Effects of pond management, tidal restoration, and annual weather on waterbird population change in salt ponds of south San Francisco Bay

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Table of Contents

Executive Summary

The purpose of this report is to identify possible drivers of changes in waterbird abundance within and around the South Bay Salt Pond Restoration Project (SBSPRP), the largest tidal marsh restoration project on the United States' Pacific Coast. The SBSPRP is restoring 15,000 acres of former salt evaporation ponds to a mix of tidal marsh and enhanced managed pond habitats. Restoration and monitoring is entering its third decade, and long-term population analyses have shown declines in some species and guilds, but evidence of cyclical trends (regular increases and decreases) in others. Trends could feasibly be driven by several factors, including but not limited to shifts in management and water quality within ponds as they have been transitioned from salt ponds to wildlife-managed ponds; habitat quality shifts as a result of restoration of ponds to tidal actions; and annual weather driving broader-scale population trends. In order to maintain baseline numbers of waterbirds in the South Bay and recover those populations that have experienced long-term declines, it is important to determine the drivers of these long-term trends to assess the likelihood that observed declines may be driven by SBSPRP actions, and the potential for changes in management within enhanced ponds to improve population trajectories.

This report is based on wintering and migratory waterbird abundance and habitat data collected by San Francisco Bay Bird Observatory from 2005–2023 and by the U.S. Geological Survey from 2002–2013. We constructed linear mixed models relating changes in waterbird abundance to changes in water quality, tidal marsh restoration, habitat features, and annual weather at six pond complexes in the South San Francisco Bay. 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 the U.S. Fish & Wildlife Service as part of the Don Edwards San Francisco Bay National Wildlife Refuge. Eden Landing Ecological Reserve ponds are owned and managed by California Department of Fish and Wildlife, except for pond CP3C which is owned by Cargill Salt.

Over the entire survey period, 28,870,480 birds were observed during 13,920 individual surveys. Data for each pond were averaged across visits within a season (winter, fall, spring, or summer) and change in abundances and water quality measures were calculated from the previous comparable season (if present). Some seasonal observations still could not be included in the models because they were missing water quality information in either the focal season, the previous season, or both. Species were grouped into five guilds (dabbling ducks, diving ducks, medium shorebirds, small shorebirds, and phalaropes) based on foraging methods and prey requirements, and we also assessed three species of special concern (eared grebes, ruddy ducks, and Bonaparte's gulls). The final dataset therefore represented 9,424 average seasonal counts of eight focal species/guilds (n = 233 to n = 2,014, depending on the species/guild).

These were compared to the following covariates: pond area, number of islands, maximum percent of levees open to public across all years, maximum percent of levees open for hunting across all years, distance to San Francisco Bay, distance to creek/slough, distance to the Tri-cities Landfill, breached (to tidal action), Muted (muted tidal), water elevation, salinity, pH, dissolved oxygen, water temperature, cumulative annual precipitation (year-to-date, last year's, and last three years), seasonal mean daily maximum temperature, and seasonal mean daily minimum temperature.

We found change in salinity most strongly predicted change in abundance across most guilds. There was one exception: decreases in Bonaparte's Gull abundance were related to decreasing salinity within ponds. Contrary to our predictions for phalaropes, however, change in salinity was inversely related to change in phalarope abundance, indicating increases in abundance occurred as salinity decreased. Dissolved oxygen, pH, and water temperature had weaker and more guild/species-specific results.

Six of the eight species/guilds we assessed show impacts of weather, and each of the five variables we assessed was supported in at least one top model. Precipitation had a positive relationship with both duck guilds, and an inverse relationship with both shorebird guilds. All relationships with temperature were "positive" in that warmer temperatures (both minimum and maximum) were associated with population increases, and colder temperatures with population decreases. We found very little support for the influence of pond characteristics or tidal restoration population changes, but caution against drawing firm conclusions from those results, as they may be the result of modeling choices and data limitations.

We stress that this model is designed to be a preliminary assessment that can be improved upon in future iterations and annual reports. Nevertheless, our results clearly highlight the importance of both salinity and weather in driving observed trends in waterbirds of South San Francisco Bay. They showcase both the clear benefits the SBSPRP has provided, while also highlighting challenges in balancing species' competing habitat needs within the region. Based on our findings, salinity management is likely to remain a key tool in achieving that balance.

List of Tables

List of Figures

Introduction

Since 2002, the U.S. Fish and Wildlife Service (USFWS) and California Department of Fish and Wildlife (CDFW, formerly California Department of Fish and Game) has been restoring 15,100 acres of former salt evaporator ponds in the South San Francisco Bay. The South Bay Salt Pond Restoration Project (SBSPRP) is restoring the area to a mix of tidal and enhanced pond habitats managed for wildlife. Additional Project goals include enhancing flood protection and improving public access to these wetland areas.

These ponds were previously managed by Cargill Salt for commercial salt production, and have been present in the San Francisco Bay for over 150 years (Ver Planck 1958). Both current and former salt ponds have significant wildlife value (Anderson 1970, Accurso 1992, Takekawa et al. 2001, Warnock et al. 2002). As a part of the Pacific Flyway, the San Francisco Bay is a key habitat for migratory and wintering waterbirds, supporting more than a million birds throughout the year (Page et al. 1999, Warnock et al. 2002). Its importance may be increased from pre-European settlement due to the loss of wetlands elsewhere, including the draining of over 90% of the wetlands in the Central Valley that form the core of the California's portion of the Pacific Flyway (Dahl 1990) the ponds may have become disproportionately important foraging and roosting areas for waterbirds. Recognizing the value of the ponds, the SBSPRP plans to retain some of these areas as enhanced pond habitats for wildlife, while restoring most to their historical state as tidal marsh. Nevertheless, some specialist guilds/species may be at-risk of declining as salt ponds altered into enhanced pond habitats and/or restored to tidal action experience changes in water quality characteristics (i.e., salinity, pH, dissolved oxygen, and temperature) that may affect invertebrate communities. Former salt ponds notably support many hypersaline specialist species, such as phalaropes, eared grebes, and Bonaparte's gull, whose saline lake habitats elsewhere in the Great Basin are imperiled by climate change and water withdrawals (Carle et al. 2023).

Management guidelines are provided by the SBSPRP Adaptive Management plan (SBSPRP 2007) to ensure the SBSPRP improves the ecological function and health of San Francisco Bay as part of the Project's NEPA/CEQA requirements. For waterbirds, annual monitoring is used based on monitoring of population abundances and comparing these to "trigger values", with higher thresholds for three-year declines and more strict thresholds for one-year declines (Table 1). Triggers were set based on each population's abundance during 2002–2005, before the start of most management activities. Ruddy Ducks (*Oxyura jamaicensis*), Eared Grebes (*Podiceps nigricollis*), Bonaparte's Gulls (*Chroicocephalus philadelphia*), diving ducks, small shorebirds, and phalaropes have had NEPA/CEQA targets defined, and guidelines have also been established for dabbling ducks and medium shorebirds based on USFWS management goals (South Bay Salt Pond Restoration Project 2007, Tarjan 2019a). Targets are set for each species/guilds based on abundances during a single focal season (except for small shorebirds, which have a target for both fall and spring migration; Table 1).

Restoration efforts have been underway for over twenty years, and analyses of long-term population trends have shown some species and guilds are declining within South San Francisco Bay. Of the nine species/guild targets, six exceeded their 2005–2007 SBSPRP baseline values (Van Schmidt 2023). For five of these species/guilds, this is attributable to increases within the Project footprint, indicating long-term benefits of the restoration and enhancement activities for waterbird populations within South San Francisco Bay.

However, several other guilds show signs of decline within the SBSPRP (Figure 1). Two species/guilds that specialize in saline habitats have declined below trigger values: phalaropes (Figure 1a) and Bonaparte's Gulls (Figure 1b; Burns et al. 2023, Van Schmidt 2023). Despite increasing within the South Bay overall, Eared Grebe counts are approaching zero within the SBSPRP footprint, with increases largely due to population growth in the salt production ponds (Figure 1c; (Van Schmidt 2023). Studies have also found that breeding waterbirds have declined within South San Francisco Bay (Hartman et al. 2021). Even for species that have increased, several have declined again in recent years, which may simply be cyclical population dynamics resulting from annual weather or density-dependent regulation (Coates et al. 2021).

For a decline to be significant under NEPA/CEQA requirements, it must be due to restoration and management activities, which has yet to be determined. (De La Cruz et al. 2018)is the only current assessment of the relationship between pond characteristics and waterbird trends within the SBSPRP for migratory and wintering species/guilds. Their study had two goals, (1) assessing trends in waterbird populations and (2) identifying habitat characteristics at the pond-scale and within ponds ("grid-scale") that predicted waterbird abundance. Scullen et al. (2013) similarly modeled how water quality parameters related to shorebird abundance, but only within the Cargill-managed ponds. Both of these studies fit shorebird count data (via a generalized linear model with a negative binomial or Poisson link function) to habitat characteristics. However, neither study has combined these various threads to explicitly assess how *changes* in pond habitats in South San Francisco Bay related to the observed *changes* in waterbird abundance across both SBSPRP and remaining commercial salt ponds. Such an analysis is promising because of the strongly divergent trends between these areas in some groups (Figure 1). One has recently been completed which related change in waterbird abundance to SBSPRP management for four breeding waterbirds at a landscape scale: American Avocets (*Recurvirostra americana*), Black-necked Stilts (*Himantopus mexicanus*), Forster's Terns (*Sterna forsteri*), and Caspian Terns (*Hydroprogne caspia*; Schacter et al. 2023). This study found increases in habitat use in managed ponds compared to salt ponds, but not for tidally restored wetlands.

While valuable, especially for informing habitat management practices to improve habitat quality for the species, no study has yet attempted to link changes in abundance for non-breeding birds to changes in habitats. Even more importantly, no study has assessed how habitat characteristics relate to abundance for the two taxa that have passed trigger points (phalaropes and Bonaparte's Gulls) at all. An analysis of the drivers of observed population changes for non-breeding migratory and wintering birds within and around the SBSPRP is therefore needed.

Report Objectives

The goal of the current research was to develop an initial model relating static pond characteristics, *changes* in pond characteristics, and annual weather to changes in waterbird abundance, in order to identify drivers of observed changes in waterbird populations. Significant resources have been spent monitoring waterbird populations within the SBSPRP, and in nearby areas still used for active salt production, since 2002. SFBBO reports this data annually to the SBSPRP Management Team. The central goal of this report was to compile relevant habitat variables, estimate annual change in population size for focal species/guilds, and create an analysis workflow for a population change model, which can then be incorporated into annual reporting and updated regularly. 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 (Burns et al. 2023), which will be the subject of a different report.

Monitoring efforts have detected declines in several key species and/or foraging guilds—most notably, Bonaparte's gull and phalaropes, two saline specialists—which have exceeded their trigger values. While we hypothesize that changes in salinity could have driven these declines, importantly, the current study was *not* designed to definitively answer this question. Such an assessment requires a more thorough investigation of the data. There are significant challenges to modeling long-term population trends, such as potential time lags, cross-scale interactions, incomplete habitat quality and/or waterbird abundance data for tidally restored wetlands, and potentially confounding relationships between drivers of waterbird abundance vs. change-in-abundance. The current report aims to form the foundation of such an analysis by compiling datasets, creating key workflows, and documenting initial findings for the SBSPRP Management Team to guide future development.

Methods

Study Area

The study area includes approximately 85 delineations of current and former salt ponds in the Santa Clara, Alameda and San Mateo counties of California. Currently and formerly monitored ponds include 28 ponds in the Alviso complex, 12 ponds in the Coyote Hills complex, 4 ponds in the Dumbarton complex, 25 ponds in the Eden Landing complex (pond CP3C is owned by Cargill Salt), 6 ponds in the Mowry complex and 10 ponds in the Ravenswood complex. Although the Coyote Hills, Dumbarton, and Mowry ponds are owned by Don Edwards San Francisco Bay National Wildlife Refuge, Cargill Salt retains salt-making rights and regulates water flow for salt production.

The salinity and depth of all ponds varies from year to year due to restoration and pond enhancement activities, and the management practices and business needs of salt production. Though not directly the target of management, this results in changes in other water quality parameters—temperature, pH, dissolved oxygen, and specific conductivity—which may also affect invertebrate communities and waterbird abundance.

Waterbird Surveys

Waterbird surveys were conducted at each pond, though the frequency and set of ponds surveyed varied over the two decades of surveys from 2002–2023. USGS conducted monthly (three per season) waterbird surveys of ponds in the SBSPRP complexes (Eden Landing, Alviso, and Ravenswood) from October 2002 through April 2013, while SFBBO conducted monthly surveys in Cargill-managed ponds (Mowry and Newark) from October 2005 through April 2015 (De La Cruz et al. 2018). From January 2014 to January 2018 (first winter survey period), SFBBO conducted 6-week long sets of surveys (two in fall, winter, and spring, and one in summer) at all actively monitored ponds. Following a one-year lapse in funding, 6-week surveys began again in January 2019 (second winter survey period), without the single summer survey period. The COVID-19 pandemic led to only a subset of ponds being surveyed from March 2020 through May 2021. Regular surveys began again in September 2021. Table 2 shows the total number of ponds surveyed at least once in each seasonal period.

There were several changes to the set of surveyed ponds over the course of the entire study period. Four tidally restored ponds are no longer surveyed due to access difficulties: E10X since 2006, A6N since 2011, and A20 and A21 since 2013. Two new ponds were added to the list of surveyed sites in summer 2010, A8S and A8W, after a tidal gate was added to the A8 complex to managed it as a muted tidal system. Pond E8X was sometimes surveyed as one pond and sometimes as two ponds (E8XN and E8XS), and was analyzed as a single pond in this report.

Surveys were performed exclusively at high tide, defined as a tide of 4.0 ft or greater at the Alameda Creek Tide Sub-Station (37° 35.70' N, 122° 08.70' W). During each survey, we observed birds from the nearest drivable road or levee using spotting scopes and binoculars. The total number of individuals of all waterbird species present on each pond were counted. Through January 2018, counts were recorded at the "grid-scale" by estimating abundance each using aerial site photos superimposed with 250x250 m^2 individually labeled grids. Bird observations were assigned to ponds only starting in January 2019. See Van Schmidt et al. (2023) for additional survey details. Whenever possible, birds were identified to the species level, with the exception of long-billed and short-billed dowitchers (identified as "dowitchers"), and greater and lesser scaup (identified as "scaup"). When species identification was not possible, we identified birds to genus (e.g., *Calidris*) or foraging guild (e.g., gulls, small shorebirds, medium shorebirds, phalaropes).

Water Quality Sampling

Water quality sampling followed methods described in Murphy et al. (2007). Water quality samples were collected from the surface of the water (depth of <0.5 m). Surveyors recorded dissolved oxygen, salinity, conductivity, pH, and temperature using a Hydrolab Minisonde (Hydrolab-Hach Company, Loveland, CO). When salinities exceeded approximately 72 ppt (the maximum value registered by the Hydrolab Minisonde), we calculated salinity using a hydrometer (Ertco, West Paterson, NJ) to measure specific gravity in combination with a temperature reading from the water sample. Barometric pressure was recorded at the beginning of the water quality sampling day and used to calibrate Hydrolab Minisonde sensors. Sampling was done at one to four (most commonly three) locations that were pre-determined at each pond. The number of sampling points per pond varied across ponds and years based on the size, configuration, accessibility of the pond, and equipment issues. Water quality data was generally collected on the day of the bird survey, but this was not perfectly consistent over the course of the full data collection period due to protocol changes, access constraints, and issues with water quality equipment (Table 2).

When present on the pond, staff gauge readings were also collected if staff gauges were present. Some ponds had multiple staff gauges, in which case only the master staff gauge (indicated by yellow paint) was used. If ponds were so dry that no water reached the staff gauge, no staff gauge reading was recorded (value of NA). SFBBO surveyors in Newark and Mowry began estimating the percent of the pond surface that was open water (vs. dry) in 2007, but this variable was not collected for SBSPRP ponds until August 2011. Because of the large number of missing values, we excluded this variable from this analysis.

Analysis

Calculating change in waterbird abundance

We modeled per-pond inter-annual change in abundance of water species or guilds with linear mixed models (package *lme4*), with a random effect for year. 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 Core Team 2017).

We modeled change in abundance for eight focal species/guilds during their focal season(s) (Table 1). A list of the species that were grouped into each foraging guild is provided in Appendix 1, Table A1.1. For this initial modeling exercise, this subset of groups was selected based on SBSPRP and USFWS management goals following Tarjan et al. (2019a). Breeding populations such as Least Tern (*Sternula antillarum)* and American Avocet were not assessed because summer surveys have ceased within this monitoring program and these taxa have been the focus of other recent studies by SFBBO (PLOVER REPORT 2022) and USGS (Hartman et al. 2021, Schacter et al. 2023). Least Terns have too low of average abundances to fit our multivariate mixed effect models to, but in future work, our models could be readily extended to assess other species/guilds.

For each species/guild, we calculated *abundance* in a given season as the average count at that pond across all surveys during that season. While this reduced the effective sample size, it resolved issues with missing observations, mismatches in water quality and waterbird count survey days, temporal autocorrelation between surveys within a season, and differences in survey frequency over the course of the study.

