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Authors: Orr, Daniel, Bonis-Ericksen, Keilani, Bautista, Camila, and Jones, Andrea

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A Shrinking Inland Sea: Trends in Avian Populations and Habitats at the Salton Sea

Daniel Orr*, Keilani Bonis-Ericksen, Camila Bautista, and Andrea Jones

Audubon California, U.S.A.

*Corresponding author; E-mail: daniel.orr@audubon.org

Abstract.—The Salton Sea, a saline lake ecosystem in southern California that is critical for birds along the Pacific migratory pathway, is rapidly changing due to changes in water use and supply. This shrinking Sea supports over 300 species of birds and millions of migratory birds annually. Reduced water inflows since 2017 have lowered water levels and increased salinity, driving ecological change, yet the impacts to birds remain largely unexamined. We analyzed bird population trends (2016–2023), changes in the extent of preferred habitats under different water management regimes, the expansion of emerging vegetation on exposed lakebed, water quality, and macroinvertebrate diversity. Bird population trends reveal both winners and losers: populations of Northern Shovelers, Least and Western Sandpipers, and Black-necked Stilts increased, while populations of American White Pelicans, nonbreeding American Avocets, Double-crested Cormorants, and Ruddy Ducks declined. Increasing shorebird population trends aligned with the expansion of emerging wetland vegetation on exposed lakebed, while decreasing trends in deep water dependent species aligned with the loss of areas of preferred deep-water habitat. Water quality was associated with waterbird abundance, but the effect of biogeochemistry on birds and their prey are not well understood and warrant further research. These findings inform strategic management efforts to sustain and restore habitats at the Salton Sea, with the goal of achieving positive outcomes for birds as wetland and saline habitats across the Western United States remain threatened. *Received 8 May 2025, accepted 4 Dec 2025.*

Key words.—avian ecology, conservation, emergent wetlands, habitat change, migratory birds, saline lakes, Salton Sea, shorebirds, water management, waterbird trends

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Across western North America, and globally, human development, agriculture, and climate change are impacting water availability and threatening the extent of wetlands, rivers, lakes, and other water-dependent habitats. These habitats form part of larger migratory flyway networks that sustain global waterbird habitat by providing essential stopover, breeding, and wintering sites as birds move between northern and southern latitudes. Inland saline lakes and wetlands are especially important habitats in the arid West of the U.S., yet they are highly vulnerable to climate-driven changes. In this region, climate induced warming and human development have led to shrinking saline lakes and rising salinity (Grimm *et al.* 1997; Larson *et al.* 2016; Reis *et al.* 2017; Wilsey *et al.* 2017), as well as shifting wetland seasonality (e.g. semi-permanent to more seasonal or temporary states) and reduced inflows and water quality (Haig *et al.* 2019; Donnelly *et al.* 2022). California alone has lost over an estimated 96% of historic wetlands over the past two centuries (Bradley and Yanega 2018; Wilson *et al.* 2022). Since millions of migratory waterbirds along the Pacific Flyway rely on a limited number of wetlands and saline lakes as a part of their annual life cycle, even small habitat changes can disproportionately affect populations (Oring and Reed 1997; Haig *et al.* 1998; Warnock *et al.* 1998; Haig 2019).

Studies have shown that climate-driven changes in wetlands and saline lakes are reshaping habitats that sustain waterbird populations along the Pacific Flyway (Larson *et al.* 2016; Senner *et al.* 2018; Haig *et al.* 2019). For example, at Lake Abert, Oregon, both periods of low water levels with high salinity and periods of high water levels with low salinity were associated with declines in invertebrates and waterbirds, showing the sensitivity of these taxa to changes in salinity (Senner *et al.* 2018). In regions along the Pacific Flyway, Donnelly *et al.* (2022) documented declines in wetland habitats and suggests that these changes are impacting stopover networks and their capacity to sustain waterbird populations. Similarly, Haig *et al.* (2019) found that climate warming is reducing wetland water availability and increasing salinity, changing wetland species community composition, limiting habitat availability, and impacting critical migratory habitats. These findings underscore the ecological significance of remaining wetland systems in the Pacific Flyway, including the Salton Sea, located in the United States in southern California.

As wetlands and saline lakes decline or disappear across the Western United States, the Salton Sea (Sea) has emerged as one of the most critical inland saline lake habitats for waterbirds (Shuford *et al.* 2002; Patten *et al.* 2003; Barnum and Johnson 2004). Saline lakes such as the Salton Sea support a variety of habitats for birds, including deltas, wetlands, mudflats, playas, islands, and open water areas (Ma *et al.* 2010; Jones *et al.* 2016). Located in the southern California Sonoran Desert and straddling Imperial and Riverside counties, the Salton Sea is a terminal saline lake covering 790 km^2 and remains the largest lake in California. The Salton Sea hosts some of the largest waterbird concentrations (Shuford *et al.* 2002; Barnum & Johnson 2004), serving as a vital stopover for migrants traveling north from southern wintering grounds to breeding sites in the Arctic, Central Valley, or other saline lakes, while also supporting southbound migratory birds coming from places such as the Great Salt Lake or Mono Lake in the fall. Today, the Salton Sea supports more than 300 species and millions of migratory birds annually, making it one of the most ecologically productive stopover, nesting, and wintering sites in the Interior West (Patten *et al* 2003; Barnum and Johnson 2004; Shuford 2014). Given this ecological significance, numerous studies have assessed its role in supporting waterbirds and documented threats from habitat decline.

Previous work has provided important baselines for establishing the Salton Sea's ecological value by documenting historical bird populations, characterizing habitats, and documenting threats faced (Jehl and Joseph 1994; Shuford *et al.* 1998; Shuford *et al.* 2002; Barnum and Johnson 2004). Historically, the Salton Sea supported, during some portion of their life cycle, roughly 30% of American White Pelicans in North America (Shuford *et al.* 2002), 25%–90% of Eared Grebes in North America (Shuford *et al.* 2002, Patten *et al.* 2003), and served as the largest wintering site and an important breeding area for Snowy Plovers in the interior West (Patten and Smith-Patten 2004; Shuford *et al.* 2004). Fluctuations in fish populations strongly influenced fish-eating bird dynamics, with populations of pelicans, cormorants, and other fish-eating birds crashing following the collapse of the tilapia population in the early 2000s (Hurlbert *et al.* 2007). These historical studies provide an essential baseline for understanding how subsequent changes in water levels have altered habitats and bird communities at the Sea.

Changes in water allocations and ecological conditions have dramatically shaped the current state of the Salton Sea. The Sea's primary water source, agricultural runoff from farms in Imperial Valley, began to decline in 2003 as a result of the Quantification Settlement Agreement (QSA). This agreement diverted roughly 300,000 acre-feet (~0.4 km^3) of Colorado River water/year from Imperial Valley agricultural use to urban use in southern California but required mitigation water deliveries (Quantification Settlement Agreement Joint Powers Authority 2003). These water deliveries to the Sea were reduced at the end of 2017 and the Sea began to shrink at a more rapid pace, exposing approximately 1,000 hectares of lakebed per year (Bradley *et al.* 2022). To date, over 15,000 hectares of lakebed (playa) have been exposed since 2003, based on USGS lake surface elevation monitoring data and our own analyses of USGS bathymetric data (United States Geological Survey 2025). This has both increased the risk of airborne pollutants due to dust from the dry playa (Frie *et al.* 2017) and resulted in the loss of critical bird habitats (Shuford *et al.* 2020).

