Habitat-based modeling of wetland bird communities: an evaluation of potential restoration alternatives for South San Francisco Bay

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EXECUTIVE SUMMARY

The 2003 acquisition of 5,471 ha of salt ponds by state and federal wildlife agencies provides an unprecedented opportunity to restore large areas of contiguous tidal wetlands in South San Francisco Bay. From an avian conservation perspective, this could represent more than a doubling of habitat for tidal marsh-associated bird species, possibly increasing overall population viability of sensitive species such as the federally-listed California Clapper Rail (*Rallus longirostris obsoletus*) and other species of conservation concern. Meanwhile, it also presents management challenges, since the existing South Bay salt ponds support large numbers and a high diversity of waterbird species that could experience local, if not population-level, declines with the loss of this managed habitat. Thus our objective was to identify habitat relationships of key avian species, and develop habitat-based models to predict avian responses to restoration and habitat change.

Based on a six year period (1999-2004) of avian surveys conducted in tidal marsh and salt pond habitats, we developed habitat relationship models for 29 focal species and seasons, and used a model-averaging approach to generate predicted densities under various habitat alternatives comprised of restored tidal marsh and managed ponds (former salt ponds managed specifically for wildlife). Models included variables representing surrounding habitat context, as well as site-level marsh and pond characteristics. We focused on three alternatives being evaluated by the South Bay salt pond restoration project team: A ("no action"), B (50% tidal restoration), and C (90% tidal restoration), as well as variations in tidal marsh pond/panne evolution and in managed pond depth within these scenarios. The action alternatives (B and C) included managed pond configurations designed to benefit a range of waterbird species, while alternative A was based on very little human intervention, other than already-completed restoration. We evaluated changes within the restoration area itself, as well as throughout the South Bay, based on existing tidal marsh and salt pond habitats, as well as current and future projected tidal flats.

Results indicated a wide range of responses by different species, confirming that restoration will involve some trade-offs among species and habitats. However, we also found many opportunities for positive solutions through a combination of intensive management, balanced habitat configurations, and phasing of restoration activities over time. Key findings are summarized below:

- Predicted foraging waterbird densities were generally lower in tidal marshes than in managed ponds. However, waterbird density in tidal marshes was almost always positively associated with the amount of open water, in the form of tidal channels, tidal ponds, and semi-tidal pannes. Black-necked Stilt (*Himantopus mexicanus*), Least Sandpiper (*Calidris minutilla*), Gadwall (*Anas strepera*), and Northern Shoveler (*A. clypeata*) were particularly responsive to increases in open water habitat. Thus habitat potential for waterbirds within restored tidal marshes could be increased by accelerating the development of large-scale open water features, such as high elevation salt pannes, via more active site engineering and construction activities. Alternatively, some restoration sites could be maintained in a state of muted tidal action, to maintain large unvegetated areas by keeping them flooded for longer periods.

- Depth and salinity conditions explained much of the variation in foraging waterbird densities within managed ponds. In general, water depth had more explanatory power than salinity for individual species, except for some high-salinity specialists—Black-necked Stilt and Eared Grebe (*Podiceps nigricollis*)—and low-salinity specialists—American White Pelican (*Pelecanus erythrorhynchus*), scaup (*Aythya* spp.) and Ruddy Duck (*Oxyura jamaicensis*). In terms of water depth, small and large shorebirds generally had much higher densities in shallow ponds (<15 cm), Eared Grebe had higher densities in deeper ponds (>1 m), and other species' responses were intermediate, with shallow ponds generally supporting more species at higher densities.

- In year 0 of any restoration alternative, soon after levees are breached and tidal action is restored, numbers of waterbirds, especially shorebirds, dabbling ducks, and some fish-eaters, are likely to increase within the restored areas, as new low-salinity, unvegetated, intertidal and subtidal foraging habitats are created. This suggests that a staggered approach to tidal marsh restoration may have the greatest opportunity to provide long-term habitat benefits for waterbirds, as newly-breached ponds may compensate for the loss of feeding opportunities in marshes that become vegetated.

- By year 50, after most restoration ponds have become vegetated, most shorebird, fish-eating, and diving duck species are expected to have higher numbers under alternatives that retain substantial areas of managed ponds (e.g., alternative B). Landbirds, rails, and dabbling ducks, however, would have highest numbers under restoration scenarios with more tidal marsh area (e.g., alternative C). Weighing the needs of a broad range of species, a mixed restoration / managed pond alternative (e.g., alternative B) appears to provide the best starting point to maximize species diversity and numbers. The alternative could be subject to modification (e.g., restoration of additional ponds) at a later point in time if avian population response warranted such a change.

- Pond management characteristics may have a greater effect on habitat capacity and overall waterbird numbers than the ratio of managed ponds to tidal marshes, to a certain point. Furthermore, intensive pond management would likely provide greater opportunities to increase waterbird numbers than the engineering of tidal marsh open water features during restoration. Managing all ponds to be shallow (<15 cm) would have a greater positive effect on more species than managing all ponds to be deep (>1 m).
- Shorebird species' responses may differ by season, and, due to overall higher use of South Bay habitats during migration periods (especially spring), migration periods have the potential to become population bottlenecks without adequate managed pond habitats. Thus, for shorebirds, it may be more appropriate to focus on pond management during these periods, when ponds are more likely to exceed their carrying capacities.
- Based on observed and modeled sediment dynamics in the South San Francisco Bay, combined with threats posed by invasive *Spartina* encroachment and sea level rise, tidal flats are most likely going to decrease in the South Bay, particularly north of the Dumbarton Bridge. This means that managed ponds and seasonal wetlands will become more important for the species that rely on tidal flats. While tidal marsh open water habitats may compensate for some of this loss, shorebird use of tidal marshes may be an order of magnitude lower than tidal flats.
- For two sensitive species, the tidal marsh-dependent Clapper Rail, and the dry pond-associated Snowy Plover (*Charadrius alexandrinus*), high variability in density among sites led to large ranges in predicted restoration responses. Using upper density estimates, alternatives A or B could support at least 500 individuals of each species, while, using lower density estimates, no alternative could simultaneously support 500 individuals of each species. For both of these species, active predator management would be an important component of any plan for species recovery.

While there are several sources of uncertainty associated with our model predictions, the principal unknown factors are the current carrying capacity of South Bay habitats, the availability of alternative habitats for bird using managed ponds, the extent to which habitat quality and availability are limiting bird population size and trajectory, and whether birds will indeed respond to change in availability of habitat in the manner that our habitat-based models assumed. Further research and monitoring of new and existing restoration sites will be needed to reduce these sources of uncertainty.

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INTRODUCTION

The 2003 acquisition of 5,471 ha of salt ponds by state and federal wildlife agencies provides an unprecedented opportunity to restore large areas of contiguous tidal wetlands in South San Francisco Bay, comparable in size only to restoration efforts in Delaware Bay (Weinstein et al. 2001). The conversion of existing salt evaporation ponds to vegetated tidal and managed marsh will create valuable new habitat for marsh-dependent birds such as the federally-listed California Clapper Rail (*Rallus longirostris obsoletus*)[1] and two California subspecies of special concern, the Alameda Song Sparrow (*Melospiza melodia pusillula*)[2] and the Salt Marsh Common Yellowthroat (*Geothlypis trichas sinuosa*)[3], increasing the long-term population viability of those species. At the same time, the restoration presents management challenges, since the existing South Bay salt ponds support large numbers and a high diversity of waterbird species that are not likely to use restored tidal marshes at similar levels as managed salt ponds (Stenzel et al. 2002, Warnock et al. 2002, Warnock 2005, Takekawa and Athearn 2006).

The trade-offs associated with restoring habitats of different types have stimulated debate among regulators, managers, scientists, citizens, and others interested in the state of the Bay as to what types of habitats should be restored and what the goals of the restoration should be (Grossinger and Baye 2004). This study represents a quantitative assessment of the effects of restoring former commercial salt pond habitat to a mixture of naturally self- sustaining habitat types (e.g., Grossinger and Baye 2004) on bird populations. PRBO Conservation Science (PRBO) has developed a predictive modeling approach called the Habitat Conversion Model (HCM, see also Stralberg et al. 2005) to help answer the overarching question:

How will the South Bay restoration project affect bird populations, and how can we ensure that the resulting habitat mix maximally benefits and supports a diverse bird community?

PHASE I FINDINGS

The first phase of our HCM study made the following recommendations based on analyses and modeling of data collected in South Bay salt ponds and tidal marshes from 1999-2001 (Stralberg et al. 2003):

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[1] All Clapper Rail species references within this report refer to subspecies *R. l. obsoletus*.
[2] All Song Sparrow species references within this report refer to subspecies *M. m. pusillula*.
[3] All Common Yellowthroat species references within this report refer to subspecies *G. t. sinuosa*.

- While songbirds and rails could benefit greatly from creation of new tidal marsh habitat, the loss of salt ponds may cause substantial reductions in mean daily waterbird numbers in those areas, especially diving ducks, fish-eaters, and shorebirds. Whether reductions or increases in bird numbers using the restored sites will result in regional changes in bird numbers is still unknown.
- The number of waterbirds that use tidal marsh habitat is strongly affected by the amount of open water habitat in the marsh (i.e., large channels and ponded areas). Thus, when new tidal marshes are created, the design, engineering and long-term management of permanent ponded areas and major sloughs (i.e., extensive open water areas) can mitigate the potential negative impact on waterbirds caused by loss of salt pond habitat. Mean daily dabbling duck numbers may even be increased through the design and management of individual restoration sites.
- Lower salinity salt ponds (< 60 ppt) support the highest number and diversity of species, while high salinity ponds (120-200 ppt) support the highest densities of shorebirds, due to their prey communities and physical characteristics. Thus retaining many low salinity ponds and a few high salinity ponds, as well as the mid-salinity ponds that complete the evaporation chain, would be most beneficial to a large number of species, if the ponds continue to function as they presently do.
- Species diversity and apparent densities can be influenced by landscape context. Increasing the amount of tidal marsh and tidal flats within the South Bay wetland landscape may help promote high species diversity, as well as increased landbird and waterbird numbers, in both salt pond and tidal marsh habitats.
- Maintaining a wetland mosaic that includes tidal marsh habitat in various successional stages, interspersed with salt ponds managed for appropriate depths and salinities, can help preserve and enhance current South Bay bird populations.
- Tidal mudflats provide important bird habitat, especially for shorebirds; assessment of bird response to habitat conversion needs to take into account availability of this habitat type.

- Trade-offs are inevitable, at least to some degree. The minimization of waterbird losses also means a smaller gain in tidal marsh landbirds, which cannot use large open-water habitat areas. There are also trade-offs among waterbirds in the management of remaining salt ponds, with respect to salinity preferences (generally low salinity for ducks and fish-eaters, high for shorebirds) and depths (shallow for shorebirds, deeper for diving ducks and fish-eaters). Trade-offs should be evaluated in the context of long-term population viability, rather than simply bird use.
- Findings were based on habitat associations and should not be directly extrapolated to future trajectories of bird populations. Future research should explore the population viability of various species and attempt to identify population bottlenecks and the habitat features that affect them, as well as the factors affecting carrying capacities of restored habitats (e.g. prey availability, microhabitat characteristics).

Results of HCM I as well as input from other scientists helped identify additional research questions and modeling directions needed to aid the restoration planning process. In light of these advancements, PRBO, in collaboration with USGS, initiated a second phase of habitat modeling work (HCM II) to begin addressing some of the following issues.

TIDAL MARSH

HCM I identified the need to promote habitat heterogeneity within tidal marshes, especially ponded areas and large tidal channels for waterbirds. Our model predictions were based on the range of conditions observed within 12 sites surveyed in 1999-2001. It is difficult to know what future restored tidal marshes will look like, given how much larger they will be than current tidal marshes. However, years of restoration experience throughout the San Francisco Bay, as well as historic records of pre-development tidal marshes, provide some indication of what may be expected within a 50-year timeframe. Herein we evaluate avian responses to realistic future restoration scenarios, based on restoration parameters provided by Phil Williams and Associates (PWA). We also use data on bird use of marsh microhabitats (vegetation, ponds/pannes, and channels) to segregate our predictions by microhabitat.

MANAGED PONDS

Managers need help in identifying the specific characteristics of managed ponds required to support the maximum species diversity and abundance of birds within as small an area as possible. The two most important variables in determining bird use in salt ponds appear to be pond salinity (a key factor driving prey diversity and abundance in the ponds), and pond depth (which determines accessibility of prey in ponds for different bird species). Previously, PRBO analyzed waterbird use of a sample of South Bay salt ponds with respect to salinity, but not water depth (Warnock et al. 2002, Stralberg et al. 2003). Recently obtained depth data collected in association with waterbird surveys conducted by USGS and PRBO now allows the examination of water depth, in conjunction with other factors, on foraging bird densities.

TIDAL FLATS

Large numbers of shorebirds forage on South Bay tidal flats during low tides (Stenzel et al. 2002). Consequently, managers need to understand how changes to salt pond and tidal marsh habitats will affect the distribution and amount of tidal flats in the South Bay, and, in turn, how the diversity and number of birds using tidal flats could change. Based on geomorphic modeling conducted by PWA, we estimate future changes in tidal flat shorebird use, based on tidal flat shorebird surveys conducted from 1988-1993 (Stenzel et al. 2002). We also incorporate changes in surrounding tidal flat extent into our predictions of managed pond and tidal marsh habitat potential.

MODELING OBJECTIVES

In addition to several data analysis needs and modeling improvements identified in Phase I, our Phase II work has focused on the evaluation of actual project alternatives, developed by the consultant team comprised of PWA and HT Harvey and Associates (HTH), in terms of their predicted effects on a suite of focal species and groups that depend on tidal marsh and/or managed pond habitats in the South Bay. The alternatives we evaluated were based on geomorphic modeling projections of sediment accretion and erosion, and marsh establishment over time (PWA 2006). We looked at a 50-year time frame to assess the immediate and long-term effects of restoration on avian densities and abundance in the South San Francisco Bay. We evaluated three alternatives (A, B, and C) to provide bookend projections. In addition, we varied managed pond depth characteristics to evaluate potential management strategies. While our

Phase I work focused on examining the effects of different types of tidal marsh restoration, we constrained this modeling exercise to the marsh conditions provided by the consultant team, and focused instead on the potential influence of different types of pond management, with respect to pond depth.

In Phase I work, we identified over 75 waterbird species regularly using South Bay salt ponds and tidal marshes. Given the complexity of understanding the dynamics of so many species, in Phase II we undertook an exercise to identify focal species that represent groups of similar responding waterbird species.

To estimate South Bay-wide effects of restoration activities, we also estimated habitat potential provided by managed pond and tidal marsh habitats outside of the restoration area (including Cargill Salt pond), evaluated overall South Bay-wide habitat potential for the suite of focal species and groups, and predicted proportional changes in bird abundance from baseline conditions.

