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Avian Response to Early Tidal Salt Marsh Restoration at Former Commercial Salt Evaporation Ponds in San Francisco Bay, California, USA

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ABSTRACT

Restoration of former commercial salt evaporation ponds in the San Francisco Bay estuary is intended to reverse a severe decline (> 79%) in tidal salt marshes. San Francisco Bay is a critical migratory stopover site and wintering area for shorebirds and waterfowl, and salt ponds are important high tide roosting and foraging areas. Conservation of past bird abundance is a stated goal of area restoration projects, and early adaptive management will be critical for achieving this objective. However, initial avian response at sites restored to tidal flow may not be indicative of long-term results. For example, winter shorebirds at a 529 ha pond breached in 2002 showed a marked increase in shorebird abundance following breaching. Shorebirds comprised 1% of area totals during 1999–2002 and increased to 46% during 2003–2008. These changes accompanied increased tidal range and sedimentation, but minimal vegetation establishment. Conversely, a fully vegetated, restored 216 ha pond in the same system consistently supported less than 2% of all waterbirds in the region. Early restoration may temporarily increase habitat, but managed ponds will be needed for long-term waterbird abundance within a restored pond-marsh system.

INTRODUCTION

During the past 200 years, over 79% of historic San Francisco Bay (SFB) salt marshes have been lost, resulting in diminished habitat for native marsh species and fragmentation of remaining marshlands (Goals Project 1999). The loss of salt marshes was in part due to construction of commercial salt ponds covering over 1.4×10^4 ha (Goals Project 1999) that have been a part of SFB's landscape since 1856 (Josselyn 1983). In the past decade, large areas of commercial salt ponds have been acquired to be restored back to tidal marsh systems. Four thousand hectares of former salt evaporation ponds and surrounding habitats along the Napa River northwest of Vallejo were purchased by the California Department of Fish and Game (DFG) in 1994, and the Napa Crystallizer Plant (567 ha) was purchased in 2003. Over 1.0×10^4 ha of commercial salt ponds in South SFB were transferred to the

ownership of DFG and the U.S. Fish and Wildlife Service (FWS) in 2003. These ponds are currently planned for tidal wetland restoration under the Napa-Sonoma Marsh Restoration Project, the Napa Plant Site Restoration Project, and the South Bay Salt Pond Restoration Project. About 4500 ha of salt evaporation ponds remain in salt production, but current restoration plans suggest that 36–64% of all SFB salt pond area will be restored to tidal marsh by 2058.

SFB estuary is an important staging and wintering area for migratory waterfowl and shorebirds in the Pacific Flyway (Harvey et al. 1992). It is recognized as a site of hemispheric importance for shorebirds because it supports at least 30% of some flyway populations (Page et al. 1999), and also up to 50% of many diving duck populations (Accurso 1992). Many of these migratory waterbirds use the baylands, which consist of the area between the historic high and low tide lines and comprise about 8.6×10^4 ha in the estuary, including all acquired salt ponds (Goals Project 1999). While salt ponds offer important opportunities for tidal marsh restoration, they have become an integral part of the landscape and provide essential habitats for large numbers of waterbirds during migration and winter (Anderson 1970; Bollman & Thelin 1970; Accurso 1992; Takekawa et al. 2001; Warnock et al. 2002). Salt ponds are unique hypersaline wetland habitats that provide dense forage for birds that feed on saline-specialist invertebrates such as brine shrimp (*Artemia* sp.) and brine flies (*Ephydra* sp.) (Anderson 1970; Takekawa et al. 2006a). Although large salt ponds were not a natural feature of the landscape, they have existed in the San Francisco Bay estuary for more than 150 years (Ver Planck 1958), and much smaller salt pannes provided hypersaline habitats in the historic landscape (Goals Report 1999). The artificial, nontidal hyperhaline ponds vary seasonally in salt content from brackish to saturated, range from a few centimeters to a few meters in depth, and are composed of relatively simple but productive assemblages of algae and invertebrates (Carpelan 1957; Lonzarich & Smith 1997). Due to extensive urbanization in SFB, few alternative habitats exist to support large numbers of migratory and wintering shorebirds and waterfowl (Takekawa et al. 2000, 2005; Warnock et al. 2002). Consequently, one goal of the salt pond restoration projects is to maintain existing ecological

value for waterbirds (e.g., Trulio et al. 2005; see also Stralberg et al. 2005) by creating a mixture of habitats: some ponds would be retained as managed ponds to support bird species currently using the ponds, while others would be restored to tidal marsh to support endangered marsh-dependent species such as the California clapper rail (*Rallus longirostris obsoletus*) and salt marsh harvest mouse (*Reithrodontomys raviventris*). Because very high bird densities have been observed on a few commercial ponds (Takekawa et al. 2006a), managers hope to optimize features and conditions of the managed ponds remaining after restoration to support past numbers of migratory and wintering birds with reduced salt pond area.