Declining water allocations have caused significant ecological shifts at the Salton Sea, including rising salinity, changes to food resources, the loss of nesting islands, and the formation of new wetlands on exposed playa. Reduced inflows due to the QSA water transfers and other factors, such as evaporation and declining inflows from Mexico, have caused salinity at the Sea to rise from 46 ppt in 2004 to 74 ppt in 2020 (United States Bureau of

Reclamation 2020), more than twice the salinity of the ocean. Salinity has reached levels where the last remaining saline-tolerant fish species that some bird species consumed, Tilapia (*Oreochromis spp.*), can no longer survive or reproduce in most areas around the Sea (Riedel *et al.* 2002; Hurlbert *et al.* 2007; Riedel 2016). Increased salinity is believed to have also altered macroinvertebrate populations (Simpson *et al.* 1998; Detwiler *et al.* 2002; Anderson *et al.* 2007; Barnum *et al.* 2017), which are critical food resources for many migratory bird species. Bradley *et al.* (2022) documented major ecological transitions since water inflows were reduced in 2017, including a shift toward invertebrate dominated food webs. Despite this evidence of ecosystem-level change, few studies have evaluated how these changes have affected specific waterbird populations and habitat extents since 2017.

The Sea's recession has also led to the loss of nesting islands and the formation of new wetlands on exposed lakebed. The loss of nesting islands, such as Mullet Island, which once supported 75% of inland California nesting Double-crested Cormorants (Shuford *et al.* 2020), has reduced reliable breeding sites for birds. Meanwhile, agricultural irrigation drains and perennial and ephemeral streams and washes that previously drained directly into the Sea, now spill out and pool onto the playa forming newly emerging wetlands. The extent and persistence of these new wetlands are not well understood. Ecological changes at the Salton Sea reveal how water management is shaping the Sea's habitats and resources, but we have limited understanding about the impact on birds. We provide a comprehensive assessment of the status of birds and their habitats at the dynamic and threatened Salton Sea.

Here, we use surveys and models of waterbird populations, along with habitat and food availability, to create a more holistic view of how the Sea is changing at an ecosystem-wide scale. We summarized National Audubon's Society's seven years of surveys, sampling, and habitat modeling at the Sea by analyzing: 1) bird population trends, 2) changes in the amount of preferred bird habitats, 3) emerging wetland habitats, 4) macroinvertebrate diversity, and 5) water quality. Specifically, we used data from waterbird surveys conducted from 2016–2023 to assess population trends of representative waterbird species to understand how bird populations are changing. We quantified acres of preferred habitats available at the Sea for representative species and compared these acreages to previous habitat assessments in 1999 and 2015, pre- and post- QSA implementation, respectively (Jones *et al.* 2016). Additionally, we used satellite imagery to quantify new emerging habitats created by irrigation and other runoff onto exposed playa. We measured dissolved oxygen at bird monitoring sites and sampled macroinvertebrates at a subset of sites over two years to better understand physical conditions and food availability. Together, these elements provide a more comprehensive understanding of the current state of the Salton Sea and its changing bird populations and habitats. This can offer insights for habitat management and planning at the Sea and serve as a model for other disappearing and managed inland saline lakes and wetlands in arid environments.

Methods

In 2016, we established 14 monitoring sites around the Sea (Fig. 1) used for waterbird surveys, water chemistry sampling, and macroinvertebrate surveys. We selected sites to represent areas where waterbirds were likely to be present and for their public and convenient accessibility to ensure reliability for long-term monitoring. We selected an additional site as a backup for instances when the Salt Creek site was inaccessible due to high stream flow.

Bird Population Trends

Surveys and Data Collection. We conducted point count Waterbird surveys from November 2016 through November 2023, at the 14 monitoring sites (Fig. 1). We define waterbirds as those from the taxonomic orders of Anseriformes (waterfowl), Charadriiformes (shorebirds), Pelecaniformes (pelicans and cormorants), Podicipediformes (diving waterbirds like grebes), Cinoniiformes (storks), Suliformes (cormorants), Gruiformes (coots and rails), Gaviiformes (loons), and Procellariiformes (petrels). Although this list includes secretive marsh birds (Gruiformes), they seldom appeared in our surveys, which did not focus on their preferred habitat. Surveys consisted of bird counts taken from a predetermined survey point along the water's edge with a spotting scope,

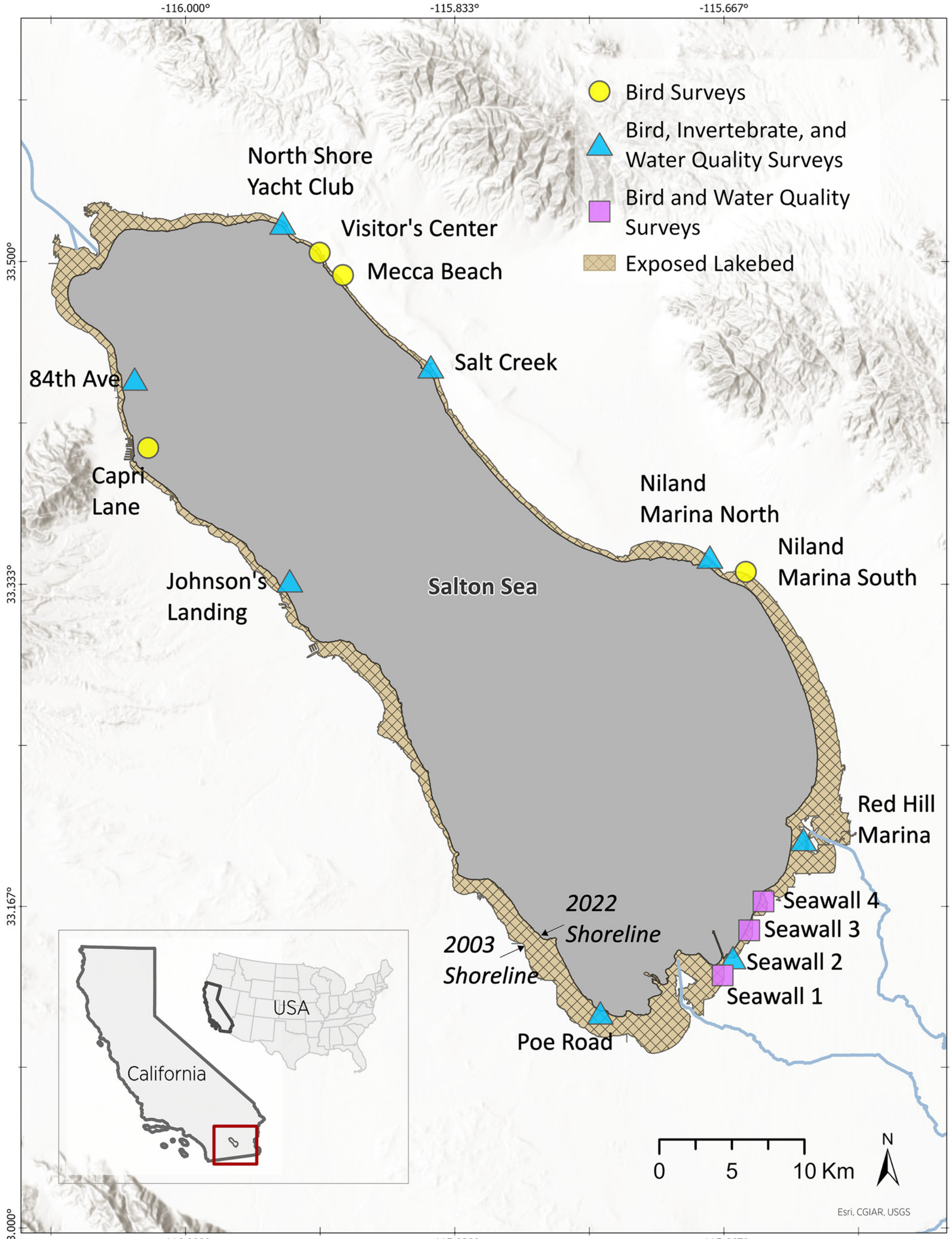

Figure 1. Map of the Salton Sea with sampling sites and playa exposed between 2003 and 2022.

covering a distance 500 m to the left and right of the survey point and 1 km offshore of the point creating a "survey box." Counts typically lasted about 15 minutes, although the exact duration varied depending on bird abundance or community participation; in all cases, observers remained on site until the survey box was thoroughly covered. Birds were identified to the species, except for the Western Sandpiper and Least Sandpiper, which were grouped in the analysis because they can be difficult to distinguish in the field and observer expertise varied across years (Table 1). Counts included birds on the shore, on the water, and in flight. Because the Sea

Table 1. List of waterbird species selected to model species population trends at the Salton Sea across survey years 2016–2023.

