Francisco Bay Estuary. San Franc Estuary Watershed Sci. 21(2):Article 2. https://doi.org/10.15447/sfews.2023v21iss2art2

Schwarz, M., LaBarbera K, Scullen J. 2024. Western Snowy Plover and California Least Tern monitoring in the San Francisco Bay annual report 2023. Report prepared for the South Bay Salt Pond Restoration Project.

South Bay Salt Pond Restoration Project. 2007. Final Environmental Impact Statement/Report. Appendix D: Adaptive Management Plan. http://www.southbayrestoration.org/pdf_files/SBSP_EIR_Final/Appendix%20D%20Final%20AMP.pdf

Takekawa, J. Y., C.T. Lu, and R.T. Pratt. 2001. Bird communities in salt evaporation ponds and baylands of the northern San Francisco Bay estuary. Hydrobiologia 466: 317–328.

Takekawa, J.Y. , D.H. Schoellhamer, A.K. Miles, G.G. Shellenbarger, N.D. Athearn, S.E. Spring, M.K. Saiki, and C.A. Jannusch. 2004. Initial biophysical changes after breaching a salt pond levee: final report on the Napa-Sonoma Wildlife Area Pond 3 breach. Unpublished Progress Report, U.S. Geological Survey, Vallejo, California, USA.

Takekawa, J.Y., J.T. Ackerman, L.A. Brand, T.R. Graham, C.A. Eagles-Smith, M.P. Herzog, B.R. Topping, G.G. Shellenbarger, J.S. Kuwabara, E. Mruz, S.L and Piotter. 2015. Unintended consequences of management actions in salt pond restoration: cascading effects in trophic interactions. PLoS One, 10(6), p.e0119345.

Tarjan, L.M. 2019a. Trends in waterbird counts in south San Francisco Bay. Report prepared for the South Bay Salt Pond Restoration Project Management Team.

Tarjan, L.M. 2019b. Migration timing and spatial distribution of phalaropes in San Francisco Bay. Report prepared for the South Bay Salt Pond Restoration Project Management Team. Revised April 2020.

Tarjan, L.M. and G. Burns. 2020. South Bay Salt Pond waterbird surveys, September 2019-February 2021. Report prepared for the South Bay Salt Pond Restoration Project Management Team.

Tarr K. & R. Carle. Phalaropes and Saline Lakes Program 2023 Report. Unpublished report by Oikonos Ecosystem Knowledge, Lee Vining, California, USA.

Van Schmidt, N. D., and A. Parsons. 2023. Effects of pond management, tidal restoration, and annual weather on waterbird population change in salt ponds of south San Francisco Bay. Page 70. Report prepared for the South Bay Salt Pond Restoration Project, San Francisco Bay Bird Observatory, Milpitas, CA.

Van Schmidt, N. D., and A. Parsons. 2024. Phalarope Migration Surveys: June–September 2023. Report prepared for the South Bay Salt Pond Restoration Project.

Van Schmidt, N. D., and A. Parsons. 2025. South Bay Salt Pond Waterbird Surveys: September 2023 - May 2024. Report prepared for the South Bay Salt Pond Restoration Project.

Van Schmidt, N. D., Carle, R., and E. Hamel. 2026. A novel Bayesian method for joint abundance estimation of multiple species of phalaropes with incomplete species identification. Report prepared for Tracy Aviary and the South Bay Salt Pond Restoration Project.

Ver Planck, W. E. 1958. Salt in California. California Division of Mines Bulletin, No. 175.

Warnock, N., G. W. Page, T. D. Ruhlen, N. Nur, J. Y. Takekawa, and J. T. Hanson. 2002. Management and conservation of San Francisco Bay salt ponds: effects of pond salinity, area, tide, and season on Pacific Flyway waterbirds. Waterbirds 25: 79–92.

Wickham, H. 2016. ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag New York.

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 2021-2025. 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	Variable 1	Variable 2
PHAL	LandfillDist_km	OpenHunting_pct
PHAL	PPT1	PPT3
PHAL	PPT1	TMN
RNPH	LandfillDist_km	OpenHunting_pct
RNPH	PPT1	PPT3
RNPH	PPT1	TMN
WIPH	LandfillDist_km	OpenHunting_pct
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 2021-2025. Model selection was carried out based on corrected Akaike Information Criterion (AICc). only up to the top 50 models are shown.