MODELING APPROACH AND ASSUMPTIONS

Our modeling approach was habitat-based, and was predicated on the general assumption that habitat is the primary limiting factor for the bird species evaluated, and that habitats were at carrying capacity at the time they were surveyed. Our empirical models are based on estimated bird densities from field surveys, and do not incorporate the effects of population dynamics or demographic processes (e.g., reproductive success and survival). Furthermore, with the exception of tidal marsh landbirds and rails, the observed salt pond and tidal marsh densities we analyzed represent mean high-tide foraging use of a habitat, rather than maximum potential densities. Because most waterbirds are highly mobile, and habitat use varies, the predictions from our models, when summed across habitats, provide "snapshot" estimates of habitat use but are likely to underestimate bird use over a longer time period (day or season). Relative densities, however, when compared across pond units, habitats, and alternatives, should be reliable indicators of relative habitat potential (rather than total habitat capacity).

In contrast, tidal flat densities were based on concurrent multiple-observer surveys, and were intended to count all individuals. Thus, tidal flat predictions are not directly comparable with salt pond and tidal marsh predictions and are treated separately here. However, our managed pond predictions are based on densities of foraging birds at high tide, when tidal flats are not available to shorebirds. Thus, while many of the same individual birds will feed on tidal flats at low tide, and in managed ponds at high tide, the two habitats represent separate foraging opportunities, both of which contribute to the energetic requirements of South Bay shorebird populations.

Our models were intended to provide a comprehensive evaluation of South Bay wetlands habitats, but did not consider other parts of the San Francisco Bay, nor any other geographic areas that may also be used by our focal species.

METHODS

STUDY AREA

Project Area

For this modeling exercise, the emphasis was on the restoration project area, defined as the 5,471 ha of salt ponds targeted for restoration and management (Figure 1). Most of the bird survey data were collected in and around the project area, i.e., south of the San Mateo Bridge. General habitats included were tidal marsh, managed ponds (managed for wildlife or salt production), and unvegetated intertidal and subtidal habitats within restored marshes.

South Bay

In our evaluation of South Bay-wide impacts of restoration, we considered the entire South San Francisco Bay south of Alameda Island, including existing and future restored tidal marshes, existing Cargill salt ponds, future ponds managed for wildlife, and tidal flats (Figure 1). We did not consider subtidal habitat except within the current salt pond areas, nor did we consider seasonal wetlands, managed freshwater wetlands, or upland habitats.

DATA COLLECTION

Salt Ponds

Bird Surveys

Salt pond bird densities were estimated from two sets of avian surveys conducted by PRBO (1999-2001) and USGS (2002-2004)[4], prior to the implementation of the Interim Study

[4] Assisted by PRBO staff in 2002 and 2003.

Period (ISP) management regime. PRBO surveys covered 21 ponds, 13 of which are now part of the Project area; the remaining eight ponds are still owned by Cargill Salt. USGS surveys covered all 54 ponds contained in the Project area (Figure 1, study area map). Conditions during this period were highly variable, encompassing a broad range of pond depth and salinity ranges (Table 1). Monthly surveys were conducted from October 1999 through February 2000, September 2000 through April 2001, and November 2002 through January 2004. Summer surveys (June, July) in 2003 were excluded from our analysis. Surveys were not conducted on bad weather days (due to access issues on the salt pond levee roads).

All birds using a pond were counted during each survey, with behaviors (foraging, roosting, and other) assigned to each group of birds recorded. USGS surveys were partitioned by smaller grid cells (Takekawa and Athearn 2006), but for our purposes, surveys were grouped at the pond level for comparability with PRBO survey data. PRBO surveys were conducted at both high and low tide, while USGS surveys were only conducted at high tide. Thus for modeling purposes, we only included high tide surveys during which ponds were not dry. For shorebirds, salt pond use is much higher at high tide due to the availability of mudflats at low tide, while other waterbird groups exhibit little difference between tides (Warnock et al. 2002, Stralberg et al. 2003). Further details on salt pond survey methods are provided in Warnock et al. (2002), Stralberg et al. (2003), and Takekawa and Athearn (2006).

Salinity Measurements

Pond salinity was measured on the same days that birds were surveyed, using the temperature and specific gravity of water samples to obtain a salinity concentration in parts per thousand (ppt). Two to four samples from different pond locations were averaged to obtain a mean salinity value for each survey (Appendix 1).

Water Depth

Salt pond water depth metrics were calculated using two sources of bathymetric data: (1) USGS boat-based depth soundings, and (2) USGS Light Detection and Ranging (LiDAR) data (Foxgrover and Jaffe 2005) for dry ponds that could not be surveyed by boat. Boat-based depth measurements were interpolated at a 5-m pixel resolution across all ponds, using an inverse-distance-weighted (IDW) algorithm (power = 2, search radius type = variable, 200 points/null maximum distance). Best available elevation surveys (USGS or Cargill) of water depth gauges, as well as gauge readings corresponding with the water depth data, were used to translate water depths into absolute elevations (NGVD29 vertical datum). Water depths during each bird survey could then be calculated by adding the value of the gauge reading to the elevation surface, with an adjustment for gauge elevation. Similarly, 1-m LiDAR-based digital elevation models (DEM) were converted to the same datum (NGVD29) as the boat-based elevation surfaces, and staff gauge readings from the closest corresponding dates to our bird surveys, were used to convert from absolute elevation to depth (see Takekawa et al. 2005 for adjustment factors). USGS bird surveys were accompanied by staff gauge readings that could be used directly. For 1999-2001 PRBO surveys, we obtained staff gauge readings from Cargill Salt and used the reading from the date closest to that of our survey.

We developed spatially-explicit bathymetric surfaces at a 5-m pixel resolution for each pond and each survey period. Those bathymetric surfaces were summarized to obtain the mean depth, shallow (< 15 cm) proportion, and deep (> 1 m) proportion of each pond for the month corresponding to each survey.

Tidal Marsh

Bird Surveys

Tidal marsh bird densities were estimated from fall, winter, and spring area surveys conducted by PRBO between September 1999 and April 2001, and from breeding season point count surveys conducted between March and May from 1999 to 2004. Several marshes were highly modified, and in some cases consisted of remnants along major tidal channels (e.g., Ravenswood Slough). Most of our study marshes were small remnant sites lacking significant tidal pannes and ponded areas, with most ponded areas forming along roads and levees.

Area surveys, which were used to estimate waterbird densities, were conducted at 12 South Bay marshes. Due to do limitations of access and detectability, we did not survey entire marshes, but sub-sampled our study sites. Because we cannot assume 100% detectability in vegetated tidal marshes, our counts must be considered an index of abundance or an *apparent density* estimate.

Point count surveys (Ralph et al. 1993), which are better suited for estimation of passerine densities, were conducted at 102 point count stations in 14 tidal marshes in the South Bay. Survey points were placed 200 m apart along permitted access routes, i.e., on peripheral levees and boardwalks. Observations within a 50-m radius were included in this analysis. Further details on point count survey methods can be found in Stralberg et al. (2003).

Tidal Marsh Habitat Characteristics

For characterization of tidal marsh habitat, we used large-scale (1:4800), high-resolution (scanned at 0.167-m pixel resolution) color-infrared photos (flown at high tide in August 2001) to map channels and natural salt ponds within our tidal marsh study sites. We used ArcInfo 8.1 (ESRI 2001) to digitize ponds and channels, classifying the channels by width category. Resulting pond and channel GIS layers were clipped to the survey areas and used to calculate pond and channel metrics for each survey marsh (Table 2).

Landscape Characteristics

To characterize salt pond and tidal marsh landscape context, we used a composite land use GIS layer, comprised of data from the San Francisco Estuary Institute's EcoAtlas (SFEI 1998) and the U.S. Geological Survey's Midcontinent Ecological Science Center (USGS MESC 1985). Within a 1-km buffer distance of each site (pond and marsh), we used ArcView's Spatial Analyst extension (ESRI 1999) to calculate the proportion of marsh, salt pond, tidal flat, urban development, and other upland land uses around that salt pond or tidal marsh (Tables 3-4).

Tidal Flats

To estimate shorebird densities on intertidal mudflats of the South Bay, we used data from PRBO's Pacific Flyway Project (Page et al. 1999). Between April 1988 and April 1993, PRBO biologists, with the help of hundreds of volunteers, conducted three fall and six spring shorebird censuses in the intertidal portion of San Francisco and San Pablo Bay and associated wetlands (Stenzel et al. 2002). This was part of a larger project whose primary goal was to obtain an overview of shorebird abundance and distribution in wetlands of the Pacific Coast of the contiguous United States (Page et al. 1999). The surveys were designed to minimize the double counting of flocks and keep track of all flock movements among plots, and involved hundreds of observers simultaneously conducting surveys at multiple locations during moderately high rising tides. Fall and spring densities were calculated by survey area (three South Bay tracts ranging in size from 1100 to 2300 ha) and averaged across years. Winter densities were based on roost counts that were assigned to tidal tracts based on fall proportions.

SELECTION OF FOCAL SPECIES

While our Phase I work focused on taxonomic and functional groups of birds, we recognized that several species within the same group may respond differently, and individual species' responses could be obscured by lumping several species together for analysis. At the same time, we knew that evaluating every species recorded with sufficient sample sizes would not only be time-consuming, but would yield results hard to synthesize and interpret. Thus, for Phase II, we developed a method for the selection of focal species by combining numerical abundances, preliminary habitat relationships, conservation status, and a published clustering technique for indicator species identification called Two-way Indicator Species Analysis (TWINSPAN) (Hill 1979).

First, we produced a list of all birds of numerical or ecological significance in either salt pond or tidal marsh habitat (Table 5). Given the relatively low number of breeding tidal marsh-dependent species, we chose to include all of them. Because the winter season accounts for the most consistent habitat use across a wide range of species, and because our data collection was more comprehensive during the winter months, we focused primarily on wintering waterbirds, and selected focal species for that period. However, for shorebird species, we also looked at fall and spring migration for numerically important shorebird species for which habitat availability within San Francisco Bay may be a limiting factor during migration. Thus, we modeled spring and/or fall relationships for shorebird species that were more abundant in the migratory seasons than in winter, according to Stenzel et al. (2002).

Winter focal species were required to be one of the 40 most abundant species in either tidal marsh or salt pond habitats (Stralberg et al. 2003), or of special conservation status, such as the federally listed Snowy Plover (*Charadrius alexandrinus nivosus*)[5] and Clapper Rail (Table 5). We then grouped species into the following broad taxonomic groups: shorebirds, waterfowl and coot, other waterbirds (grebes, terns, cormorants, pelicans, and gulls), herons and egrets, and raptors and corvids. Herons and egrets, which had similar tidal marsh and salt pond densities, were modeled as a group, rather than selecting focal species, while raptors and corvids did not have adequate data for modeling and were excluded. For each of the remaining three groups, we performed a TWINSPAN (Hill 1979) analysis to separate the species into four representative groups per taxon, based on winter high tide abundance (Table 6). These groups represented species that tended to be similar both in their densities and spatial distribution in the salt ponds and/or tidal marshes, suggesting that their habitat relationships were similar. Within these groups

[5] All Snowy Plover species references within this report refer to subspecies *C. a. nivosus*, often called the Western Snowy Plover.

of similar species, we selected one or more focal species for modeling purposes. With our salt pond and tidal marsh datasets, we log-transformed density estimates for each species and conducted regression analyses, using salt pond salinity, tidal marsh pond/channel characteristics, and surrounding landscape characteristics as independent variables. We selected the species with the highest model R^2 in each habitat (salt pond and tidal marsh) from each group, based on the premise that these species would be better indicators of change within those habitats.

From this list, we removed species that are not thought to depend on either salt pond or tidal marsh habitat for foraging (Western Gull [*Larus occidentalis*], Herring Gull [*L. argentatus*], Ring-billed Gull [*L. delawarensis*], Double-crested Cormorant [*Phalacrocorax auritus*], Western/Clark's Grebe [*Aechmophorous* spp.], and Black-bellied Plover [*Pluvialis squatarola*]). We also removed a species with similar densities in salt pond and tidal marsh (American Wigeon [*Anas americanus*], based on the premise that it would experience little effect of restoration. Finally, we added species of conservation concern based on a declining population trend or special conservation status. Each of these focal species was modeled in both tidal marsh and salt pond habitats unless their mean densities in that habitat were below 0.01 birds/ha (Table 6).

Because we assumed that foraging habitat, but not roosting habitat, is limited for waterbirds in the South Bay, we restricted our analyses to foraging waterbirds, excluding roosting and other non-foraging observations. For resident tidal marsh landbirds and rails, our density estimates were based on breeding season surveys, and were assumed to provide an index of breeding densities (Nur et al. 1997).

DATA ANALYSIS

Core Depth and Salinity Range Analysis

To facilitate further understanding of salt pond conditions most suitable for each focal species, we calculated core salt pond salinity and depth ranges for each species, illustrated with two-way standard deviation plots. We defined the core depth and salinity ranges to be equal to the mean plus or minus one standard deviation of all feeding observations of a species. For this purpose, an observation of a species was considered a single pond count containing at least one feeding individual of that species. That is, depth and salinity means and standard errors were based on the set of ponds and dates used by a species; they were not weighted by species abundance.

Model Selection

For each focal species and group, as well as each habitat type (salt pond and tidal marsh) we delineated a list of candidate site and landscape variables that we believed could influence the density of birds using an area (Tables 7-8). Due to large interannual differences in salt pond bird numbers, which were collected over a four-year period, we included a year effect with the list of salt pond variables. Subsequently, using this list of candidate variables, we constructed all possible linear models (i.e., all possible combinations of candidate variables), with the log-transformed density of each focal species as the response variable. We used natural log (density + 1) as the transformation. We used log transformation not only because densities tended to be lognormally distributed, but also because we considered density to be influenced in a multiplicative fashion by the independent variables rather than an additive fashion (Nur et al. 1999). Thus we assumed that a unit increase in a predictor variable resulted in a proportionate (percentage) increase or decrease in density rather than in a fixed increase or decrease in number of birds ha^{-1}. Whereas the magnitude of the constant to be added (to prevent taking the log of zero) is somewhat arbitrary, we carried out preliminary analysis to confirm that adding +1 was most suitable over the range of all species analyzed. In this way, we were able to use the same modeling approach for all species.

This analysis was performed uniquely for each habitat and season that a focal species was of interest. The process was automated using a custom program written in SAS, using Proc Mixed. All variables were considered fixed effects, and a normal distribution of the residuals (of log-transformed values) was assumed. For tidal marsh models, which were based on data from just 12 sites, we limited the models to one site variable (either linear channel density, channel area, or pond proportion) and two landscape variables.

For each candidate model analyzed, we calculated a weight based on the adjusted Akaike Information Criterion (AICc), a measure of model suitability and parsimony. This criterion quantifies model fit but also penalizes more complex models, thus quantifying the trade off between simplicity and model fit (Burnham and Anderson 2002). For each focal species-season-habitat combination analyzed, the combined AICc weights of all models summed to one, with the most parsimonious (or “best”) model having the highest weight. The entire suite of models (for each species-season-habitat) was used to predict log-transformed density within a pond unit, by calculating the prediction of each model, and producing a weighted average of predictions, based on the AICc weights. The predictions were generated using a custom program developed in Visual Basic for Applications (VBA) for ArcGIS 9.1. The model-averaged standard errors for each prediction incorporated not only error around the prediction but also the model uncertainty (Burnham and Anderson 2002).