Common name	Scientific name	Breeding/non-breeding	Habitat use
Ruddy Duck	*Oxyura jamaicensis*	Non-breeding	Deep water
Northern Shoveler	*Spatula clypeata*	Non-breeding	Mid-depth water
Least and Western Sandpiper	*Calidris minutilla and Calidris mauri*	Non-breeding	Mudflats/shallow water
Black-necked Stilt	*Himantopus mexicanus*	Breeding and non-breeding	Mudflats/shallow water
American Avocet	*Recurvirostra americana*	Breeding and non-breeding	Mudflats/shallow water
American White Pelican	*Pelecanus erythrorhynchos*	Nonbreeding	Deep water
Double-crested Cormorant	*Nannopterum auritum*	Breeding and Nonbreeding	Deep water
Eared Grebe	*Podiceps nigricollis*	Nonbreeding	Deep water

was rapidly receding in some areas it was necessary to move survey points towards the water's edge to ensure the surveys were covering the intended habitat. In some cases, the original survey point is now over a mile from the shoreline.

Surveys were performed every two months from November 2016 through November 2018. From January 2019 to September 2022, surveys were increased to monthly, except for June during all years due to low bird counts and except for March and April of 2020 because of the COVID-19 pandemic. Beginning in September 2022, bird surveys were decreased again to quarterly. Surveys began in the morning at the same starting site and then proceeded in a consistent order through the remaining sites, typically continuing into the late afternoon. Survey times at individual sites varied within this daily sequence depending on survey duration, volunteer participation, weather, or site accessibility.

Data Analyses and Modeling. Species are present at the Sea during different time periods, representing breeding populations, nonbreeding populations, or, in some cases, both (Table 1). We assessed each species during the months when it was expected to be present at the Sea based its breeding or nonbreeding season, the timing of their migration and examination of these raw data. For example, the American Avocet and Black-necked Stilt have both breeding and nonbreeding populations at the Sea (Shuford 2014), so we modeled trends for both breeding and nonbreeding populations. We did this by separating data for the American Avocet and Black-necked Stilt into breeding season (April–July) and nonbreeding (August–March). We assumed only resident birds would be counted during breeding season and both populations would be counted during nonbreeding season. In contrast, the Least and Western Sandpiper are present only during nonbreeding season (August–April). The Least and Western Sandpiper are lumped in this analysis because they can be difficult for some surveyors to distinguish between in the field. The Ruddy Duck (October–May) and Northern Shoveler (July–April) are present during their nonbreeding season, The American White Pelican (August–March) is present for the nonbreeding season as is the Eared Grebe (November–May). The Double-crested Cormorant is present year-round. We also modeled trends for total waterbirds across all years, derived by calculating the sum of all waterbirds counted during each survey.

To control for differences in survey frequency among years, surveys were grouped into seasonal replicates: Fall (October–December), Winter (January–March), Spring (April–June), and Summer (July–September). These seasons were selected to group surveys into subsamples, but they do not necessarily represent breeding or nonbreeding periods for all migratory or resident species at the Sea.

We developed generalized mixed-effects models in R 4.1.1 (R Development Core Team 2021) for each species as well as for all waterbirds combined using the glmmTMB package (Brooks *et al.* 2017) with a negative binomial log link function to assess annual population trends. The trend was estimated as a log-linear change between 2016 and 2023. We used season and site as random effects to account for correlated counts within sites and seasons. We then sampled the posterior of the distribution of the trend estimate 5000 times, transformed samples using trend =[exp(sample)−1] x 100, and calculated the 0.025th, 0.5th, and 0.975th quantiles of the transformed samples to get the posterior median (± 95% credible interval) percent change of the population per year. Because we expected some credible intervals to be large, we also used the transformed samples to calculate the probability of a positive or negative trend by taking the sum of samples that were above or below 0 (depending on a negative or positive trend) and dividing it by the number of samples from the posterior distribution (5000). Probability values for trends were used to determine if a positive or negative trend was clear (≥ 0.95), likely (≥ 0.85), or possible (≥ 0.75). This means that a model's 95% credible interval could overlap zero and still have a likely trend given that 85% or more of the model's predictions (out of 5000) are consistent with the predicted trend. This modeling approach of simulating the posterior of the distribution for model parameters is described in Gelman & Hill (2006) and is common in Bayesian data analysis. We believe this approach is useful for assessing populations for conservation because it gives managers a sense of the range of possible trends within an established credible interval but also a gauge for assessing the probability of the trend's accuracy given a wide credible interval. This can aid in decision making and has been used in other population assessments developed to inform conservation (Robinson *et al.* 2018).

Changes in Preferred Habitats

Surveys and Data Collection. Quantifying preferred avian habitat at the Sea is critical to understanding how habitats are changing over time as shifts in water allocations drive changes in water level and the amount of habitat available to birds. Jones *et al.* (2016) quantified preferred avian habitats at the Sea for 1999 and 2015, pre- and post- QSA implementation. Our methods described here follow Jones *et al.* (2016) so that we can make direct comparisons in this paper. They identified four key waterbird habitats at the Sea defined as *playa*: the alkali flats, dry barnacle and sand beaches of the exposed sea bed, *mudflats and shallow water*: 0–15 cm depth, *mid-depth water*: 15–30 cm depth, and *deep water*: 30–200 cm depth (Jones *et al.* 2016). We paired these habitat definitions, with associated indicator species for each habitat (Table 2), and used model variables (24 variables; Appendix A) outlined by Jones *et al.* (2016), to derive updated estimates of the amount preferred habitats for 2023. This analysis relied on updated land cover and bathymetric GIS layers (Appendix A) while adhering to their methodological framework.

Data from Cornell Lab of Ornithology's online data portal, eBird (Sullivan *et al.* 2009) and Point Blue Conservation Science's Pacific Flyway Shorebird Survey (PFSS) (Reiter 2011), collected between 2011 and 2015, were used for habitat modeling by Jones *et al.* (2016) described below. The PFSS is an early winter survey conducted by volunteer scientists using established protocols across much of the Pacific Flyway (Reiter 2025). EBird is an online database of checklists or observations reported by birders which includes a geographic location and species identification and often how many of a given species were counted (Fink *et al.* 2024). Data from eBird can be problematic and introduce error associated with sampling distance from bird species (counting birds that may be far off from the observer's actual location), and the varying amount of time or effort spent counting birds. To help minimize errors associated with these data, Jones *et al.* (2016) applied several filters to eBird as other studies have done. Data were filtered to include only approved records (those vetted by eBird's reviewers). Jones *et al.* (2016) also filtered to include only records from survey events (where all species detected were reported) and records using either traveling count, stationary count, exhaustive area count and random location count protocols after Fink *et al.* (2010) and Hurlbert and Liang (2012). To help control for imprecision of traveling checklists within the 500 m model resolution, eBird records that used a traveling protocol were filtered to include only those that were less than 500 m. To avoid false multiple counts from groups where multiple people report a checklist, only one checklist was included from a known group survey where multiple observers entered the same checklist.

A 500x500 meter grid was developed and habitat variable data including acres of our key habitat types within each cell were added to this grid as were PFSS and eBird data. The grid spanned the extent of the 2003 Salton Sea shoreline and 1 km beyond the shoreline to capture additional bird counts. This was done because at that time, eBird data was less geographically precise and this allowed for the inclusion of more data to be used in the model. Preferred habitat does not refer to the amount of available habitat but instead refers to the amount of habitat where most of the indicator species for that habitat are predicted to be present.