Adaptive management has been heavily emphasized in salt pond restoration planning because of inherent uncertainties in restoration outcomes and because of potential impacts to endangered species and to entire flyway populations of migratory birds (see Trulio et al. 2005). Restoration monitoring is critical for adaptive management to be successful because it can provide early detection of unintended project consequences, which can then be addressed with corrective management action. However, initial avian response to levee breaching may not be indicative of long-term project success in former evaporative salt pond systems, as breached salt ponds are transitional habitats. They are open to tidal flow and thus to the hydrologic processes that may bring about the re-establishment of marsh vegetation, but vegetation establishment can take many years to occur (Williams & Orr 2002).

We examined bird use at six former commercial salt evaporation ponds over a ten year period. One pond was breached and mostly re-vegetated prior to the study period and provided a reference site for a restored marsh in this system, while other ponds were breached or remained intact during the study period. This study examines how salt pond bird use may change in both restored and managed ponds during the restoration process and how avian response may guide adaptive management at salt pond restoration sites.

STUDY AREA

We examined salt ponds in the North Bay (38.12°N-38.21°N; 122.28°W-122.37°W) subregion of the San Francisco Bay estuary (Figure 1). The salt ponds of the Napa-Sonoma Marshes Wildlife Area (NSM) were located 5 km northwest of Vallejo, California and comprised about 3400 ha. They were acquired in 1994 (2917 ha) and 2003 (470 ha) by the California Department of Fish and Game and became part of the NSM. Although commercial salt production ceased in 1993, the system remained intact for

nearly a decade with lower salinities and muted tidal flow in primary ponds and higher salinity and very little tidal flow in ponds farther inland (Lionberger et al. 2004; Takekawa et al. 2006a). Salinity generally increased from lower to higher numbered ponds as evaporation caused the water to become progressively more concentrated as it moved through the system. Water historically entered the system by tidal influence through a one-way gate from the North Bay into Pond 1 (P1). Water was pumped into Pond 2 (P2) intermittently during the study period (T. Huffman, California Department of Fish and Game, personal communication). Increased water elevation in P2 supplied water into Pond 3 (P3) via an inverted siphon except when density differences and small hydraulic head prevented flow (Lionberger et al. 2004). Density differences and small hydraulic head prior to the 2002 breach at P3 usually prevented flow through an inverted siphon between P3 and Pond 4 (P4) (Lionberger et al. 2004). During salt production, the supernate was removed from the brine and stored in Pond 7 (P7), the bittern pond.

System hydrology changed in late 2002 after an unauthorized breach on P3 (529 ha) resulted in direct tidal flow in the breached pond and reduced salinities in P3 and P4 (382 ha), which received water flow directly from P3. In March 2006, the initiation of the Napa-Sonoma Marsh Restoration Project saw additional breaches constructed at P3, P4, and P5, opening a total of 1233 ha (36% of the total North Bay salt pond area) to tidal flow.

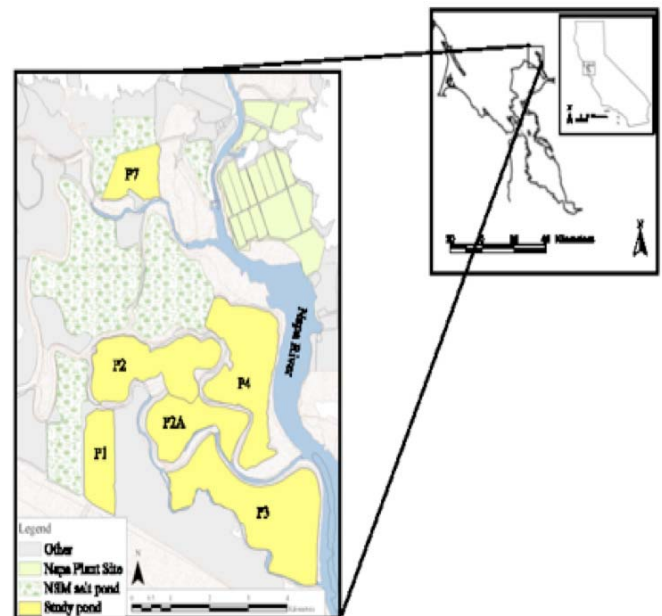


Figure 1—Former salt evaporation ponds, including 6 study ponds, in the Napa-Sonoma Marshes (NSM) Wildlife Area located 5 km northwest of Vallejo, California, USA on the northern edge of San Pablo Bay in the San Francisco Bay estuary.

METHODS

A subsample of six NSM salt ponds was selected for study between 1999 and 2008 to be representative of the salinity gradient in the salt pond system (Figure 1). We superimposed a 250 m x 250 m (6.25 ha) Universal Transverse Mercator (UTM) grid upon the sampled ponds to provide a framework for integrated sampling (Takekawa et al. 2006a). This grid system provided the basis for identifying bird locations in each pond (Matveev 1995; Posey et al. 1995). Ponds P1, P2, P3, P4, and P7 ranged in size from 123 ha to 529 ha and varied in mean salinity from 23 g l⁻¹ to 224 g l⁻¹ (Table 1), similar to the salinity range found in commercial salt production systems. Additionally, Pond 2A (P2A) was a 215 ha former salt pond that had been breached in 1995 and was mostly vegetated when the study began (Williams & Orr 2002).