Assessing Variable Importance and Model Fit

The importance of each site and landscape variable was assessed for the full suite of models used for prediction. For each variable, we calculated an importance score by summing up the AICc weights of each model in which it occurred. Thus, variables that appeared in all of the high-weight models had importance scores close to 1; those that appeared only in low-weight models had scores close to 0. To assess the explanatory power of each variable, we calculated partial R^2 statistics for each of the variables contained in the "best" (highest AICc weight) models.

SCENARIO EVALUATION

Project Area

We evaluated three alternatives (A, B, C) developed by PWA and HTH for two time periods: year 0 and year 50. Each year 50 alternative was evaluated using two different assumptions about the formation of tidal marsh ponds/pannes provided by HTH: (1) extensive development of tidal marsh ponds/pannes, characteristic of high elevation ancient marshes ("max"); and (2) less extensive development of tidal marsh ponds/pannes ("min"). These alternatives varied in the ratio of restored tidal marsh to managed ponds, as well as the managed pond characteristics (Table 9). However, tidal marsh characteristics did not differ across alternatives. That is, the development of an individual tidal marsh was assumed to be the same at year 50, regardless of how many other South Bay ponds were restored to tidal action. Most restored ponds were assumed to be completely vegetated by year 50, but small amounts of intertidal mudflat and subtidal habitat were predicted to remain within the restoration area. Predicted conditions at year 0 within the restored ponds (post-breach) consisted mostly of intertidal mudflat and subtidal habitats. Any areas predicted to be vegetated in year 0 were not considered bird habitat, as they were assumed to be *Spartina*-dominated without any channel development. Interannual variation, as represented by the year effect in our salt pond models, was addressed by assuming that conditions under each alternative were a composite of all four years surveyed (i.e., each of the four years was given a weight of 0.25).

For each alternative we used the suite of models described above to produce model-averaged density predictions and error bounds (by focal species / group, habitat, and season) at the pond level. These density estimates were then translated into predicted, lower (-1 SE), and upper bound (+1 SE) abundance estimates for each pond (by back-transforming log-transformed density to obtain density, and multiplying by pond area). For subtidal and intertidal portions of restored marshes, we did not use managed pond or tidal marsh models, but instead applied constant mean densities (±1 SE) derived from very low salinity (<45 ppt) salt pond surveys from the datasets described previously. For subtidal and intertidal habitats, we calculated mean densities across all salt ponds described as "deep" and "shallow," respectively (Table 8). We used geometric means of these surveys to account for the high variation among surveys and to make the densities comparable to the model-derived density predictions, described above.

Abundance predictions were summed up across ponds within the project. Due to the asymmetry of standard errors on abundance predictions that are based on log-transformed density models, upper and lower bounds were obtained by calculating separately the upper and lower standard error estimates. Each standard error was calculated by taking the square-root of the sum, across all ponds, of the squared difference between the prediction and lower or upper bound. For example, the lower bound standard error was calculated as

$\sqrt{\sum_{i=ponds}\left(\overline{x_i} - \sigma_{i,lower}\right)^2}$, where $\overline{x_i}$ is the predicted abundance of pond i, and $\sigma_{i,lower}$ is the lower bound of that prediction based on 1 standard error.

For each species and season, the project-level abundance index for each alternative and each year was divided by the abundance index for alternative A, year 0, which was considered the "baseline" condition. This index of proportional change (and associated 90% confidence interval) was compared across species and alternatives to assess general trends in restoration responses within the project area.

Evaluation of Sensitive Species with Sparse Data

For two species, the Clapper Rail and the Snowy Plover, data available were limited and difficult to model directly with habitat and landscape features. Instead of modeling densities with respect to site and landscape characteristics, mean values were estimated from published South Bay surveys and then extrapolated over the available habitat provided by each alternative.

Clapper Rail density estimates were based on two years (2005, 2006) of surveys conducted over a large portion of the San Francisco Estuary. We used methods identical to those used to estimate relative density for landbirds, except that the radius of detection for Clapper Rail was assumed to be 200 m rather than 50 m. We used only sites surveyed south of San Pablo Bay that were larger than 10 ha. We calculated the mean density, as well as 90th and 10th percentiles of density estimates to represent the mean, upper and lower density estimates, respectively, that might be observed in restored marshes. These densities were then multiplied by the amount of tidal marsh habitat within each alternative to produce upper and lower estimates for a predicted Clapper Rail abundance index. Relative density is generally an underestimate of actual density and can be prone to additional biases (Nur et al. 1997, Buckland et al. 2001). However, the upper and lower estimates should represent a reasonable range for possible Clapper Rail numbers.

For the Snowy Plover, San Francisco Bay surveys showed dramatic spatial and annual variability in density (Appendix 2). However, many of the low density sites were most likely regions of poor plover habitat, which would not be characteristic of ponds managed specifically for Snowy Plover. Thus we made the assumption that after 50 years, all ponds designated as “summer dry” ponds (Table 9) would be managed to produce quality Snowy Plover habitat and densities would be similar to those found in San Francisco Bay in the 1980s (Feeney 1991) or recently in Moss Landing at a site managed specifically for high quality Snowy Plover habitat (PRBO unpubl. data). Thus we used mean nesting densities from Feeney (1991) as an upper density estimate, and mean nesting estimates across all sources as a lower density estimate. These mean and upper densities were then multiplied by the amount of habitat designated as “dry pond” within each alternative to produce upper and lower estimates for a predicted Snowy Plover abundance index.

South Bay

To evaluate the overall change in abundance predicted for each alternative over the entire South San Francisco Bay, we first added the estimated total abundance index for each species (and season) *within* the project area to an abundance index for that species (and season) *outside* the project area in the South Bay. The abundance indices for areas outside the project areas were derived by multiplying areas of existing tidal marsh and Cargill-managed salt ponds by mean density estimates for those habitats based on our survey datasets. Cargill densities were based solely on surveys conducted within Cargill ponds between 1999 and 2001 (Table 10). Tidal marsh mean densities were based on all available South Bay tidal marsh survey data (PRBO unpubl. data) (Table 11). We used geometric means of these surveys to account for the high variation among surveys and to make our approach comparable to that of the project analysis described above.

As was done within the project area, the overall South Bay abundance index for each alternative and each year was divided by a South Bay abundance index for alternative A, year 0, which was considered the "baseline" condition. This index of proportional change (and associated 90% confidence interval) was compared across species and alternatives to assess general trends in restoration responses at the South Bay level. If, for a particular species, the 90% confidence interval overlapped 1.0, then we inferred no change in that species' South Bay abundance index. A 90% confidence interval with a lower bound greater than 1.0 was an interpreted as an increase, while an upper bound less than 1.0 was interpreted as a decrease.

Evaluation of Intensive Management Actions

To assess the sensitivity of the system to various management options, and compare the effects of pond management with the effects of tidal marsh engineering, we varied two different parameters within the alternatives. First we evaluated the South Bay-wide proportional changes in focal species using two different levels of tidal marsh "pondedness" based on two potential scenarios provided to us by the consultant team as "minimum" and "maximum" levels of natural pond formation. Next we varied managed pond depth, comparing the given managed pond characteristics for each alternative with two hypothetical alternatives. The first was designed to create optimal shorebird habitat within the ponds by managing all ponds to be 50% shallow (<15 cm water depth), 50% intermediate (15 cm – 1 m water depth) and 0% deep (>1 m water depth), holding salinity constant ("shallow" ponds). The second was designed to create deeper foraging habitat for fish-eaters and diving ducks by managing all ponds to be 75% deep, 23% intermediate, and 2% shallow, holding salinity constant ("deep" ponds). The "shallow" ponds were assumed to have a mean depth of 0.15 m; the "deep" ponds were assumed to have a mean depth of 1 m. These parameters were based on the characteristics of individual ponds included in restoration alternative B.

Evaluation of Changes in Tidal Flat Shorebird Habitat Potential

Due to differences in field survey and density estimation methods, as well as the daily movement of shorebirds among tidal flats, salt ponds, and other bay habitats, we did not attempt to sum or otherwise combine tidal flat estimates with predictions for managed ponds and tidal marshes within the restoration area. However, we did multiply predictions of future tidal flat area provided by PWA (Table 12) by mean fall and spring tidal flat shorebird densities (Table 13), in order to estimate the percent change in abundance for each shorebird focal species/season. In contrast with managed pond and tidal marsh predictions, we did not model densities based on site or landscape conditions, but used mean densities (over 5 years from 1998 to 1993), calculated separately for each bridge-bounded tidal flat area in the South Bay (Figure 1).

RESULTS

CORE SALT POND DEPTH AND SALINITY RANGES

Our analysis of salt pond depth and salinity ranges used by each focal species revealed that most non-phalarope shorebird species used a broader range of salinity conditions than depth conditions (Figure 2), while other waterbirds species tended to use a similar range of salinities and depths (Figure 3). Mean (± SD) depths varied from 0.025 ± 0.30 m (Snowy Plover) to 0.69 ± 0.71 m (Wilson's Phalarope [*Phalaropus tricolor*]) across shorebird species (Figure 2), and from 0.34 ± 0.44 m (Forster's Tern [*Sterna forsteri*]) to 0.98 ± 0.61 m (Eared Grebe [*Podiceps nigricollis*]) across other waterbird species (Figure 3). Mean (± SD) salinities varied from 75 ± 56 ppt (Semipalmated Plover [*Charadrius semipalmatus*]) to 103 ± 43 ppt (Red-necked Phalarope [*Phalaropus lobatus*]) across shorebird species (Figure 2), and from 42 ± 13 ppt (Northern Pintail [*Anas acuta*]) to 84 ± 37 ppt (Eared Grebe) across other waterbird species (Figure 3).

SITE/LANDSCAPE HABITAT ASSOCIATIONS

Salt Pond Models

Salinity (linear and quadratic terms) and depth variables (mean depth and shallow proportion) were generally the most predictive of apparent bird densities in salt pond focal species models, with one or more variables in each category entering into most top models for most species (Figures 4-7). In general, depth variables were more important than salinity variables. Pond size (ha) was also important for many species, especially shorebirds, with mostly negative relationships. Landscape variables were generally less important, less commonly entering into all top models for a species. Comparing the single top models for each species (based on the lowest AICc value), salinity and depth variables also had the highest partial R^2 statistics for most species (Table 14). Model R^2 values ranged from 0 for winter Gadwall (*Anas strepera*) to 0.63 for Western Sandpiper (*Calidris mauri*) in spring. While some species' models had high explanatory power, high variability in bird densities generally led to low R^2 values.

Group models generally had higher explanatory power than individual focal species' models, ranging from $R^2 = 0.08$ for fish-eaters to $R^2 = 0.81$ for spring shorebirds (Table 15). Individual model variables also generally had higher explanatory power (in terms of partial R^2 values) in group models (Table 15). Depth and salinity variables were most important across groups, with salinity having the highest partial R^2 for winter dabbling ducks (0.42) and small shorebirds (0.28). Shallow proportion was the variable with the highest explanatory power for spring small shorebirds (0.50), while mean depth was highest for fall small shorebirds (0.51). The year effect (0.15), followed by salinity (0.13), was most important for diving ducks, while the proportion of natural upland was most important for waders (0.17).

Small Shorebirds

For small shorebirds, the variables with highest support were pond size (negative effect), mean pond depth (negative effect), proportion of shallow habitat within a pond (positive effect), and, in spring, salinity (negative quadratic effect) (Figure 4). All species had mean depth and/or shallow proportion in their top models across all seasons, but there was only strong support for the salinity variables in the spring models. The explanatory power of both depth and salinity (expressed as partial R^2) was highest during the spring season (Table 14). Surrounding tidal marsh proportion (negative effect) and mudflat proportion (positive effect) were important for the Least Sandpiper (*Calidris minutilla*) in spring.

Large Shorebirds

Large shorebirds also responded to depth and salinity, with strong support for the depth variables across all species, but only Black-necked Stilt (*Himantopus mexicanus*) had salinity (quadratic response) in all of its top models (Figure 5). Pond size (negative effect) and surrounding mudflat proportion (positive effect) were important variables for American Avocet (*Recurvirostra americana*) and Willet (*Tringa semipalmatus*) in winter, while surrounding tidal marsh proportion (positive effect) was important for Willet in fall and winter. The explanatory power of individual variables (as measured by partial R^2 values), was generally lower for large shorebirds than for small shorebirds (Table 14).

Dabbling and Diving Ducks

Waterfowl species displayed variability in their responses to site and landscape variables. The strongest support across all seasons was for models including salinity (negative effect, positive quadratic for Ruddy Duck [*Oxyura jamaicensis*]) (Figure 6). For Mallard (*Anas platyrhynchos*), pond size (negative effect), surrounding marsh proportion (positive effect), and mean depth (negative effect) had strong support, appearing in all top models. Landscape variables were less important, but surrounding tidal and non-tidal marsh proportion had relatively high support in Northern Shoveler models; Ruddy Duck exhibited a positive response to surrounding bay proportion. Explanatory power was not high for any the variables in waterfowl models (Table 14).

Fish-eaters, Eared Grebe, and Phalaropes

Depth and salinity also had strong support in several other waterbird models (Figure 7). For fish-eaters (American White Pelican [*Pelecanus erythrorhynchus*] and Forster's Tern), there was a fairly strong negative effect of salinity, while Eared Grebe exhibited a quadratic response to salinity, a positive response to mean depth, and positive responses to surrounding proportion of salt ponds and open bay. Variable support was not high for phalaropes, except for pond size (positive) for Red-necked Phalarope, and shallow proportion (positive) for Wilson's Phalarope. Variable explanatory power was generally low except for mean depth in Eared Grebe models (partial $R^2 = 0.29$) and pond size in Red-necked Phalarope models (partial $R^2 = 0.23$) (Table 14).

Tidal Marsh Models

Tidal marsh waterbird models were based primarily on categorical microhabitat type (pond/panne, channel, vegetation), which were forced into all models, but these models were also influenced somewhat by channel density, channel area, pond proportion, and surrounding landscape variables (Figures 8-10). For the single top models of each focal species, R^2 values were quite low, mostly less than 0.10, with a maximum of 0.15 for Greater Yellowlegs (*Tringa melanoleuca*) in winter (Table 16). Similarly, tidal marsh landbird models demonstrated weak effects of channel density as well as surrounding landscape composition (Figure 11). R^2 values ranged from 0 for the Common Yellowthroat to 0.50 for the Marsh Wren (*Cistothorus palustris*). Group models performed slightly better, with R^2 values up to 0.19 (Table 17).

Small Shorebirds

Small shorebirds were observed in higher densities within pond/panne and channel microhabitats (compared to the vegetation microhabitat) except for spring Western Sandpiper, which had lower densities in tidal marsh channels (Figure 8). There was also strong support for the pond/panne proportion variable in winter Western Sandpiper models (positive effect). In terms of landscape variables, the proportion of surrounding and mudflats and salt ponds were both present in most top models for winter Dunlin (*Calidris alpina*) and Western Sandpiper (positive effects). Explanatory power was generally low for small shorebird tidal marsh models (Table 16).

Large Shorebirds

Large shorebirds, similarly, had higher densities within the pond/panne and channel microhabitat variables (Figure 9). There was also strong support for channel area in winter Greater Yellowlegs models, and for pond/panne proportion in winter Black-necked Stilt models. There was not strong support for any landscape variable in large shorebird models. Overall, explanatory power of most variables was relatively low (Table 16).