Data Analyses and Modeling. We quantified preferred bird habitats using habitat suitability models for each indicator species chosen for each of the four habitat types identified by Jones *et al.* (2016). While the

Table 2. List of waterbird indicator species (common name and scientific name) for five key habitats at the Salton Sea. We quantified the acreages of each habitat type that are used by these indicator species, then compared this to habitats used by indicator species for each habitat in 1999 and 2015.

Common name	Scientific name	Habitat use
Snowy Plover	*Charadrius nivosus*	Playa
American Avocet	*Recurvirostra americana*	Mudflats and Shallow Water
Marbled Godwit	*Limosa fedoa*	Mudflats and Shallow Water
Dowitcher spp.	*Limnodromus griseus*	Mudflats and Shallow Water
Dunlin	*Calidris alpina*	Mudflats and Shallow Water
Western Sandpiper	*Calidris mauri*	Mudflats and Shallow Water
Least Sandpiper	*Calidris minutilla*	Mudflats and Shallow Water
Snowy Egret	*Egretta thula*	Mid-Depth Water
Gadwall	*Mareca strepera*	Mid-Depth Water
Northern Shoveler	*Spatula clypeata*	Mid-Depth Water
Eared Grebe	*Podiceps nigricollis*	Deep Water
Ruddy Duck	*Oxyura jamaicensis*	Deep Water
American White Pelican	*Pelecanus erythrorhynchos*	Deep Water
Double-crested Cormorant	*Nannopterum auritum*	Deep Water

Jones *et al.* (2016) habitat suitability models appeared in a technical report that was not refereed by a journal, it went through a peer review process that is outlined and detailed in the technical report. Their model was designed to be used in the future to continue to assess habitats and make comparisons to the past. We used habitat suitability models developed by Jones *et al.* (2016), to allow us to make direct comparisons with their pre- and post-QSA implementation habitat quantities. The Jones *et al.* (2016) habitat suitability models used a boosted regression trees modeling framework in R version 3.2.1 to predict the probability of occurrence of each indicator species based on 24 habitat variables (Appendix A) and the bird data described above.

We used R 4.1.1 (R Core Team 2021) and gbm (Greenwell *et al.* 2022) and dismo (Hijmans *et al.* 2022) packages to run the Jones *et al.* (2016) models with our updated landscape variables (Appendix A) and then convert probability of occurrence model predictions to presence/absence predictions. Presence/absence was determined by calculating the prevalence of each indicator species in each cell based on probability of occurrence scores after Liu *et al.* (2005), Lobo *et al.* (2008), and Jones *et al.* (2016). These represent threshold values of probability of occurrence above which a species was assumed to be present and below which a species was presumed to be absent based on habitat suitability model predictions.

We used presence cells from each indicator species to calculate a range of estimates including the mean, lower bound, and upper bound preferred habitat areas, after Jones *et al.* (2016). The lower bound estimate includes preferred habitat acreage for all indicator species of a particular habitat, the upper bound estimate is the acreage of preferred habitat for half or more of the species for that habitat, and the weighted mean preferred habitat available is the mean area of the lower and upper bound estimates weighted by the number of species in each estimate. For example, the lower bound estimate for mudflats and shallow water habitat would include acres from presence cells where all indicator species for that habitat were predicted to be present, while the upper bound would include acres were half or more of those species were predicted to be present. These preferred habitat areas were then compared to values from the Jones *et al.* (2016) study for pre- (1999) and post- (2015) QSA implementation as the methods we used here are the same as theirs.

Emerging Wetland Habitats

Surveys and Data Collection. To quantify new emerging wetlands on exposed playa and measure changes in the persistence of these wetlands to understand how habitat availability is changing we used 10 m resolution multispectral imagery from the European Space Agency's Sentinel 2 satellite (atmospherically corrected L2A products). We selected imagery from January 2020 and 2022 with less than 9% cloud cover to measure and assess vegetation on exposed playa. We used ArcGIS Pro (Esri 3.4.3) to create a playa shapefile with which to limit imagery analyses to the playa area only (defined as the area between the 2003 water level and the water level from the date of the imagery), to exclude areas above and below the playa. From this imagery we developed a Normalized Difference Vegetation Index (NDVI) (Rouse *et al.* 1973) for each year. NDVI is a commonly used index to identify and assess the density and health or robustness of vegetation in multispectral imagery (Huete and Liu 1994; Leprieur *et al.* 2000) and is useful for identifying vegetation in arid environments (Richard and Poccard 1998; Jafari & Arman 2014; Amiri & Tabatabaie 2010; Alsharrah *et al.* 2015). NDVI uses a mathematical equation including visible and near-infrared light wavelengths reflected by photosynthetic organisms to assess vegetation and is measured in values that range from −1 to 1. We considered values above 0.25 as playa vegetation based on Alsharrah *et al.* (2015) and visual inspection of results.

Data Analyses and Modeling. We identified and quantified emerging vegetation on exposed playa at the Sea using the playa vegetation data derived from the NDVI data in ArcGIS Pro (Esri 2.4.3). Along the margins of the Sea where water flows in shallow sheets across the playa, algal mats can form. In addition to vascular plants, NDVI analyses also identify this algal sheet flow vegetation. Although potentially important foraging habitat, this algal sheet flow vegetation did not represent the vascular plant vegetation primarily targeted by this study. We visually identified algal vegetation areas by analyzing the imagery used in these analyses along with higher-resolution Google Earth imagery. This process facilitated the visual detection of algal sheet flow and its differentiation from vascular plant vegetated areas further inland. The identified areas were then digitized to provide the best spatial estimate of algal vegetation coverage. We then derived and compared total area of plant vegetated areas (not including algal vegetation), and total vegetated areas (including algal vegetation) for 2020 and 2022. We mapped changes in the persistence, expansion, and contraction of vegetation on exposed playa between those years.

Macroinvertebrate Diversity

Surveys and Data Collection. Macroinvertebrate diversity can be an indicator of the health of an aquatic system. Here we expected that low macroinvertebrate diversity would be an indication of an unhealthy aquatic ecosystem, while high diversity would be an indication that the system is more stable. We conducted macroinvertebrate surveys quarterly from July 2020 through July 2022 to assess macroinvertebrate population dynamics as an indicator of food availability for shorebirds and other waterbirds, many of whom depend on macroinvertebrates as a food source, particularly during migration. Macroinvertebrates were collected at 7 of the 14 bird survey sites, selected as representative of different regions (i.e., north, south, east, and west part of the Sea), substrate types, and shoreline slopes (Fig. 1). We selected three core sampling locations at three 30 m lengths of shoreline per site, for a total of nine samples per site per visit. We selected these locations by observing where shorebirds were feeding at a site, then walked into the shallow standing water in that area and took three core samples per 30 m transect, one in the middle and one at each end. We used a 10 cm diameter clam gun to take core samples of sediment in water no deeper than 15 cm. We placed the substrate from a core sample through a 0.5 mm sieve, rinsing with fresh water until all mud was removed. We picked out all macroinvertebrates from the core sample using forceps and placed them into a 50 ml jar with a 70% ethanol mixture mixed with distilled water at a 1:1 ratio. All samples were sent to an independent laboratory (ESA Environmental Services) where macroinvertebrates were identified to taxonomic family at minimum and species when possible and counted from all samples.

Data Analyses and Modeling. We sorted macroinvertebrate survey data by family and then we recorded the number of macroinvertebrates per family per sample and calculated a total count per site per sample interval or day sampled. These are eight samples for each of seven sites, except for Poe Road (four samples) and Sea Wall (three samples) because of site access issues due to weather conditions ($n = 55$).