For each guild, we explored two possible response variables. First, *change in abundance* was calculated as the mean *abundance* of the focal species/guild in its target season in year *t* minus its *abundance* in the previous year *t - 1*. The second measure was *percent change in abundance*, which we calculated as:

$$\textit{\% Change in Abundance} = \mathrm{Ln}(1 + ((Abundance_t - Abundance_{t-1}) / (Abundance_{t-1} + Adjustment)))$$

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 0, we added an adjustment factor to allow us to capture new colonizations, which could be important for rare species. We tested two adjustment factors, one of which was simply *Adjustment* = 1, and one which was:

$$Adjustment = \mathrm{Max}(1, \mathrm{Median}(Abundance)) / 10$$

In initial modeling we found little difference in model results or performance tests (described below) between these two adjustment factors, so we elected to keep the simpler *Adjustment* = 1.

Zero change could indicate either a stable population, or no population (*Abundance* = 0 in both year *t* and year *t-1*). The latter is a different process and likely conflates large amounts of habitat that may be wholly unsuitable. Unlike previous studies (e.g., De La Cruz et al. 2018), we sought explicitly to model drivers of observed changes in populations, *not* abundance or habitat selection. We therefore removed all rows that were 0 in both years from the dataset. It is important to keep in mind when interpreting our results that factors that might be very important in rendering a habitat suitable or unsuitable overall were by design excluded from the scope of our assessment.

Model design

Due to the large number of habitat variables assessed, we did not include a random effect for site. A full list of variables we assessed is in Table 3. We separated variables into five categories: (1) site characteristics, (2) distances from ponds to landscape features, (3) annual changes in pond hydrology, (4) annual changes in pond water quality, and (5) annual weather within South San Francisco Bay.

To increase model parsimony and build on past studies, we constrained our modelsets for the first two groups (site characteristics and location measures) based on results from De La Cruz et al. (2018). They modeled abundance of focal guilds/species for two behavior types (counts of foraging or roosting birds) at two spatial scales (ponds or within-pond grids) based on site characteristics. Counts at the grid-scale were discounted in 2019, and the goal of this analysis was to inform contemporary population monitoring efforts, so we modeled only pond-scale abundance. We also grouped foraging and roosting counts in our models because (1) we sought to assess broader drivers of population trends (i.e., effects of annual weather) rather than exclusively habitat use and (2) De La Cruz et al. (2018) found variability in site characteristics and species/guild generally accounted for more variation in count than foraging vs. roosting habitat use did. They assessed the following covariates: pond area, pond bottom topography, water depth, presence of islands, salinity, whether it had been restored to tidal action, percentage of pond levees open to the public, percentage of pond levees open to hunting, and distance to San Francisco Bay, urban areas, creeks/sloughs, the Tri-Cities Landfill, and levees (grid-scale only).

In our analysis, we set for each species/guild a “base” model, which included covariates De La Cruz et al. (2018) found to significantly affect abundances of either foraging- or roosting birds at either the pond- or grid-scale. There were three exceptions to this: (1) we excluded dynamic measures (pond salinity, water elevation, and whether it had been restored to tidal action) because relating changes in these measures was of particular interest; (2) we did not include topography because data was lacking for sites outside the SBSPRP footprint, (3) we did not include distance to levee because this covariate was not meaningful at the pond-scale (i.e., all ponds had distance to levee = 0 km). Phalaropes and Bonaparte's gulls were not assessed by De La Cruz et al. 2019, so we fit a "base" model for these covariates as an initial stage of model selection (described below).

The following sections describe our methods for calculating each covariate from field-collected data or geospatial data layers. The final set of covariates were then standardized (scaled to their mean and standard deviation) to allow their effect sizes to be compared in final models to identify the relative importance of each in driving changes in abundance. As in De La Cruz et al. (2018), only a subset of the ponds in any given season/year could be included in the final models because they were missing pond or habitat data in the current year, previous year, or both.

Site characteristics

We obtained a list of site characteristics for SBSPRP ponds from De La Cruz et al. (2018). We performed area calculations using the package *stars* in R (version 4.3.1), along with a high resolution (10 m per pixel) digital elevation model. We estimated the number of islands by counting the number of exposed land masses within each pond in satellite imagery in Google Earth Pro (v7.3.6), using multiple dates and at least one with ponds at high fill levels to resolve ambiguity. We characterized islands as landmasses that are surrounded by water and remain above water throughout the year.

We used data from De La Cruz et al. (2018) on the percent of each levee that was open to the public or hunting across years (the maximum value observed in any month throughout the study period), and reviewed state and federal maps for those ponds that were not included in their dataset. Data from De La Cruz et al. (2018) was also used to represent distances to various other features including San Francisco Bay, the nearest urban area, the nearest creek or slough, and the nearest landfill (which are gull foraging areas). For sites they did not assess, we filled gaps in this dataset by measuring the distance from pond centroids using satellite imagery in Google Earth Pro. San Francisco Bay was defined as the nearest point on the boundary of the open water of the Bay; the nearest urban area was defined as the closest paved area; and the nearest creek was the closest named creek; and the landfill distance was to the Tri Cities landfill located in Fremont, CA (37.490 °N, 121.990 °W).

Water quality variables

Water quality parameters affect prey availability of foraging birds and thus contribute to guild distribution patterns (Velasquez 1992, Warnock et al. 2002, Takekawa et al. 2006). For each pond, we calculated seasonal average salinity, temperature, dissolved oxygen, pH, and specific conductivity by taking the mean values measured across all sampling locations and visits within a pond each season. We also calculated change in each of these values as $Value_{year2} - Value_{year2}$.

Hydrology variables

We examined two potential management effects: breaching ponds to restore them to tidal marsh (*Breached*), or the addition of gates to manage a wetland as muted tidal (*Muted*). Lists of sites and the dates they were either tidally breached or changed to muted tidal hydrology were obtained by reviewing published reports and aerial imagery (Appendix 1, Table A1.2). Kelly and Condeso (2017) found that shorebird populations in southern Tomales Bay, California, continued increasing in response to tidal restoration for up to 8 years after levees were breached. We therefore treated *Breached* and *Muted* as binary variables (0 before breach/gate installation, 1 after), assuming that after a breach event, waterbird abundance would continue to change for multiple years in response to the habitat changes and development.

De La Cruz et al. (2018) found water depth significantly predicted the abundance of numerous guild species. We examined both percent open water and measured staff gauge (pond surface elevation) as variables to capture changes in hydrology. However, both measures proved problematic, as they were missing from over half of observations (even after aggregating to the pond-season level). Furthermore, staff gauges are not standardized and in many cases represent different vertical datum models (NGDV29 or NADV88). Translating staff gauge value to pond depth also requires bathymetry data to calculate the difference from the water surface to the pond bottom, however, this data is not available outside of the SBSPRP footprint (De La Cruz et al. 2018). We attempted to use adjustment factors recorded in the database for standardizing staff gauge values across ponds to the NADV88 datum and subtract these values from a coastal digital elevation model to calculate depth, but this resulted in negative values and thus appeared unreliable. Because of the lack of standardization in staff gauge values across ponds, we were also concerned about comparing them within the model. Therefore, we did not use percent open water, water depth, or raw staff gauge values as models. We included one term–change in water elevation –that seemed less unreliable because it relied only on observations of relative change (i.e., it could be calculated just as StaffGaugeValue$_{t-1}$ - StaffGaugeValuet$_{t-1}$. This was still missing for over half of observations, so we filled missing values with 0 (no observed change), and report this as *observed change in water elevation* and note that the measure underestimates true change in water elevation.

Climate covariates

We obtained climate covariates from v8 of the Basin Characterization Model (BCM; Flint and Flint 2014). The BCM is a model of current and future California weather and surface hydrology, which spatially downscales historical and projected global climate variables temperature and precipitation to 270 m. It integrates spatial data on soils, geology, and monthly climate to estimate change in surface runoff, among other variables, which could provide the basis for a more thorough investigation of hydrological impacts of weather on the SBSPRP system. BCM data is not yet available for the 2023 water year, so we obtained PRISM data (PRISM Climate Group 2014) to fill this gap. Provisional PRISM data was used for March–August, and stable historical data for January–February.

For this study, we assessed only three weather effects: precipitation, minimum temperature, and maximum temperature. For each species/guild model, we calculated (package *stars*) maximum and minimum temperature by taking the mean of the three month's daily means for the target season for that species/guild (i.e., winter dabbling ducks in 2012 used the mean of monthly Dec 2011–Feb 2012 mean daily minimum temperature). Studies of shorebirds in coastal California have found that abundances can be affected by cumulative precipitation with up to three-year time lags (Kelly and Condeso 2017, Warnock et al. 2021). We therefore hypothesized that precipitation might be a central driving factor in observed population dynamics, and tested three different variables. The first, Precip0, captured the water-year-to-date cumulative precipitation for the year of the waterbird count (i.e., Sept–Oct for a model of fall abundance, and Sept–May for a model of spring abundance). Precip1 was a one-year lagged cumulative precipitation term (i.e., sum Sept–Aug precipitation in the previous year), and Precip3 was a three-year lagged cumulative precipitation term (i.e, sum of Sept–Aug precipitation in the previous three years).

Model fitting and evaluation

We checked for multicollinearity among variables by calculating variance inflation factors (VIF; package *performance*) and correlation coefficients (package *stats*) for all pairs of variables within the final dataset for all guilds, and flagging VIF >3 or correlation >0.7 as problematic (Zuur et al. 2010). We found specific conductivity and salinity were highly correlated (0.842 for raw values and 0.712 for change in values) so we excluded specific conductivity from the analyses as salinity is a key parameter being managed for in salt production ponds. For the fall-and-spring model of small shorebirds, mean monthly maximum temperature (TMX) was highly correlated with both mean monthly minimum temperature (TMN) and year-to-date precipitation (PPT0), so we removed TMX from this guild's modelset. All other variables passed these checks.

We examined quartile-quartile plots and histograms of residuals (*base* R) to assess whether the linear mixed model's assumptions of normally distributed residuals and heteroscedasticity were valid. We found evidence of non-normal residuals in the *change in abundance* model, but both *% change in abundance* measures had linear quartile-quartile plots with normally distributed residuals for all guilds. We therefore used *% change in abundance* as our measure for our analysis analysis.

We carried out a model selection exercise based on corrected Akaike's Information Criterion (AICc; calculated in package *AICcmodavg*). "Base" models from De La Cruz et al. (2018) were not available for Bonaparte's Gulls and phalaropes, so we first ran a full modelset of all combinations of these variables (Table 2, "base" in column *Stage*). We selected the top model from this assessment to carry forward as the base model. The top base model for Bonaparte's Gull included only the *Dist_Landfill_km* covariate and included no covariates for Phalaropes (intercept only). The model selection tables are in Appendix 2. The base model for small shorebirds also included an additional dummy variable, *Spring*, indicating the season. These base model variables were carried forward into all models in the next stage of model selection.

Model selection on the remaining variables was the main focus of our study. To reduce the size of the modelset and avoid model ambiguity, our modelsets included *Change in [variable]* terms only paired with the original [*variable*]. Therefore, the full modelset was all permutations of column variables in the "main", appended to a single (static) "base" model that was unique to each focal species/guild.

This resulted in up to 4,096 models assessed per focal species/guild. Bonaparte's Gulls, phalaropes, and small shorebirds had fewer because their models required dropping some variables. As mentioned, small shorebirds had TMX removed due to collinearity. Despite tests not flagging it as highly correlated, we found that including both *PPT0* and *PPT1* caused convergence problems in the phalarope dataset—possibly because in the summer season, this was the same variable, just lagged one year. We therefore dropped from the phalarope modelset all models that included both of these terms (either was allowed individually). *Breached* and *Muted* proved problematic for Bonaparte's Gulls and phalaropes, causing their models to fail to converge because there were <30 change observations of these taxa in these ponds within the dataset. Therefore, we grouped these variables as a single binary variable, *BreachedOrMuted* in their modelsets.

Model interpretation

Interpreting the results of our model is 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.

When framing our presentation of our results, we have generally interpreted variables based on observed changes. For example, we know that most guilds have increased, but phalaropes, Bonaparte's Gulls, and Eared Grebes (within the SBSPRP footprint) have declined, so we interpret the results for the former as what *drove the increases* and for the latter what *drove the declines*. We do likewise where there are known habitat or climate impacts—for instance, salinity of ponds was greatly reduced across much of the study area, so we discuss the impact of *decreasing salinity*. In other instances interpretation of the modeled relationship is ambiguous; California has both experienced severe droughts over the past two decades, as well as unusually intense winter storms in 2023. When possible, we draw on previous papers' findings (e.g., Warnock et al. 2021, De La Cruz et al. 2019) to aid in interpretation of likely drivers of effects. For all variables, however, it is important to keep in mind that the observed relationships may have multiple potential explanations.

Results

Over the entire survey period, 28,870,480 birds were observed during 13,920 individual surveys. The final dataset therefore represented 9,424 average seasonal counts of the eight focal species/guilds. Sample sizes are listed in Figures 2–9, and ranged $n = 233$ to $n = 2,014$, depending on the species/guild. Table 4 lists for each species/guild whether each covariate was supported by AICc model selection, the direction of their effect, and whether their 95% confidence intervals overlapped zero.

Dabbling ducks

Increases in dabbling duck abundance were most strongly driven by reductions in pond salinity (Figure 2). De La Cruz et al. (2018) found that dabbling ducks were most abundant on ponds with low salinity (≤33 ppt), though Scullen et al. (2013) did not find support for such a relationship in the salt production pond complexes. Higher precipitation in the previous year was also significantly positively related to change in abundance of dabbling ducks. The other covariates had 95% confidence intervals that overlapped zero, though a positive relationship between dissolved oxygen and increases in abundance was still supported by AICc model selection (Table 5). A positive relationship with dissolved oxygen may be the result of its benefits for invertebrate prey abundance (Scullen et al. 2013). For both salinity and dissolved oxygen, changes in water quality parameters more strongly drove changes in abundance than the parameters themselves. Neither of the base pond characteristics (islands or hunting percent) found to predict abundance by De La Cruz et al. (2018) were strongly supported for relating to changes in abundance, though both parameters did have their signs in the expected direction (islands increased abundance, and hunting decreased it).

Diving ducks

Change in diving duck abundance was very strongly related to decreases in salinity, and to a lesser degree, precipitation in the focal year (Sept–Feb), with rainier years showing increased abundance (Figure 3). Less saline ponds were slightly more likely to see population growth, but confidence intervals overlapped zero, echoing previous studies that have found that less saline ponds have higher overall abundance of diving ducks (Scullen et al. 2013, De La Cruz et al. 2019). Surprisingly, an inverse relationship with dissolved oxygen was supported in model selection (Table 6), though weak (Figure 3). This was unexpected because previous reports found abundance was higher in ponds with higher dissolved oxygen (Scullen et al. 2013, De La Cruz et al. 2019). As in dabbling ducks, the base pond characteristics showed little influence on change in abundance.

Ruddy Ducks

Surprisingly, despite Ruddy Ducks being nested within diving ducks, their best models differed notably (Figures 3, 4). In agreement with the broader guild's trends, increasing salinity most strongly predicted increases in abundance, followed by decreasing pH (Figure 4). Ponds with higher dissolved oxygen levels had more increases in Ruddy Duck abundance over the course of the study period; this more strongly related to change in abundance than *increase* in dissolved oxygen did (a pattern rarely seen in the other species/guilds). Model selection supported that Ruddy Duck population growth was higher as water depth in ponds increased (Table 7), which is in line with expectations for a species of diving duck; however confidence intervals overlapped zero (Figure 4). Like dabbling and diving ducks, base site characteristics did not predict percent changes in abundance.

Eared Grebes

Despite strongly divergent patterns of decline within the SBSPRP footprint and increase within the salt production ponds, change in Eared Grebes abundance was not strongly related to any variable except a positive relationship with interannual increases in water elevation (a proxy for increasing water depth; Figure 5). Both De La Cruz (2018) and Scullen et al. (2013) found that Eared Grebe abundances were higher at ponds with greater staff gauge values. This suggests possibly that the restoration activities or pond drawdowns could be a driving factor in the observed changes, but more analyses are necessary to disentangle whether this relationship could explain the decline within the SBSPRP, the increase within salt production ponds, or both. There was also support for mean maximum monthly winter temperatures and changes in dissolved oxygen explaining changes in Eared Grebe abundance (Table 8), but with large confidence intervals that overlapped zero (Figure 5). Similarly, De La Cruz et al. (2018) did not find any relationships between abundance and water quality measures, though Scullen et al. (2013) found support for higher abundances at ponds with higher pH and salinity. Changes in Eared Grebe abundance were not strongly related to pond characteristics.

Bonaparte’s Gulls

All assessed variables had confidence intervals that overlapped zero for Bonaparte’s Gulls, indicating our models had weak explanatory power for identifying drivers of changes in abundance of this species (Figure 6). However, support for several variables was observed based on AICc (Table 9). Decreasing salinity was the variable with the strongest support for the observed decline, and more saline sites were more likely to experience population growth (Figure 6). Temperature showed the largest effect sizes, but effects were inconsistent, with trends showing that increasing minimum temperature related to increased population size but warmer water temperatures related to decreased population size (or vice-versa). Benefits from higher dissolved oxygen were weakly supported, with very large confidence intervals. Model selection on the base covariates supported a weak relationship with distance-to-landfill, with ponds closer to the Tri-Cities landfill showing greater Bonaparte Gull population growth over time, as expected (Figure 6).