Table 1—Mean winter salinity ± SE at salt ponds of the Napa-Sonoma Marshes Wildlife Area, California, USA, prior to (1999–2002) and following (2003–2008) the breach at P3.

Pond	Salinity (g l ⁻¹) ± SE	
	Winter 1999-2002	Winter 2003-2008
P1	18.5 ± 3.1	17.6 ± 1.8
P2	23.2 ± 2.4	18.0 ± 1.9
P3	40.4 ± 1.9	10.5 ± 1.5
P4	141.9 ± 27.0	30.3 ± 7.7
P7	223.7 ± 18.0	219.6 ± 20.9

Salinity

Salinity was measured monthly in P1, P2, P3, P4 and P7 from February 1999 until November 2001 and monthly from August 2003 through February 2008. Measurements were timed to occur within a week of bird surveys and locations were referenced to the 250 x 250 m UTM grid. Water levels were recorded from staff gages installed at a single location within each pond. Because pond salinities often exceeded the calibration capabilities of water quality meters, specific gravity was used for comparability to calculate salinity in all ponds and was measured with a hydrometer (Ertco, West Paterson, New Jersey) scaled for the appropriate range. These data were corrected for temperature and converted to salinity. Where pond salinity was less than 70 g l⁻¹, a Hydrolab Minisonde[®] (Hydrolab-Hach Company, Loveland, CO) was additionally used to measure specific conductance (internally converted to salinity with the 1978 Practical Salinity Scale) to check for consistency with concurrently collected specific gravity measurements.

Bird Surveys

Complete counts of the six ponds were conducted monthly from January 1999 to June 2001, bimonthly through August 2002, and monthly thereafter through February 2008. Observers conducted counts of species with binoculars and spotting scopes from vantage points on pond levees, and locations of waterbirds were placed within the grid cells of each pond to examine the spatial distribution of birds. More detailed location data were recorded to indicate when birds were on levees or islands rather than open water, and behavior was also recorded to indicate whether birds were foraging or roosting. Surveys were conducted during the day within 3 hours of the highest high tide when the largest number of waterbirds was roosting in the salt ponds. A low tide survey was additionally conducted on P3 in December 2007 for comparison with the high tide count conducted in the same month. Low tide access restrictions and the limited tide window made it possible only to count a portion of this large pond during this period, but the superimposed grid system allowed partial-pond comparisons between high and low tide counts.

Identified waterbirds were separated into guilds to examine differences among foraging groups rather than differences among species (Table 2). These foraging guilds included: 1) dabbling ducks—fed in the upper water column, e.g., *Anas clypeata* (northern shoveler); 2) diving ducks—fed in deeper water on benthic invertebrates, e.g., *Oxyura jamaicensis* (ruddy duck); 3) fish-eating birds—fish consumers, e.g., *Sterna forsteri* (Forster’s tern); 4) herons—herons and egrets, e.g., *Ardea alba* (great egret); 5) small shorebirds—foraged in the top layer (< 3 cm) of sediments, e.g., *Calidris mauri* (western sandpiper); 6) medium shorebirds—reached deeper into the substratum than small shorebirds, e.g., *Limosa fedoa* (marbled godwit); and 7) gulls, e.g., *Larus californicus* (California gull).

Analyses

Differences in salt ponds were examined during the winter (Dec–Feb), spring (Mar–May), summer (June–Aug), and fall (Sep–Nov) seasons. Months were assigned to seasons to encompass bird migration chronology; these provided replicated measures for seasonal abundance comparisons between ponds. Because of higher and more consistent bird abundance during winter, winter bird abundances and salinity data were used for among-pond comparisons and for low and high tide comparisons. Bird densities were calculated to account for differences in pond area. Density differences between the winters prior to (1999–2002) and following (2003–2008) the P3 breach at different ponds were investigated on square-root transformed data using two-way analysis of variance (ANOVA) (SAS Institute 1990). Although ANOVA is reasonably robust to failed normality assumptions, we additionally performed the test on rank-transformed data to compare results when data did not meet the normality assumption (Zar 1999).