Water Quality

Surveys and Data Collection. Dissolved Oxygen (DO) is a measure of aquatic health and may impact fish and invertebrate availability and/or quality of bird habitat (United States Environmental Protection Agency 1986). We expected that DO would vary across sites and would include close to or hypoxic conditions. To better capture the overall health of the aquatic ecosystem and variation in habitat conditions across sites we measured DO concentrations (mg/L) monthly at 11 monitoring sites (Fig. 1) from February 2020 through June 2021. We collected data using an Aqua Troll 400 Multiparameter Probe and prior to each survey date, the probe's sensors were calibrated following the manufacturer's instructions. At each site, we placed the probe in the water at a minimum depth sufficient to submerge the sensor, approximately 3 inches when the probe was standing upright, or less when slanted. Once the probe was positioned at the appropriate depth, we allowed it to settle for approximately 2 minutes to reduce interference from sediment disturbance before recording measurements. We recorded data via the probe's corresponding mobile application.

Data Analyses and Modeling. We assessed water quality survey data as a measure for aquatic health at the Sea. DO (mg/L) concentrations were compared over the survey period and between sites ($n = 115$). To do this we examined mean DO levels at each site and compared them against U.S. EPA standards for aquatic health that list levels below 5 mg/L as cause for concern for aquatic life (United States Environmental Protection Agency 1986). If a sampling site had a mean DO concentrations below 5mg/L, we classified it as low DO and if a site had mean concentrations above that value, we classified it as moderate DO. We then used these categorical rankings to characterize this subset of sites for additional analysis to understand if waterbirds responded to these conditions.

Waterbird Response to Dissolved Oxygen

Surveys and Data Collection. To see if and how waterbird abundance varied with DO concentrations (a measure of aquatic health) we used total waterbird counts across all species for each site and survey where water quality samples were also taken ($n = 21$). We expected that waterbird abundance would increase with DO. Sites were classified as having moderate or low DO by mean DO concentrations as described above. We used only waterbird survey data that aligned with the DO sampling time period and sites where DO was sampled. We limited the amount of waterbird data to this time period because of the uncertainty around interannual variation in DO concentrations and long-term trends of DO at the Salton Sea.

Data Analyses and Modeling. We analyzed variation in waterbird relative abundance with DO concentrations using the same modeling and trend distribution sampling approach described for waterbirds above. For waterbird abundance we again used site as a random effect, and we used DO site categories (low or moderate) as an independent variable.

RESULTS

Bird Population Trends

The waterbird trend analyses we report here include a predicted positive or negative trend along with the median percent change (± 95% credible interval) of the population per year. We expected some credible intervals (CI) to be large, so to provide more insight and interpretability into these trends we also calculated a probability from the sampling of the posterior of the distribution of the trend estimate to determine if the trend was clear (≥ 0.95), likely (≥ 0.85), or possible (≥ 0.75). Waterbirds considered as a single group and assessed using all surveys from every month surveys were available, had a clear increasing trend (probability = 0.999) with a median population increase of 15.13%/year (CI = 5.52%–25.19%). Waterfowl species trends varied. Nonbreeding (October–May) Ruddy Ducks had a possible negative trend (probability = 0.81) with a median population trend of −6.95%/year (CI = −21.08%–9.51%; Fig. 2). In contrast, nonbreeding (July–April) Northern Shovelers had a clear increasing trend (probability = 1) with a median population trend of 53.47%/year (CI = 20.56%–94.55%). Shorebird trends varied between species and breeding/nonbreeding populations. Nonbreeding

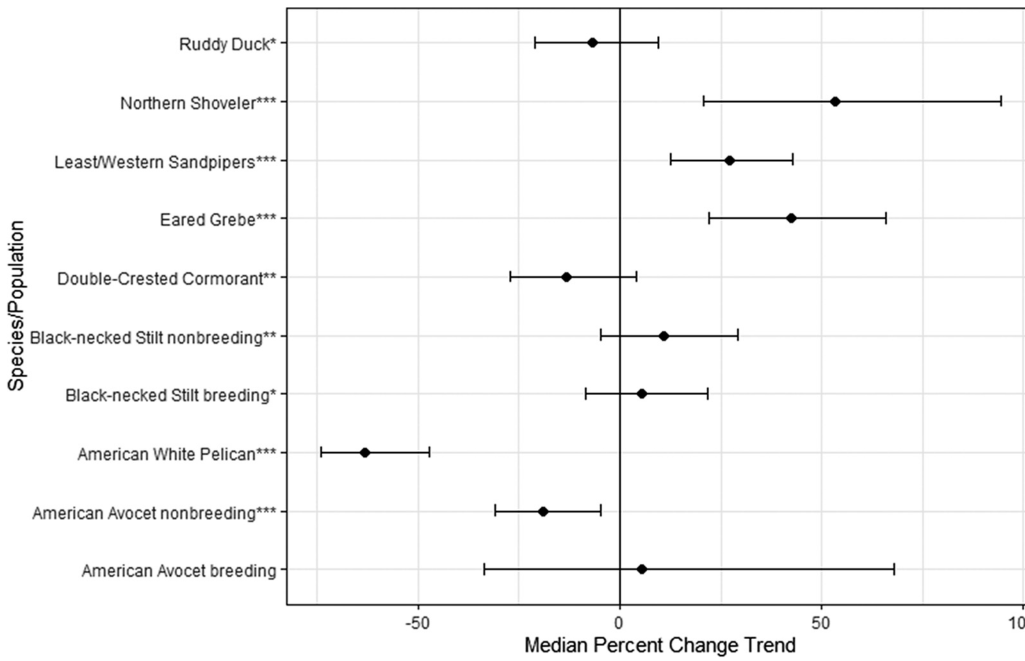

Figure 2. Modeled Salton Sea waterbird trends from 2016–2023. Dots are the median values; error bars are 95% credible interval. Probability of the trend is indicated by * (placed next the Species/Population name), where * is a clear trend (probability ≥ 0.95), ** is a likely trend (probability = 0.85–0.95), and * is a probable trend (probability = 0.75–0.85). Here we did not consider probabilities below 0.75 as probable trends. Percent change is on the x-axis and represents predicted annual trends.**

(August–April) Least and Western sandpipers (assessed as a single group because of difficulty consistently identifying them at distance) had a likely positive trend (probability = 0.90) with a median population trend of 27.18%/year (CI = 12.65%–42.92%). Nonbreeding (August–March) Black-necked Stilts were likely to be increasing (probability = 0.90) with a median population trend of 10.98%/year (CI = −4.74%–29.21%). Likewise, breeding (April–July) populations also had a possible increasing trend (probability = 0.77) with a median population increase of 5.40%/year (CI = −8.51%– 21.82%). American Avocet nonbreeding populations (August–March) had a clear decreasing population trend (probability = 1) with a median trend of −19.21%/year (CI = −30.97%– −4.91%). Breeding (April–July) American Avocet populations essentially had no trend (probability = 0.59) with a predicted increase somewhere between −33.65% and 68.06%. Nonbreeding (August–March) American White Pelicans had a clear negative trend (probability = 1) with a predicted decrease of −63.12%/year (CI = −74.13%– −47.29%). The Double-crested Cormorant population (present year-round at the Sea and so was assessed using all survey months available) at the Sea had a likely decreasing trend (probability = 0.94) with a prediction of −13.12%/year (CI = −27.03%–3.93%). Nonbreeding (November–May) Eared Grebe populations at the Sea had a clear increasing trend over the study period (probability = 1) with a median increase of 42.30%/year (CI = 22.23%–65.90%).