Phalaropes

Salinity was the only predictor of the observed decreases in phalarope populations with confidence intervals not overlapping zero—but surprisingly, decreases in salinity corresponded with increases in phalaropes, opposite of predictions (Figure 7). The phalarope dataset has low statistical power because of high variability in counts during their summer migration (Tarjan 2019b), resulting in several large average effect sizes which had low confidence. Decreasing dissolved oxygen levels also increased phalaropes, as did decreasing water levels. A reduction in population growth one year after rainier years was supported in model selection, albeit weakly (Table 10). Lag effects of one year have also been seen in other migratory phalarope populations (Jehl et al. 1999). Time-lagged effects occur because influences on adult condition can take years to trickle down through juvenile recruitment to eventual population size (Metzger et al. 2010). No site covariates were even weakly supported for predicting changes in abundance, with the “base” model for phalaropes being an intercept-only model. Neither previous study assessed phalaropes (Scullen et al. 2013, De La Cruz et al. 2018).

Medium shorebirds

Reductions in water levels, which presumably resulted in a greater extent of mudflats for foraging, were the most significant predictor of changes in medium shorebird populations (Figure 8). The largest effect was lower mean minimum temperatures during winter significantly predicting increases in abundance (Figure 8). Conversely, greater year-to-date rain (Sept–Feb) correlated to reduced abundance, indicating negative impacts of storms or positive impacts of drought on abundance within South San Francisco Bay, but this covariate's confidence intervals overlapped zero. As seen in other guilds, decreasing salinity was a strong predictor of population growth of medium shorebirds. Negative relationships between change in percent abundance and both dissolved oxygen and change in dissolved oxygen were supported in model selection (Table 11), but had weak effects that overlapped zero (Figure 8). As in other species/guilds, pond characteristics had little influence, but islands did weakly predict increasing abundance, as expected (De La Cruz et al. 2018).

Small shorebirds

We fit a single model for small shorebirds that encompassed both spring and fall migration periods, and found that spring migration had significantly higher population growth (Figure 9). Results for small shorebirds were markedly consistent with those of medium shorebirds: decreasing water elevation was the most significant predictor of increases in small shorebird abundance during migration, but weather was also important. Change in minimum temperatures had the largest effect size and precipitation was inversely related to abundance (Figure 9). Unlike medium shorebirds, small shorebirds showed lag effects of precipitation for up to three years, in line with findings of Warnock et al. (2021). Similar to most guilds, decreases in pond salinity was a strong predictor of increasing population size. There was weak support for change in abundance having a negative relationship with dissolved oxygen and a positive relationship with change in pH (Table 12, Figure 9). These results were contrary to expectations, as Scullen et al. (2013) found small shorebirds were more abundant at ponds with higher salinity and less abundant at ponds with higher pH. De La Cruz et al. (2018) found higher abundance of small shorebirds in breached ponds, but we did not find any evidence of this. Because our surveys are conducted at or near high tide, they may underestimate small shorebird use of tidally restored ponds, which is likely higher during low tide when mudflats are exposed. However, De La Cruz et al. (2018) did find a positive effect of tidal restoration, so even with this limitation the high-tide surveys are able to capture small shorebird use of tidal ponds to some degree. Lastly, results weakly supported that sites with closer proximities to creeks/sloughs showed increased growth, but population growth was not notably predicted by islands (Figure 9).

Discussion

Influence of pond management and restoration

We were able to directly relate changing water quality parameters, climate, and management of ponds to observed changes in waterbird populations within the SBSPRP and nearby commercial salt pond complexes. We note that several factors (discussed below) limit the scope of this model, and that this is a first-pass at designing a more robust analysis. Most importantly, it is difficult for a model to prove a negative: not finding a significant relationship could be because there is no significant relationship, or it can be because the model is mispecified. With that caveat, we can highlight several clear trends in results.

Salinity

One of the biggest changes in management within the SBSPRP has been the reduction of hypersaline ponds as part of the Initial Stewardship Plan. As early as summer 2004, actions were taken to reduce salinity in hypersaline ponds to transition them away from salt production, and most ponds had transitioned by fall 2005 (SBSPRP 2007). Ponds A12–A15, E5, and E6 were retained as high salinity ponds. Our results clearly demonstrate that reductions in salinities directly corresponded to statistically significant increases in abundances of most assessed guilds. Of the eight target species/guilds assessed, six showed a statistically significantly increase in abundance in relation to decreases in salinities (or vice versa). Surprisingly, this included phalaropes, a specialist in saline habitats. Only Bonaparte's Gull showed a relationship between decreasing abundance and decreasing salinity. Eared Grebes, another species frequently associated with salt ponds, did not show support for a relationship with salinity, echoing the findings of De La Cruz et al. (2018).

Species below the trigger threshold

The increase in abundance after the ponds were transition from salt production for most of these species is a major success of the project. However, the finding that Bonaparte's Gulls showed a direct relationship between changes in salinity and changes in abundance at a pond scale supports the hypothesis that SBSPRP management actions have contributed to the decline of this species within South San Francisco Bay. Due to the species' rarity, our models had low statistical power, which has also been an issue with phalaropes (Tarjan 2019b). We recommend future studies interrogate this relationship in more depth (e.g., attempting to fit quadratic or non-linear forms, etc.) to attempt to establish a statistically significant link.

Our results contribute to an emerging impression that these species' habitat needs within the SBSPRP may be more complicated than initially believed. Phalaropes generally prefer high-salinity environments such as saline lakes (Jehl 1988, 1999, Frank and Conover 2022, Carle et al. 2023), so it would be expected that they would decrease as salinity decreases. Nevertheless, we found that they *increased* at sites as salinity decreased, even as their total population *decreased* regionally. This finding agrees with results from our Phalarope Migration Surveys, where >90% of birds were most found in low- or medium-salinity ponds (Burns et al. 2023). Although the high variability in the phalarope dataset reduces its statistical power (Tarjan 2019b), our model had the highest density of usable data for this species during the years of the Initial Stewardship Plan (Figure 10), which was during their initial decline (Figure 1a). The annual report on 2023 Phalarope Migration Surveys will apply this analysis model to that dataset, but a more careful interrogation of the historic dataset may provide more useful information about the period of decline.

Two phalarope species use the SBSPRP as migratory stopover grounds. Wilson's Phalaropes (*Phalaropus tricolor*) almost exclusively utilize high-salinity lakes and ponds during migration (Carle et al. 2023), and make up a notable minority of current observations (Burns et al. 2023, LaBarbera et al. 2023). Red-necked Phalaropes (*P. lobatus)* are believed to have more plasticity in foraging habitats, utilizing coastal habitats during migration rather than exclusively saline lakes (Rubega and Inouye 1994). We hypothesize divergent patterns between the two species, with Red-necked Phalarope proportionally increasing as salinity decreased, and the population-level decline driven by early loss of Wilson's Phalaropes. We plant to analyze these species separately in future research, including in the next Phalarope Migration Surveys report.

Tidal restoration

We found no statistically significant relationships between changes in waterbird abundance after ponds were breached for tidal restoration or had tidal gates installed. However, our analysis of tidal breach impacts is likely not robust due to data limitations. Unfortunately, two ponds (E10X, A6N) were never surveyed post-breach because tidal restoration rendered the sites inaccessible, and two more ponds

(E8AE, E8AW) never had their water quality points measured due to inaccessibility of the water. Therefore, data on how waterbird abundances changed in response to the water quality shifts in these ponds could not be included in our analysis. We were able to model only five tidally breached ponds that continued to have accessible water quality points: A17, A19, A20, A21, E8X. Breached ponds therefore represented only 6% of the surveyed ponds, and even less of the final dataset because A20 and A21 have not been surveyed since 2013. Because of these limitations and sampling biases, our analysis is better suited to assessing changes in managed/enhanced ponds than in tidally restored ponds.

Small shorebirds were the only focal guild for which De La Cruz et al. (2018) found support for tidal breach improving habitat, though their dataset shares the limitations of ours. Kelly & Condeso (2017) also observed a steady increase in shorebird abundance following tidal marsh restoration, which reached a threefold increase 8-years post breach. They found little influence of restoration outside of the immediate area of restoration, though we did not assess spillover effects of tidal restoration on nearby ponds.

There are several other reasons that tidal breaching could not show an effect in our dataset. Ponds are also drawn down before breaching, which could also temporarily increase small shorebird use the year *before* breaching, complicating inference in our year-to-year analysis model. There may be more complex changes as the hydrology and vegetation of the ponds changes as they are drawn down, breached, and gradually flooded, which our current binary breached vs. not-breached variable did not capture. A time-since-breach effect or year-before-breach effect might also benefit future analysis.

Pond water level

As expected, we saw strong significant effects of water depth. Falling water elevations increased abundances of both small shorebirds, medium shorebirds, phalaropes (guilds which all use shallow water to forage; Figs. 7–9) and increasing water elevations increased abundance of Eared Grebes and Ruddy Ducks (species which dive to forage; Figs. 4–5). Surprisingly, a positive effect of increasing water elevation was not supported for the diving duck guild overall. Nevertheless, changes in water elevation proved to be a key management factor driving changes in abundance, in agreement with De La Cruz et al. (2018) who also found water depth was central to guild abundance patterns.

Because we lacked bathymetry data for all ponds, and there were many missing staff gauge values, we could not include true water depth as a variable in our model. Bathymetry is available for SBSPRP ponds and De La Cruz et al. (2018) was therefore able to calculate and assess true water depth because they did not assess non-SBSPRP ponds; but for Eared Grebes they also assessed Cargill ponds, and then they too had to use staff gauge values as a proxy for water depth. In this study, we felt it crucial to include both SBSPRP and Cargill ponds because the Cargill ponds serve as a “control group” for management actions, since they should have experienced less dramatic changes in pond characteristics. We therefore used observed change in water elevation (i.e., change in staff gauge value) as a proxy for water depth. This should be a reasonable representation for change in elevation because the relative change should be the same regardless (i.e., falling depth should either benefit small shorebirds to some degree regardless of the pond’s average starting depth). The validity of this approach is supported by the strong significance of this variable in several of our models. Nevertheless, our model’s statistical power is likely reduced because of this limitation, which may explain why some guilds (e.g., diving ducks) did not show significant effects.

Other water quality characteristics

Higher levels of dissolved oxygen were related to increase in population size for Bonaparte’s Gull, Ruddy Duck, and dabbling ducks, but were negatively related for phalaropes and shorebirds. These differing responses likely reflect different targets by each foraging guilds prey, as changes in dissolved oxygen can have cascading and complex effects on prey relative abundance and trophic webs (for an extreme example, see Takekawa et al. 2015). Aside from changes in salinity, a negative relationship with dissolved oxygen was one of the strongest relationships for phalaropes, and represents a viable alternative hypothesis for the decline. Invertebrate sampling of phalarope ponds vs. non-ponds across a gradient of dissolved oxygen concentrations could assess this hypothesis.

Water temperature and pH generally had weak effects in models. However, Ruddy Duck showed a strong inverse relationship with pH. The reason for this is unclear, and we could not find any potential explanations in the literature. Most pond habitat characteristics fluctuate together seasonally (Van Schmidt et al. 2023), potentially complicating inference.

We found moderate support for changes in water depth relating to changes in bird populations in some species. However, De La Cruz et al. (2018) found this variable highly predictive for overall waterbird abundance. Because of the many missing staff gauge values and lack of bathymetry data outside of the SBSPRP ponds, our model of water depth likely lacked statistical power, and we caution against drawing strong conclusions based on results for this variable. Compilation of data on past water level drawdowns and drying would greatly benefit future versions of this model.

Pond characteristics

Unlike De La Cruz et al. (2018), we found very little support for the influence of pond characteristics on our change model. We recommend that inferences about the value of pond area, islands, and other static characteristics be based on their report. Their model related abundance and habitat selection to these and found significant results; but our paper *not* finding significant results does not mean these characteristics are not important. Because these pond characteristics were largely static, they did not predict changes in the abundance, but they could have strong influences on whether sites were suitable or unsuitable overall (i.e., habitat selection).

Weather effects

Outside of management activities, weather and broader flyway-scale populations trends unrelated to local factors are the two leading hypotheses for what could be driving population dynamics. Assessing the influence of flyway-scale population trends is beyond the scope of this paper, but could be assessed by Bayesian modeling following methods outlined in Coates et al. (2021). However, our findings do confirm the hypothesis that annual weather plays a role in driving trends in observed trends. Six of the eight species/guilds we assessed show impacts from weather, and each of the five variables we assessed was supported in at least one top model.

Precipitation had a positive relationship with both duck guilds (Figures 2, 3), and a negative one with both shorebird guilds (Figures 8, 9). During the California drought from 2013–2015, drying wetlands in the Central Valley correlated to decreases in both Central Valley and San Francisco Bay populations of yellowlegs (*Tringa* spp.) and dowitchers (*Limnodromus* spp.), but evidence of a shift in distribution from the Central Valley to the San Francisco Bay in Dunlin (*Calidris alpine;* Barbaree et al. 2020). Our results partially concurred with this finding, with precipitation decreases strongly predicting increases in small shorebirds (or vice versa; Figure 9), but also in medium shorebirds (albeit weakly). Conversely, precipitation effects could be the result of birds moving inland to escape winter storms; Warnock and Takekawa (1995) documented shifts in Dunlin from coastal wetlands to inland wetlands in rainy years.

All relationships with temperature were "positive" in that warmer temperatures (both minimum and maximum) were associated with population increases, and colder temperatures with population decreases. Responsiveness was higher to minimum temperature (three of the four species/guilds) than maximum temperature (only Eared Grebe, and confidence intervals overlapped zero). Temperatures were much more strongly related to shorebird abundance for both guilds (Figures 8, 9). There was strong correlation between maximum monthly temperature and both minimum monthly temperature and precipitation during small shorebirds' fall and spring migration, so this relationship could alternatively represent the impacts of higher maximum temperatures. However, this seems the less likely explanation given the strong evidence from other studies that drought and winter storms affect annual shorebird distribution patterns along the Pacific Flyway.

Management recommendations and future directions

The strongest relationships found in our study were related to the transition from ultra-high salinity salt production ponds to enhanced managed pond habitats. The SBSPRP has clearly improved habitat quality for many waterbirds guilds in San Francisco Bay. Given the clear evidence that habitat management efforts have created benefits for most guilds, we recommend focusing on identifying the habitat needs of the remaining taxa in need of support—phalaropes and Bonaparte's Gulls.

Our results support the conclusion that Bonaparte's Gull has declined in part due to the reduction of salinity in the SBSPRP. The retention of some high-salinity ponds is therefore advisable to allow this species to persist with South San Francisco Bay. Given that there is a decade of data, future research could easily dig deeper into the habitat relationships to identify best management practices. For phalaropes, we avoid making recommendations in this report, as we will do so in the forthcoming Phalarope Migration Survey report (see also the recommendations in last year's report, Burns et al. 2023).

We emphasize strongly that the current study was conducted as a data exploration exercise to build a foundational model using a small amount of residual reporting funds. The models presented here need additional development before they can conclusively answer questions about the factors driving waterbird population changes within this region. Notable improvements would be to:

1) Calculate model averaged coefficients from model selection results. We have presented the top models due to the limited time and funding but AIC tables (Tables 5–12) show relatively flat AIC weights, indicating multiple competitive alternative models.
2) Assess (1) interaction terms between variables and between raw measures and change-over-time measures, and (2) potential non-linear relationships between predictor variables and rates of waterbird population change (e.g., quadratic terms or generalized additive models).
3) Integrate time lags for multiple variables (not just precipitation).
4) Create model predictions, with and without key covariates to demonstrate more conclusively the amount of each population trajectory that the observed relationships account for. For example, we could remove the salinity changes from the input data and re-predict the Bonaparte's Gull population trajectories to determine how important this is to the overall trend, and then compare that to the triggers.
5) Improve annual measures of management changes (island construction dates, hunting access, water drawdowns) and disturbance from construction.
6) Gather water depth measures based on bathymetry, and interpolate or estimate missing values. Over half of records were missing staff gauge readings or percent open water records. Because of the importance of water depth for most taxa's abundance patterns (De La Cruz et al. 2018), we recommend that staff gauges be installed at all ponds in a standardized way, so that water levels can be measured more consistently across the survey area and related to waterbird use. This would be especially fruitful at ponds that are currently or were formerly used by Bonaparte's Gulls and phalaropes, to try and understand drivers of change.

7) A complementary assessment of site extirpation. Our model captures *annual* change rates, but if a site becomes wholly unsuitable, despite this seriously harming the ecosystem, it is registered as only a single loss of abundance, because it is thereafter only “zero abundance” (no change).
8) Assess a broader set of species and foraging guilds.