Dabbling and Diving Ducks

Waterfowl were predicted to have higher densities within pond/panne and channel microhabitats than within the vegetation (Figure 10). Winter Gadwall and Northern Shoveler also had strong support for pond/panne proportion (positive effect) in their models. For Gadwall, surrounding tidal marsh and non-tidal marsh proportion were also important (positive effect). No variables had strong support in diving duck models except for the categorical microhabitat variables. Variable/model explanatory power was generally low across waterfowl species (Table 16).

Landbirds

For landbirds, variable support was generally low, except for small channels (negative effect) and surrounding open bay proportion (positive effect) for the Marsh Wren, and surrounding tidal marsh proportion (positive effect) for the Song Sparrow (Figure 11). Model explanatory power was also low except for the Marsh Wren and the previously named variables (Table 16).

SCENARIO EVALUATION

Modeling Caveat

It is important to remember that the models we have developed predict indices of survey-level foraging or breeding density and abundance (i.e., mean daily values). The predictions are based on the assumption that habitat is limiting for modeled species and was at carrying capacity when surveyed. Our predictions should not be interpreted as actual instantaneous or cumulative population numbers for the South Bay. Furthermore, because roosting birds were excluded from analysis, abundance indices will underestimate total numbers of all non-breeding species.

Project Area

Within the project area, across the three primary alternatives evaluated, alternative C resulted in the highest year 0 abundance indices for diving ducks, waders, medium shorebirds, and small shorebirds in winter, as well as small shorebirds in fall (Table 18). Alternative B had the highest spring small shorebird abundance index, and alternative A had the highest dabbling duck, fish-eater, and phalarope abundance index in year 0. In year 50, alternative C provided the most suitable conditions for landbirds and dabbling ducks, while alternative B was most suitable for all other species groups (Table 18).

In terms of focal species, alternative C resulted in the highest (not necessarily statistically significant) predicted abundance for most species (11 of 29) in year 0, followed by alternative A (9 species) (Table 19). In year 50, alternative B resulted in the highest predicted abundance for most species (22 of 29), followed by alternative C (6 species) (Table 19).

Alternative C resulted in the highest mean managed pond densities for dabbling ducks, medium shorebirds, and small shorebirds in winter, as well as phalaropes and small shorebirds in fall, and small shorebirds in spring (Table 18). Alternative A had the highest predicted managed pond density of fish-eaters, and alternative B had the highest predicted diving duck density. Predicted wader densities in managed ponds were very similar across alternatives.

Pond-level maps of predicted densities for most species and seasons can be found in Appendix 3.

Landbirds

Group and Focal Species Predictions -- Abundance

No landbirds (Common Yellowthroat, Song Sparrow, or Marsh Wren) were predicted to occur in the restoration areas in year 0, due to the lack of vegetation in the initial stage of restoration (Figures 12-15). We predict, in year 50, landbirds as a group would experience the largest increase in abundance under alternative C (Figure 12), with Song Sparrow showing the most potential for increase (Figure 13), followed by Marsh Wren (Figure 14). Common Yellowthroat showed similar responses as Song Sparrow to the alternatives, although the differences between the alternatives were not as large and had large error bounds associated with them (Figure 15). Alternatives A and B also resulted in increased abundance indices, somewhat in proportion to the amount of tidal marsh habitat available. Managed ponds provided no habitat potential for these species.

Group and Focal Species Predictions -- Density

In terms of overall predicted densities for landbirds as a group, alternative C provided the highest potential for suitable habitat, followed by alternatives A and B, which were quite similar (Table 20). Common Yellowthroat predicted density varied very little by alternative, while the highest predicted Marsh Wren densities were under alternative B, and the highest predicted Song Sparrow Density was under alternative C (Table 21). Landbird predictions were confined to vegetated tidal marsh (not ponds/pannes or channels). For all landbird and rail focal species, tidal marsh density predictions were the same for “max” and “min” assumptions about pond/panne formation.

Small Shorebirds

Group Predictions -- Abundance

In year 0, small shorebirds in the fall and winter had the highest predicted abundance index under alternative C, due to large areas of intertidal mudflat habitat within the restoration area (Figure 16, Figure 17). In the spring, however, alternative B performed best, with most habitat potential provided by managed ponds (Figure 18). In year 50, alternative B performed best across all three seasons, mostly based on managed pond habitat. “Max” and “min” tidal marsh pond/panne assumptions produced equivalent predictions except in the fall, when “max” predictions were higher. In general, the small shorebird abundance index was predicted to decrease between year 0 and year 50, except for Alternative B during the spring, for which the two endpoints had similar predictions.

Density

Group Predictions -- Density

In the winter, predicted small shorebird density was highest within the intertidal mudflat portion of restored marshes, while managed pond density was much higher in the spring (Table 20). In the fall, the two habitats were similar, and the ranking changed depending on the alternative. Overall, managed pond densities increased substantially from alternative A to alternative C, following the increase in shallow pond management intensity. Tidal marsh pond and channel bird densities were an order of magnitude lower than intertidal mudflat or managed pond bird densities. In the fall and winter, density predictions for the "max" tidal marsh pond/panne assumption were higher than for the "min" assumption within the same alternative.

Focal Species Predictions -- Abundance

In year 0, Alternative C was predicted to result in the highest abundance index for all four small shorebird species (Dunlin, Least Sandpiper, Semipalmated Plover, Western Sandpiper) during the winter season, followed by alternative B (Figures 19-22). The same was true for Least Sandpiper in the fall (Figure 23), and for all species except for Least Sandpiper (highest in alternative A) in the spring (Figures 25-27). The habitat feature that provided the highest potential increase in abundance was unvegetated intertidal mudflat within restored marshes. In year 50, managed ponds made the largest contribution to small shorebird abundance, and alternative B performed best for all of these species in all seasons, followed by alternative C. Tidal marsh open water habitats only played a significant role for Least Sandpiper (all seasons) and Western Sandpiper in the fall (Figure 24). Overall, habitat for small shorebirds was predicted to decrease between year 0 and year 50 within the restoration area.

Focal Species Predictions -- Density

Managed pond densities were much higher than tidal marsh densities for all of the small shorebirds except Least Sandpiper, for which similar densities were predicted in each habitat (Table 21). Intertidal mudflat densities were comparable to managed pond densities in most cases, and were higher than managed pond densities for Semipalmated Plover, Least Sandpiper, and Western Sandpiper in the fall (alternatives A and B), Dunlin in the winter (alternative A), and Dunlin and Western Sandpiper in the spring (alternative A),

In general, predicted densities within managed ponds increased with management intensity from alternatives A through C, except for the Least Sandpiper in spring, which had the highest predicted density under alternative B. For this species, tidal marsh densities were similar across alternatives. In the spring and winter, tidal marsh channels had higher predicted densities than ponds/pannes, which in turn had higher densities than vegetated areas. In the fall, ponds/pannes were predicted to have the highest Least Sandpiper densities. For Western Sandpiper, Dunlin, and Semipalmated Plover, pond/panne densities were generally higher than channel and vegetation densities. Thus, in most cases, predicted densities were higher under the "max" tidal marsh pond/panne assumptions than under the "min" assumptions.

Large Shorebirds

Group Predictions -- Abundance

In year 0, the highest large shorebird numbers were predicted under alternative C during the winter season. This was mostly due to the amount of intertidal mudflat habitat within the restoration area (Figure 28). However, at year 50, in the fall, alternative B ("max") performed best, followed by alternative C ("max"), which did slightly better than B ("min"). In the winter, alternatives B and C were equivalent. In alternative B, managed ponds provided the most habitat potential, while more birds were predicted in tidal marsh habitat in alternative C. Across all three alternatives, the abundance of large shorebirds was predicted to decrease between year 0 and year 50.

Group Predictions -- Density

In general, predicted winter densities were highest on intertidal mudflat portions of restored marshes, followed by managed ponds (Table 20). Tidal marsh pond/panne and channel densities were often comparable to managed pond densities, and were higher for some alternatives and seasons. Densities under "max" tidal marsh pond/panne assumptions were generally predicted to be higher than under "min" assumptions. Tidal marsh densities did not vary much by alternative.

Focal Species Predictions

In year 0, Alternative C yielded the highest predicted abundance for the American Avocet and Willet in winter (Figure 29, Figure 32) while alternative B performed best for Black-necked Stilt and Greater Yellowlegs in winter (Figures 30-31), and Willet in fall (Figure 33). In year 50, alternative B performed the best for all large shorebirds except Greater Yellowlegs, which had the highest predicted abundance under alternative C. Managed ponds provided most of the habitat potential for large shorebirds in year 50, although tidal marshes were more important for Greater Yellowlegs. Overall, abundances decreased from year 0 to year 50 for most large shorebirds (except Greater Yellowlegs).

For American Avocet, Black-necked Stilt, and Willet, predicted managed pond density increased with management intensity from alternative A to alternative C (Table 21). Managed pond densities were higher than tidal marsh densities for all of these species, but Black-necked Stilt had relatively high marsh pond/panne densities (higher under “max” assumptions than “min” assumptions), and Willet had relatively high predicted densities in tidal marsh channels. Greater Yellowlegs had highest predicted densities in channels, and much lower densities within vegetation and in ponds/pannes (Table 21). In the winter, Willet and American Avocet intertidal mudflat densities were higher than managed pond densities.

Phalaropes

Group Predictions -- Abundance

In year 0, fall phalarope abundance was predicted to be highest under alternative A, but very similar across alternatives, with large error bounds around all predictions (Figure 34). In year 50, predictions were also similar. Due to large error bounds around the predictions, no increases or decreases could be determined.

Group Predictions -- Density

Phalaropes were only predicted to occur in managed pond habitats, with highest densities predicted under alternatives A and C (Table 20).

Focal Species Predictions – Abundance

Predicted abundances of Red-necked and Wilson’s Phalaropes were highest in year 0 for alternative A, and highest in year 50 for Alternative B (Figure 35, Figure 36). Habitat availability for both species was largely dependent on the area of high salinity managed ponds. Thus the overall predicted abundance index remained unchanged between year 0 and year 50 for the action alternatives (B and C), but decreased under the no action alternative (A).

Focal Species Predictions – Density

In terms of density, Red-necked Phalarope density was predicted to be highest under alternative A and lowest under alternative C (Table 21), while the opposite was true for Wilson's Phalaropes (Table 21).

Dabbling Ducks

Group Predictions -- Abundance

In year 0, dabbling duck abundance was predicted to be highest under alternative A, followed by alternatives B and C (Figure 37). In year 50, however, our predictions of dabbling duck abundance under alternative A decreased, while alternatives B ("max") and C ("max") performed best, and predicted similar numbers of dabbling ducks.

Group Predictions -- Density

Predicted dabbling duck densities were highest in tidal marsh ponds and channels under "max" pond/panne assumptions, followed by managed ponds, then tidal marsh ponds and channels under "min" assumptions (Table 20). Tidal marsh vegetation densities were higher than both subtidal and intertidal mudflat habitats. The highest managed pond and tidal marsh densities occurred under alternative C.

Focal Species Predictions -- Abundance

In year 0, Northern Pintail had the highest predicted abundance under alternative C (Figure 40), while Gadwall did marginally better under alternative A (Figure 38), and Northern Shoveler did slightly better under alternative B (Figure 41). Mallard predictions were similar for alternatives B and C (Figure 39). By year 50, Gadwall and Northern Shoveler were predicted to do better under alternative C, while for Northern Pintail and Mallard, alternative B was marginally better. Tidal marsh and managed pond habitats were both important for dabbling ducks, the relative value of each varying by species. Across alternatives, predicted abundance stayed fairly similar between year 0 and year 50 for all species except Northern Pintail, for which a decline was predicted.

Focal Species Predictions -- Density

Within the dabbling duck group, density trends varied by species. Gadwall had much higher predicted tidal marsh densities than managed pond densities, with similar pond/panne and channel density predictions (Table 21). Mallard also had higher tidal marsh than managed pond densities, but for this species, marsh channels were more important than marsh ponds/pannes, and intertidal mudflat densities were similar to managed pond densities (Table 21). Northern Pintail had highest density predictions in intertidal mudflat areas within restored marshes, followed by managed ponds, then marsh channels and ponds/pannes (Table 21). Northern Shoveler had highest predicted densities in tidal marsh habitats, followed by managed ponds, subtidal areas, and intertidal areas within restored marshes. Among tidal marsh microhabitats, pond/panne and channel densities were similar, and both were higher under "max" than under "min" assumptions (Table 21). For Northern Shoveler, predicted tidal marsh channel, pond/panne, and vegetation densities increased from alternative A to alternative B to alternative C, whereas other species had similar densities across alternatives.

Diving Ducks

Group Predictions -- Abundance

In year 0, diving duck abundance was predicted to be highest under alternative C, followed by alternatives B and A (Figure 42). Most of the habitat capacity in alternative C was provided by intertidal mudflat habitats within the newly breached ponds. In year 50, however, when marshes were almost fully vegetated, alternative B performed best ("max" and "min" assumptions were equivalent). Overall, diving duck abundance was predicted to decrease between year 0 and year 50 under all three alternatives.

Group Predictions -- Density

Diving duck densities were predicted to be highest in subtidal portions of restored marshes, followed by intertidal mudflats, managed salt ponds, and tidal marsh channels (Table 20). Densities were very low in tidal marsh ponds and vegetated areas. In managed ponds, predicted densities of diving ducks were similar for alternatives B and C (and higher than in alternative A), whereas in tidal marsh channel and pond/panne habitats, densities were more similar in alternatives A and C (and higher than in alternative B).

Focal Species Predictions -- Abundance

At the onset of restoration (year 0), scaup (*Aythya* spp.) numbers were predicted to be highest under Alternative A (Figure 44), while Ruddy Duck predictions were very similar across all three alternatives (Figure 43). For both diving duck species, alternative B performed best in year 50. Managed ponds were responsible for most diving duck habitat potential, but tidal marsh open water (primarily subtidal portions of newly breached ponds) were also important in year 0, especially for Ruddy Duck.

Focal Species Predictions -- Density

Ruddy Duck densities were highest in subtidal portions of restored marshes, followed by managed ponds in alternative A, and intertidal mudflat portions of restored marshes (Table 21). Among tidal marsh microhabitats, channel densities were highest.

Herons and Egrets

Group Predictions -- Abundance

In year 0, winter herons and egrets had the highest predicted abundance under alternatives B and C, with subtidal and intertidal habitats in the restoration area compensating for the loss of managed ponds (Figure 45). In year 50, alternative B performed best and alternatives A and C were similar. Most of the habitat potential in alternatives A and B was provided by managed ponds, whereas alternative C habitat consisted mostly of tidal marsh open water (ponds/pannes and channels).

Group Predictions -- Density

The highest predicted heron and egret densities occurred in intertidal mudflat habitat and tidal marsh channels, followed by subtidal areas, and then managed ponds (Table 20). Tidal marsh pond and vegetation microhabitats had similar predicted densities. Comparing the “min” and “max” ponds assumption within the tidal marsh habitat, no differences were predicted for tidal marsh channels and vegetated areas, while “min” assumptions produced higher pond/panne densities.