Changes in Preferred Habitats

Quantifying preferred habitats across five habitats in 2023 showed changes from previous pre- and post- QSA implementation preferred habitat estimates (Fig. 4) conducted by Jones *et al.* (2016). Preferred habitat refers to the amount of habitat where most of the indicator species (Table 2) of a given habitat have been predicted to be present, as described in the methods, and does not represent total available habitat. Because playa habitat was

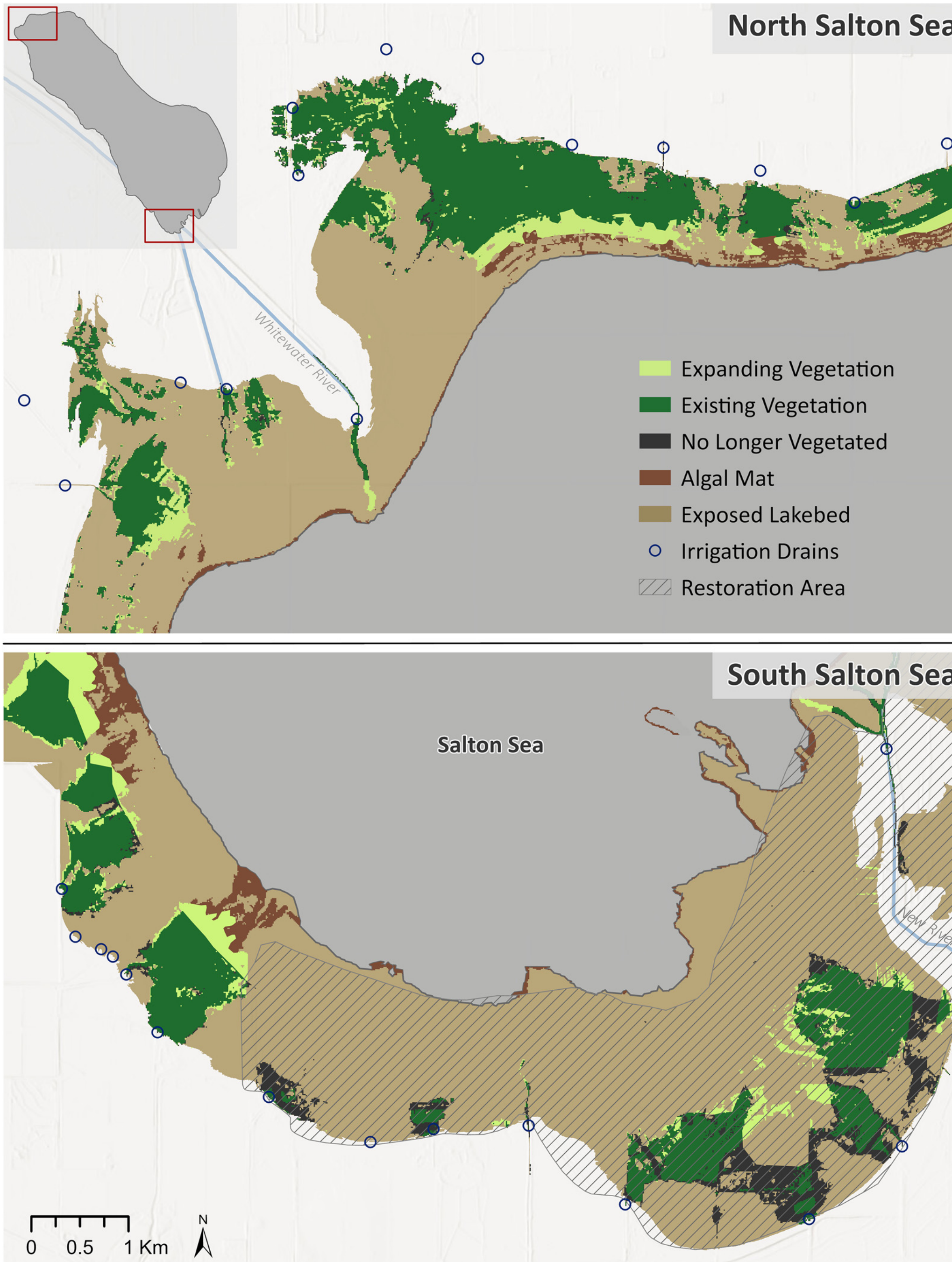

Figure 3. Showing the change in emerging vegetated wetlands (2020 and 2022) at the northern and southern parts of the Salton Sea as well as algal areas. The dark green indicates persistent wetland habitat between years, the light green indicates expanding or new vegetation, black indicates loss of vegetation, dark brown areas represent algal areas, light tan areas are exposed playa. The dots represent the location of irrigation drains.

represented by only a single species (Table 2), there was no weighted mean (WM) upper (UB) or lower bound (LB) estimates but only the presence estimate for that species (Snowy Plover, *Anarhynchus nivosus*). The amount of preferred playa habitat increased to 15,601

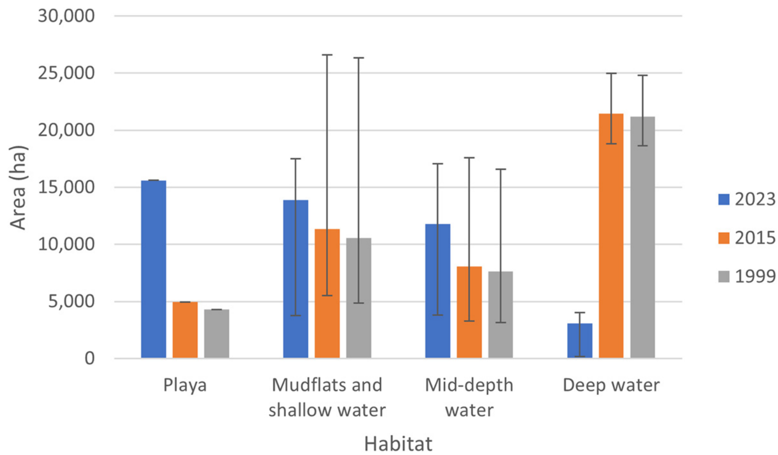

Figure 4. Hectares of estimated preferred habitat across years at the Salton Sea. Values used are WM estimates and error bars are UB and LB of estimates. Blue are results of this study (2023 estimates) compared to estimates from Jones *et al.* (2016) for 2015 (gray) and 1999 (orange).

ha in 2023 compared to 4,290 ha in 1999 and 4,937 ha in 2015. There were marginal increases in mudflats and shallow water habitat in 2023 (WM = 13,348 ha, UB = 17,495 ha, LB = 3,761 ha) compared to1999 (WM = 10,652 ha, UB = 26,345 ha, LB = 4,856 ha) and 2015 (WM = 11,331 ha, UB = 26,588 ha, LB = 5,504 ha). Mid-depth water habitat also showed marginal increases in our estimates for 2023 (WM = 11,785 ha, UB = 17,045 ha, LB = 3,796 ha) compared to both 1999 (WM = 7,649 ha, UB = 16,592 ha, LB = 3,157 ha) and 2015 (WM = 8,053 ha, UB = 17,563 ha, LB = 3,278 ha). Deep water habitats showed a large decrease in 2023 (WM = 3,069 ha, UB = 4,034 ha, LB = 174 ha) compared to 1999 (WM = 21,206 ha, UB = 24,807 ha, LB = 18,616 ha) and 2015 (WM = 21,448 ha, UB = 24,969 ha, LB = 18,818 ha).

Emerging Wetland Habitats

Emerging vegetation on exposed playa increased between years (Fig.3). We identified 2,732 ha of overall vegetation (vascular plant and algal vegetation) surrounding the Salton Sea in 2020 and 2,959 ha in 2022. Specifically, vascular plant vegetation increased from 2,189 ha in 2020 to 2,513 ha in 2022, while algal type vegetated areas decreased from 543 ha to 446 ha, respectively.

Macroinvertebrate Diversity

Almost all (97.5%) macroinvertebrates counted throughout the sampling period were identified as *Trichocorixa reticulata* or generally as the family Corixidae (water boatman). Overall, 12 taxonomic families were present in samples: Staphylinidae, Carabidae, Corophiidae, Coleoptera, Diptera, Syrphidae, Hemiptera, Dytiscidae, Ephydridae, Ephydra, and Corixidae.