A major issue is the lack of data on breached ponds post-restoration. This question of how waterbirds populations will respond to these changes will become increasingly important as breached ponds take up an ever-larger proportion of the landscape. Furthermore, as breached ponds transition from mudflat to more fully vegetated tidal marsh, their suitability may once again shift (Athearn et al. 2009). We therefore strongly recommend an increase in monitoring efforts to survey the currently breached ponds to fill this key data gap. Simply gathering water quality information from A8W, E8AE, and E8AW would greatly improve our ability to use the already-monitored waterbird counts. Year-to-year analysis is also confounded by the fact that construction may disrupt abundances before the actual “breach” event takes place. We had to compile data on breach dates from reports because there is no unified database of such habitat changes, but data on construction disturbances is even harder to find, and staff gauge and “percent open water” records were too inconsistently recorded to effectively use. Better data collection on disturbances from construction would also greatly improve models’ ability to assess post-breach changes. Efforts could be made to compile datasets on past construction activities into a unified, searchable database.

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Tables

Table 1. Summary of recent three-year average (2021–2023) non-breeding waterbird trends compared with SBSPRP targets or baseline values (2005–2007; Van Schmidt 2023, Burns and Van Schmidt 2023). Season = the season in which the species/guild counts are highest; SBSPRP target = baseline count defined by the SBSPRP Adaptive Management Plan (SBSPRP 2007). Targets for dabbling ducks and medium shorebirds were not defined in the Adaptive Management Plan, so we assumed that baseline values were the mean count per survey in 2005–2007 (denoted by *); Threshold = NEPA/CEQA significance threshold; Percent change = percent difference between recent counts (most recent three-year average) and SBSPRP targets or baseline values; Trigger = true if a trigger was detected, where two out of the most recent three consecutive years had counts below baseline values for most species/guilds. The trigger for PHAL, BOGU, and EAGR was three consecutive years more than 25% below NEPA/CEQA baseline, or any single year more than 50% below NEPA/CEQA baseline.

Species/Guild	**Season**	**SBSPRP Target**	**Threshold**	**Percent Change (2021–2023)**	**Trigger**
Ruddy ducks	Winter	12602	-15%	159%	FALSE
diving ducks	Winter	39645	-20%	53%	FALSE
small shorebirds	Fall	60623	-20%	70%	FALSE
small shorebirds	Spring	73728	-20%	27%	FALSE
Eared grebes	Winter	5640	-50% / -25%	76% / 48%	FALSE
phalaropes	Summer	3225	-50% / -25%	-77% / -75%	TRUE
Bonaparte's gulls	Winter	1270	-50% / -25%	40% / -62%	TRUE
dabbling ducks	Winter	48524*	NA	-18%	FALSE
medium shorebirds	Winter	23312*	NA	-2%	FALSE

Table 2. Number of ponds that had one or more waterbird counts, water quality surveys, and complete cases (count survey *and* water quality sampling in both the current and previous season).

SeasonID	Season	WaterYear	WaterbirdCounts	WaterQualitySamples	CompleteCases
1	Fall	2003	61	7	0
2	Winter	2003	61	8	0
3	Spring	2003	61	8	0
4	Summer	2003	61	21	0
5	Fall	2004	61	58	7
6	Winter	2004	61	60	8
7	Spring	2004	61	59	6
8	Summer	2004	61	58	9
9	Fall	2005	61	58	54
10	Winter	2005	61	58	58
11	Spring	2005	61	58	54
12	Summer	2005	61	58	26
13	Fall	2006	83	80	55
14	Winter	2006	83	82	57
15	Spring	2006	83	81	56
16	Summer	2006	83	81	27
17	Fall	2007	82	79	78
18	Winter	2007	82	79	79
19	Spring	2007	82	77	55
20	Summer	2007	82	79	27
21	Fall	2008	82	80	74
22	Winter	2008	82	79	70
23	Spring	2008	82	81	55
24	Summer	2008	82	76	21
25	Fall	2009	82	75	73
26	Winter	2009	82	81	71
27	Spring	2009	82	81	78
28	Summer	2009	82	77	21
29	Fall	2010	82	77	74
30	Winter	2010	82	81	80
31	Spring	2010	82	80	77
32	Summer	2010	81	75	19
33	Fall	2011	84	77	74
34	Winter	2011	84	79	76
35	Spring	2011	81	78	73
36	Summer	2011	84	81	18
37	Fall	2012	82	77	71
38	Winter	2012	83	79	75
39	Spring	2012	83	78	69
40	Summer	2012	83	73	12
41	Fall	2013	83	73	68

42	Winter	2013	83	80	77
43	Spring	2013	83	78	69
44	Summer	2013	83	74	15
45	Fall	2014	83	69	58
46	Winter	2014	81	74	74
47	Spring	2014	81	78	71
48	Summer	2014	81	72	16
49	Fall	2015	81	77	59
50	Winter	2015	81	78	74
51	Spring	2015	81	78	75
52	Summer	2015	81	74	10
53	Fall	2016	81	76	72
54	Winter	2016	81	78	78
55	Spring	2016	81	78	76
56	Summer	2016	80	74	6
57	Fall	2017	81	76	68
58	Winter	2017	81	78	78
59	Spring	2017	81	76	72
60	Summer	2017	81	75	6
61	Fall	2018	81	75	68
62	Winter	2018	81	75	57
63	Spring	2018	0	0	0
64	Summer	2018	0	0	0
65	Fall	2019	0	0	0
66	Winter	2019	81	72	51
67	Spring	2019	81	75	0
68	Summer	2019	0	0	0
69	Fall	2020	81	77	0
70	Winter	2020	81	78	69
71	Spring	2020	44	37	24
72	Summer	2020	0	0	0
73	Fall	2021	0	0	0
74	Winter	2021	24	22	22
75	Spring	2021	0	0	0
76	Summer	2021	0	0	0
77	Fall	2022	81	78	0
78	Winter	2022	81	78	22
79	Spring	2022	81	78	0
80	Summer	2022	0	0	0
81	Fall	2023	81	78	76
82	Winter	2023	81	78	78
83	Spring	2023	81	78	75

Table 3. Predictor variables used to predict change in seasonal mean waterbird abundance within 85 current and former salt ponds of the South San Francisco Bay. “Base” stage indicates the variable were included in all models significant for a species/guild in De La Cruz et al. (2018), while “Main” indicates AICc model selection was carried on out on the variable. Change *model* column indicates whether the variable varied was static across years, varied annually, or was modeled as the annual change in this variable from its value in the previous year (“Change”, with name suffix “*_AC*”). Three variables were excluded from consideration: specific conductivity, because it was highly correlated with salinity, percent open water, because it was not collected during the first half of the study period, and distance to urban area, because an earlier study found it did not significantly affect any guild (De La Cruz et al. 2018).

Stage	Class	Name	Description	Units	Change model
Base	Pond characteristics	Area_km2	Pond area	km^2	Static
Base		Islands_num	Number of islands	count	Static
Base		OpenPublic_pct	Maximum % of levees open to public across all years	%	Static
Base		OpenHunting_pct	Maximum % of levees open for hunting across all years	%	Static
Base	Pond location	BayDist_km	Distance to San Francisco Bay	km	Static
Base		CreekDist_km	Distance to creek/slough	km	Static
Base		LandfillDist_km	Distance to Tri-cities Landfill	km	Static
Main	Hydrology	Breached	Breached to tidal action	Binary 1 / 0	Annual
Main		Muted	Gated culvert to muted tidal	Binary 1 / 0	Annual
Main		WaterElev_m	Mean water depth	cm	Change
Main	Water quality	Salinity	Mean salinity	ppt	Annual + Change
Main		pH	Mean pH	pH scale	Annual + Change
Main		LDO_mgL	Mean dissolved oxygen	mg/L	Annual + Change
Main		Temp_C	Mean water temperature	°C	Annual + Change
Main	Climate	PPT0	Cumulative annual precipitation – year-to-date	mm	Annual
Main		PPT1	Cumulative annual precipitation – last year’s only	mm	Annual
Main		PPT3	Cumulative annual precipitation – last three years	mm	Annual
Main		TMX	Mean daily maximum temperature for the season	°C	Annual
Main		TMN	Mean daily minimum temperature for the season	°C	Annual

Table 4. Number of times each main predictor variable was positively or negatively related to inter-annual percent change in mean seasonal abundance of eight species or foraging guilds within 85 current and former salt ponds of the South San Francisco Bay. + indicates positive, ++ indicates positive and 95% confidence intervals not-overlapping zero, 0 indicates negligible effect, - indicates negative, -- indicatives negative and 95% confidence intervals not overlapping zero. Blanks indicate the variable was not in the top model.

Variable	Dabbling duck	Diving duck	Ruddy Duck	Eared Grebe	Bonaparte's Gull	Phala-ropes	Medium shorebird	Small shorebird
Area_km2		0	0	+				
Islands_num	+	0	–	+		+	+	0
OpenPublic_pct								0
OpenHunting_pct	–							
BayDist_km		0						
CreekDist_km						–	0	–
LandfillDist_km								
Breached								
Muted								
WaterElev_m_AC			+	+ +	–	– –	– –	– –
Salinity	+	–	0		+	0	0	+
Salinity_AC	– –	– –	– –		– –	– –	– –	– –
pH			0					0
pH_AC			– –					+
LDO_mgL	0	–	+ +	0	0	+	–	–
LDO_mgL_AC	+	–	+	+	–	+	–	0
Temp_C								
Temp_C_AC								
PPT0		+				–	–	
PPT1	+ +				–			
PPT3								– –
TMX				+				
TMN					+	+ +	+ +	+ +

Table 5. Model selection table for interannual change in abundance of dabbling ducks during winter (Dec–Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003–2023. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only models with AICc weight ≥0.01 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**	**ΔAICc**	**AICc**	**-2LogLike**	**w_i**
Abun_PC ~ Islands_num + OpenHunting_pct + LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC	10	0.00	5275.73	-2627.77	0.05
Abun_PC ~ Islands_num + OpenHunting_pct + LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC + TMN	11	1.67	5277.40	-2627.59	0.02
Abun_PC ~ Islands_num + OpenHunting_pct + LDO_mgL + LDO_mgL_AC + PPT0 + PPT1 + Salinity + Salinity_AC	11	1.76	5277.49	-2627.63	0.02
Abun_PC ~ Islands_num + OpenHunting_pct + Breached + LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC	11	1.83	5277.56	-2627.67	0.02
Abun_PC ~ Islands_num + OpenHunting_pct + LDO_mgL + LDO_mgL_AC + PPT1 + PPT3 + Salinity + Salinity_AC	11	1.86	5277.59	-2627.68	0.02
Abun_PC ~ Islands_num + OpenHunting_pct + LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC + WaterElev_m_AC	11	1.91	5277.64	-2627.70	0.02
Abun_PC ~ Islands_num + OpenHunting_pct + LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC + TMX	11	1.91	5277.64	-2627.71	0.02
Abun_PC ~ Islands_num + OpenHunting_pct + Muted + LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC	11	2.01	5277.75	-2627.76	0.02
Abun_PC ~ Islands_num + OpenHunting_pct + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT1 + Salinity + Salinity_AC	12	2.33	5278.07	-2626.90	0.02
Abun_PC ~ Islands_num + OpenHunting_pct + LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC + Temp_C + Temp_C_AC	12	2.92	5278.65	-2627.19	0.01
Abun_PC ~ Islands_num + OpenHunting_pct + LDO_mgL + LDO_mgL_AC + PPT0 + PPT1 + Salinity + Salinity_AC + TMX	12	3.16	5278.90	-2627.31	0.01
Abun_PC ~ Islands_num + OpenHunting_pct + LDO_mgL + LDO_mgL_AC + Salinity + Salinity_AC	9	3.17	5278.91	-2630.38	0.01

Table 6. Model selection table for interannual change in abundance of diving ducks during winter (Dec–Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only models with AICc weight ≥0.01 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**	**ΔAICc**	**AICc**	**-2LogLike**	**w_i**
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + PPT0 + Salinity + Salinity_AC	11	0.00	5083.10	-2530.44	0.02
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + PPT0 + Salinity + Salinity_AC + TMN	12	0.79	5083.89	-2529.82	0.01
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + PPT0 + Salinity + Salinity_AC + WaterElev_m_AC	12	0.87	5083.97	-2529.86	0.01
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + Salinity + Salinity_AC + TMN + TMX	12	0.99	5084.10	-2529.92	0.01

Table 7. Model selection table for interannual change in abundance of Ruddy Ducks during winter (Dec–Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only models with AICc weight ≥0.01 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	ΔAICc	AICc	-2LogLike	w_i
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + Salinity + Salinity_AC + WaterElev_m_AC	13	0.00	4771.21	-2372.42	0.02
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + Salinity + Salinity_AC + WaterElev_m_AC	14	0.25	4771.46	-2371.52	0.02
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + PPT1 + Salinity + Salinity_AC + WaterElev_m_AC	15	0.52	4771.73	-2370.62	0.02
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + Salinity + Salinity_AC + TMN + WaterElev_m_AC	14	0.55	4771.76	-2371.67	0.02
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + Salinity + Salinity_AC	13	0.61	4771.82	-2372.73	0.02
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + Salinity + Salinity_AC	12	0.80	4772.01	-2373.85	0.02
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT1 + Salinity + Salinity_AC + WaterElev_m_AC	14	0.93	4772.14	-2371.86	0.02
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + PPT1 + Salinity + Salinity_AC	14	1.14	4772.35	-2371.97	0.01
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + Salinity + Salinity_AC + TMN	13	1.30	4772.51	-2373.07	0.01
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	15	1.50	4772.72	-2371.12	0.01
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT1 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	15	1.51	4772.72	-2371.12	0.01
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT3 + Salinity + Salinity_AC + WaterElev_m_AC	14	1.53	4772.74	-2372.16	0.01

Formula	**K**	**ΔAICc**	**AICc**	**-2LogLike**	**w_i**
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + Salinity + Salinity_AC + TMX + WaterElev_m_AC	15	1.55	4772.76	-2371.14	0.01
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + PPT1 + Salinity + Salinity_AC + TMX + WaterElev_m_AC	16	1.62	4772.83	-2370.14	0.01
Abun_PC ~ BayDist_km + Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + Salinity + Salinity_AC + TMX	14	1.83	4773.04	-2372.31	0.01

Table 8. Model selection table for interannual change in abundance of Eared Grebes during winter (Dec–Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only models with AICc weight ≥0.01 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**	**ΔAICc**	**AICc**	**-2LogLike**	**w_i**
Abun_PC ~ Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + TMX + WaterElev_m_AC	9	0.00	4014.34	-1998.08	0.02
Abun_PC ~ Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + TMN + WaterElev_m_AC	9	0.19	4014.53	-1998.17	0.02
Abun_PC ~ Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + WaterElev_m_AC	8	0.24	4014.58	-1999.21	0.02
Abun_PC ~ Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + PPT0 + TMX + WaterElev_m_AC	10	0.61	4014.95	-1997.36	0.01
Abun_PC ~ Islands_num + Area_km2 + LDO_mgL + LDO_mgL_AC + TMN + TMX + WaterElev_m_AC	10	0.88	4015.22	-1997.50	0.01
Abun_PC ~ Islands_num + Area_km2 + Breached + LDO_mgL + LDO_mgL_AC + TMX + WaterElev_m_AC	10	0.94	4015.28	-1997.53	0.01

Table 9. Model selection table for interannual change in abundance of Bonaparte's Gulls during winter (Dec–Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only models with AICc weight ≥0.01 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**	**ΔAICc**	**AICc**	**-2LogLike**	**w_i**
Abun_PC ~ LandfillDist_km + LDO_mgL + LDO_mgL_AC + Salinity + Salinity_AC + Temp_C + Temp_C_AC + TMN	11	0.00	1885.72	-931.48	0.03
Abun_PC ~ LandfillDist_km + LDO_mgL + LDO_mgL_AC + Salinity + Salinity_AC + Temp_C + Temp_C_AC	10	0.20	1885.92	-932.64	0.02
Abun_PC ~ LandfillDist_km + LDO_mgL + LDO_mgL_AC + Temp_C + Temp_C_AC + TMN	9	0.56	1886.28	-933.88	0.02
Abun_PC ~ LandfillDist_km + LDO_mgL + LDO_mgL_AC + Temp_C + Temp_C_AC	8	0.77	1886.49	-935.04	0.02
Abun_PC ~ LandfillDist_km + BreachedOrMuted + LDO_mgL + LDO_mgL_AC + Salinity + Salinity_AC + Temp_C + Temp_C_AC + TMN	12	1.70	1887.41	-931.26	0.01
Abun_PC ~ LandfillDist_km + LDO_mgL + LDO_mgL_AC + PPT3 + Salinity + Salinity_AC + Temp_C + Temp_C_AC + TMN	12	1.73	1887.45	-931.27	0.01
Abun_PC ~ LandfillDist_km + BreachedOrMuted + LDO_mgL + LDO_mgL_AC + Salinity + Salinity_AC + Temp_C + Temp_C_AC	11	1.84	1887.56	-932.40	0.01
Abun_PC ~ LandfillDist_km + LDO_mgL + LDO_mgL_AC + PPT3 + Temp_C + Temp_C_AC + TMN	10	1.85	1887.57	-933.47	0.01