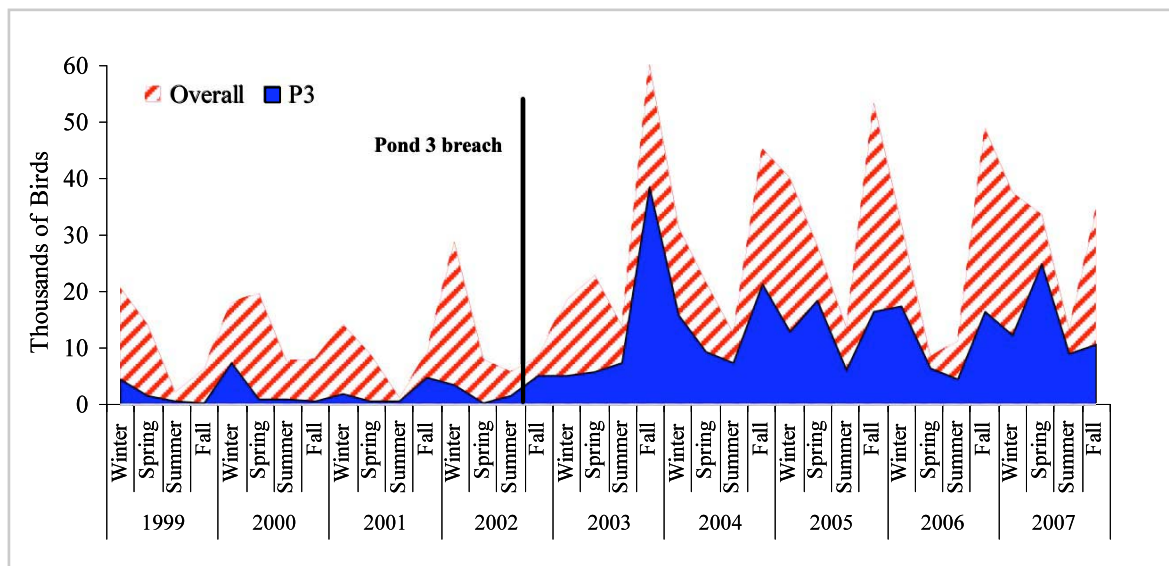


Figure 2—Combined seasonal bird abundances in study ponds P1, P2, P2A, P3, P4, and P7 from winter 1999 through fall 2007, Napa-Sonoma Marshes Wildlife Area, California, USA. Bird abundances in the breached pond, P3, are shown as a proportion of the total.

Because most bird species were migratory and exhibited high annual variability in overall abundance, counts were additionally standardized to total system abundance by calculating the proportional abundance at each salt pond (i.e., the sum of proportional abundances of a group of birds at all ponds totaled 100%). Changes in proportional abundance were interpreted as shifts in pond selection by groups of birds over time.

RESULTS

Total bird abundance increased after the breach of P3 (Figure 2). Highest numbers were observed during spring and fall migration periods and during winter, when overall bird numbers were on average 5.7 times higher than the preceding summer.

Pond salinity during winter was generally consistent, with standard deviation (SD) < 8 g l⁻¹, but was more variable (SD 18-72 g l⁻¹) in the higher salinity ponds P4 and P7. Salinity declined at all salt ponds during the 5 years following the initial breach but this was most pronounced at P3, which decreased from 40 to 10 g l⁻¹, and at P4, which decreased from 142 to 30 g l⁻¹ (Table 1).

Two-way ANOVA results on square-root transformed data suggested that overall winter bird densities differed between ponds ($F_{6,166} = 52.81$, $P < 0.0001$) and had a significant time (before vs. after breaching) effect as well ($F_{1,166} = 8.83$, $P = 0.0034$). However, data failed to meet the normality assumption (Shapiro-Wilk $W = 0.902$, $P < 0.0001$). Similar results were obtained when the test was performed on rank-transformed data (pond effect $F_{6,166} = 70.86$, $P < 0.0001$; time effect $F_{6,166} = 4.02$, $P = 0.0466$). The interaction between pond and time in

relation to breaching was not found to be significant when the ANOVA was performed on square-root transformed data ($F_{6,166} = 1.87$, $P = 0.0887$), but when the test was performed on rank-transformed data, there was a significant interaction effect ($F_{6,166} = 2.85$, $P = 0.0115$).

Winter densities increased 71% from 12.0 birds ha⁻¹ during 1999–2002 (pre-breach) to 20.6 birds ha⁻¹ during 2003–2008 (post-breach). Bird densities at individual ponds also increased during this period, except at P2A, which declined from 1.3 birds ha⁻¹ to 0.5 birds ha⁻¹, and P7, the bittern pond, which supported very few birds overall (Figure 3). The low salinity intake pond, P1, had the highest mean winter bird densities in the system both before (21.2 birds ha⁻¹) and after (34.7 birds ha⁻¹) the breach. P1 was followed by P4, which also had higher bird densities overall than both before (20.7 birds ha⁻¹) and after (27.6 birds ha⁻¹) the breach. Total bird abundance at P3 increased 178% after the breach, proportionally more than expected based on the 71% increase in overall system bird numbers. Before the breach, the density of birds at P3 was 8.3 birds ha⁻¹, only 70% of overall system density. After breaching, however, there were slightly more birds than would be expected if proportional abundance remained constant (23.0 vs. 20.6 birds ha⁻¹, or about 110%).

Pond 3 bird densities were lower than the rest of the system in the winters just prior to the breach, but following the breach, pond bird density increased to a level consistent with the other ponds. Pond 2A, the vegetated pond, had consistently low bird densities, generally 1-2 birds ha⁻¹, when compared to other ponds through all 10 years of the study (Figure 4).