Fish-eaters

Group Predictions -- Abundance

In year 0, fish-eaters had the highest predicted abundance under alternative A, with most of the habitat provided by managed ponds (Figure 46). For alternatives B and C diving ducks had similar predicted abundances in year 0, with subtidal habitats making up for the loss of managed ponds in alternative C. In year 50, alternative B performed best ("max" and "min" assumptions were equivalent), with most of the habitat still provided by managed ponds. Similar to diving ducks, the number of fish-eaters was predicted to decrease between year 0 and year 50 under all three alternatives, as the amount of open water habitat decreases.

Group Predictions -- Density

Highest potential densities were predicted in subtidal portions of restored marshes, followed by managed ponds, intertidal portions of restored marshes, and tidal marsh channels (Table 20). Tidal marsh pond and vegetation densities were very low. Alternative A had the highest managed pond densities, while tidal marsh channel densities were similar across all alternatives.

Focal Species Predictions -- Abundance

In year 0, we predicted the highest abundance for American White Pelican within alternative A (mostly managed ponds), while alternatives B and C (mostly subtidal portions of restored tidal marsh) were similar (Figure 47). Forster's Tern, however, had the highest predicted abundance under alternative C (mostly intertidal and subtidal portions of restored tidal marsh) (Figure 48). In year 50, alternative B appeared to be best for both of these species. However, across all alternatives, substantial decreases in abundance were predicted between year 0 and year 50.

Focal Species Predictions -- Density

Predicted American White Pelican densities were highest in subtidal habitat, followed by managed ponds and intertidal mudflat portions of restored marshes (Table 21). Forster's Tern densities were highest in intertidal mudflat portions of restored marshes, followed by subtidal habitat and managed ponds.

Eared Grebe

In year 0, alternative A predicted the highest abundance index for Eared Grebe, with major reductions predicted under alternatives B and C (Figure 49). By year 50, alternative B, which contained the most managed pond habitat, performed best of the three. Eared Grebe were only predicted to occur in managed ponds, with the highest densities also occurring in alternative B, which contained deeper ponds (Table 21). However, across all alternatives, large decreases in Eared Grebe abundance were predicted.

Clapper Rail

We obtained lower and upper breeding density estimates of 0.06 birds/ha and 1.15 birds/ha, respectively, yielding predicted abundance indices ranging from estimates of 108 to 2,063 birds within the project area under alternative A to estimates of 274 to 5,256 birds under alternative C (Table 22). Based on our estimate of currently available tidal marsh habitat in South San Francisco Bay, we estimated that other tidal marsh areas outside the project would contribute an additional 234 to 4,487 Clapper Rail, based on the lower and upper density estimates, respectively.

Our mean density estimates of 0.54 birds/ha led to predictions within the restoration area ranging from 969 birds under alternative A to 2,468 birds under alternative C, and another 2,107 birds in the rest of the South Bay.

Snowy Plover

Predicted abundance indices were highest for alternative B, ranging from 135 birds using the minimum density estimate (0.2 birds/ha) to 945 birds using the maximum density estimate (1.4 birds/ha) (Table 23). Alternative C contained no dry pond habitat, and thus resulted in predictions of 0 birds under both density assumptions. Using the maximum density estimate (1.4 birds/ha), approximately 360 ha of dry pond habitat would be needed throughout the South Bay in the summer to support the current goal of maintaining a population 500 Snowy Plovers in South San Francisco Bay (Table 23). Using the minimum estimate (0.2 birds/ha), 2500 ha of dry pond habitat would be required to maintain a population of 500 breeding Snowy Plovers.

Proportional Change

Comparing each alternative with the “baseline” (alternative A, year 0), and evaluating the 90% confidence intervals of predicted proportional change, alternatives B and C had the same number of species/seasons (out of 26) predicted to increase (12), and almost the same number predicted to decrease (3 and 4, respectively) in year 0 (Figure 50). However, the magnitudes of proportional change (positive or negative) were generally much greater under alternative C. In year 50, alternatives B ("max" ponds) and C ("max" ponds) predicted the same number of increases (seven species/seasons, compared to five for alternative A), but alternative B ("max") predicted the fewest decreases (eight species/seasons, compared to 16 under alternative A ("max") and 13 under alternative C "max" (Figure 51). The directions of change were similar under the "min" tidal marsh pond assumptions, except for fall Least Sandpiper and winter Black-necked Stilt, which decreased rather than stayed the same, and winter Northern Shoveler, which stayed the same rather than increased (Figure 52) in alternative C.

By year 50, under alternatives B and C ("max"), the highest proportional increase was predicted for breeding (spring) Song Sparrow, followed by breeding Marsh Wren, breeding Common Yellowthroat, winter Greater Yellowlegs, and winter Gadwall (Table 24). Under alternative B, the highest proportional decrease was predicted for spring Least Sandpiper (-85%), followed by winter American Avocet (-71%), fall Red-necked Phalarope (-68%), and winter Eared Grebe (-64%) (Table 24). Under alternative C, the highest proportional decrease was predicted for fall Red-necked Phalarope (-98%), followed by winter Eared Grebe (-93%), spring Least Sandpiper (-90%), and winter Forster's Tern (-89%) (Table 24).

Across all alternatives, by year 50, landbirds and dabbling ducks mostly increased; diving ducks, Eared Grebe, phalaropes, and fish-eaters mostly decreased; and shorebirds had mixed changes.

South Bay

Comparing alternative B with the "baseline" (alternative A, year 0), eight out of 26 species were predicted to increase and two were predicted to decrease in year 0 (Figure 53). For alternative C, ten species were predicted to increase, while four were predicted to decrease in year 0 (Figure 53). The magnitudes of proportional change (positive or negative) were generally much greater under alternative C. In year 50, alternative C ("max") predicted the most increases (five species/seasons, compared to four for alternatives A and B), but alternative B ("max") predicted the fewest decreases (eight species/seasons, compared to 14 under alternatives A ("max ") and C "max") (Figure 54). There was no difference in the direction of change between the "min" and "max" tidal marsh open water assumptions, except for Northern Shoveler, which, under alternative C, was predicted to increase under "max" assumptions, but stay the same under "min" conditions.

By year 50, under alternatives B and C, the highest percent increase was predicted for winter Gadwall (305% and 533%, respectively), followed by winter Greater Yellowlegs (115% and 168%), and breeding Song Sparrow (101% and 179%), Marsh Wren (84% and 113%), and Common Yellowthroat (73% and 121%) (Table 25). Under alternative B, the highest percent decrease was predicted for fall Red-necked Phalarope (-63%), followed by winter Forster's Tern (-57%), winter American White Pelican (-55%), and spring Least Sandpiper (-50%) (Table 25). Under alternative C, the highest percent decrease was predicted for fall Red-necked Phalarope (-90%), followed by winter American White Pelican (-81%), Forster's Tern (-80%), and Ruddy Duck (-68%) (Table 25).

Across all alternatives, by year 50, landbirds increased, diving ducks and Eared Grebe decreased, phalaropes and fish-eaters mostly decreased, dabbling ducks mostly increased, and other groups had mixed changes.

Evaluation of Intensive Management Actions

Comparing predicted proportional change for the "shallow" management regime (all managed ponds 50% shallow) with the management regimes provided by the consultant team, the "shallow" regime did better for several species under alternative B, but did not make much difference under alternative C (Figure 55). Specifically, species that switched from a decrease to an increase were spring Dunlin, spring and fall Western Sandpiper, and fall Willet. Winter Western Sandpiper, Dunlin, and American Avocet went from a decrease to no change. Under alternative C, only fall Western Sandpiper and Spring Dunlin went from a decrease to no change.

The "deep" management regime resulted in no new predicted increases, although under alternative B, one species, Eared Grebe, went from a predicted decrease to no predicted change (Figure 56).

Evaluation of Changes in Tidal Flat Shorebird Habitat Potential

Geomorphic modeling (PWA 2006) predicted a decline in tidal flat area over a 50-year time period, regardless of restoration alternative (Table 12). Consequently, the total number of shorebirds predicted in tidal flat habitat decreased somewhat in proportion to the loss of tidal flats, which was greatest under Alternative C, followed by Alternative B. However, due to variations in individual species' densities across tidal tract sections, the proportional changes varied (Table 26). Yellowlegs, phalaropes, and Willet were predicted to have the largest proportional change (decrease) across all three alternatives (from 0.46 of baseline under alternative A to 0.42 of baseline under alternative C). American Avocet had the smallest proportional change (from 0.99 of baseline under Alternative A to 0.60 of baseline under alternative C). In terms of numerical abundance, the highest potential for declines would be for spring Western Sandpiper, Least Sandpiper, and Dunlin (grouped during surveys), for which the difference in the predicted abundance index for year 0 vs. year 50 was over 300,000 birds (a 53% decrease from year 0) (Table 27).

DISCUSSION AND RECOMMENDATIONS

FOCAL SPECIES

Given the diversity of bird species that use San Francisco Bay (Bollman 1970, Takekawa et al. 2001, Stenzel et al. 2002) and the complexity of responses to various habitat and landscape features (Stralberg et al. 2003), we found the focal species approach useful for identifying species that are likely to represent the responses of other similar species to restoration. In this study, we did not assert that the species we selected are “indicator” species in the sense that they reflect ecosystem responses and represent other non-avian taxonomic groups (Lawler 2003), overall biodiversity (Noss 1990), natural communities (Kremen 1992), environmental changes (Temple and Wiens 1989), or ecosystem health (Hilty and Merenlender 2000). We modeled the responses of bird species and groups as conservation targets in their own right, and thus used the term “focal” species in a broad sense -- bird species of special interest, many of which represent other similar bird species (Lambeck 1997, Chase and Geupel 2005).

Focal species models allowed us to describe a broader range of potential responses than those predicted for taxonomic groups, and identify species that are likely to respond differently from the group to which they belong. For example, within small shorebirds, the Least Sandpiper, particularly during fall and spring migration, had higher predicted tidal marsh densities than the other species, resulting in similar predictions for alternatives B and C in year 50, whereas alternative B clearly performed better for other small shorebirds. A similar pattern was seen with Greater Yellowlegs and Willet, compared to other large shorebirds. For other groups, including the landbirds, dabbling ducks, diving ducks and fish-eaters, focal species patterns tended to reflect those of the overall group. In general, group models performed better than focal species models, in terms of overall and partial R^2 values.

SITE/LANDSCAPE HABITAT ASSOCIATIONS

While the primary focus of this modeling project was to predict responses to various restoration alternatives, not to evaluate avian habitat relationships in managed pond and tidal marsh habitats, our model diagnostics and variable importance measures provided some insight into which variables were most important, as well as the direction of response (positive or negative). Furthermore, the incorporation of managed pond water depth data revealed new habitat relationships that were not evaluated in our Phase I effort. Core depth and salinity range analyses provided additional understanding of waterbird managed pond use. Finally, the use of focal species allowed us to examine species-specific differences as well as group-level trends.

Tidal Marsh

Due to the small number of tidal marsh sites surveyed and high variability among surveys (most species were absent on most surveys), tidal marsh waterbird models generally had low explanatory power. However, as reported in Phase I (Stralberg et al. 2003), waterbird density in tidal marshes was almost always positively associated with the amount of open water, in the form of tidal channels, tidal ponds, and semi-tidal pannes. In general, but with a few exceptions, shorebird and dabbling ducks densities were highest in ponds/pannes, where food may be more accessible than in surrounding vegetated areas and the approach of predators more easily detected (Leger and Nelson 1982). Diving ducks, fish-eaters, and waders, however, had higher densities in channels, which provide valuable habitat for fish foraging and reproduction (Boesch and Turner 1984, Williams and Zedler 1999, West and Zedler 2000). Furthermore, open water densities of some species appeared to increase as the proportion of channels or ponds within a marsh increased, suggesting that larger or more extensive open water areas are likely to attract higher densities of birds.

Landbird densities did not seem to be greatly affected by channel density at the site level examined here. However, other studies have demonstrated the affinity of Song Sparrow (Spautz et al. in press) and Clapper Rail (Evens et al. 1996, Foin et al. 1997) to tidal sloughs, the former for nesting, the latter for foraging, suggesting that marshes with high channel density could support higher densities of landbirds and rails.

These findings emphasize the importance of encouraging natural, dendritic slough development in restoring tidal marshes, rather than maintaining deep borrow ditches that may not provide similar foraging (and nesting) potential. They also highlight the need to prevent further encroachment of invasive *Spartina alterniflora* hybrids, which can reduce channel development, further reducing foraging areas for waterbirds (Callaway and Josselyn 1992). While *S. alterniflora* has also been shown to attract high densities of nesting Clapper Rail (PRBO unpublished data), there is no research or documentation of long-term breeding success within invaded marshes.

In addition, while large ponds/pannes are likely to develop in large marshes over time, as has been documented in maps and descriptions of pre-settlement marshes (Collins and Grossinger 2005), it may take over 50 years for enough sediment to accumulate to create high marsh pannes. However, it may be possible to accelerate the development of large-scale open water features, such as high elevation salt pannes, via more active site engineering and construction activities. In addition, restoration sites may be maintained in a state of muted tidal action, to maintain large unvegetated areas by keeping them flooded for longer periods.

In managed marsh systems, managers have long manipulated the open water / vegetation ratio in order to enhance waterbird populations. On the east coast, the creation of open water ponds within salt marshes has been shown to benefit a wide range of waterbird species (Erwin et al. 1994). Baye (2005) suggested that "If the potential for restored salt marshes to develop internal shorebird habitats is neglected, we may overlook important opportunities to reduce the artificial segregation of managed pond and tidal marsh habitats." Indeed, the dichotomous view of salt ponds vs. tidal marsh should rather be a continuum of tidal action and management intensiveness, ranging from non-tidal, intensively managed high salinity ponds, to muted tidal low salinity wetlands, to fully tidal vegetated marsh. Ultimately, however, a successful restoration will benefit from strategic engineering of restored sites to provide maximum habitat potential for a wide range of species.

Managed Ponds

Depth and salinity were able to explain much of the variation in foraging waterbird densities within managed ponds, especially when aggregated at the group level. However, at the individual species level, much variation remained unexplained, reflecting the high variability of bird use within managed ponds, and suggesting that (a) pond use is highly ephemeral, perhaps because available foraging habitat is superabundant; and/or (b) other factors that we did not measure are also important in determining variation in prey abundance and availability. While species composition and salinity relationships of salt pond invertebrates is fairly well described (Lonzarich 1997, Takekawa et al. 2005, Herbst 2006), temporal and spatial aspects of invertebrate variability, and issues of accessibility to waterbirds are not well understood. Waterbird foraging activity may be triggered by episodic invertebrate "blooms" or specific water depth conditions that may change quickly following a natural rainfall or artificial water manipulation event (Helmers 1992, Anderson and Smith 2000).

While both salinity and water depth variables were important, we found that in general, water depth had more explanatory power than salinity for individual species. Exceptions tended to be high-salinity specialists—Black-necked Stilt and Eared Grebe—or low-salinity specialists—American White Pelican, scaup and Ruddy Duck. The salinity preferences of these species appear to be driven by the prey that can tolerate (or not) respective salinities (low salinity = fish and high invertebrate diversity, high salinity = brine shrimp and flies and high biomass). However, salinity was a better predictor for most groups: dabbling and diving ducks, fish-eaters, and small and large shorebirds in the winter. For small shorebirds in the spring and fall, salinity and depth variables seemed to be of similar importance.