Water Quality

DO varied across all sites and years, with many sites exhibiting hypoxic levels at the surface. Johnson's Landing and Niland Marina had the highest DO concentrations, with mean values of 8.70 mg/L and 9.06 mg/L respectively, both being categorized as moderate DO sites. Poe Road and Red Hill Bay exhibited the lowest DO concentrations with mean values of 3.03 mg/L and 0.53 mg/L, respectively, and categorized as low DO sites. The Yacht Club (4.07 mg/L) and 84th (3.61 mg/L) were the only other low DO sites with a mean DO concentration of 4.91 mg/L. Salt Creek (5.13 mg/L) and Sea Wall 1 (5.81 mg/L) were both moderate DO sites.

Waterbird Response to Dissolved Oxygen

Waterbird abundance had a clear negative relationship with DO concentration (probability = 1). DO concentrations had a strong negative effect with a median estimate of −86.90 (CI −94.40–−69.65). This corresponds to a modeled mean prediction of relative waterbird abundance of 687.11 (SD = ± 45.21) individuals at low DO sites and 88.97 (SD = ± 6.66) waterbirds at moderate DO sites.

DISCUSSION

These results show dramatic changes in avian habitats and populations at the Salton Sea. Waterbird populations largely shifted with changes in the amount of preferred habitat available. While the availability of habitats remained relatively static during QSA mitigation deliveries (Jones *et al.* 2016), after mitigation flows ended in 2017, the Sea began to rapidly shrink, and we now see changes in habitats and winners and losers in avian populations.

Although waterbird populations as a group have increased over the study period, this is not true for all waterbird species. The trend seems to be driven primarily by shorebirds, waterfowl, and grebes. Increases in nonbreeding Northern Shovelers were large compared to other species we analyzed, indicating that the Sea continues to provide good habitat for some waterfowl. Status and trends models at varying scales are now easily accessible through the eBird website, where users can quarry species, populations (breeding or nonbreeding) and regions and view eBird status and trends maps and information. We used these models developed by Fink *et al.* (2024) as a tool to compare our results to broader scale modeled trends to gain insight into how much of what we are seeing at the Sea may be local trends or if they are more likely tied to broader trends. For national trends we selected the United States as the region and California for subregion to view results at the state level. For Salton Sea specific eBird trends we could examine the results from the cells covering that specific area. Corroborating our model, 2011–2021 eBird trends for nonbreeding Northern Shovelers show increases in the northern part of the Sea but uncertain trends in other areas of the Sea (Fink *et al.* 2024). Broader national trends from eBird trend models for the same time period indicate decreases in this population (Fink *et al.* 2024). Ruddy Ducks did see declines over the study period but there is evidence that this is a wider trend (California Department of Fish and Wildlife 2024; Fink *et al.* 2024) and not necessarily a response to changes at the Sea. However, Ruddy Ducks are diving ducks and are likely more dependent on macroinvertebrates (Hohman *et al.* 1992) than Northern Shovelers, which are dabbling ducks feeding on the surface on more algal and plant matter during the nonbreeding season (Tietje and Teer 1996). So, it is possible that we are seeing different responses from different waterfowl species depending on their feeding preference. Low macroinvertebrate diversity could be a compounding effect contributing to the declining trend of Ruddy Duck populations at the Sea.

While 62% of shorebird species are in decline across North America, having lost 37.4% of shorebirds since the 1970's (Rosenberg *et al.* 2019), overall shorebird trends at the Sea were largely influenced by increases in nonbreeding Least and Western Sandpipers. This trend in Least and Western Sandpiper populations may reflect improved habitat quantity or quality. EBird trends from 2011–2021 also indicate increases in these species at the Sea, despite decreasing national trends (Fink *et al.* 2024). There is preliminary evidence that biofilm, found to be a significant food resource for some Least and Western Sandpiper populations (Elner *et al.* 2005; Kuwae *et al.* 2008), is also present at the Sea (Walters *et al.* 2024). This could be an important emerging resource for these small shorebirds. As more playa has been exposed, irrigation drains and other sources of freshwater have mixed with salt water in expanding mudflat areas, a key process for biofilm development (Schnurr *et al.* 2019), which has likely increased the availability of biofilm as a resource (Walters *et al.* 2024).

Black-necked Stilt trends were also positive but with less certainty and magnitude. This contrasts with national and statewide trends, which appear negative (Fink *et al.* 2024). American Avocet population trends varied between breeding and nonbreeding populations. Breeding population had essentially no trend, indicating that the population is not changing. However, the nonbreeding population had a strong negative trend that is most likely a larger flyway trend (Reiter 2025). This is also supported by eBird trends that indicated negative trends at the national level and California state level (Fink *et al.* 2024). Since breeding populations of American Avocets at the Sea appear to be stable, the trend seen in the nonbreeding population may not be connected to conditions at the Sea but to broader, region-wide habitat loss trends (Thomas *et al.* 2006). These population trends indicate that the Sea continues to provide habitat and resources for migratory and resident shorebird populations and in some cases, those habitats and/or resources may be increasing in quantity and/or quality.

Populations of the piscivorous species in this study, nonbreeding American White Pelicans and year-round Double-crested Cormorants, both exhibit strong negative trends. This is expected because fish have largely died off in the Sea and those that are left are likely only reproducing in small numbers where fresh water meets the Sea, because the current salinity of the Sea overall is more than twice that of California coastal waters (United States Bureau of Reclamation 2020). Prior short-term fish die offs have led to decreases in these species in the past (Hurlbert *et al.* 2007). However, this more permanent loss may only be remedied by habitat restoration that provides water quality and habitat conditions sufficient to support fish populations. In addition to loss of food resources, nesting opportunities for Double-crested Cormorants have also diminished, as the Sea has receded and exposed important inland nesting islands like Mullet Island to land-based predators (Shuford *et al.* 2002). These population trends are evidence that habitats supporting larger migratory and resident piscivorous birds at the Sea are decreasing in quantity and/or quality. While the Double-crested cormorant national and state trends are also negative, the American White Pelican has a relatively neutral national trend and negative state eBird trend (Fink *et al.* 2024). However, neither of these larger scale trends reach the magnitude of the decline our models produce for the Sea.

Nonbreeding Eared Grebe populations at the Sea showed a strong positive trend over the study period. Although trends indicated declines in the first two years of surveys, over time they increased. Eared Grebes are thought to have fed mainly on pile worms (*Neanthes succinea;* Jehl and McKernan 2002) at the Sea, which may have died off with rising salinity (Bradley *et al.* 2022; Anderson *et al.* 2007). The increase in Eared Grebes at the Sea corresponded with the arrival of massive numbers of water boatman. However, Eared Grebe trends presented here should be interpreted with caution. While they were thought to occur only along the edges of the Sea (Shuford *et al.* 2002), it has become clear through our observations and aerial survey efforts by Oasis Bird Observatory (R. McKernan, pers. commun.) that they can be found congregating in large rafts towards the center of the Sea. The surveys used here were shore-based surveys that did not capture areas beyond 1 km offshore where Eared Grebes may be more numerous and so they may not accurately represent the true trend of Eared Grebes at the Sea. In addition, our surveys began in 2016, when Eared Grebes may already have declined. Earlier counts estimated Eared Grebe numbers at the Sea to be greater than 1 million (Jehl and McKernan 2002), while more current aerial surveys by Oasis Bird Observatory capture counts closer to 500,000 at a time the annual population at the Sea would be expected to peak (R. McKernan, pers. commun.). This indicates that while the species is still present at the Sea in large numbers, they are present in numbers far below historic counts. Although we believe our trend analyses likely does not capture an accurate trend for Eared Grebes, we have included the species here because its life history and migration are strongly tied to saline lakes which are facing declines across the Western United States and changes in these habitats may adversely impact the species.