Formula	K	dAICc	AICc	-2LL	w
Abun_PC ~ 1	2	0.00	465.86	-230.86	0.12
Abun_PC ~ Area_km2	3	1.58	467.44	-230.57	0.06
Abun_PC ~ BayDist_km	3	2.05	467.90	-230.80	0.04
Abun_PC ~ Islands_num	3	2.08	467.94	-230.82	0.04
Abun_PC ~ OpenHunting_pct	3	2.13	467.99	-230.85	0.04
Abun_PC ~ OpenPublic_pct	3	2.13	467.99	-230.85	0.04
Abun_PC ~ LandfillDist_km	3	2.15	468.00	-230.85	0.04
Abun_PC ~ CreekDist_km	3	2.15	468.01	-230.86	0.04
Abun_PC ~ Area_km2 + BayDist_km	4	3.23	469.09	-230.29	0.02
Abun_PC ~ Area_km2 + OpenHunting_pct	4	3.67	469.53	-230.51	0.02
Abun_PC ~ Area_km2 + LandfillDist_km	4	3.74	469.60	-230.55	0.02
Abun_PC ~ Area_km2 + Islands_num	4	3.74	469.60	-230.55	0.02
Abun_PC ~ Area_km2 + CreekDist_km	4	3.76	469.62	-230.56	0.02
Abun_PC ~ Area_km2 + OpenPublic_pct	4	3.78	469.64	-230.57	0.02
Abun_PC ~ BayDist_km + Islands_num	4	4.14	470.00	-230.75	0.02
Abun_PC ~ BayDist_km + OpenPublic_pct	4	4.15	470.01	-230.76	0.02
Abun_PC ~ BayDist_km + OpenHunting_pct	4	4.22	470.08	-230.79	0.01
Abun_PC ~ BayDist_km + CreekDist_km	4	4.23	470.09	-230.80	0.01
Abun_PC ~ Islands_num + OpenHunting_pct	4	4.24	470.10	-230.80	0.01
Abun_PC ~ BayDist_km + LandfillDist_km	4	4.24	470.10	-230.80	0.01
Abun_PC ~ Islands_num + OpenPublic_pct	4	4.28	470.14	-230.82	0.01
Abun_PC ~ CreekDist_km + Islands_num	4	4.28	470.14	-230.82	0.01
Abun_PC ~ Islands_num + LandfillDist_km	4	4.28	470.14	-230.82	0.01
Abun_PC ~ LandfillDist_km + OpenPublic_pct	4	4.32	470.18	-230.84	0.01
Abun_PC ~ OpenHunting_pct + OpenPublic_pct	4	4.33	470.19	-230.84	0.01
Abun_PC ~ CreekDist_km + OpenHunting_pct	4	4.33	470.19	-230.85	0.01
Abun_PC ~ CreekDist_km + OpenPublic_pct	4	4.34	470.19	-230.85	0.01
Abun_PC ~ CreekDist_km + LandfillDist_km	4	4.35	470.21	-230.85	0.01
Abun_PC ~ Area_km2 + BayDist_km + OpenHunting_pct	5	5.25	471.11	-230.17	0.01
Abun_PC ~ Area_km2 + BayDist_km + LandfillDist_km	5	5.35	471.21	-230.22	0.01