Table 10. Model selection table for interannual change in abundance of phalaropes during winter (Dec–Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only models with AICc weight ≥0.01 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	ΔAICc	AICc	-2LogLike	w_i
Abun_PC ~ LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC + WaterElev_m_AC	9	0.00	1290.76	-635.98	0.03
Abun_PC ~ LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	10	0.15	1290.92	-634.96	0.03
Abun_PC ~ LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC	8	0.77	1291.53	-637.44	0.02
Abun_PC ~ LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC + TMN	9	0.80	1291.56	-636.38	0.02
Abun_PC ~ LDO_mgL + LDO_mgL_AC + Salinity + Salinity_AC + WaterElev_m_AC	8	1.17	1291.94	-637.65	0.02
Abun_PC ~ LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC + TMX + WaterElev_m_AC	10	1.43	1292.19	-635.60	0.01
Abun_PC ~ LDO_mgL + LDO_mgL_AC + Salinity + Salinity_AC	7	1.76	1292.53	-639.01	0.01
Abun_PC ~ BreachedOrMuted + LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC + WaterElev_m_AC	10	1.92	1292.69	-635.85	0.01
Abun_PC ~ LDO_mgL + LDO_mgL_AC + PPT0 + Salinity + Salinity_AC + WaterElev_m_AC	9	1.97	1292.74	-636.97	0.01
Abun_PC ~ LDO_mgL + LDO_mgL_AC + PPT1 + PPT3 + Salinity + Salinity_AC + WaterElev_m_AC	10	1.99	1292.75	-635.88	0.01
Abun_PC ~ LDO_mgL + LDO_mgL_AC + PPT0 + PPT3 + Salinity + Salinity_AC + WaterElev_m_AC	10	2.00	1292.76	-635.88	0.01

Table 11. Model selection table for interannual change in abundance of medium shorebirds during winter (Dec–Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only models with AICc weight ≥0.01 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	ΔAICc	AICc	-2LogLike	w_i
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + PPT0 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	12	0.00	5808.51	-2892.13	0.04
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + Salinity + Salinity_AC + TMN + WaterElev_m_AC	11	0.55	5809.06	-2893.42	0.03
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + PPT0 + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	13	0.65	5809.16	-2891.43	0.03
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	12	0.85	5809.36	-2892.55	0.03
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + PPT0 + PPT1 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	13	0.89	5809.40	-2891.55	0.02
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + PPT0 + PPT3 + Salinity + Salinity_AC + TMN + TMX + WaterElev_m_AC	14	1.03	5809.54	-2890.60	0.02
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + PPT0 + Salinity + Salinity_AC + TMN + TMX + WaterElev_m_AC	13	1.35	5809.86	-2891.78	0.02
Abun_PC ~ CreekDist_km + Islands_num + Breached + LDO_mgL + LDO_mgL_AC + PPT0 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	13	1.75	5810.27	-2891.98	0.02
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + PPT0 + PPT1 + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	14	1.78	5810.30	-2890.97	0.02
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + Salinity + Salinity_AC + WaterElev_m_AC	10	1.96	5810.47	-2895.14	0.01
Abun_PC ~ CreekDist_km + Islands_num + Muted + LDO_mgL + LDO_mgL_AC + PPT0 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	13	2.04	5810.56	-2892.13	0.01
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + Salinity + Salinity_AC + TMN + TMX + WaterElev_m_AC	12	2.24	5810.75	-2893.25	0.01

Formula	K	ΔAICc	AICc	-2LogLike	w_i
Abun_PC ~ CreekDist_km + Islands_num + Breached + LDO_mgL + LDO_mgL_AC + Salinity + Salinity_AC + TMN + WaterElev_m_AC	12	2.27	5810.79	-2893.26	0.01
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + PPT0 + PPT1 + Salinity + Salinity_AC + TMN + TMX + WaterElev_m_AC	14	2.36	5810.88	-2891.26	0.01
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	12	2.43	5810.95	-2893.34	0.01
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + PPT0 + PPT1 + PPT3 + Salinity + Salinity_AC + TMN + TMX + WaterElev_m_AC	15	2.44	5810.96	-2890.28	0.01
Abun_PC ~ CreekDist_km + Islands_num + Breached + LDO_mgL + LDO_mgL_AC + PPT0 + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	14	2.45	5810.97	-2891.31	0.01
Abun_PC ~ CreekDist_km + Islands_num + Breached + LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	13	2.54	5811.05	-2892.37	0.01
Abun_PC ~ CreekDist_km + Islands_num + Muted + LDO_mgL + LDO_mgL_AC + Salinity + Salinity_AC + TMN + WaterElev_m_AC	12	2.59	5811.10	-2893.42	0.01
Abun_PC ~ CreekDist_km + Islands_num + Breached + LDO_mgL + LDO_mgL_AC + PPT0 + PPT1 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	14	2.61	5811.12	-2891.39	0.01
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + PPT1 + Salinity + Salinity_AC + TMN + TMX + WaterElev_m_AC	13	2.62	5811.14	-2892.42	0.01
Abun_PC ~ CreekDist_km + Islands_num + LDO_mgL + LDO_mgL_AC + Salinity + Salinity_AC + TMX + WaterElev_m_AC	11	2.67	5811.19	-2894.48	0.01
Abun_PC ~ CreekDist_km + Islands_num + Muted + LDO_mgL + LDO_mgL_AC + PPT0 + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC	14	2.68	5811.20	-2891.42	0.01

Table 12. Model selection table for interannual change in abundance of small shorebirds during winter (Dec–Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only models with AICc weight ≥0.01 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**	**ΔAICc**	**AICc**	**-2LogLike**	**w_i**
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC + Spring	16	0.00	10036.87	-5002.30	0.13
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC + Spring	17	0.98	10037.85	-5001.77	0.08
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + Muted + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC + Spring	17	1.79	10038.66	-5002.18	0.05
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT1 + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC + Spring	17	1.96	10038.83	-5002.26	0.05
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + Breached + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC + Spring	17	1.97	10038.83	-5002.26	0.05
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT3 + TMN + WaterElev_m_AC + Spring	14	2.04	10038.91	-5005.35	0.05
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + Muted + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC + Spring	18	2.71	10039.58	-5001.62	0.03
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + Breached + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC + Spring	18	2.93	10039.80	-5001.73	0.03
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + PPT1 + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC + Spring	18	2.98	10039.85	-5001.75	0.03

Formula	K	ΔAICc	AICc	-2LogLike	w_i
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT3 + Salinity + Salinity_AC + Temp_C + Temp_C_AC + TMN + WaterElev_m_AC + Spring	18	3.25	10040.11	-5001.89	0.03
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + Muted + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT1 + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC + Spring	18	3.77	10040.64	-5002.15	0.02
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + Breached + Muted + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC + Spring	18	3.78	10040.65	-5002.15	0.02
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + PPT3 + TMN + WaterElev_m_AC + Spring	15	3.79	10040.66	-5005.21	0.02
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + Muted + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT3 + TMN + WaterElev_m_AC + Spring	15	3.88	10040.74	-5005.25	0.02
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + Breached + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT1 + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC + Spring	18	3.93	10040.80	-5002.23	0.02
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + Breached + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT3 + TMN + WaterElev_m_AC + Spring	15	3.98	10040.85	-5005.30	0.02
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT1 + PPT3 + TMN + WaterElev_m_AC + Spring	15	4.02	10040.89	-5005.32	0.02
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + PPT3 + Salinity + Salinity_AC + Temp_C + Temp_C_AC + TMN + WaterElev_m_AC + Spring	19	4.08	10040.94	-5001.28	0.02
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + Breached + Muted + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC + Spring	19	4.69	10041.56	-5001.59	0.01
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + Muted + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + PPT1 + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC + Spring	19	4.69	10041.56	-5001.59	0.01

Formula	K	ΔAICc	AICc	-2LogLike	w_i
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT3 + Temp_C + Temp_C_AC + TMN + WaterElev_m_AC + Spring	16	4.80	10041.67	-5004.70	0.01
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + Breached + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT0 + PPT1 + PPT3 + Salinity + Salinity_AC + TMN + WaterElev_m_AC + Spring	19	4.93	10041.80	-5001.71	0.01
Abun_PC ~ CreekDist_km + Islands_num + OpenPublic_pct + Muted + LDO_mgL + LDO_mgL_AC + pH + pH_AC + PPT3 + Salinity + Salinity_AC + Temp_C + Temp_C_AC + TMN + WaterElev_m_AC + Spring	19	4.94	10041.81	-5001.71	0.01

Figures

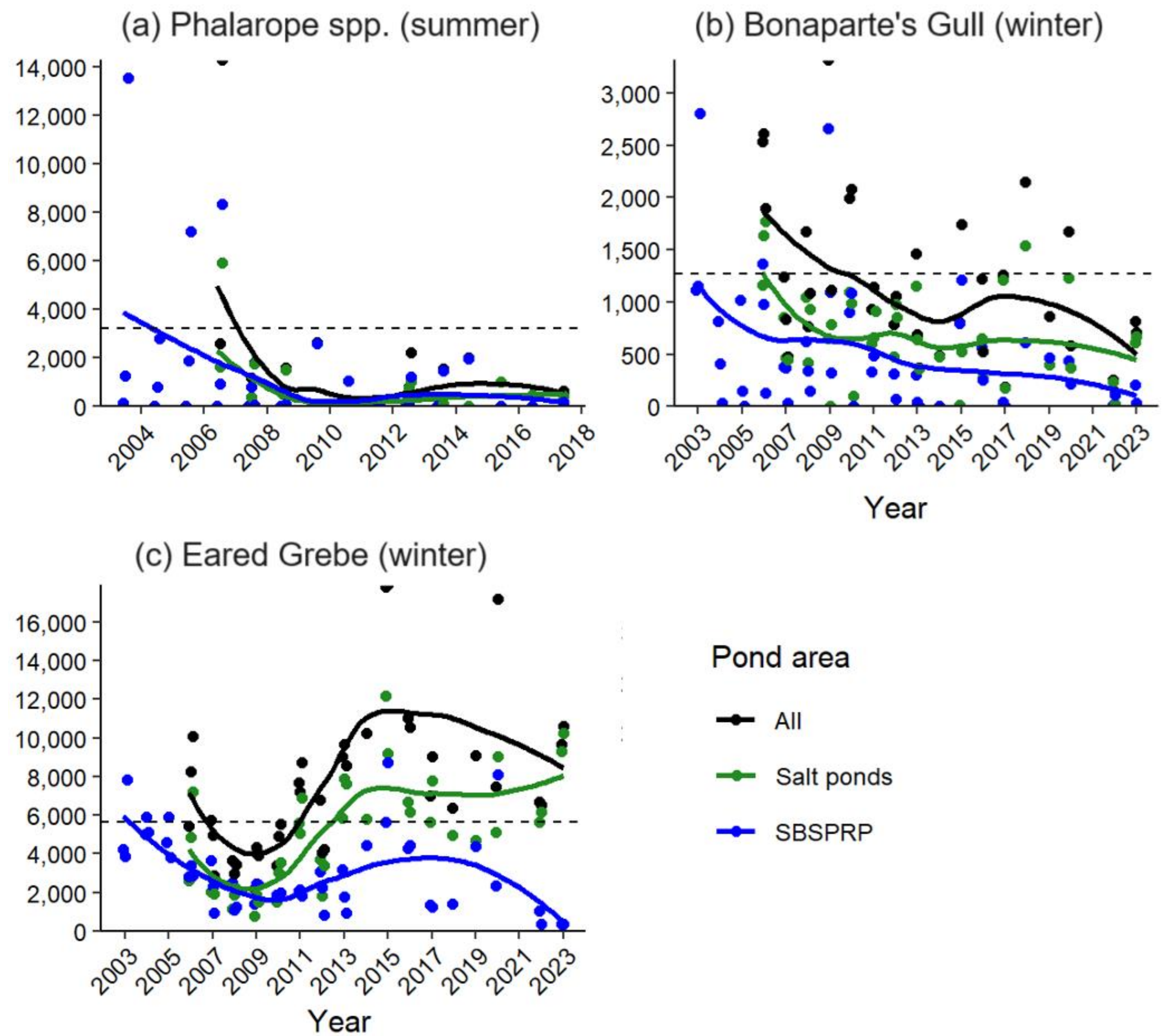

Figure 1. Counts of (a) phalaropes, (b) Bonaparte's Gulls and (c) Eared Grebe during peak seasons within the South Bay Salt Pond Restoration Project (SBSPRP) and salt production ponds. Lines represent LOESS curves and the dashed lines denote SBSPRP targets or baseline values (average counts from 2005–2007; Table 1). Adapted from Van Schmidt et al. (2023).

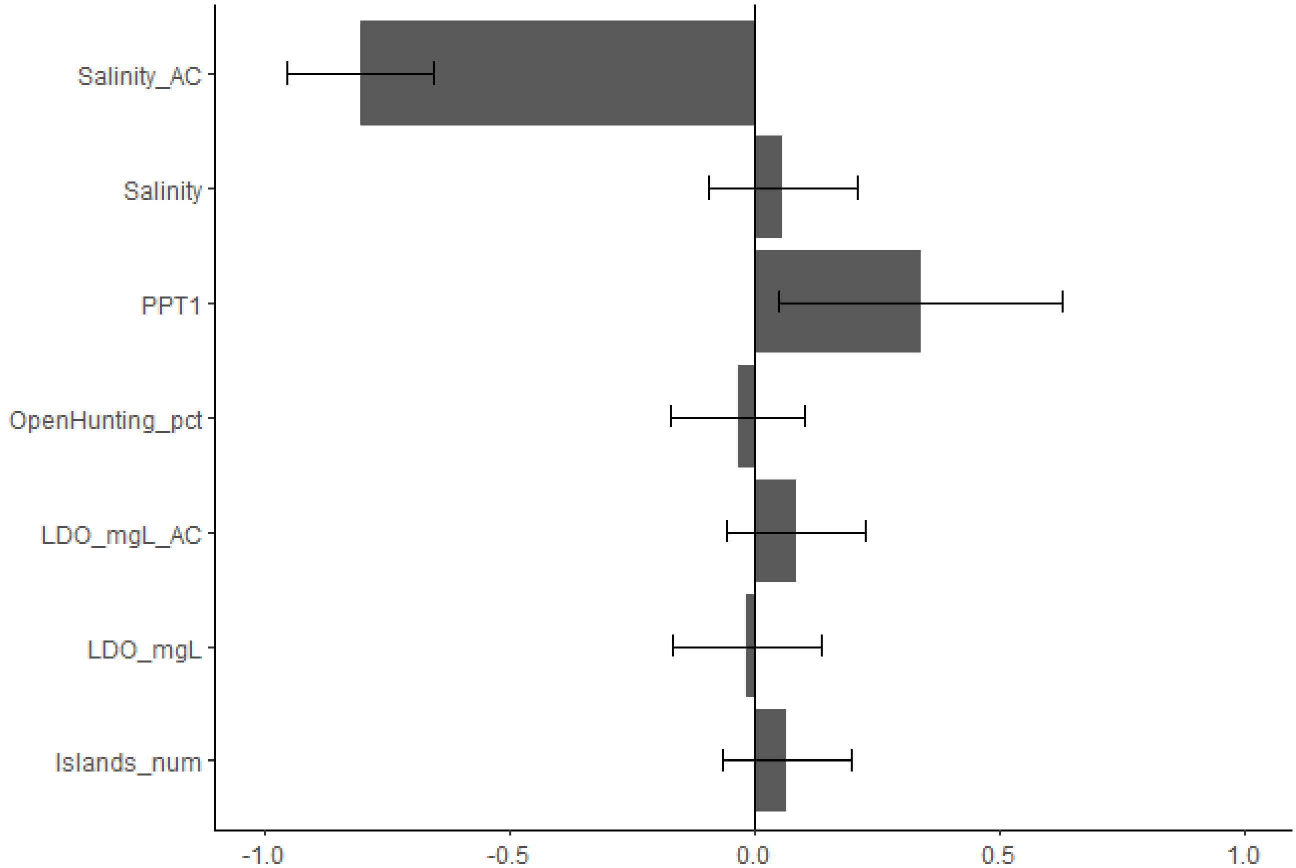

Figure 2. Fixed effects for the top model for interannual change in abundance of dabbling ducks during winter (Dec–Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003–2023 (n = 1,172). Dark bars show coefficients and error bars show 95% confidence intervals.