Changes in preferred habitat extent between pre- and post- QSA implementation aligned with our bird population trend results. The expansion of preferred mudflat and shallow water habitats, which primarily supports shorebirds, was especially pronounced in the north and south, where the Sea's shallow bathymetry exposed more playa as water receded. This increase in preferred mudflat and shallow water area corresponded with stable and increasing shorebird population trends. Similarly, increases in preferred mid-depth water habitat, measured using waterfowl as an indicator species, aligned with increasing trends in Northern Shovelers. In addition, declines in preferred deep-water habitat aligned with decreasing population trends of American White Pelicans and Double-crested Cormorants, both deep water habitat species.

Emerging vegetation analyses showed persistent and large amounts of algal vegetation in areas with expansive mudflats and shallow to mid-depth water areas and large amounts of agricultural drainage (Fig. 3). Agricultural ditches that used to drain directly into the Sea now spill out onto open playa and create wetlands areas and algal vegetation closer to the interface between the runoff and the Sea. The wetland vegetation and pooling water creates habitat for shorebirds, waterfowl, and marsh birds. These newly emerging habitats may also contribute to the increases in shorebirds and waterfowl over the study period. These wetland areas have been relatively consistent spatially and expand towards the shoreline as the edge of the Sea continues to retreat (Fig. 3).

On-the-ground visits to these wetlands confirmed that they consist of a mixture of native wetland vegetation and dense stands of Tamarisk, an aggressively invasive plant species that is common to the area. Our observations indicate that Tamarisk provides more of a wooded riparian habitat for songbirds compared to the native wetland vegetation, which provides habitat for waterbirds. This is consistent with the findings of Flannery *et al.* (2004) whose study suggested that these wooded riparian areas serve as important stopover sites for Neotropical migrant passerines, like the Wilson's Warbler. We do not know the relative proportions of these different vegetations in the emerging wetlands. Vegetation surveys would provide more detailed information on the relative quantities or amount of variation in habitats represented by these emerging wetlands. Further research is also need on the water demands and impacts on water quality of these emerging habitats. Additional studies could be used to inform potential maintenance of habitats in lieu of restoration or predictions of how long these emerging habitats may continue to persist.

Macroinvertebrate populations at the Salton Sea during our sampling period were dominated by *Trichocorixa reticulata,* or generally the water boatman family, which comprised 97.5% of all individuals counted. Low macroinvertebrate diversity is often a sign of poor aquatic health (United States Environmental Protection Agency 2024), and our findings may suggest that the macroinvertebrate community at the Salton Sea may already reflect the effects of rising salinity and other stressors. Continued increases in salinity could drive further decreases in macroinvertebrate populations, which could impact waterbird populations reliant on these food resources, particularly shorebirds and some waterfowl (Senner *et al.* 2018). While biofilm could be an additional resource for shorebirds, it is primarily small shorebirds like sandpipers that are adapted to exploiting it, leaving other shorebird species potentially vulnerable to changes in macroinvertebrate availability (Kuwae *et al.* 2021). While it could be argued that some saline lakes support low-diversity but high-abundance macroinvertebrate communities, such as with brine shrimp dominated systems in Utah's Great Salt Lake, it is unclear whether this will be the case at the Salton Sea because of interactions between macroinvertebrate species (Bradley *et al.* 2022). More research is needed to understand what food resources waterbirds are exploiting at the Sea to better understand current food web dynamics at the Sea and the potential impacts of environmental change.

Water quality levels could also be a cause for concern. Declines in DO could lead to deterioration of aquatic habitat and loss of aquatic food resources for birds through more frequent or longer-lasting hypoxic conditions (Saari *et al.* 2018, Cumming *et al.* 2012). All sites experienced hypoxic conditions at times but southern sites that have more extensive mudflat areas with shallow slopes to the shoreline (Poe Road and Red Hill Bay) experienced lower mean DO concentrations than sites with steeper sloped shorelines (Johnson's Landing and Niland Marina), where there could be more mixing between deeper and shallower water. These sites with lower mean DO are also more impacted by agricultural drainage. Both Poe Road and Red Hill Bay are close to river inputs that are largely agricultural drains and are surrounded by agricultural ditches that drain directly onto the playa and flow into the Sea. This nutrient loading in shallow lakes surrounded by agriculture is known to cause eutrophication leading to low DO (Dondajewska *et al.* 2019).

The negative relationship between waterbird abundance and DO may be habitat and resource driven rather than a direct relationship with DO. More extensive mudflat and shallow to mid-depth water likely provide more foraging areas for shorebirds and waterfowl. The nutrient rich freshwater input into shallow pools that flow to the Sea create an opportunity for greater algal growth than other areas. We have observed large aggregations of shorebirds foraging in these areas with algal growth. In addition, this combination of freshwater mixing and nutrients likely contributes to these sites having higher concentrations of biofilm (Walters *et al.* 2024; Stal and De Brouwer 2003), which is a likely food resource for large numbers of Least and Western Sandpipers. More research is needed to understand how biogeochemical processes influence food resources at the Sea and how environmental change may alter those processes and resources for better or worse.

Management Implications

While deep water piscivorous species are in steep decline at the Sea, populations of some waterfowl and shorebirds remain stable or are increasing. This study provides evidence that changes in habitat availability may be key to explaining these population trends, and although not directly supported by our results, we know from other studies (Walters *et al.* 2024, Anderson *et al.* 2007, Hurlbert *et al.* 2007) that changes in food resources also likely play a key role. Shorebirds are in decline globally and in the United States they are experiencing accelerated losses (North American Bird Conservation Initiative 2025). Places like the Salton Sea, where habitats will be managed into the future and shorebirds seem to be doing well, offer opportunities to create stable and reliable habitats to aid the survival and potential recovery of many of these species. Shorebirds depend on very shallow water habitats which require less water than many other waterbird habitats. As water availability will no doubt continue to decrease these factors should be considered. So, while we encourage the creation and restoration of diverse habitats, maintaining a focus or priority on shorebird habitat may be most beneficial, as well as realistic.

As California resource management agencies grapple with how to mitigate for changes at the Sea, this study describes the status of bird populations and how available habitat, and food resources have been influenced by water use changes. The Salton Sea will continue to recede until around 2047, with the current projected water supplies at around −78.669 m NAVD88 (compared to -69.129 m in 2003), at a salinity of about 271,885 mg/L and over 39,300 ha of exposed playa (CH2M Hill 2018). While emerging habitats are currently expanding, changes in water availability and quality and increases in agricultural irrigation efficiency make the future of emerging wetland habitats at the Sea an unknown. Future research is needed to determine the relative quantities of different vegetation in these emerging wetlands, what habitat values they offer and how much water they will demand. This will be important to support and enhance these emerging habitats which can and should be prioritized as opportunities for lower effort and less invasive restoration. Additionally, continued monitoring of bird populations, food resource availability and quality, and habitats is needed to continue to document changes in this dynamic ecosystem.

Saline lakes and other wetland habitats have been disappearing from the Pacific Flyway for over 100 years (Bradley and Yanega 2018, Jehl and Joseph 1994, Wilson *et al.* 2022), with this trend continuing to play out across the Western United States and globally (Donnelly *et al.* 2022, Wurtsbaugh *et al.* 2017). The analyses and results from this study shed light onto how bird populations and habitats are changing under current water management policy and practices. This study offers broad views of how changes in water allocations and use can influence avian populations and habitats. It describes what species may be at risk and identifies opportunities for the State of California and its partners to create long-term solutions by maintaining and stabilizing emerging habitats and food resources to provide dependable habitats into the future. It will be important to create diverse habitats but to also focus on species, like shorebirds, that are doing well at the Sea while their broader populations are in decline as wetland and saline lake habitats globally remain threatened.

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