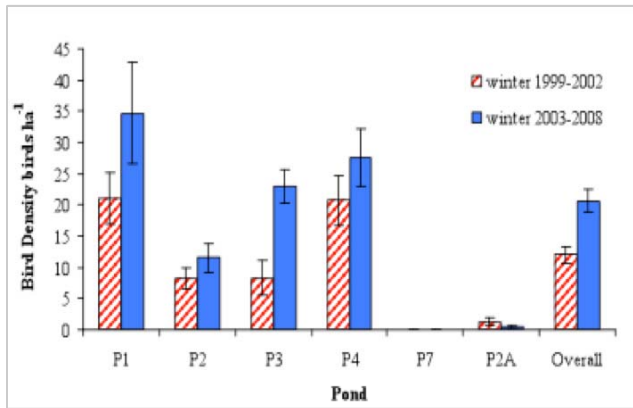


Figure 3—Overall bird density (mean ± SE) at the Napa-Sonoma Marshes Wildlife Area, California, USA, during the winters prior to (1999–2002; n = 9) and following (2003–2008; n = 17) the breach at P3. Mean bird density at P7 was 0.

The highest observed bird densities at single surveys were at P3 and P4 during winters following the P3 breach and were over 124 birds ha⁻¹; most of these were small shorebirds. Although both small and medium shorebirds have increased proportionately more following the breach than other guilds in P3 during high tide surveys (Table 3), low tide abundances may be even higher. While dabbling and diving ducks declined at P3 during low tide, shorebirds more than doubled (Figure 5). Of these shorebirds, 98% were observed foraging, whereas only 57% were observed foraging during the high tide survey.

The high tide counts demonstrated that shorebird guilds were also highly responsive to system-level change. Prior to the breach during the winters of 1999–2002, 87% of all small shorebirds and 79% of all medium shorebirds were counted in P4 (Table 3). During the winters of 2003–2008, the proportion of all shorebirds on P4 declined to 32%, with subsequent increases on P3 for small shorebirds (1% to 46%) and medium shorebirds (2% to 37%). Diving ducks declined at P1 and P3 from a combined total of 57% of area diving ducks to 38% after the breach; this decline was reflected in an increase at P4 from 8% to 29% of area totals. Dabbling ducks also declined from 36% to 12% of area totals at P1 and increased from a shared total of 60% to 84% at P3 and P4. Forty-five percent of all fish-eating birds were counted at P3 before the breach, but this declined to 16% after breaching, while fish-eating birds increased from 0% at P4 before breaching to 22% afterwards. Similarly, herons declined at P2 from 56% to 20% and increased at P3 and P4 from 16% to 57%. Seventy-four percent of gulls were counted at P4 before the breach, but after the breach this number declined by half at P4 and increased by 30% at P2 and 11% at P3 (Table 3).

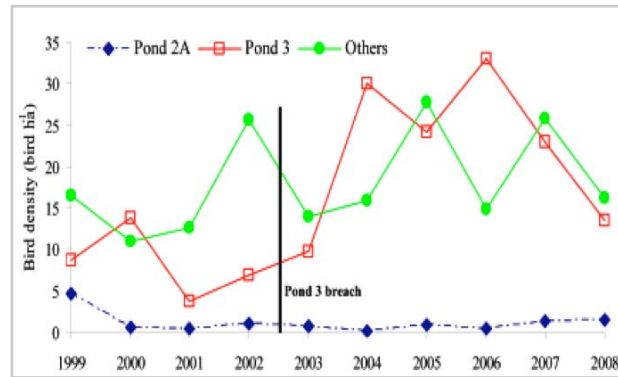


Figure 4—Mean winter bird density at P2A and P3, and combined average density at other study ponds (P1, P2, P4, and P7) at the Napa-Sonoma Marshes Wildlife Area, California, USA, from 1999 through 2008.

DISCUSSION

Waterbird Use of Salt Ponds

Salt ponds provide unique habitats that support large populations of migratory waterbirds (Takekawa et al. 2001, 2006a; Paracuellos et al. 2002; Warnock et al. 2002). Shorebirds in particular use salt ponds as important upland roosts during high tide, when adjacent mudflats are unavailable; insufficient availability of nearby roosts may constrain the ability of shorebirds to exploit mudflats as foraging habitats (Dias et al. 2006). Although large salt ponds are not natural features in SFB, their presence for the past 150 years (Ver Planck 1958) as well as their functional replacement of natural estuarine wetlands such as salt pannes has made them critical habitats for waterbirds. Salt ponds have provided these species with multiple advantages. The large expanses of water facilitated taking flight and predator avoidance, while the shallow, sheltered impoundments likely created a favorable microclimate for roosting (Warnock & Takekawa 1996). These conditions also favored foraging, and salt pond uplands have contributed significantly to shorebird foraging that have been reduced in natural estuarine habitats (Velasquez & Hockey 1991; Masero & Pérez-Hurtado 2001).

Velasquez (1992) noted that the abundance and species composition of macroinvertebrate prey was related to salinity, while the availability of prey to particular species of birds depended on depth. Lower salinity salt ponds support benthic invertebrates such as those used by shorebirds in shallow water and diving ducks in deeper water, and they are also favorable conditions for birds that consume fish, which generally cannot survive in salinities > 80 g l⁻¹ (Takekawa et al. 2006a). However, hypersaline ponds may be particularly valuable for many shorebirds and other species that can forage on the dense populations of *Artemia* and *Ephydra* that thrive there. In mid-hypersaline

SFB ponds, *Artemia* represent an important food resource for species that exploit this prey, with biomass exceeding the combined macroinvertebrate biomass of other pond by several orders of magnitude (Takekawa et al. 2006a). *Ephydra* have likewise been found to be an important prey species, and are more likely fed on by many species of waterbirds than are *Artemia* (Anderson 1970).