Formula	K	dAICc	AICc	-2LL	w
Abun_PC ~ Area_km2 + BayDist_km + Islands_num	5	5.41	471.26	-230.25	0.01
Abun_PC ~ Area_km2 + BayDist_km + OpenPublic_pct	5	5.42	471.28	-230.26	0.01
Abun_PC ~ Area_km2 + BayDist_km + CreekDist_km	5	5.47	471.33	-230.28	0.01
Abun_PC ~ Area_km2 + OpenHunting_pct + OpenPublic_pct	5	5.76	471.62	-230.43	0.01
Abun_PC ~ Area_km2 + Islands_num + OpenHunting_pct	5	5.89	471.75	-230.50	0.01
Abun_PC ~ Area_km2 + CreekDist_km + OpenHunting_pct	5	5.92	471.78	-230.51	0.01
Abun_PC ~ Area_km2 + LandfillDist_km + OpenPublic_pct	5	5.95	471.81	-230.52	0.01
Abun_PC ~ Area_km2 + CreekDist_km + Islands_num	5	5.95	471.81	-230.52	0.01
Abun_PC ~ Area_km2 + Islands_num + LandfillDist_km	5	5.96	471.81	-230.53	0.01
Abun_PC ~ Area_km2 + CreekDist_km + LandfillDist_km	5	5.98	471.84	-230.54	0.01
Abun_PC ~ Area_km2 + Islands_num + OpenPublic_pct	5	6.00	471.86	-230.55	0.01
Abun_PC ~ Area_km2 + CreekDist_km + OpenPublic_pct	5	6.01	471.87	-230.56	0.01
Abun_PC ~ BayDist_km + CreekDist_km + OpenPublic_pct	5	6.31	472.17	-230.70	0.01
Abun_PC ~ BayDist_km + Islands_num + OpenHunting_pct	5	6.34	472.20	-230.72	0.01
Abun_PC ~ BayDist_km + Islands_num + OpenPublic_pct	5	6.34	472.20	-230.72	0.01
Abun_PC ~ BayDist_km + CreekDist_km + Islands_num	5	6.35	472.21	-230.72	0.01
Abun_PC ~ BayDist_km + LandfillDist_km + OpenPublic_pct	5	6.35	472.21	-230.72	0.01
Abun_PC ~ BayDist_km + Islands_num + LandfillDist_km	5	6.40	472.25	-230.75	0.01
Abun_PC ~ BayDist_km + OpenHunting_pct + OpenPublic_pct	5	6.41	472.27	-230.75	0.00
Abun_PC ~ BayDist_km + CreekDist_km + OpenHunting_pct	5	6.45	472.31	-230.77	0.00

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 2021-2025. Model selection was carried out based on corrected Akaike Information Criterion (AICc). only up to the top 50 models are shown.

Formula	**K**	**dAICc**	**AICc**	**-2LL**	**w**
Abun_PC ~ 1	2	0.00	452.15	-224.00	0.12
Abun_PC ~ OpenPublic_pct	3	1.85	453.99	-223.84	0.05
Abun_PC ~ BayDist_km	3	1.89	454.04	-223.87	0.05
Abun_PC ~ Area_km2	3	1.94	454.09	-223.89	0.05
Abun_PC ~ LandfillDist_km	3	2.03	454.17	-223.93	0.05
Abun_PC ~ OpenHunting_pct	3	2.06	454.21	-223.95	0.04
Abun_PC ~ CreekDist_km	3	2.15	454.30	-223.99	0.04
Abun_PC ~ Islands_num	3	2.15	454.30	-224.00	0.04
Abun_PC ~ Area_km2 + BayDist_km	4	3.50	455.65	-223.56	0.02
Abun_PC ~ Area_km2 + OpenPublic_pct	4	3.71	455.86	-223.67	0.02
Abun_PC ~ BayDist_km + OpenPublic_pct	4	3.98	456.12	-223.80	0.02
Abun_PC ~ BayDist_km + LandfillDist_km	4	3.98	456.12	-223.80	0.02
Abun_PC ~ CreekDist_km + OpenPublic_pct	4	4.01	456.15	-223.82	0.02
Abun_PC ~ Islands_num + OpenPublic_pct	4	4.01	456.16	-223.82	0.02
Abun_PC ~ BayDist_km + OpenHunting_pct	4	4.03	456.17	-223.83	0.02
Abun_PC ~ LandfillDist_km + OpenPublic_pct	4	4.04	456.19	-223.83	0.02
Abun_PC ~ OpenHunting_pct + OpenPublic_pct	4	4.06	456.20	-223.84	0.02
Abun_PC ~ Area_km2 + Islands_num	4	4.07	456.22	-223.85	0.02
Abun_PC ~ BayDist_km + CreekDist_km	4	4.08	456.23	-223.85	0.02
Abun_PC ~ BayDist_km + Islands_num	4	4.09	456.24	-223.86	0.02
Abun_PC ~ Area_km2 + LandfillDist_km	4	4.11	456.26	-223.87	0.02
Abun_PC ~ Area_km2 + OpenHunting_pct	4	4.12	456.27	-223.87	0.02
Abun_PC ~ Area_km2 + CreekDist_km	4	4.14	456.28	-223.88	0.02
Abun_PC ~ CreekDist_km + LandfillDist_km	4	4.21	456.36	-223.92	0.02
Abun_PC ~ Islands_num + LandfillDist_km	4	4.23	456.38	-223.93	0.01
Abun_PC ~ CreekDist_km + OpenHunting_pct	4	4.25	456.39	-223.94	0.01
Abun_PC ~ Islands_num + OpenHunting_pct	4	4.27	456.42	-223.95	0.01
Abun_PC ~ CreekDist_km + Islands_num	4	4.36	456.51	-223.99	0.01
Abun_PC ~ Area_km2 + BayDist_km + Islands_num	5	5.62	457.76	-223.49	0.01
Abun_PC ~ Area_km2 + BayDist_km + OpenPublic_pct	5	5.65	457.80	-223.50	0.01