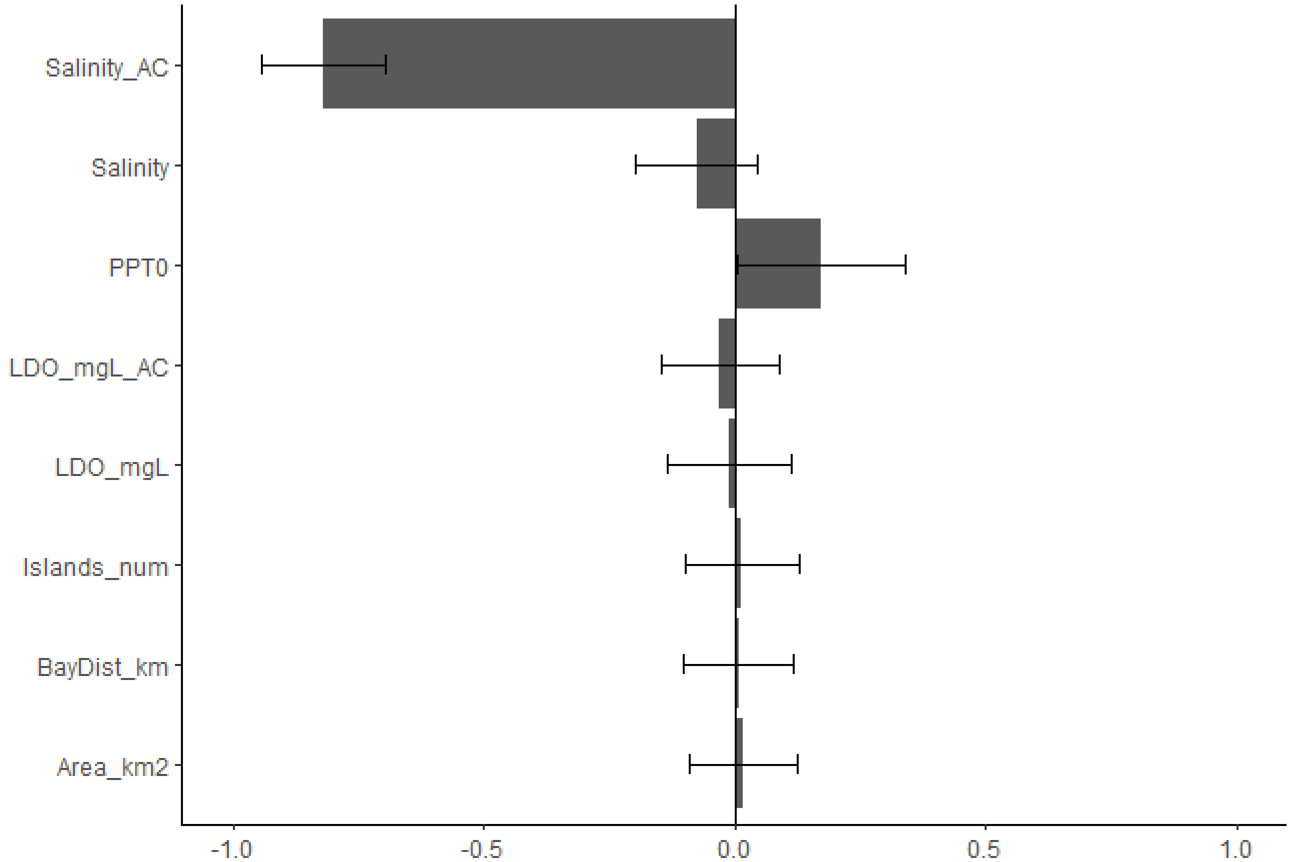

Figure 3. Fixed effects for the top model for interannual change in abundance of diving ducks during winter (Dec–Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003–2023 (n = 1,227). Dark bars show coefficients and error bars show 95% confidence intervals.

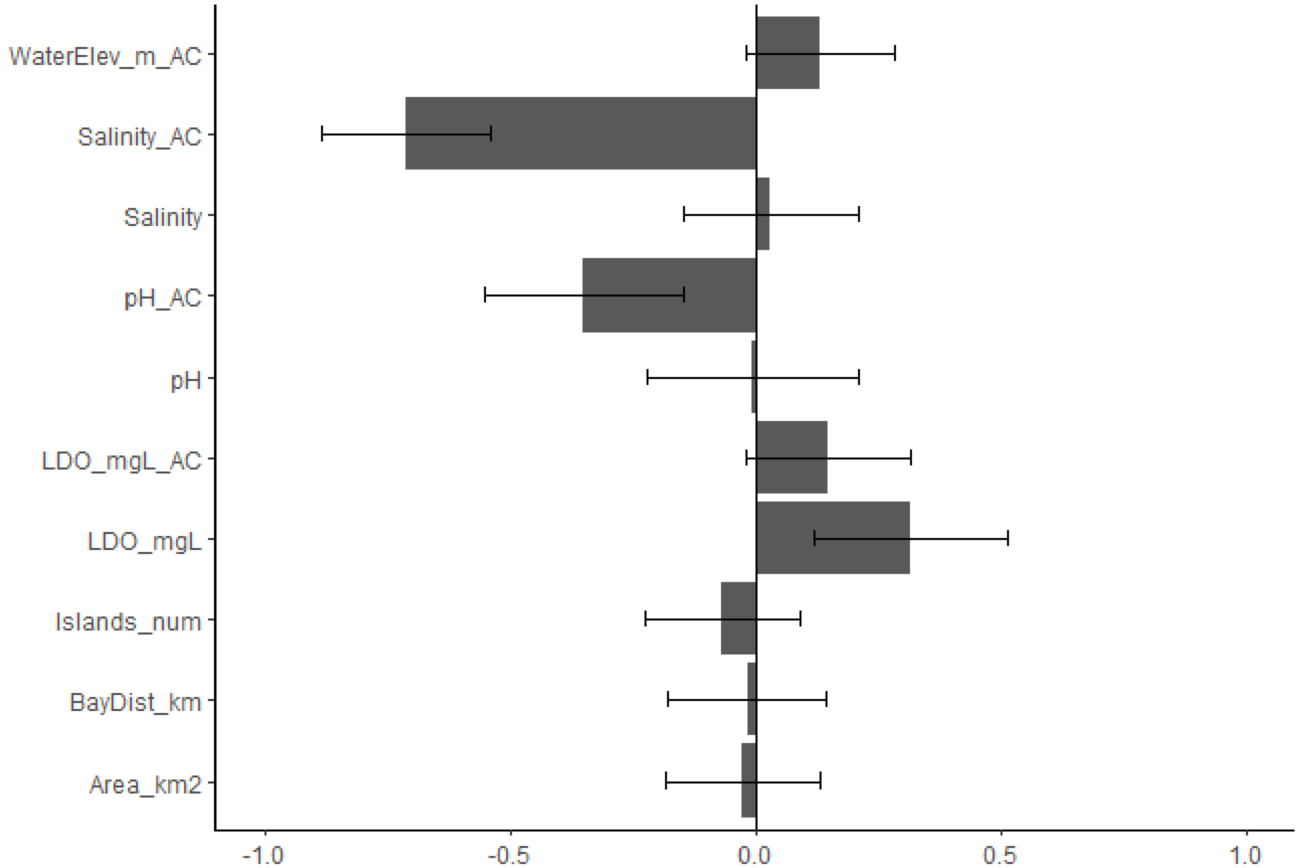

Figure 4. Fixed effects for the top model for interannual change in abundance of Ruddy Ducks during winter (Dec–Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003–2023 (n = 1,016). Dark bars show coefficients and error bars show 95% confidence intervals.

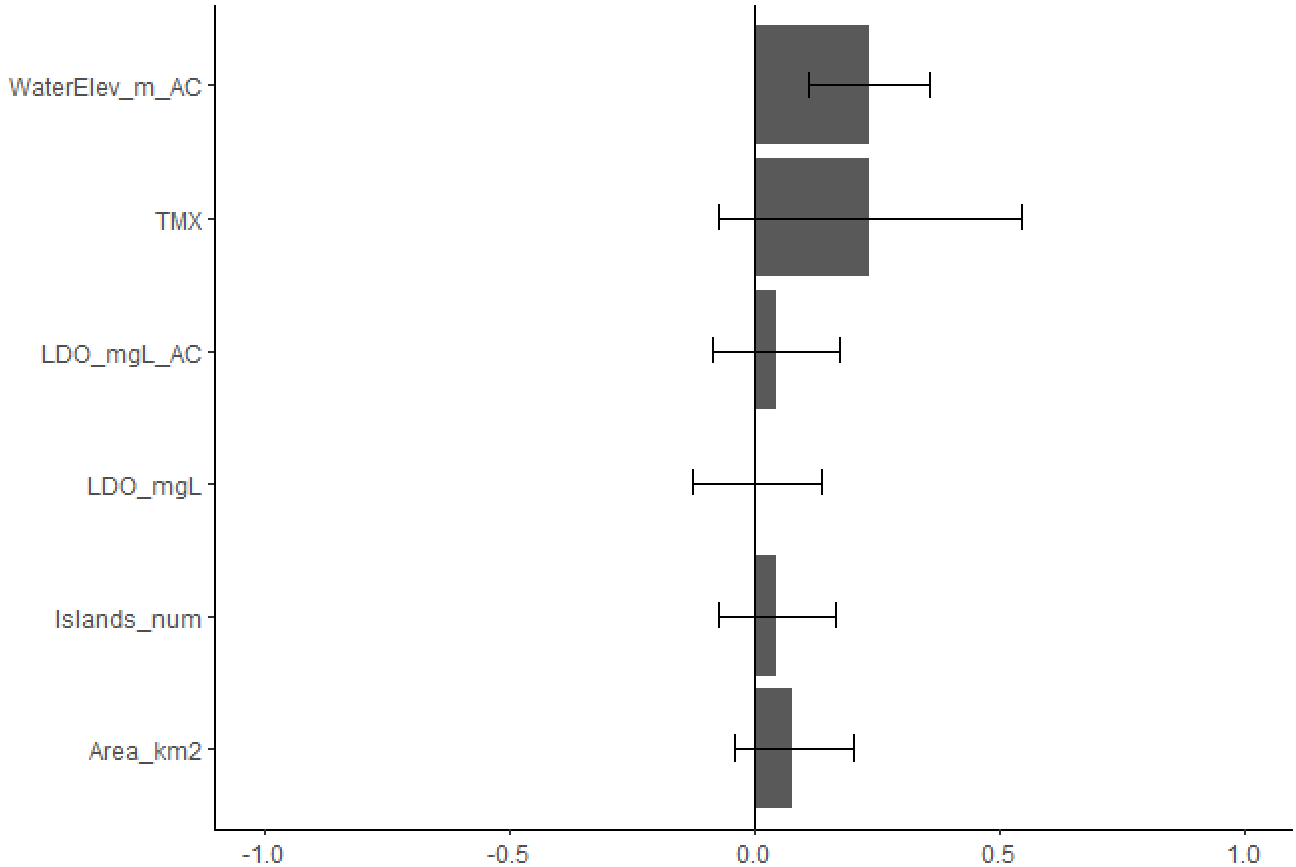

Figure 5. Fixed effects for the top model for interannual change in abundance of Eared Grebes during winter (Dec–Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003–2023 (*n* = 969). Dark bars show coefficients and error bars show 95% confidence intervals.

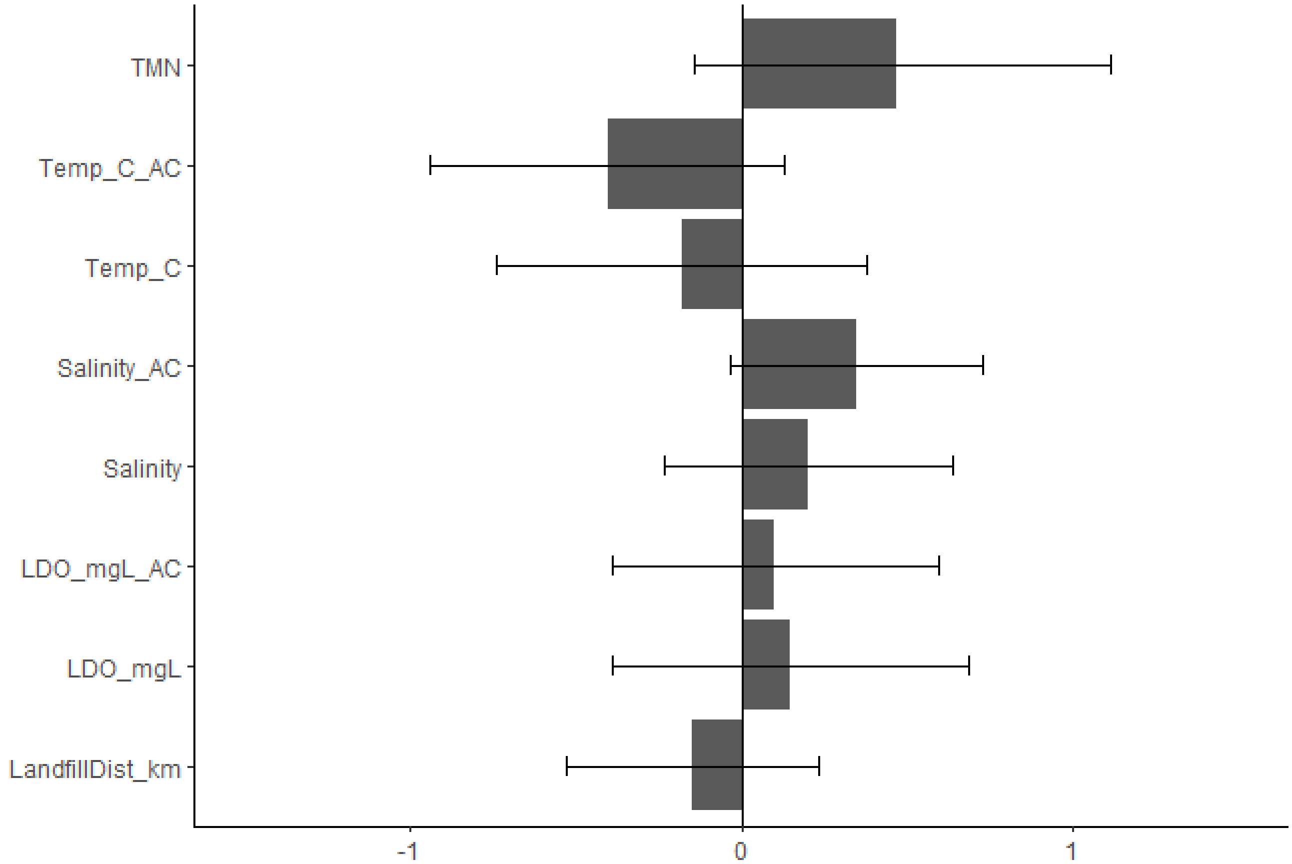

Figure 6. Fixed effects for the top model for interannual change in abundance of Bonaparte's Gulls during winter (Dec–Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003–2023 (n = 360). Dark bars show coefficients and error bars show 95% confidence intervals.

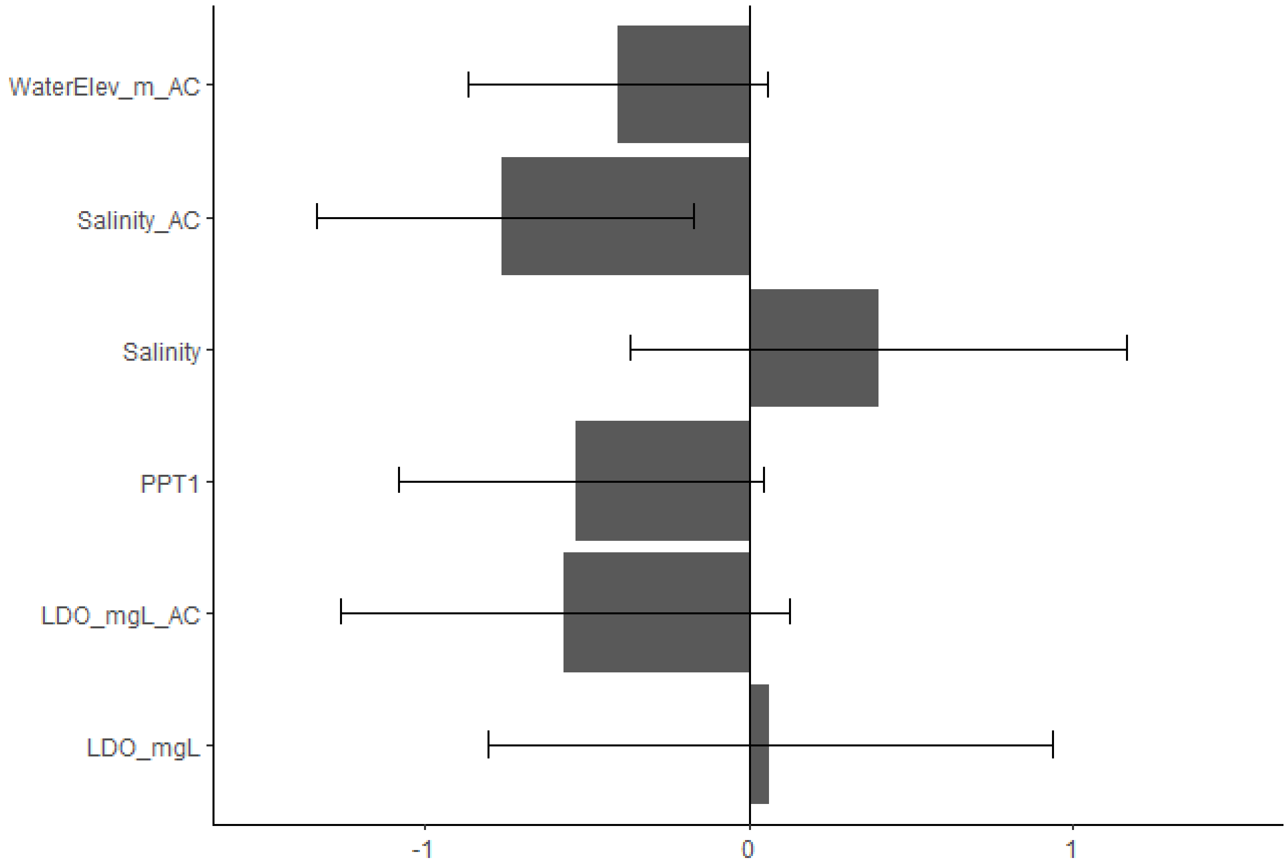

Figure 7. Fixed effects for the top model for interannual change in abundance of phalaropes during summer (Jun–Aug) in current and former salt ponds in South San Francisco Bay, California, water year 2003–2023 (*n* = 233). Dark bars show coefficients and error bars show 95% confidence intervals.