In terms of water depth, small and large shorebirds had higher densities in shallow ponds, across all species except for Black-necked Stilt and Wilson's Phalarope, both of which had non-linear depth relationships. Of the other waterbird species, only Eared Grebe had higher densities in deeper ponds. This diving species has been shown to feed mainly on brine flies and shrimp in South Bay saltponds (Anderson 1970), prey that can occur deep in the water column (Larson 2000). Other species' responses were mixed, generally exhibiting weak non-linear depth relationships. In particular, the deep proportion variable did not seem to be important for any species, suggesting that minimal amounts of "deep" habitat are enough for the species who forage at depths greater than one meter. In general, the shallower ponds supported more bird species at higher densities.

Several other researchers have demonstrated the importance of water level on waterbird foraging density and diversity. In the Florida Everglades, Great Blue Heron (*Ardea herodias*), Great Egret (*Egretta alba*), Wood Stork (*Mycteria americana*) and White Ibis (*Eudocimus albus*) were shown to decline above a certain species-specific depth threshold (Bancroft et al. 2002).

Velasquez (1992) found that species diversity increased when water levels in managed salt pans were reduced. Erwin (1996) demonstrated the importance of shallow open water habitats for shorebirds and other waterbirds. Elphick and Oring (1998) found that, while depth was a poor predictor of waterbird density in flooded rice fields, depths of 15-20 cm were used most frequently by the greatest number of species, and most species' use occurred at depths less than 30 cm (less than 15 cm for shorebirds). Our pond-level analyses resulted in broader depth ranges due the coarse unit of analysis, and the variable microtopography within individual salt ponds. However, it seems clear that salinity and depth affect bird abundance and distribution in managed ponds by affecting prey communities (low salinity = high invertebrate diversity and fish presence; high salinity = no fish, low invertebrate diversity but high invertebrate biomass (Warnock et al. 2002)) and prey accessibility.

The pond size variable we examined produced perplexing results—smaller ponds being used by more foraging waterbirds than larger ponds—suggesting that this variable may be confounded with other habitat conditions preferable to shorebirds, or that smaller ponds are more likely to be fully used by shorebirds.

PREDICTION UNCERTAINTY

In general, salt pond models explained much more of the variability in foraging bird density than did the tidal marsh models, but even the salt pond models often explained less than 50% of variation in density at the survey level. While this means we should be cautious about interpreting tidal marsh site- and landscape-level habitat relationships, it does not reduce our ability to make broad-scale comparisons across scenarios. First, several steps were taken to improve the predictive power of our models: (1) Variable selection was conservative and based on *a priori* understanding of bird habitat requirements. (2) The model-averaging approach allowed us to use more of the information available to us, reduce the chance that any one variable would have disproportionate influence in the predictions, and incorporate the uncertainty of model selection into our estimates. Also, the differences in mean densities between managed pond and tidal marsh habitats were still quite large for most species and groups, and these differences in mean densities were primarily what differentiated one alternative from another. Our variable and model uncertainties are included in the error bounds for our predictions at all levels: site, restoration project, and South Bay. Interpretation of predictions and the comparison of alternatives must consider the associated error bounds, as well as the general trends.

Our predictions for restored tidal marshes were based on a limited sample of remnant marshes in the South Bay, which in many ways may differ from the marshes that will be created from current salt ponds. They were smaller, and because many were located along major sloughs or along the bay edge, they may have had more adjacent bay and shallow open water habitat than future restoration marshes will have. Thus it is possible that our models may be less accurate when applied to the range of conditions represented by future restoration projects.

To generate predictions for a given alternative, we aggregated site-level predictions across the entire restoration area, and then across the entire South Bay, resulting in fairly large error bounds for some species. However, that uncertainty could be reduced with additional data collection to improve sample sizes and the range of conditions surveyed, especially in tidal marsh habitats. Furthermore, sub-pond analysis of water depth relationships may provide tighter managed pond prediction ranges. For future research, a Bayesian approach might help to explicitly incorporate the uncertainty present in our models (Ellison 1996).

PROJECT AREA SCENARIO EVALUATION

At the project level, several key findings were apparent from our scenario evaluation exercise, leading to some general recommendations.

Year 0: Value of Newly Restored Marshes

In year 0 of any restoration alternative, soon after levees are breached and tidal action is restored, waterbird numbers are likely to increase, as new unvegetated intertidal and subtidal zones are created. Our model predictions for year 0, which used low salinity salt pond data to estimate future intertidal and subtidal densities, reflected the value of these shallow low salinity habitats for many species. However, not all waterbirds would be expected to respond positively; Eared Grebe, some fish-eaters, and some shorebird species—species that use high salinity and/or deeper ponds—would likely experience a decrease in habitat potential. Several researchers have demonstrated the importance of newly restored tidal marshes to shorebirds and other waterbirds (Brawley et al. 1998, Atkinson et al. 2001, Nur et al. 2004). Indeed, the reported increase in waterbirds in certain ponds during the ISP may demonstrate this phenomenon at some level (Takekawa and Athearn 2006). This suggests that a staggered approach to tidal marsh restoration may have the greatest opportunity to provide long-term habitat benefits for waterbirds, as newly-breached ponds may compensate for marshes that become vegetated. Extending the restoration period would also provide more time for the first restoration sites to develop high marsh ponds/pannes that could replace the foraging potential provided by intertidal mudflats in early restoration projects. Modeling of various staggered approaches could help managers identify strategies for the timing of restoration.

Year 50: Striking a Balance

By year 50, after most restoration ponds have become vegetated, most shorebird, fish-eating, and diving duck species are expected to have higher numbers under alternatives that retain substantial areas of managed ponds (e.g., alternative B). Landbirds, rails, and dabbling ducks, however, would have highest numbers under restoration scenarios with the most tidal marsh area (e.g., alternative C). Of the alternatives we analyzed, weighing the needs of a broad range of species, a mixed restoration / managed pond alternative (e.g., alternative B) appears to provide the best starting point to maximize species diversity and numbers. The alternative could be subject to modification (e.g., restoration of additional ponds) at a later point in time if avian population response warranted such a change.

Under a mixed restoration / managed pond alternative with intensive pond management, several waterbird species, particularly diving ducks (Ruddy Duck and Scaup), fish-eaters, (American White Pelican and Forster's Tern), and Eared Grebe, would still likely lose up to 50% of their winter foraging habitat potential within the restoration area. However, this loss could be partially compensated for by an increase in or restoration of subtidal habitats in the South Bay. While certain shorebirds (Black-necked Stilt, Greater Yellowlegs) have the potential to increase in numbers with the restoration of salt ponds to tidal marsh, most shorebird species are likely to experience some decrease in numbers as their open water foraging habitat is restored to some mixture of vegetated marsh, depending on how remaining open water habitat is managed. Under current conditions, intertidal mudflats might be expected to compensate partially for this loss, but future predictions of South Bay sediment dynamics (PWA ref.) suggest that intertidal mudflats will decrease in area, regardless of restoration alternatives. Invasive *Spartina* hybrids may also encroach upon remaining mudflats, further reducing the foraging areas available for shorebirds (Stralberg et al. submitted). Thus it will be important to refine predictions of future mudflat change, and manage remaining ponds as carefully as possible to accommodate these species.

Importance of Pond Management

Pond management characteristics may have a greater effect on habitat capacity and overall waterbird numbers than the ratio of managed ponds to tidal marshes. Several waterbird species (primarily shorebirds) had similar predicted managed pond abundance indices under alternative B as under alternative A, despite the 50% reduction in overall pond area, due to higher mean predicted densities. Alternative C, which contained a higher proportion of shallow ponds, generally yielded higher mean predicted densities than alternative B. Exceptions included Eared Grebe and fish-eaters (American White Pelican and Forster's Tern), which generally use deeper ponds. Shallow ponds are likely to be more difficult to manage, especially at lower depths, due to issues of vegetation encroachment and evaporation. However, they appear to provide greater differential benefits to more species, while deeper ponds support fewer species. Those species may be more affected by salinity (low for fish-eaters, high for Eared Grebe) than depth. Clearly a combination of pond depths and salinities will be needed.

SOUTH BAY-WIDE SCENARIO EVALUATION

For the entire South Bay, inclusive of the restoration area, additional insights were provided by key findings from our South Bay-wide scenario evaluation:

Pond Management vs. Tidal Marsh Engineering

Intensive pond management would likely provide greater opportunities to increase waterbird numbers than the engineering of tidal marsh open water features during restoration. At the South Bay scale, only one species (Northern Shoveler) differed in its proportional change response between the minimum and maximum tidal marsh open water assumptions. Conversely, several shorebird species (Western Sandpiper and Wilson's Phalarope in fall; Dunlin, Semipalmated Plover, and Willet in winter) switched from predicted decreases to predicted increases or no change in numbers when 50% shallow (<15 cm) managed pond conditions were modeled. Managing ponds to be 75% deep (>1 m) increased predictions for a few species (Eared Grebe, American White Pelican), but the negative impact on shorebirds seemed to outweigh the positive benefits. While managing all ponds to be shallow is not a feasible solution for all species, it indicates that an improvement in the management of shallow ponds could dramatically increase habitat potential (and thus, most likely, shorebird numbers), much more so than engineering high marsh pannes, or limiting tidal intake to create muted marsh habitat. While these measures should also be pursued, they may be more costly, especially in proportion to the number of birds expected to benefit.

Optimal Managed Pond / Tidal Marsh Configurations

With respect to avian populations, one of the biggest questions to be answered about the restoration is “how much is enough?” That is, how many managed ponds must be maintained to support or increase waterbird numbers and diversity, and how much tidal marsh must be created to increase tidal marsh bird numbers and support viable populations over the long term? Neither of these questions is easily answered, and trying to answer both creates even bigger challenges for restoration planning. Our habitat-based model predictions suggest that, assuming ponds are currently at carrying capacity, approximately 50% of current managed ponds would have to be retained and intensively managed to maintain current overall waterbird usage within the South Bay. But depending on the efficacy of pond management, as well as the carrying capacities of managed ponds and other habitats, this percent could be substantially lower. The evaluation of habitat carrying capacities, relating changes in habitat to impacts on the demography of species, and developing population viability analyses (PVA) for key species could help identify the species and seasons for which habitat is limiting in the South Bay. Ongoing monitoring efforts may also be used to detect declines early.

Techniques to identify optimal proportions of managed ponds and tidal marshes, as well as optimal pond salinities and depths, given a range of conservation targets, are also needed. They can more easily provide insight into the range of habitat proportions and pond management conditions that would provide the most efficient benefits to a broad range of species. They can also highlight which species play the largest role in determining optimal solutions, providing additional focus for restoration efforts.

In terms of tidal marsh, any increase in habitat represents a potential gain for rails, landbirds, and many dabbling ducks. For Clapper Rail, survival of the species may depend on the amount of tidal marsh restored in the South Bay, but it also appears likely that other factors, such as predation and habitat quality, may be more important than amount of habitat in determining the future viability of the Clapper Rail population, similar to the situation for Snowy Plovers (Nur et al. 2001).

Seasonal Differences

Our results showed that shorebird species' foraging responses to habitat change may differ by season, and fall and spring migration periods have the potential to become population bottlenecks without adequate managed pond habitats. We found that, at the South Bay scale, under alternative B, the abundances of most shorebird species were expected to either increase or remain stable in the winter, but decrease during the fall and spring seasons, unless all ponds were managed as 50% shallow. This probably reflects the overall higher use of South Bay habitats during migration periods, especially in the spring (Davidson and Evans 1986, Warnock and Takekawa 1996, Takekawa et al. 2001). Furthermore, salinity and depth conditions vary throughout the year based on rainfall and other weather elements, such that salinity and depth may play different roles during different periods, as invertebrate prey communities are also changing. During migration, especially during the spring, when the migration period is condensed, shorebirds require additional fuel stores to successfully migrate and breed (Piersma 2002). Additionally, numbers of birds such as the Western Sandpiper may increase by over 200% in the spring, with additional time being spent feeding in salt ponds (in addition to intertidal mudflats) to put on extra fuel for the migration to breeding grounds (Warnock and Takekawa 1995). Thus, for shorebirds at least, it is appropriate to focus on this bottleneck period, where bird use is more likely to exceed habitat carrying capacities. Loss of higher salinity ponded habitat with high invertebrate biomass may be especially damaging to some waterbirds during this period since energy intake may be higher on some salt ponds than on intertidal mudflats (Masero 2003).

Tidal Flat Declines

While tidal marsh open water habitats may compensate for some loss of foraging opportunities provided by tidal flats, shorebird use of tidal marshes may be an order of magnitude lower than tidal flats (Reinert and Mello 1995, Burger 1997). Based on observed (Foxgrover et al. 2004) and modeled (PWA 2006) sediment dynamics in the South San Francisco Bay, combined with threats posed by invasive *Spartina* encroachment (Stralberg et al. 2004) and sea level rise (Galbraith et al. 2002, Orr et al. 2003), tidal flats are most likely going to decrease in the South Bay, particularly north of the Dumbarton Bridge. Tidal marsh restoration will contribute to the sediment deficit, but even under alternative C, would be a small component of the overall loss. This means that managed open water habitats and seasonal wetlands will become more important for the species that rely on tidal flat foraging opportunities. Particularly in the face of sea level rise, managed ponds will be the most reliable foraging habitats available for many species. Thus it will be important to plan for additional habitat capacity within remaining managed ponds.

Special Status Species

Clapper Rail predictions were linked directly to the amount of tidal marsh habitat within the alternatives. We would expect substantially more tidal marsh, and hence more breeding Clapper Rail under alternative C, a 250% increase over alternative A and a 163% increase over alternative B (by year 50); thus, a 53% increase in numbers comparing alternative C to B. Predicted Clapper Rail numbers would be in addition to Clapper Rails using marshes outside of the restoration project. Whereas it is easy to compare proportional change in numbers, evaluating expected Clapper Rail numbers relative to any desired absolute criterion of population size is difficult. Such an evaluation is especially problematic because of the substantial difference between the upper and lower density estimates, due to high observed variability across sites and years.

Assuming that the restoration process creates tidal marsh habitat that is suitable for Clapper Rail in the 50 year time frame, it is probable that our lower estimates of density (10^{th} percentile) are excessively pessimistic. Instead, using the mean density estimate (0.54 birds/ha) suggests that alternatives B and C have an excellent chance of providing enough high quality Clapper Rail habitat to meet one of the stated recovery goals, a population of ~1125 individuals across the South Bay (USFWS, pers. comm.), especially in conjunction with currently available habitat in the region. An upcoming PRBO report will present the results of the recently completed Bay-wide Clapper Rail survey, provide refined density estimates, and allow us to provide a more informed estimate of the current Clapper Rail population.

Even with better density estimates, active predator management would still be an important component of this species' recovery efforts. Studies of Clapper Rail breeding biology have shown that high nest and adult predation are critical limiting factors for the population in the Bay, with introduced Red Fox comprising a large proportion of the predator community (Foin et al. 1997).