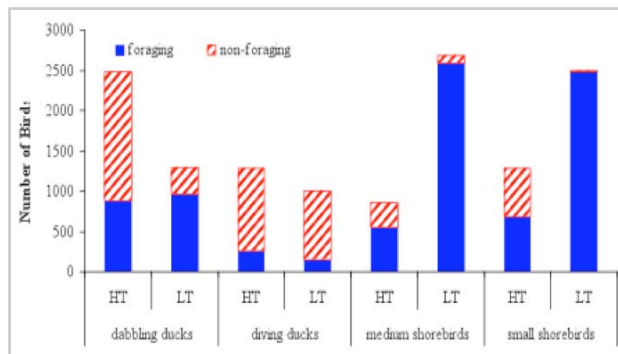


Figure 5–Winter 2008 bird abundance and activity (foraging or non-foraging) of dominant bird guilds at pond P3, Napa-Sonoma Marshes Wildlife Area, California, USA, during high (HT) and low tide (LT).

Bird use at the NSM largely conformed to expectations based on salinity and water depth. Prior to the P3 breach in summer 2002, highest winter bird densities were at P1 and P4. P1 was a mixohaline salt pond with variable water depth, and supported shorebirds at the shallow southern end and waterfowl at the deeper, northern end. Because of the low salinity, the pond supported benthic macroinvertebrates and many fish (Takekawa et al. 2006a), and was the only pond to support high proportions (> 10%) of every foraging guild examined. P4 was also somewhat variable in water depth, but it was a mid-hyperhaline pond and did not support fishes (Takekawa et al. 2006a). Although the pond supported about 10% of ducks in the system and many gulls, these species were primarily roosting at the pond and not foraging. The high bird densities at P4 were primarily due to shorebirds that were able to exploit the combination of variable, shallow water depth and dense populations of *Artemia* and *Ephydra* that were present at the pond during this period (Takekawa et al. 2006a). P2 and P3 had generally low bird density and similar composition. P2 was a mixohaline salt pond with consistent water depth and supported primarily diving ducks and fish-eating birds, which use deeper water, and herons, which forage along pond levees. P3 was a shallower, low-hyperhaline pond and supported nearly half of the dabbling ducks in the system as well as large proportions of diving ducks and fish-eating birds. P7, the bittern pond, was a high-hyperhaline pond and was too saline to support macroinvertebrates or fish, so it was not surprising that few birds were observed at the pond.

Changes in Pond Conditions Following Breaching

A key factor for restoring wetlands is to restore the hydrologic conditions that drive the structure and function of the wetland (Odum et al. 1995; Mitsch & Gosselink 2007). In tidal salt marshes, wetland hydrology is largely a function of the frequency and duration of tidal inundation, which is determined by the elevation of the site relative to tidal fluctuations (Montalto & Steenhuis 2004). In some salt ponds, such as P2A, restoration may be as simple as breaching levees according to hydrological patterns in the neighboring sloughs or the bay. Reintroduction of tidal action to a salt pond can encourage sedimentation and eventual vegetation establishment until the marsh reaches maturity. Initially, however, the expected effects of breaching are to reduce salinity to ambient conditions and to create tidally fluctuating water levels. As water depth and salinity are important parameters for determining habitat quality for birds, birds should respond immediately to these changes. Waterbirds are highly mobile and readily move between habitats in response to tidal fluctuations and changing foraging conditions (Burger et al. 1977).

Annual variation in bird numbers was high, but results of ANOVA tests on square-root and rank transformed bird density data suggested that bird densities increased following the breach. Additionally, a possible interaction effect between ponds and time relative to breaching suggested that some ponds may have changed more than others. This result suggests that changes in the system may have been related to changes at a few ponds rather than simply an increase in system bird numbers overall, and observed changes in physical characteristics and proportional bird distributions at P3 and P4 support this conclusion.

Immediately following the breach, P3 shifted from a low-hyperhaline pond to a mesohaline pond, the least saline pond in the system. Similarly, P4, which received inflow from P3, shifted from a mid- to high-hyperhaline pond to a low-hyperhaline pond. This reduction in salinity was likely responsible for some shifts in bird use, as diving ducks, fish-eating birds, and herons began to use P4 after the breach. Salinity and water level changes were most immediate in P3 because it was initially the only pond to receive direct tidal flow, and it became a dynamic habitat with continually shifting water levels. The addition of shallowly-inundated, open mudflat habitat at P3 and the concurrent loss of *Artemia* and *Ephydra* at P4 provoked a shift in some of the proportional distribution of shorebirds from P4 to P3 following the breach. Despite the redistribution of many birds to P3 and P4, the highest densities in the system remained at P1 following the breach, and the increased density at P4 was not inconsistent with the increase expected based on system-wide bird abundance increases in the years following the 2002 breach. What is notable is that P2 density increased very little. The

invariable bathymetry at P2 is likely responsible, as the pond is able to support a smaller subset of bird species than more variable ponds such as P1. Conversely, density at P3 increased much more than expected based on past proportional abundance, suggesting habitat quality improvements particularly for shorebirds.