Formula	K	dAICc	AICc	-2LL	w
Abun_PC ~ Area_km2 + BayDist_km + CreekDist_km	5	5.73	457.88	-223.54	0.01
Abun_PC ~ Area_km2 + BayDist_km + LandfillDist_km	5	5.77	457.91	-223.56	0.01
Abun_PC ~ Area_km2 + BayDist_km + OpenHunting_pct	5	5.77	457.91	-223.56	0.01
Abun_PC ~ Area_km2 + CreekDist_km + OpenPublic_pct	5	5.86	458.00	-223.61	0.01
Abun_PC ~ Area_km2 + OpenHunting_pct + OpenPublic_pct	5	5.86	458.01	-223.61	0.01
Abun_PC ~ Area_km2 + LandfillDist_km + OpenPublic_pct	5	5.93	458.08	-223.65	0.01
Abun_PC ~ Area_km2 + Islands_num + OpenPublic_pct	5	5.97	458.11	-223.66	0.01
Abun_PC ~ BayDist_km + Islands_num + OpenPublic_pct	5	6.20	458.34	-223.78	0.01
Abun_PC ~ BayDist_km + LandfillDist_km + OpenPublic_pct	5	6.21	458.35	-223.78	0.01
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct	5	6.24	458.39	-223.80	0.01
Abun_PC ~ BayDist_km + OpenHunting_pct + OpenPublic_pct	5	6.24	458.39	-223.80	0.01
Abun_PC ~ BayDist_km + CreekDist_km + LandfillDist_km	5	6.24	458.39	-223.80	0.01
Abun_PC ~ BayDist_km + CreekDist_km + OpenPublic_pct	5	6.24	458.39	-223.80	0.01
Abun_PC ~ BayDist_km + Islands_num + LandfillDist_km	5	6.25	458.39	-223.80	0.01
Abun_PC ~ CreekDist_km + LandfillDist_km + OpenPublic_pct	5	6.26	458.40	-223.81	0.01
Abun_PC ~ Islands_num + OpenHunting_pct + OpenPublic_pct	5	6.26	458.41	-223.81	0.01
Abun_PC ~ Area_km2 + CreekDist_km + Islands_num	5	6.28	458.42	-223.82	0.01
Abun_PC ~ CreekDist_km + OpenHunting_pct + OpenPublic_pct	5	6.28	458.42	-223.82	0.01
Abun_PC ~ Islands_num + LandfillDist_km + OpenPublic_pct	5	6.28	458.43	-223.82	0.01
Abun_PC ~ BayDist_km + Islands_num + OpenHunting_pct	5	6.29	458.44	-223.83	0.01

Table A1.4. Model selection table for base model of pond characteristics related to change in abundance of Wilson's Phalaropes during summer in current and former salt ponds in South San Francisco Bay, California, summer 2021-2025. Base model selection was not carried out for this species because of the limited sample size available for fitting parameters.

Formula	K	dAICc	AICc	-2LL	w
Abun_PC ~ 1	2	0	167.32	-81.47	1

Table A1.5. 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 2021-2025. Model selection was carried out based on corrected Akaike Information Criterion (AICc). Only up to the top 50 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 ~ PPT1	3	0.00	463.95	-228.83	0.28
Abun_PC ~ pH_AC + PPT0 + TMN	5	0.79	464.74	-226.99	0.19
Abun_PC ~ TMN	3	1.12	465.08	-229.39	0.16
Abun_PC ~ PPT3	3	1.25	465.21	-229.45	0.15
Abun_PC ~ Temp_C	3	1.52	465.47	-229.59	0.13
Abun_PC ~ 1	2	1.91	465.86	-230.86	0.11