Similarly, breeding Snowy Plover densities were linked directly to the estimated number of ponds that would be dry during the summer, under the assumption that all of these dry ponds will be managed for Snowy Plover. Alternative B would provide nearly twice as much dry pond habitat in the summer as alternative A, and thus would be the preferred option for maximizing breeding Snowy Plover habitat. Alternative C would provide no dry pond habitat within the restoration area, and thus would not provide any additional support to the endangered plover population. In fact, current Snowy Plover habitat may be converted to tidal marsh under any of the alternatives, causing a reduction in available habitat.

Again, for this species, as well as for Clapper Rail, high variability in density among sites led to large ranges in our abundance estimates. Using the upper density estimate, we concluded that a population of more than 500 Snowy Plovers—the recovery plan target—could be maintained under either alternative A or B. However, using the lower density estimate, none of the alternatives were adequate. Intensive management should be able to produce habitat quality (and corresponding Snowy Plover densities) more closely resembling our upper estimates. However, the lower estimates highlight the importance of intensive management in maintaining the population. Indeed, if lower estimates were correct, then our calculations suggest that it would be impossible to simultaneously maintain 500 Snowy Plovers and 1,125 Clapper Rail in the South Bay. Based on upper density estimates, either alternative A or B could accommodate those numbers.

It is also important to recognize that, while substantial amounts of dry pond habitat may be available at year 0 under alternative B, much of this would not be managed specifically for Snowy Plover, nor would we necessarily expect Snowy Plover populations to increase to fill the available habitat without significant predator management activities. A population viability analysis (PVA) developed for the Pacific Coast population of the Snowy Plover (Nur et al. 2001) showed that reproductive success was the most important limiting factor for this species. Given high predation rates for this species (Page et al. 1995), active predator management needs to be an important component of any plan for species recovery.

RESEARCH NEEDS

While the results of our scenario evaluation provide new quantitative and qualitative information about potential effects of tidal marsh restoration on South Bay bird communities, there are still many unanswered questions and data gaps that need to be filled.

Population Dynamics

We assume, to a certain extent, that foraging birds are limited by the amount of suitable habitat in the Bay, but they may instead be limited by additional factors, including reproductive success on their breeding grounds, over-winter survival, or food availability along their migratory paths (Evans 1991, Sillett et al. 2000, Sillett and Holmes 2002). Due to the limitations associated with using habitat availability as a surrogate for bird population numbers, it will be necessary to assess the short- and long-term viability for populations of several species of interest, under various restoration scenarios. Population viability analyses (PVA) are one way to identify the limiting demographic parameters (e.g., reproductive success, recruitment and survival) for a population, and assess the population's probability of long-term survival under various scenarios (Boyce 1992, Nur and Sydeman 1999). In general, adult survival has been shown to be the most important limiting factor across shorebird taxa (Sandercock 2003). Reproductive success, however, is thought to be limiting for many other species, as it fluctuates widely, compared to the relatively stable adult survival, and may be easier to influence through management (Nur et al. 2001). Threatened and endangered species, such as the Clapper Rail and Snowy Plover, are particularly important to model in this way, but demographic analyses of wintering waterbird species should also be conducted.

Habitat Carrying Capacity and Energetics

Assuming that birds are habitat-limited, we still do not know if and when (seasonally and diurnally) bird numbers in existing ponds and tidal marshes are close to the population carrying capacity for those habitats. For migratory shorebirds, our survey data suggest that ponds and tidal flats are closest to feeding carrying during spring migration. However, we don't know how prey availability may vary across seasons, perhaps changing the carrying capacity for foraging birds. Additional energetics studies are needed to determine those carrying capacities and we need to know the demographic consequences of energetic differences; birds demonstrate some resilience to energetic fluctuations that do not always impact survival or reproduction.

It is possible that high-quality ponds could hold higher bird numbers if current habitat were lost, and that remaining ponds could be managed to support higher numbers of birds. Fundamentally, we need to gain a better understanding of salt pond prey availability, waterbird foraging requirements, and carrying capacities for various species (Goss-Custard et al. 1996). We assumed in this exercise that all ponds were being used to their maximum potential.

However, it may be that more individuals can be accommodated within just a few high-quality ponds. Conversely, if habitats are already being used at carrying capacity, behavioral responses of birds and their prey to habitat reduction could create a decline in bird numbers even greater than in direct proportion to the area of habitat lost (Goss-Custard 2003).

Existing data on bird energetics and prey availability can be used to estimate a range of carrying capacities for each salt pond and other habitats. In Spain, using an energetic approach combined with an understanding of prey selection, Masero (2003) found that salt ponds (salinas) contributed 25% of the daily prey consumption of waterbirds in the winter and 79% during the pre-migration period compared with intertidal mudflats. Durell et al. (2006) found, through individual-based energetics models, that non-tidal habitats were important for overwinter survival of large shorebird species, with reductions in overwinter survival rates predicted for Black-tailed Godwit and curlews when habitat areas fell below 200 ha.

Finally, we assumed that roosting habitat was not limited for the focal species and groups analyzed. This is an assumption worth testing further.

Regional Habitat Availability

In addition, we need to learn more about regional habitat availability for the species we modeled, some of which rely predominantly on San Francisco Bay during the winter or as stopover habitat during migration. Species whose populations depend heavily upon San Francisco Bay, such as Western Sandpiper (Butler et al. 1996, Iverson et al. 1996, Bishop and Warnock 1998), Dunlin (Warnock and Gill 1996), and Canvasback (Accurso 1992) may be more strongly affected by changes in habitat availability if they are unable to adapt by using other coastal wetland areas instead. Models of west coast stopover and wintering habitat availability for several key species would help managers prioritize habitats for conservation and restoration, based on their regional importance (Warnock and Bishop 1996, Takekawa et al. 2002, Warnock et al. 2002). Spatial models can be combined with energetics information to characterize the suitability and relative value of migration stopover sites (Simons et al. 2000).

Non-Native Species

The introduction of non-native flora and fauna to salt ponds, tidal marshes and tidal flats has the potential to greatly influence birds using these habitats. Habitat-based modeling has demonstrated the potential for loss of up to 80% of South Bay tidal flat shorebird foraging potential if the invasive *Spartina alterniflora* (and hybrids with the native *S. foliosa*) spreads to occupy its full range of tidal inundation tolerance (Stralberg et al. submitted). The invasive *S. alterniflora* has already reduced significantly the available shorebird foraging habitat in Willapa Bay, Washington (Jaques 2002), and in Europe, the invasive *S. anglica* has reduced numbers of Dunlin using English coastal wetlands (Goss-Custard and Moser 1988). Managers need to better understand the spread of the non-native *S. alterniflora* in San Francisco Bay and its possible effects on birds of the Bay before they can make informed decisions about how to deal with this invasive species.

Climate Change and Rising Sea Level

It is now well-recognized that increasing levels of carbon dioxide and other greenhouse emissions in the earth's atmosphere are leading to increasing global temperatures and other changes in climate patterns (Houghton et al. 2001). For coastal regions, the most dramatic result of future climate change is likely to be a rise in sea level caused by the melting of glaciers and polar ice caps (Pernetta and Elder 1992, Michener et al. 1997). The Intergovernmental Panel on Climate Change (IPCC 2001) has estimated an overall sea level rise (SLR) of 10 to 90 cm by the year 2100. While data from current restoration projects suggest that tidal marsh restoration may be able to keep up with SLR (Orr et al. 2003), this depends on the rate of SLR and sediment availability, both of which have large uncertainties associated with them. In the South Bay, where surrounding development limits the expansion of intertidal habitats, SLR may have more dramatic effects than in the North Bay, where much of the adjacent uplands are undeveloped. Modeling based on current elevation and historic rates of SLR suggest that 83% of South Bay tidal marshes and tidal flats could be submerged by SLR by 2100 (Galbraith et al. 2002). Clearly, such dramatic changes would greatly reduce the predicted abundances of tidal marsh-dependent species, such as the Clapper Rail, and to a lesser extent, tidal flat-dependent foraging shorebirds, ultimately decreasing the value of salt pond restoration in general.

Further spatially-explicit geomorphic modeling of South Bay sediment dynamics, incorporating the effects of SLR, are needed, in order to fully assess the South Bay's future habitat potential. Depending on the outcome, diked and managed habitats may, out of necessity, play a greater role in the provision of habitat for South Bay waterbirds.

CONCLUSION

Of the restoration alternatives we examined, all would be expected to have major effects on South Bay bird communities. Even alternative A, the "no action" alternative, would result in major habitat changes over a 50-year time horizon, and would likely lead to changes (mostly declines) in bird habitat use. In the short term, we expect that alternative C would lead to the most increases in waterbird numbers, especially shorebirds, dabbling ducks, and fish-eaters, due to the creation of large new low salinity open water habitats. In the long-term (over 50 years), we project that alternative B would provide the best balance of habitats, leading to large increases in numbers of landbirds, rails, and dabbling ducks, while most other waterbird species would not experience significant increases or decreases in numbers. Small shorebirds during spring migration, phalaropes during fall, and winter Dunlin, American Avocet, Ruddy Duck, and Eared Grebe would be most vulnerable to declines under both alternatives B and C, and are good candidates for intensive management within managed ponds.

While there are several sources of uncertainty associated with our model predictions, the principal unknown factors are the current carrying capacity of South Bay habitats, the availability of alternative habitats for bird using managed ponds, the extent to which habitat quality and availability are limiting bird population size and trajectory, and whether birds will indeed respond to change in availability of habitat in the manner that our habitat-based models assumed. Further research and monitoring of new and existing restoration sites will be needed to reduce these sources of uncertainty.

Nonetheless, our model results suggest that it is possible to achieve a balanced solution for all bird species with the use of intensive pond management, and, to a lesser extent, engineering of new tidal marsh restoration trajectories to provide more open water habitat for waterbirds. Depending on habitat carrying capacities, these solutions may be easier or more difficult to achieve than our models suggest. However, future changes in bay habitats, including climate change, mudflat declines and potential *Spartina* invasion, may change carrying capacities dramatically and alter the playing field, making adaptive management and phased restoration efforts a necessary component of this project.

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FIGURES

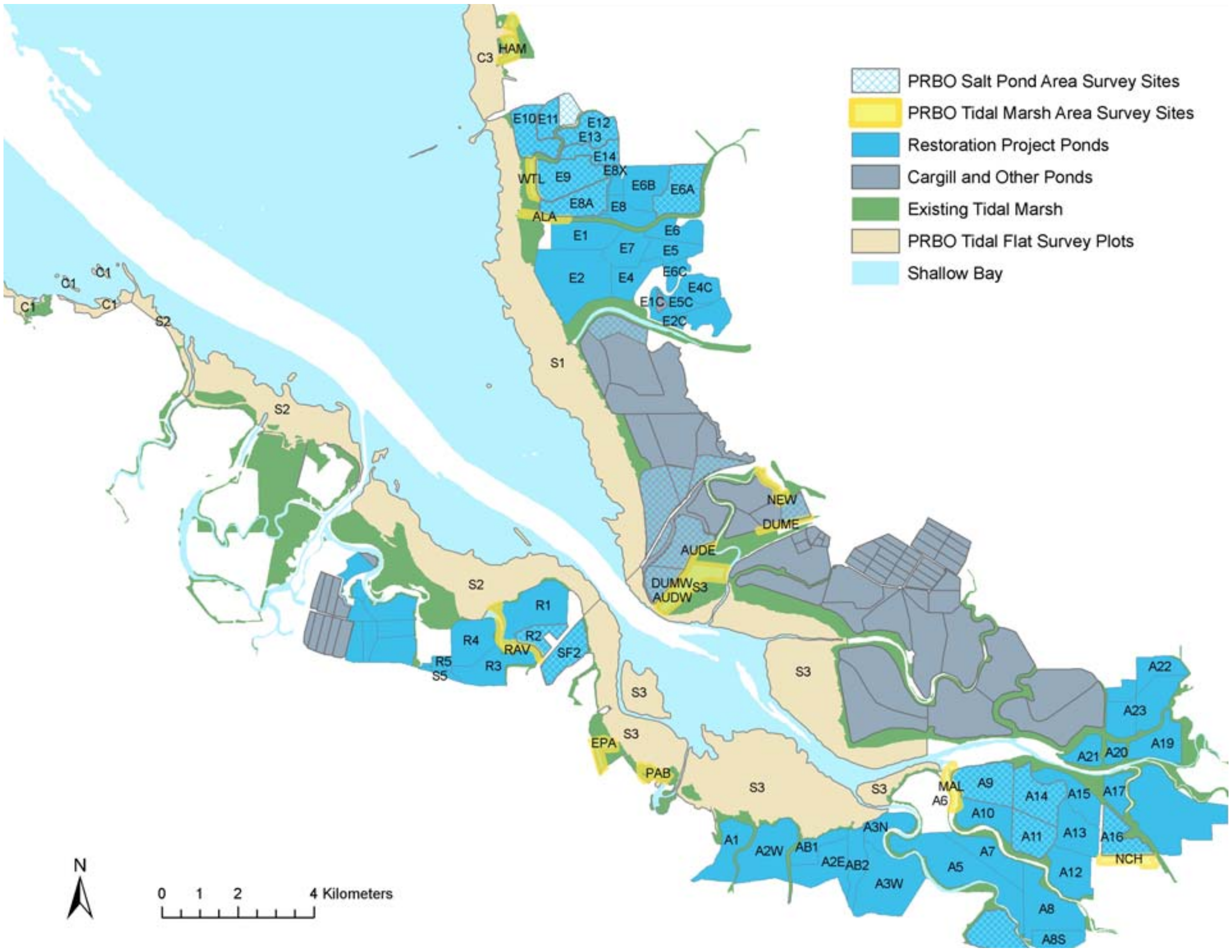

Figure 1. Map of South San Francisco Bay study area.
Restoration project ponds labeled and shown in blue. All restoration project ponds were surveyed by USGS and/or PRBO, and data were used to develop salt pond models. Other PRBO pond survey locations, used to calculate mean densities for Cargill ponds, also shown. PRBO tidal marsh study sites (indicated in yellow and labeled) were used to develop tidal marsh models. Tidal flat survey tracts (indicated in beige and labeled) were used to derive tidal flat shorebird densities. Existing tidal marsh and Cargill and other ponds were used to develop South Bay-wide estimates of abundance indices. Other small tidal marsh and tidal flat habitat areas south of Alameda Island were also included in South-Bay wide estimates.