Breached ponds provide unique low-tide foraging opportunities for shorebirds. Breached salt ponds are essentially temporary mudflats that may be inundated for different periods than mudflats in the SFB; thus, they may provide a longer foraging window for birds moving between habitats. Additionally, rapid early sedimentation, as has occurred at P3 (Takekawa et al. 2006b) may also make the pond favorable for rapid colonization by macroinvertebrates that can then be exploited by foraging shorebirds.

Waterbird Use of Restored Ponds

Breached salt ponds, while they may be heavily used by waterbirds, are a transitional habitat between two long-term states: managed salt ponds and mature tidal salt marsh. However, it is not always easy to predict how long the transition will last. P2A was > 80% vegetated 3 years after breaching (Goals Project 1999), but 6 years after its initial breach, adjacent P3 has < 5% vegetative cover (Athearn & Takekawa, unpublished data). Williams & Orr (2002) noted that in addition to initial pond elevation, the primary constraints on marsh development in SFB were restricted tidal action, limited sediment supply, and high wind-wave energy. Tidal action was initially restricted at P3, as the pond was more than twice the size of P2A yet contained only a single breach (at first only 0.25 m, eventually around 30 m wide by 2006) before additional construction opened up the pond to full tidal action in 2006 (Takekawa et al. 2006b). It is possible that high wave energy could inhibit marsh development at P3, but although adjacent pond P2A had the highest wave power index of all sites examined by Williams & Orr (2002), it also had among the highest vegetation establishment rates. Sedimentation was significant in the years immediately following the breach (Takekawa et al. 2006b), so it is likely that sedimentation will continue and that vegetation establishment may be rapid once appropriate elevations are obtained.

Mature salt marsh supports some of the same bird species as salt ponds, but at much lower densities (Stralberg et al. 2005). Although P2A supported some dabbling ducks, overall waterbird densities were consistently low throughout the study period. P2A was the only pond that showed a decrease in overall bird density after the P3 breach despite a 71% increase in bird density overall. However, the species composition at P2A included marsh dependent birds not found at managed or recently breached

salt ponds, including endangered California clapper rails (*Rallus longirostris obsoletus*) and many marsh passerines.

CONCLUSIONS

Salt ponds, although not part of the natural SFB landscape, have become an important resource for waterbirds in the estuary. Converting salt ponds to tidal marsh will likely benefit some species at the expense of others. Most shorebirds prefer more open habitats rather than tidal marsh plain habitats (Warnock & Takekawa 1995). Development of coastal zones and interior valley wetlands have resulted in fewer areas available for migratory waterbirds in the flyway, and alternative wetlands may not exist outside of the SFB estuary to compensate for loss of waterbird habitats in the ecosystem.

An important consequence of this loss of habitat is that careful planning is needed to determine the number and configuration of ponds restored during a given timeframe to ensure that sufficient habitat is available for waterbirds. Although restoration is intended to benefit endangered tidal marsh species, the process of restoration may result in habitat loss for bird species that heavily use salt ponds while also remaining unsuitable for tidal marsh species until plants reach sufficient maturity to provide cover and forage. Early reintroduction of tidal action during salt pond restoration may initially result in a strong avian response to newly restored ponds, and consequently a negative response in managed ponds even if habitat quality in managed ponds remains constant. Adaptive monitoring of salt pond systems that include transitional, breached pond habitats will require careful interpretation rather than simple numerical evaluation.

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Table 2—Bird species and associated foraging guilds observed at the Napa-Sonoma Marshes Wildlife Area, California, USA, during the winters prior to (1999–2002) and following (2003–2008) the breach at P3.