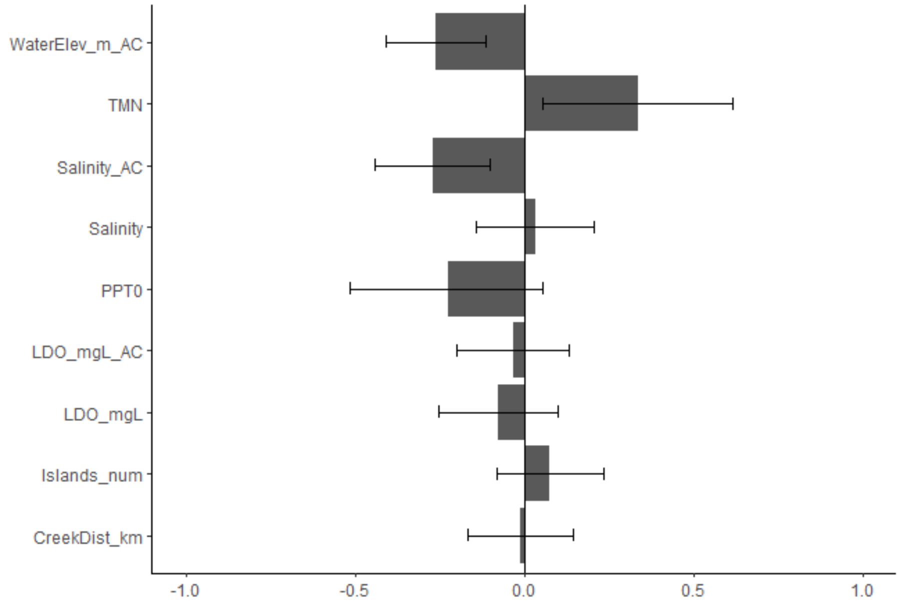

Figure 8. Fixed effects for the top model for interannual change in abundance of medium shorebirds during winter (Dec–Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003–2023 (n = 1,207). Dark bars show coefficients and error bars show 95% confidence intervals.

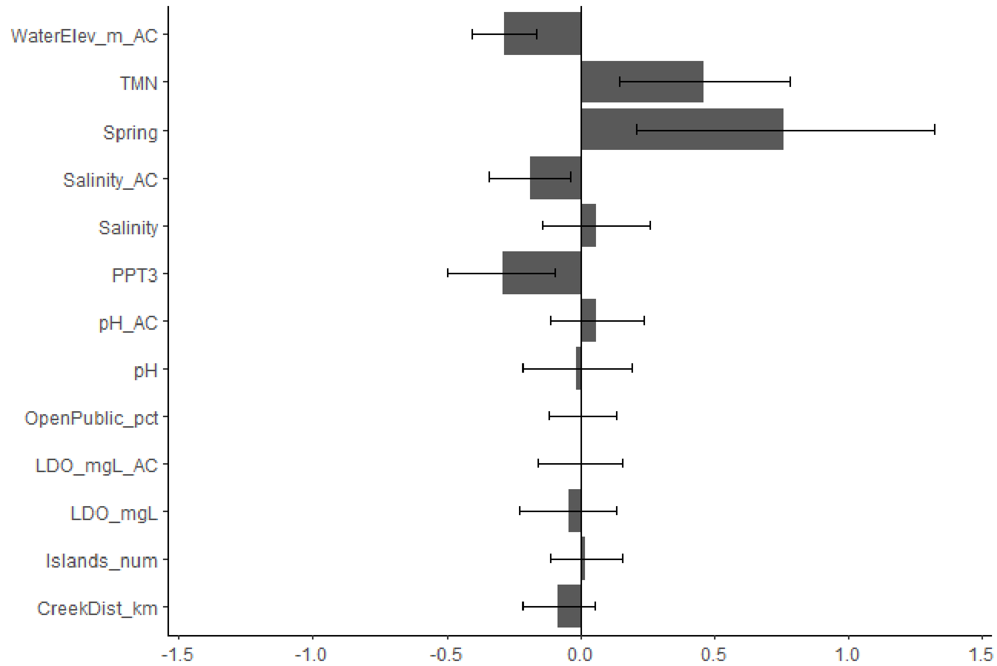

Figure 9. Fixed effects for the top model for interannual change in abundance of small shorebirds during fall (Sept–Nov) or spring (Mar–May) in current and former salt ponds in South San Francisco Bay, California, water year 2003–2023 (*n* = 2,014). Dark bars show coefficients and error bars show 95% confidence intervals.

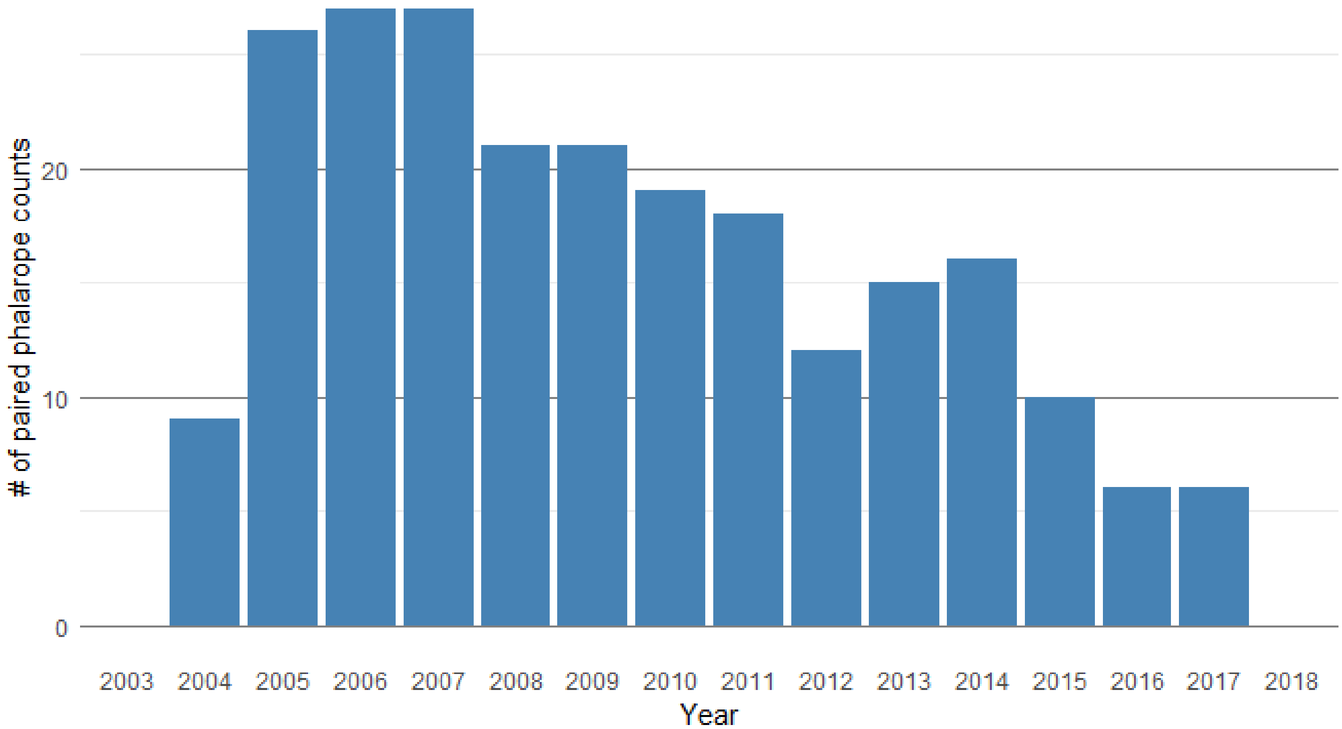

Figure 10. Number of paired counts of phalarope (year-to-year change) within the South San Francisco Bay in each summer of the study, until summer surveys were ceased in 2018. Data sampling was highest in the earliest years of the study, when the most dramatic changes in salinity occurred as part of the Initial Stewardship Plan.

Appendix 1

Table A1.1. Species assignments to foraging guilds. Guilds included dabblers, divers, Eared Grebes, fisheaters, gulls, herons, medium shorebirds, phalaropes, small shorebirds, and terns.

Common Name	Species Code	Scientific Name	Guild
American Coot	AMCO	*Fulica americana*	DABBLER
American Green-winged Teal	AGWT	*Anas crecca*	DABBLER
American Wigeon	AMWI	*Anas americana*	DABBLER
Blue-winged Teal	BWTE	*Anas discors*	DABBLER
Cinnamon Teal	CITE	*Anas cyanoptera*	DABBLER
Common Moorhen	COMO	*Gallinula chloropus*	DABBLER
Domestic Mallard	DOMA	*Anas spp*	DABBLER
Eurasian Wigeon	EUWI	*Anas penelope*	DABBLER
Gadwall	GADW	*Anas strepera*	DABBLER
Green-winged Teal	GWTE	*Anas crecca*	DABBLER
Long-tailed Duck	LTDU	*Clangula hyemalis*	DABBLER
Mallard	MALL	*Anas platyrhynchos*	DABBLER
Northern Pintail	NOPI	*Anas acuta*	DABBLER
Northern Shoveler	NSHO	*Anas clypeata*	DABBLER
Unidentified dabbling duck	DABB	*dabbling duck spp.*	DABBLER
Barrow's Goldeneye	BAGO	*Bucephala islandica*	DIVER
Bufflehead	BUFF	*Bucephala albeola*	DIVER
Canvasback	CANV	*Aythya valisineria*	DIVER
Common Goldeneye	COGO	*Bucephala clangula*	DIVER
Greater Scaup	GRSC	*Aythya marila*	DIVER
Lesser Scaup	LESC	*Aythya affinis*	DIVER
Redhead	REDH	*Aythya americana*	DIVER
Ring-necked Duck	RNDU	*Aythya collaris*	DIVER
Ruddy Duck	RUDU	*Oxyura jamaicensis*	DIVER
Surf Scoter	SUSC	*Melanitta perspicillata*	DIVER

Common Name	Species Code	Scientific Name	Guild
Tufted Duck	TUDU	*Aythya fuligula*	DIVER
Unidentified diving duck	DIVE	*diving duck spp.*	DIVER
Unidentified scaup	SCAU	*Aythya spp.*	DIVER
White-winged scoter	WWSC	*Melanitta fusca*	DIVER
Eared Grebe	EAGR	*Podiceps nigricollis*	EAREDGR
American White Pelican	AWPE	*Pelecanus erythrorhynchos*	FISHEAT
Belted Kingfisher	BEKI	*Ceryle alcyon*	FISHEAT
Black Skimmer	BLSK	*Rhynchops niger*	FISHEAT
Brown Booby	BRBO	*Sula leucogaster*	FISHEAT
Brown Pelican	BRPE	*Pelecanus occidentalis*	FISHEAT
Clark's Grebe	CLGR	*Aechmophorus clarkii*	FISHEAT
Common Loon	COLO	*Gavia immer*	FISHEAT
Common Merganser	COME	*Mergus merganser*	FISHEAT
Double-crested Cormorant	DCCO	*Phalacrocorax auritus*	FISHEAT
Hooded Merganser	HOME	*Lophodytes cucullatus*	FISHEAT
Horned Grebe	HOGR	*Podiceps auritus*	FISHEAT
Long-tailed Jaeger	LTJA	*Stercorarius longicaudus*	FISHEAT
Pacific Loon	PALO	*Gavia pacifica*	FISHEAT
Pelagic Cormorant	PECO	*Phalacrocorax pelagicus*	FISHEAT
Pied-billed Grebe	PBGR	*Podilymbus podiceps*	FISHEAT
Red-breasted Merganser	RBME	*Mergus serrator*	FISHEAT
Red-necked Grebe	RNGR	*Podiceps grisegena*	FISHEAT
Red-throated Loon	RTLO	*Gavia stellata*	FISHEAT
Unidentified Cormorant	CORM	*Phalacrocorax* spp.	FISHEAT
Unidentified grebe	GREBE	N/A	FISHEAT
Western Grebe	WEGR	*Aechmophorus occidentalis*	FISHEAT
Western Grebe or Clark's Grebe	WEGR/CLGR	*Aechmophorus* spp.	FISHEAT

Common Name	**Species Code**	**Scientific Name**	**Guild**
Bonaparte's Gull	BOGU	*Larus philadelphia*	GULL
California Gull	CAGU	*Larus californicus*	GULL
California Gull or Ring-billed Gull	CAGU/RBGU	*Larus* spp.	GULL
Franklin's Gull	FRGU	*Larus pipixcan*	GULL
Glaucous-winged Gull	GWGU	*Larus glaucescens*	GULL
Glaucous Gull	GLGU	*Larus hyperboreus*	GULL
Herring Gull	HERG	*Larus argentatus*	GULL
Mew Gull	MEGU	*Larus canus*	GULL
Ring-billed Gull	RBGU	*Larus delawarensis*	GULL
Sabine's Gull	SAGU	*Xena sabini*	GULL
Slaty-backed Gull	SBGU	*Larus schistisagus*	GULL
Thayer's Gull	THGU	*Larus thayeri*	GULL
Unidentified gull	GULL	*Larus* spp.	GULL
Western Gull	WEGU	*Larus occidentalis*	GULL
American Bittern	AMBI	*Botarus lentiginosus*	HERON
Black-crowned Night-Heron	BCNH	*Nycticorax nycticorax*	HERON
Cattle Egret	CAEG	*Bubulcus ibis*	HERON
Great Blue Heron	GBHE	*Ardea herodias*	HERON
Great Egret	GREG	*Ardea alba*	HERON
Green Heron	GRHE	*Butorides virescens*	HERON
Little Blue Heron	LBHE	*Egretta caerulea*	HERON
Snowy Egret	SNEG	*Egretta thula*	HERON
White-faced Ibis	WFIB	*Plegadis chihi*	HERON
American Avocet	AMAV	*Recurvirostra americana*	MEDSHORE
Black-bellied Plover	BBPL	*Pluvialis squatarola*	MEDSHORE
Black-necked Stilt	BNST	*Himantopus mexicanus*	MEDSHORE
Black Oystercatcher	BLOY	*Haematopus bachmani*	MEDSHORE

Common Name	Species Code	Scientific Name	Guild
Black Turnstone	BLTU	*Arenaria melanocephala*	MEDSHORE
Common Snipe	COSN	*Gallinago gallinago*	MEDSHORE
Golden Plover	GOPL	*Pluvialis spp.*	MEDSHORE
Greater Yellowlegs	GRYE	*Tringa melanoleuca*	MEDSHORE
Killdeer	KILL	*Charadrius vociferus*	MEDSHORE
Lesser Yellowlegs	LEYE	*Tringa flavipes*	MEDSHORE
Long-billed Curlew	LBCU	*Numenius americanus*	MEDSHORE
Marbled Godwit	MAGO	*Limosa fedoa*	MEDSHORE
Pacific Golden-Plover	PAGP	*Pluvialis fulva*	MEDSHORE
Red Knot	REKN	*Calidris canutus*	MEDSHORE
Ruddy Turnstone	RUTU	*Arenaria interpres*	MEDSHORE
Ruff	RUFF	*Philomachus pugnax*	MEDSHORE
Spotted Redshank	SPRE	*Tringa erythropus*	MEDSHORE
Stilt Sandpiper	STSA	*Calidris himantopus*	MEDSHORE
Surfbird	SURF	*Aphriza virgata*	MEDSHORE
Unidentifed yellowlegs	YELL	*Tringa spp.*	MEDSHORE
Unidentified medium shorebird	SHOR	*med shorebird spp.*	MEDSHORE
Wandering Tattler	WATA	*Tringa incana*	MEDSHORE
Whimbrel	WHIM	*Numenius phaeopus*	MEDSHORE
Willet	WILL	*Catoptrophorus semipalmatus*	MEDSHORE
Red-necked Phalarope	RNPH	*Phalaropus lobatus*	PHAL
Red Phalarope	REPH	*Phalaropus fulicaria*	PHAL
Unidentified phalarope	PHAL	*Phalaropus spp.*	PHAL
Wilson's Phalarope	WIPH	*Phalaropus tricolor*	PHAL
Baird's Sandpiper	BASA	*Calidris bairdii*	SMSHORE
Dunlin	DUNL	*Calidris alpina*	SMSHORE
Least Sandpiper	LESA	*Calidris minutilla*	SMSHORE

Common Name	**Species Code**	**Scientific Name**	**Guild**
Long-billed Dowitcher	LBDO	*Limnodromus scolopaceus*	SMSHORE
Pectoral Sandpiper	PESA	*Calidris melanotos*	SMSHORE
Sanderling	SAND	*Calidris alba*	SMSHORE
Semipalmated Plover	SEPL	*Charadrius semipalmatus*	SMSHORE
Semipalmated Sandpiper	SESA	*Calidris pusilla*	SMSHORE
Short-billed Dowitcher	SBDO	*Limnodromus griseus*	SMSHORE
Snowy Plover	SNPL	*Charadrius alexandrinus*	SMSHORE
Spotted Sandpiper	SPSA	*Actitis macularia*	SMSHORE
Unidentified Dowitcher	DOWI	*Limnodromus* spp.	SMSHORE
Unidentified peeps	PEEP	*Calidris* spp.	SMSHORE
Western Sandpiper	WESA	*Calidris mauri*	SMSHORE
Western Sandpiper or Dunlin	WESA/DUNL	*Calidris* spp.	SMSHORE
Western Sandpiper or Least Sandpiper	WESA/LESA	*Calidris* spp.	SMSHORE
Arctic Tern	ARTE	*Sterna paradiaea*	TERN
Black Tern	BLTE	*Chlidonias niger*	TERN
Caspian Tern	CATE	*Sterna caspia*	TERN
Common Tern	COTE	*Sterna hirundo*	TERN
Elegant Tern	ELTE	*Sterna elegans*	TERN
Forster's Tern	FOTE	*Sterna forsteri*	TERN
Least Tern	LETE	*Sterna antillarum browni*	TERN
Unidentified tern	TERN	*Sterna* spp.	TERN

Table A1.2. Table of estimated dates of levee breaches and changes to muted tidal hydrology, which were used in the model to estimate *Muted* and *Breached* covariates. Dates were estimated from published reports and aerial imagery, and may have inaccuracies.