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 2021-2025. Model selection was carried out based on corrected Akaike Information Criterion (AICc). Only up to the top 50 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 + PPT1	4	0.00	444.70	-218.09	0.30
Abun_PC ~ pH_AC + PPT3 + TMX	5	1.63	446.34	-217.77	0.13
Abun_PC ~ pH_AC + PPT3 + TMN	5	2.03	446.73	-217.97	0.11
Abun_PC ~ PPT1 + Salinity_AC	4	3.04	447.74	-219.61	0.07
Abun_PC ~ pH_AC + PPT0 + TMN	5	3.10	447.81	-218.51	0.06
Abun_PC ~ pH_AC + Salinity_AC + TMN	5	3.63	448.33	-218.77	0.05
Abun_PC ~ pH_AC + TMN	4	3.92	448.62	-220.05	0.04
Abun_PC ~ PPT1	3	4.01	448.71	-221.20	0.04
Abun_PC ~ pH_AC + PPT3	4	4.18	448.89	-220.18	0.04
Abun_PC ~ Salinity_AC + TMN	4	4.32	449.02	-220.25	0.04
Abun_PC ~ PPT3 + Salinity_AC + WaterElev_m_AC	5	4.70	449.40	-219.31	0.03
Abun_PC ~ PPT3 + Salinity_AC	4	4.86	449.56	-220.52	0.03
Abun_PC ~ PPT3	3	5.46	450.17	-221.93	0.02
Abun_PC ~ Salinity_AC	3	5.58	450.28	-221.99	0.02
Abun_PC ~ TMN	3	6.34	451.04	-222.37	0.01
Abun_PC ~ 1	2	7.44	452.15	-224.00	0.01

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 2021-2025. Model selection was carried out based on corrected Akaike Information Criterion (AICc). Only up to the top 50 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 ~ PPT3 + Temp_C	4	0.00	161.00	-75.83	0.54
Abun_PC ~ Temp_C	3	0.70	161.70	-77.46	0.38
Abun_PC ~ PPT3 + Temp_C_AC + TMN	5	6.02	167.02	-77.47	0.03
Abun_PC ~ Temp_C_AC	3	6.06	167.06	-80.14	0.03
Abun_PC ~ 1	2	6.33	167.32	-81.47	0.02

Appendix 2

This appendix presents a table of the mean water quality measures of 2025. Measures were taken between September 2 and October 15.

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, 2025.

Location	Pond	Salinity (ppt)	DO (mg/L)	pH	Temp (C)	Staff gauge (ft)	ΔSalinity (ppt)	ΔDO (mg/L)	ΔpH	ΔTemp (C)	ΔStaff gauge (ft)
Baylands	Alviso Marina	96.50	7.16	8.71	21.32		-48.50	0.91	0.59	-8.41	
	Coyote Hills Regional Park	108.00	7.95	7.96	26.10		84.72	-7.11	-0.87	-2.32	
	Crittenden Marsh	138.00	112.20	7.74	12.45		113.70	102.34	-1.39	-13.81	
	New Chicago Marsh	51.63	7.36	9.26	19.89		17.33	-5.47	0.31	-11.25	
	Northeast of N1										
	Spreckles Marsh	45.20	9.63	9.69	19.51		-49.20	1.08	1.10	-9.92	
	Sunnyvale WPCP	1.94	65.65	8.19	27.97		1.34	59.01	-0.14	-3.33	
Don Edwards	A10	42.21	10.33	8.55	24.30	3.52	9.90	-2.16	-0.23	-4.11	
	A11	48.69	9.60	8.49	24.19	3.56	16.86	-1.73	-0.16	-3.29	
	A12		3.91	6.83	27.66			-0.78	-0.35	-2.16	
	A13	312.00	4.88	7.35	27.03		24.00		-0.23	-0.48	
	A14	52.52	6.22	9.08	25.40	3.70	20.33	-1.34	1.19	6.58	
	A15		2.66	7.12	26.95				0.14	-2.09	
	A16	20.52	5.61	8.67	23.39	3.30	2.29	-3.16	-0.25	-0.01	-0.30
	A22										
	A23										
	A3N	46.74	6.18	8.55	21.33	2.10	-0.81	-4.71	-0.07	-6.64	0.30
	A3W	27.25	7.50	8.77	22.97	2.00	1.69	-4.34	-0.01	-3.57	3.20
	A5	22.56	15.36	8.80	26.31	4.76	1.92		0.28	3.83	2.56
	A7	21.77	12.38	8.57	27.76	4.80	2.63		-0.02	5.08	2.50
	A8	17.75	5.38	7.77	23.20	4.88	2.79	-2.59	0.71	-0.90	
	A9	49.35	4.29	8.76	29.30	3.56	8.17	-1.18	0.22	0.01	2.16
	R1	32.19	8.72	8.07	27.75		-6.52	1.64	-0.34	4.11	
	R2	334.00	3.74	6.99	32.13		27.50	0.27	0.00	5.35	
	R3										
	R4										
	R5										