Guild	Common Name	Scientific Name
Dabbling Ducks	American coot	<i>Fulica americana</i>
	American wigeon	<i>Anas americana</i>
	cinnamon teal	<i>Anas cyanoptera</i>
	Eurasian wigeon	<i>Anas penelope</i>
	gadwell	<i>Anas strepera</i>
	green-winged teal	<i>Anas crecca</i>
	mallard	<i>Anas platyrhynchos</i>
	northern pintail	<i>Anas acuta</i>
	northern shoveler	<i>Anas clypeata</i>
Diving Ducks	Barrow's goldeneye	<i>Bucephala islandica</i>
	bufflehead	<i>Bucephala albeola</i>
	canvasback	<i>Aythya valisineria</i>
	common goldeneye	<i>Bucephala clangula</i>
	greater scaup	<i>Aythya marila</i>
	lesser scaup	<i>Aythya affinis</i>
	long-tailed duck	<i>Clangula hyemalis</i>
	redhead	<i>Aythya americana</i>
	ruddy duck	<i>Oxyura jamaicensis</i>
scaup (greater, lesser)	<i>A. marila, A. affinis</i>	
Eared Grebes	eared grebe	<i>Podiceps nigricollis</i>
Fisheaters	American white pelican	<i>Pelecanus erythrorhynchos</i>
	black tern	<i>Chlidonias niger</i>
	Brandt's cormorant	<i>Phalacrocorax penicillatus</i>
	Caspian tern	<i>Sterna caspia</i>
	Clark's grebe	<i>Aechmophorus clarkii</i>
	common merganser	<i>Mergus merganser</i>
	double-crested cormorant	<i>Phalacrocorax auritus</i>
	Forster's tern	<i>Sterna forsteri</i>
	horned grebe	<i>Podiceps auritus</i>
	least tern	<i>Sterna antillarum</i>
	red-breasted merganser	<i>Mergus serrator</i>
Hérons	black-crowned night-heron	<i>Nycticorax nycticorax</i>
	great blue heron	<i>Ardea herodias</i>
	great egret	<i>Casmerodius albus</i>
	snowy egret	<i>Egretta thula</i>
Small Shorebirds	Baird's sandpiper	<i>Calidris bairdii</i>
	dowitcher (long, short-billed)	<i>Limnodromus scolopaceus, L. griseus</i>
	dunlin	<i>Calidris alpina</i>
	least sandpiper	<i>Calidris minutilla</i>
	semipalmated plover	<i>Charadrius semipalmatus</i>
	snowy plover	<i>Charadrius alexandrinus</i>
	spotted sandpiper	<i>Actitis macularia</i>
western sandpiper	<i>Calidris mauri</i>	
Medium Shorebirds	American avocet	<i>Recurvirostra americana</i>
	black-bellied plover	<i>Pluvialis squatarola</i>
	black-necked stilt	<i>Himantopus mexicanus</i>
	greater yellowlegs	<i>Tringa melanoleuca</i>
	killdeer	<i>Charadrius vociferous</i>
	long-billed curlew	<i>Numenius americanus</i>
	lesser yellowlegs	<i>Tringa flavipes</i>
	marbled godwit	<i>Limosa fedoa</i>
	stilt sandpiper	<i>Calidris himantopus</i>
	whimbrel	<i>Numenius phaeopus</i>
	willet	<i>Catoptrophorus semipalmatus</i>
Phalaropes	red-necked phalarope	<i>Phalaropus lobatus</i>
Gulls	Bonaparte's gull	<i>Larus philadelphia</i>
	California gull	<i>Larus californicus</i>
	glaucous-winged gull	<i>Larus glaucescens</i>
	herring gull	<i>Larus argentatus</i>
	mew gull	<i>Larus canus</i>
	ring-billed gull	<i>Larus delawarensis</i>
	Sabine's gull	<i>Xema sabini</i>

Table 3—The proportion of total birds within each foraging guild (mean ± SE) that was observed at each study pond in the Napa-Sonoma Marshes Wildlife Area, California, USA, during the winters prior to (1999–2002) and following (2003–2008) the breach at P3.

	P1		P2		P3		P4		P7	
	99-02	03-08	99-02	03-08	99-02	03-08	99-02	03-08	99-02	03-08
Dabbling ducks	0.36 ± 0.13	0.12 ± 0.05	0.03 ± 0.01	0.04 ± 0.04	0.49 ± 0.11	0.65 ± 0.04	0.11 ± 0.11	0.19 ± 0.04	0.00 ± 0.00	0.00 ± 0.00
Diving ducks	0.21 ± 0.02	0.15 ± 0.03	0.34 ± 0.09	0.33 ± 0.07	0.36 ± 0.09	0.23 ± 0.03	0.08 ± 0.04	0.29 ± 0.06	0.00 ± 0.00	0.00 ± 0.00
Fish eaters	0.28 ± 0.12	0.27 ± 0.10	0.27 ± 0.16	0.35 ± 0.09	0.45 ± 0.21	0.16 ± 0.04	0.00 ± 0.00	0.22 ± 0.10	0.00 ± 0.00	0.00 ± 0.00
Gulls	0.17 ± 0.12	0.12 ± 0.04	0.01 ± 0.00	0.31 ± 0.16	0.08 ± 0.05	0.19 ± 0.08	0.74 ± 0.16	0.37 ± 0.15	0.00 ± 0.00	0.01 ± 0.01
Hérons	0.28 ± 0.17	0.23 ± 0.05	0.56 ± 0.18	0.20 ± 0.09	0.08 ± 0.05	0.17 ± 0.05	0.08 ± 0.05	0.40 ± 0.11	0.01 ± 0.01	0.01 ± 0.01
Medium shorebirds	0.18 ± 0.08	0.31 ± 0.10	0.00 ± 0.00	0.00 ± 0.00	0.02 ± 0.01	0.37 ± 0.06	0.79 ± 0.08	0.32 ± 0.09	0.00 ± 0.00	0.00 ± 0.00
Small shorebirds	0.12 ± 0.06	0.14 ± 0.09	0.01 ± 0.01	0.08 ± 0.08	0.01 ± 0.01	0.46 ± 0.12	0.87 ± 0.06	0.32 ± 0.14	0.00 ± 0.00	0.00 ± 0.00

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