Pond	Breached date	Muted tidal date
A16		
A17	10/31/2012	3/31/2005
A18		
A19	3/7/2006	
A20	3/12/2006	
A21	3/29/2006	
E1		8/11/2004
E2		8/11/2004
E4		8/11/2004
E5		8/11/2004
E6		8/11/2004
E7		8/11/2004
E10		
A6	12/6/2010	
A5		4/22/2011
A7		4/22/2011
A8		4/22/2011
A8W		4/22/2011
A8S		4/22/2011
E8A		4/29/2005
E8AE	9/13/2011	
E8AW	9/13/2011	
E8XS	3/1/2011	
E8SN	7/1/2011	
E8X	3/1/2011	
E10X	10/26/2006	

Table A1.2. Table of estimated levee breach and muted tidal management dates used in the model to estimate *Muted* and *Breached* covariates. Dates were estimated from published reports and aerial imagery, and may be inaccurate .

Appendix 2

Table A2.1. Model selection table for base model of pond characteristics related to interannual change in abundance of Bonaparte's Gull during winter (Dec-Feb) in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only models with AICc weight ≥0.01 are shown.

Formula	K	ΔAICc	AICc	-2LogLike	w_i
Abun_PC ~ LandfillDist_km	4	0.00	2114.37	-1053.13	0.08
Abun_PC ~ CreekDist_km + LandfillDist_km	5	1.03	2115.40	-1052.62	0.05
Abun_PC ~ LandfillDist_km + OpenHunting_pct	5	1.58	2115.94	-1052.90	0.04
Abun_PC ~ LandfillDist_km + OpenPublic_pct	5	1.62	2115.98	-1052.92	0.04
Abun_PC ~ Area_km2 + LandfillDist_km	5	1.66	2116.03	-1052.94	0.04
Abun_PC ~ Islands_num + LandfillDist_km	5	1.75	2116.12	-1052.98	0.03
Abun_PC ~ BayDist_km + LandfillDist_km	5	2.04	2116.41	-1053.13	0.03
Abun_PC ~ Area_km2 + CreekDist_km + LandfillDist_km	6	2.54	2116.90	-1052.35	0.02
Abun_PC ~ Area_km2	4	2.59	2116.95	-1054.43	0.02
Abun_PC ~ 1	3	2.59	2116.96	-1055.45	0.02
Abun_PC ~ CreekDist_km + Islands_num + LandfillDist_km	6	2.64	2117.01	-1052.40	0.02
Abun_PC ~ CreekDist_km + LandfillDist_km + OpenPublic_pct	6	2.74	2117.11	-1052.45	0.02
Abun_PC ~ CreekDist_km + LandfillDist_km + OpenHunting_pct	6	2.81	2117.18	-1052.48	0.02
Abun_PC ~ BayDist_km + CreekDist_km + LandfillDist_km	6	3.09	2117.46	-1052.62	0.02
Abun_PC ~ Islands_num + LandfillDist_km + OpenHunting_pct	6	3.27	2117.63	-1052.71	0.02
Abun_PC ~ LandfillDist_km + OpenHunting_pct + OpenPublic_pct	6	3.41	2117.78	-1052.78	0.01
Abun_PC ~ Area_km2 + LandfillDist_km + OpenHunting_pct	6	3.42	2117.78	-1052.79	0.01
Abun_PC ~ Area_km2 + LandfillDist_km + OpenPublic_pct	6	3.50	2117.87	-1052.83	0.01
Abun_PC ~ Area_km2 + Islands_num + LandfillDist_km	6	3.52	2117.88	-1052.84	0.01
Abun_PC ~ Islands_num + LandfillDist_km + OpenPublic_pct	6	3.52	2117.89	-1052.84	0.01

Formula	K	ΔAICc	AICc	-2LogLike	w_i
Abun_PC ~ BayDist_km + LandfillDist_km + OpenHunting_pct	6	3.63	2118.00	-1052.89	0.01
Abun_PC ~ Area_km2 + BayDist_km + LandfillDist_km	6	3.64	2118.00	-1052.89	0.01
Abun_PC ~ BayDist_km + LandfillDist_km + OpenPublic_pct	6	3.65	2118.01	-1052.90	0.01
Abun_PC ~ Area_km2 + BayDist_km	5	3.71	2118.07	-1053.96	0.01
Abun_PC ~ BayDist_km + Islands_num + LandfillDist_km	6	3.79	2118.16	-1052.97	0.01
Abun_PC ~ Islands_num	4	3.90	2118.26	-1055.08	0.01
Abun_PC ~ BayDist_km	4	4.09	2118.46	-1055.18	0.01

Table A2.2. Model selection table for base model of pond characteristics related to interannual change in abundance of phalaropes during summer (Jun-Aug) in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023. Model selection was carried out based on corrected Akaike Information Criterion (AICc); only models with AICc weight ≥0.01 are shown.

	Formula	K	ΔAICc	AICc	-2LogLike	w_i
1	Abun_PC ~ 1	3	0.00	1460.66	-727.28	0.11
4	Abun_PC ~ CreekDist_km	4	1.58	1462.24	-727.04	0.05
8	Abun_PC ~ OpenPublic_pct	4	1.84	1462.49	-727.17	0.04
2	Abun_PC ~ Area_km2	4	2.02	1462.67	-727.26	0.04
3	Abun_PC ~ BayDist_km	4	2.04	1462.70	-727.27	0.04
6	Abun_PC ~ LandfillDist_km	4	2.05	1462.71	-727.28	0.04
7	Abun_PC ~ OpenHunting_pct	4	2.06	1462.71	-727.28	0.04
5	Abun_PC ~ Islands_num	4	2.06	1462.72	-727.28	0.04
10	Abun_PC ~ Area_km2 + CreekDist_km	5	3.34	1464.00	-726.88	0.02
23	Abun_PC ~ CreekDist_km + OpenPublic_pct	5	3.50	1464.16	-726.96	0.02
21	Abun_PC ~ CreekDist_km + LandfillDist_km	5	3.61	1464.26	-727.02	0.02
20	Abun_PC ~ CreekDist_km + Islands_num	5	3.64	1464.30	-727.03	0.02
22	Abun_PC ~ CreekDist_km + OpenHunting_pct	5	3.66	1464.31	-727.04	0.02
15	Abun_PC ~ BayDist_km + CreekDist_km	5	3.66	1464.31	-727.04	0.02
28	Abun_PC ~ LandfillDist_km + OpenPublic_pct	5	3.74	1464.39	-727.08	0.02
19	Abun_PC ~ BayDist_km + OpenPublic_pct	5	3.78	1464.44	-727.10	0.02
14	Abun_PC ~ Area_km2 + OpenPublic_pct	5	3.88	1464.53	-727.15	0.02
29	Abun_PC ~ OpenHunting_pct + OpenPublic_pct	5	3.89	1464.55	-727.16	0.02
26	Abun_PC ~ Islands_num + OpenPublic_pct	5	3.91	1464.57	-727.17	0.01
9	Abun_PC ~ Area_km2 + BayDist_km	5	4.06	1464.72	-727.24	0.01
13	Abun_PC ~ Area_km2 + OpenHunting_pct	5	4.09	1464.75	-727.26	0.01
11	Abun_PC ~ Area_km2 + Islands_num	5	4.09	1464.75	-727.26	0.01

	Formula	K	ΔAICc	AICc	-2LogLike	w_i
12	Abun_PC ~ Area_km2 + LandfillDist_km	5	4.09	1464.75	-727.26	0.01
27	Abun_PC ~ LandfillDist_km + OpenHunting_pct	5	4.10	1464.76	-727.26	0.01
18	Abun_PC ~ BayDist_km + OpenHunting_pct	5	4.11	1464.77	-727.27	0.01
16	Abun_PC ~ BayDist_km + Islands_num	5	4.12	1464.77	-727.27	0.01
17	Abun_PC ~ BayDist_km + LandfillDist_km	5	4.12	1464.78	-727.27	0.01
24	Abun_PC ~ Islands_num + LandfillDist_km	5	4.13	1464.79	-727.28	0.01
25	Abun_PC ~ Islands_num + OpenHunting_pct	5	4.13	1464.79	-727.28	0.01

Table A2.3. Top model for interannual change in abundance of dabbling ducks in current and former salt ponds in South San Francisco Bay, California, water year 2003–2023.

	Estimate	SE	t-value	Lower 95% CI	Upper 95% CI
(Intercept)	0.2558	0.1514	1.6891	-0.0533	0.5726
Islands_num	0.0653	0.0666	0.9803	-0.0653	0.1959
OpenHunting_pct	-0.0334	0.0696	-0.4798	-0.1698	0.1031
LDO_mgL	-0.0146	0.0768	-0.1908	-0.1654	0.1361
LDO_mgL_AC	0.0859	0.0725	1.1841	-0.0564	0.2281
PPT1	0.3394	0.1410	2.4071	0.0520	0.6265
Salinity	0.0592	0.0776	0.7627	-0.0931	0.2115
Salinity_AC	-0.8040	0.0755	-10.6468	-0.9525	-0.6555

Table A2.4. Top model for interannual change in abundance of diving ducks in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023.

	Estimate	SE	t-value	Lower 95% CI	Upper 95% CI
(Intercept)	0.0801	0.0810	0.9881	-0.0875	0.2466
BayDist_km	0.0086	0.0558	0.1549	-0.1008	0.1181
Islands_num	0.0146	0.0569	0.2573	-0.0969	0.1262
Area_km2	0.0162	0.0552	0.2936	-0.0921	0.1245
LDO_mgL	-0.0102	0.0631	-0.1609	-0.1341	0.1138
LDO_mgL_AC	-0.0299	0.0601	-0.4975	-0.1479	0.0881
PPT0	0.1700	0.0814	2.0882	0.0055	0.3404
Salinity	-0.0761	0.0622	-1.2228	-0.1982	0.0460
Salinity_AC	-0.8185	0.0616	-13.2806	-0.9405	-0.6965

Table A2.5. Top model for interannual change in abundance of Ruddy Ducks in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023.

	Estimate	**SE**	**t-value**	**Lower 95% CI**	**Upper 95% CI**
(Intercept)	0.0910	0.1606	0.5666	-0.2439	0.4186
BayDist_km	-0.0176	0.0816	-0.2154	-0.1777	0.1425
Islands_num	-0.0677	0.0801	-0.8462	-0.2248	0.0893
Area_km2	-0.0270	0.0804	-0.3358	-0.1847	0.1308
LDO_mgL	0.3175	0.1001	3.1710	0.1208	0.5142
LDO_mgL_AC	0.1488	0.0858	1.7335	-0.0196	0.3173
pH	-0.0063	0.1096	-0.0574	-0.2213	0.2088
pH_AC	-0.3496	0.1032	-3.3874	-0.5525	-0.1469
Salinity	0.0305	0.0909	0.3359	-0.1478	0.2089
Salinity_AC	-0.7119	0.0882	-8.0762	-0.8856	-0.5382
WaterElev_m_AC	0.1316	0.0775	1.6982	-0.0211	0.2842

Table A2.6. Top model for interannual change in abundance of Eared Grebes in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023.

	Estimate	SE	t-value	Lower 95% CI	Upper 95% CI
(Intercept)	-0.1108	0.1551	-0.7146	-0.4296	0.2073
Islands_num	0.0467	0.0607	0.7699	-0.0723	0.1657
Area_km2	0.0803	0.0622	1.2908	-0.0418	0.2025
LDO_mgL	0.0039	0.0666	0.0587	-0.1268	0.1346
LDO_mgL_AC	0.0446	0.0651	0.6849	-0.0832	0.1725
TMX	0.2348	0.1528	1.5372	-0.0745	0.5464
WaterElev_m_AC	0.2354	0.0621	3.7912	0.1131	0.3577

Table A2.7. Top model for interannual change in abundance of Bonaparte's Gulls in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023.

	Estimate	SE	t-value	Lower 95% CI	Upper 95% CI
(Intercept)	0.0446	0.2912	0.1533	-0.5575	0.6690
LandfillDist_km	-0.1479	0.1929	-0.7664	-0.5271	0.2313
LDO_mgL	0.1452	0.2723	0.5333	-0.3920	0.6815
LDO_mgL_AC	0.1004	0.2500	0.4015	-0.3924	0.5943
Salinity	0.2018	0.2217	0.9105	-0.2339	0.6374
Salinity_AC	0.3466	0.1907	1.8171	-0.0334	0.7260
Temp_C	-0.1826	0.2830	-0.6452	-0.7393	0.3784
Temp_C_AC	-0.4046	0.2706	-1.4955	-0.9385	0.1273
TMN	0.4674	0.2999	1.5588	-0.1440	1.1100

Table A2.8. Top model for interannual change in abundance of phalaropes in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023.

	Estimate	SE	t-value	Lower 95% CI	Upper 95% CI
(Intercept)	-0.4707	0.2617	-1.7988	-1.0425	0.0849
LDO_mgL	0.0648	0.4364	0.1484	-0.8032	0.9331
LDO_mgL_AC	-0.5684	0.3510	-1.6194	-1.2606	0.1224
PPT1	-0.5362	0.2606	-2.0575	-1.0803	0.0464
Salinity	0.3993	0.3860	1.0344	-0.3677	1.1660
Salinity_AC	-0.7592	0.2755	-2.7559	-1.3299	-0.1717
WaterElev_m_AC	-0.4026	0.2344	-1.7175	-0.8643	0.0588

Table A2.9. Top model for interannual change in abundance of medium shorebirds in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023.

	Estimate	SE	t-value	Lower 95% CI	Upper 95% CI
(Intercept)	0.0876	0.1317	0.6652	-0.1828	0.3649
CreekDist_km	-0.0100	0.0795	-0.1259	-0.1659	0.1458
Islands_num	0.0767	0.0799	0.9599	-0.0800	0.2333
LDO_mgL	-0.0775	0.0893	-0.8678	-0.2534	0.0985
LDO_mgL_AC	-0.0337	0.0840	-0.4010	-0.1986	0.1312
PPT0	-0.2255	0.1375	-1.6398	-0.5161	0.0531
Salinity	0.0317	0.0888	0.3569	-0.1427	0.2061
Salinity_AC	-0.2704	0.0862	-3.1369	-0.4397	-0.1010
TMN	0.3375	0.1356	2.4889	0.0540	0.6166
WaterElev_m_AC	-0.2602	0.0750	-3.4692	-0.4075	-0.1129

Table A2.10. Top model for interannual change in abundance of small shorebirds in current and former salt ponds in South San Francisco Bay, California, water year 2003-2023.

	Estimate	SE	t-value	Lower 95% CI	Upper 95% CI
(Intercept)	-0.2521	0.2017	-1.2495	-0.6595	0.1647
CreekDist_km	-0.0840	0.0683	-1.2293	-0.2180	0.0500
Islands_num	0.0201	0.0688	0.2917	-0.1149	0.1551
OpenPublic_pct	0.0046	0.0644	0.0709	-0.1218	0.1310
LDO_mgL	-0.0451	0.0921	-0.4895	-0.2258	0.1355
LDO_mgL_AC	-0.0013	0.0806	-0.0161	-0.1594	0.1568
pH	-0.0132	0.1051	-0.1253	-0.2194	0.1930
pH_AC	0.0608	0.0887	0.6858	-0.1131	0.2348
PPT3	-0.2922	0.0912	-3.2034	-0.4980	-0.0990
Salinity	0.0592	0.1016	0.5830	-0.1405	0.2590
Salinity_AC	-0.1901	0.0776	-2.4504	-0.3423	-0.0380
TMN	0.4578	0.1591	2.8781	0.1458	0.7789
WaterElev_m_AC	-0.2855	0.0623	-4.5863	-0.4076	-0.1635
Spring	0.7611	0.2812	2.7066	0.2097	1.3236