Location	Pond	Salinity (ppt)	DO (mg/L)	pH	Temp (C)	Staff gauge (ft)	ΔSalinity (ppt)	ΔDO (mg/L)	ΔpH	ΔTemp (C)	ΔStaff gauge (ft)
	RSF2U2	34.12	14.58	8.49	25.77	5.66	1.89	3.31	0.19	3.13	-0.24
	RSF2U3	264.50		7.61	28.46	2.00	10.00		0.20	4.14	
Eden	CP3C	48.61	7.90	8.60	23.60	5.66	-1.18	-3.63	-0.08	-7.38	1.96
Landing	E10	40.71	3.51	7.85	19.08	5.40	-0.04	-5.92	-0.67	0.25	-0.50
	E11										
	E12	39.58	8.02	8.23	27.06	6.26	-2.70	1.22	0.35	2.01	-0.44
	E13	34.98	11.55	8.52	28.67	5.68	-0.69	2.04	-0.12	3.04	-0.02
	E14	37.17	15.51	8.53	22.42	4.68	3.50	4.03	0.32	-2.30	-0.32
	E1C	190.00	4.40	7.92	28.38		105.36	-1.19	-0.21	-0.76	
	E2	73.30	3.28	8.70	22.68	5.08	21.17	-4.22	0.22	-3.46	1.78
	E2C	32.31	10.95	8.48	23.37	6.29	-0.88	2.04	0.18	-4.50	2.59
	E4	117.65	2.05	8.27	21.14	3.80	46.99	-9.67	-0.48	-9.92	0.70
	E4C	145.00	5.25	7.98	23.77	5.82	23.00	-2.15	-0.19	-8.04	
	E5	133.13	8.66	8.43	22.40	5.68	-23.54	6.94	0.79	-8.87	2.18
	E5C	36.46	11.46	8.69	24.39	3.70	-4.78	2.59	0.29	-5.74	-0.10
	E6	45.92	14.50	9.39	23.57	3.60	-51.15	8.20	0.51	-1.76	
	E6A	71.92	3.28	8.53	27.59	3.80	33.36	-7.04	-0.09	3.17	1.70
	E6B	50.52	11.06	8.21	24.62	3.62	7.07	0.98	-0.45	0.54	1.72
	E6C	228.50	4.73	7.76	24.91	5.58	-20.00	2.41	0.22	-3.60	2.13
	E7	47.44	19.07	9.50	26.34		-31.06	9.79	1.01	1.56	
	E8	56.80	8.46	8.73	30.86	3.88	21.68	0.54	0.81	11.75	0.98
Salt	M1	38.77	9.58	8.59	24.56	2.50	-29.20		0.00	-2.17	0.80
Ponds	M2	50.05	8.57	8.69	25.81	3.40	-18.92		0.06	-1.21	0.60
	M3	98.45	5.35	8.38	25.88	8.06	-33.80		0.42	-2.02	5.16
	M4	116.77	3.48	8.09	22.81	2.60	-81.23	0.79	0.40	-6.46	0.80
	M5	192.67	2.20	7.77	23.69	2.60	-55.33	-0.22	0.21	-5.83	-0.20
	M6	238.67	2.04	7.58	26.39	2.60	-24.66	-0.02	0.11	-2.59	0.80
	N1	0.56		8.00	25.65	0.60	-114.77		-0.37	-0.44	0.30
	N2	115.25	5.16	8.39	24.37	1.80	21.55	-0.70	0.00	2.08	-0.90
	N3	83.93	7.43	8.54	22.60		6.73	1.35	-0.01	0.17	
	N4	77.84	4.22	8.34	23.15	2.00	26.00	-1.16	-0.31	3.62	0.30
	NPP1			7.61	25.06	6.70			-0.37	-1.73	