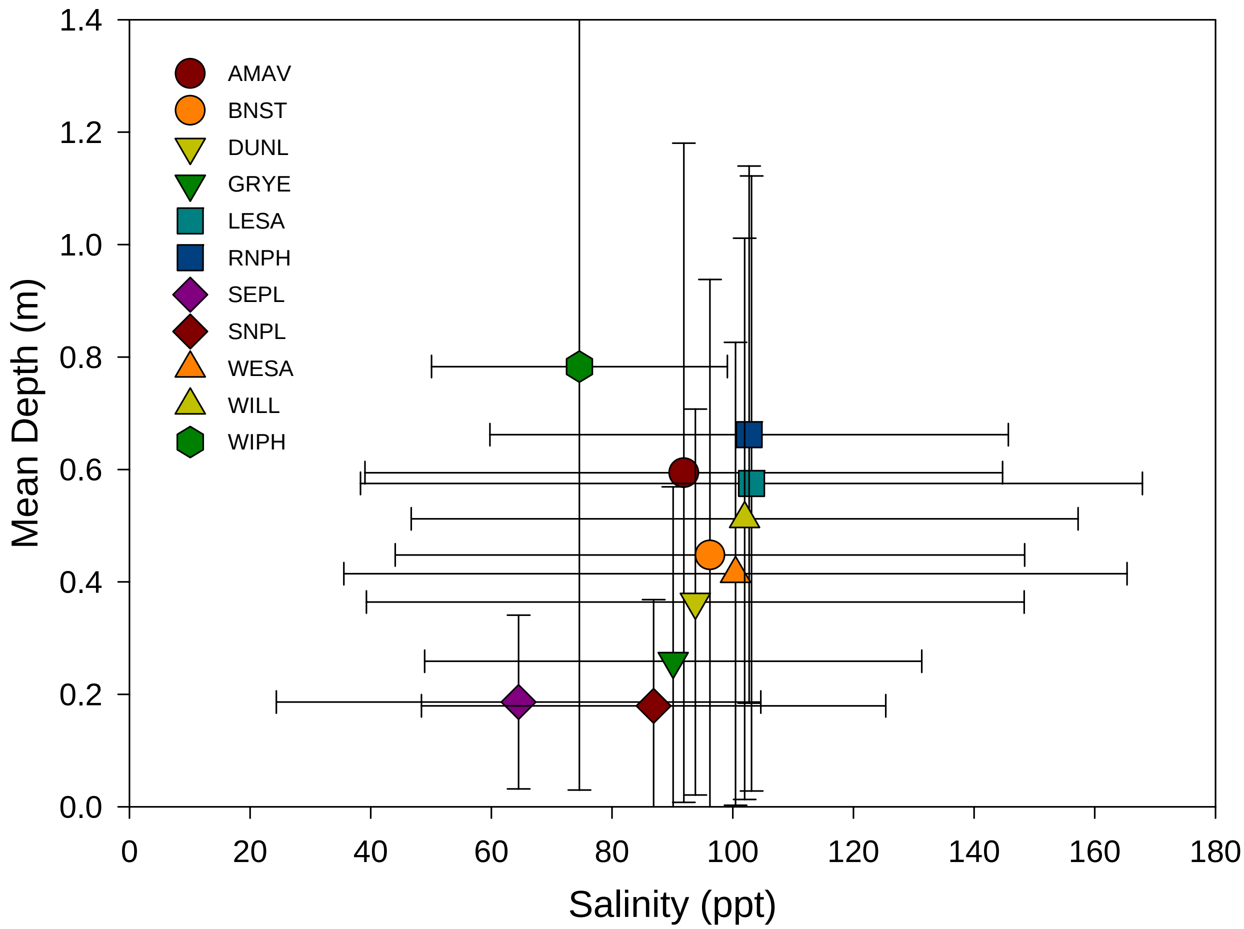

Figure 2. Shorebird salinity/depth graph.
Means (shapes) and standard deviations (error bars) of mean pond depth and salinity for shorebird species (including phalaropes) across all surveys. Means and standard deviations were calculated at the pond level (i.e., each pond in which at least one feeding individual was observed). Species 4-letter codes are defined in Table 6.

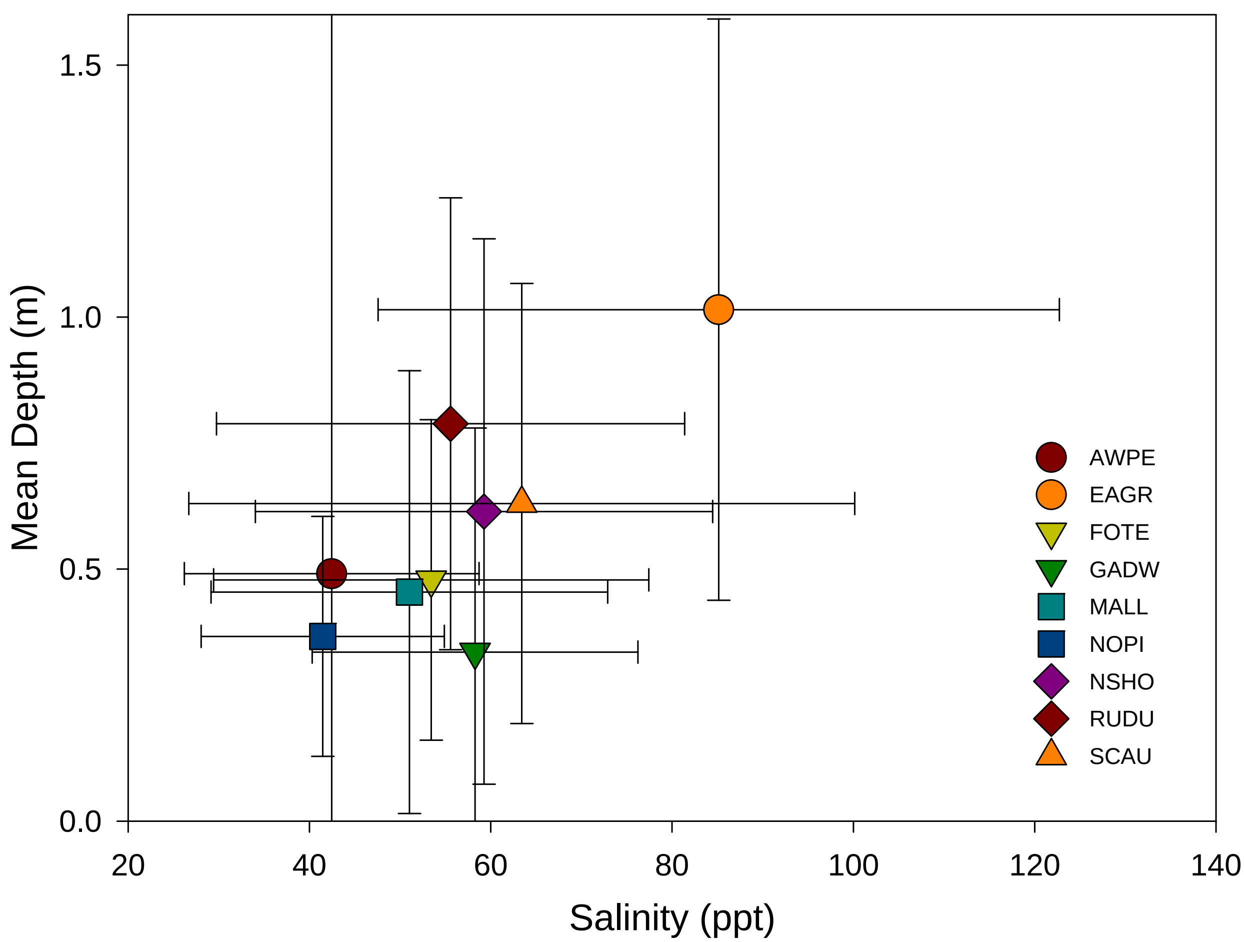

Figure 3. Other waterbird salinity/depth graph.
Means (shapes) and standard deviations (error bars) of mean pond depth and salinity for waterfowl and other waterbird species across all surveys. Means and standard deviations were calculated at the pond level (i.e., each pond in which at least one feeding individual was observed). Species 4-letter codes are defined in Table 6.

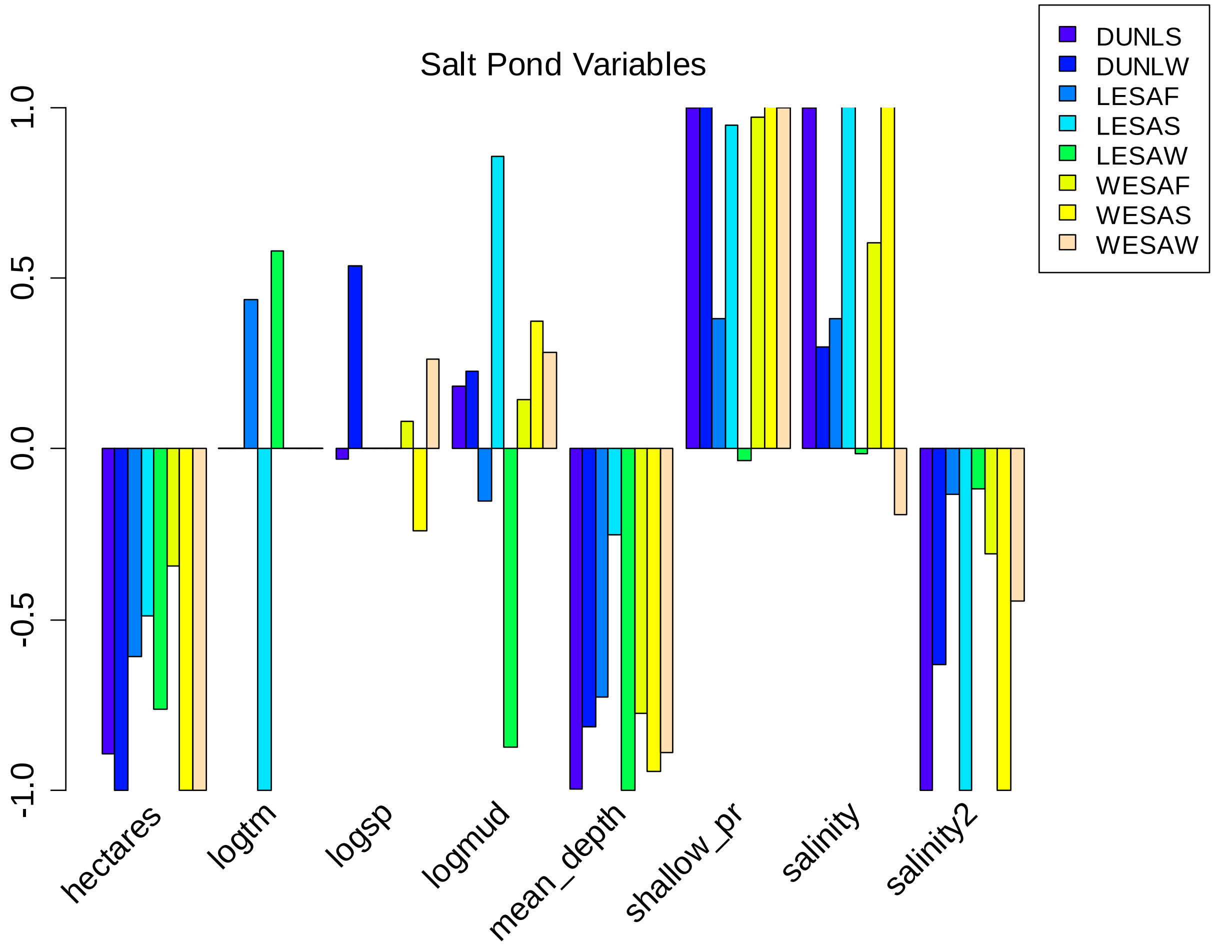

Figure 4. Variable support for small shorebird salt pond models.
Variable support represents the proportion of top models, weighted by AICc weight, in which a variable occurred. Variables with a support of 1.0 were included in all "top" (AICc weight ≥ 0.01) models. Figure legend = 4-letter species code (for abbreviations see Table 6) + season (S = spring, F = fall, W = winter). Data collected 1999-2004 at south San Francisco Bay salt ponds.

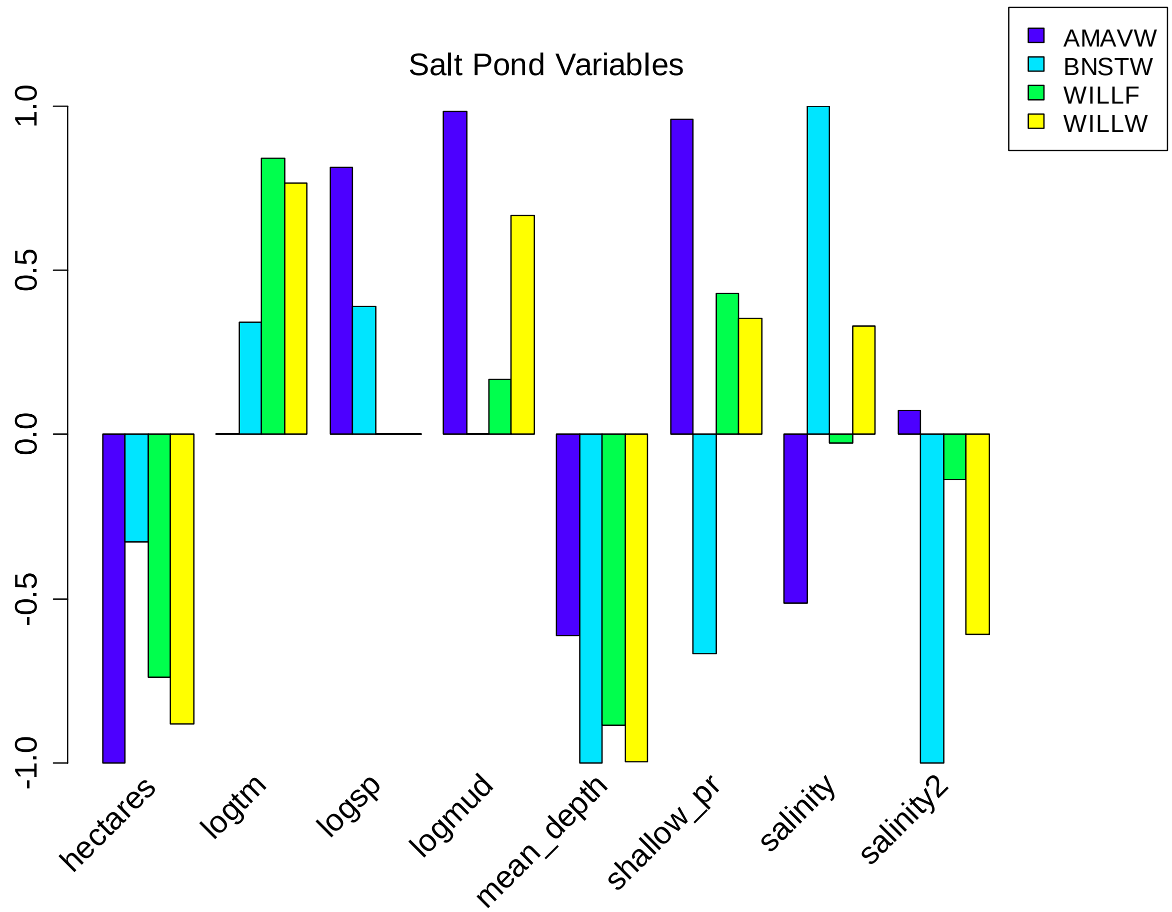

Figure 5. Variable support for large shorebird salt pond models.
Variable support represents the proportion of top models, weighted by AICc weight, in which a variable occurred. Variables with a support of 1.0 were included in all "top" (AICc weight ≥ 0.01) models. Figure legend = 4-letter species code (for abbreviations see Table 6) + season (S = spring, F = fall, W = winter). Data collected 1999-2004 at south San Francisco Bay salt ponds.

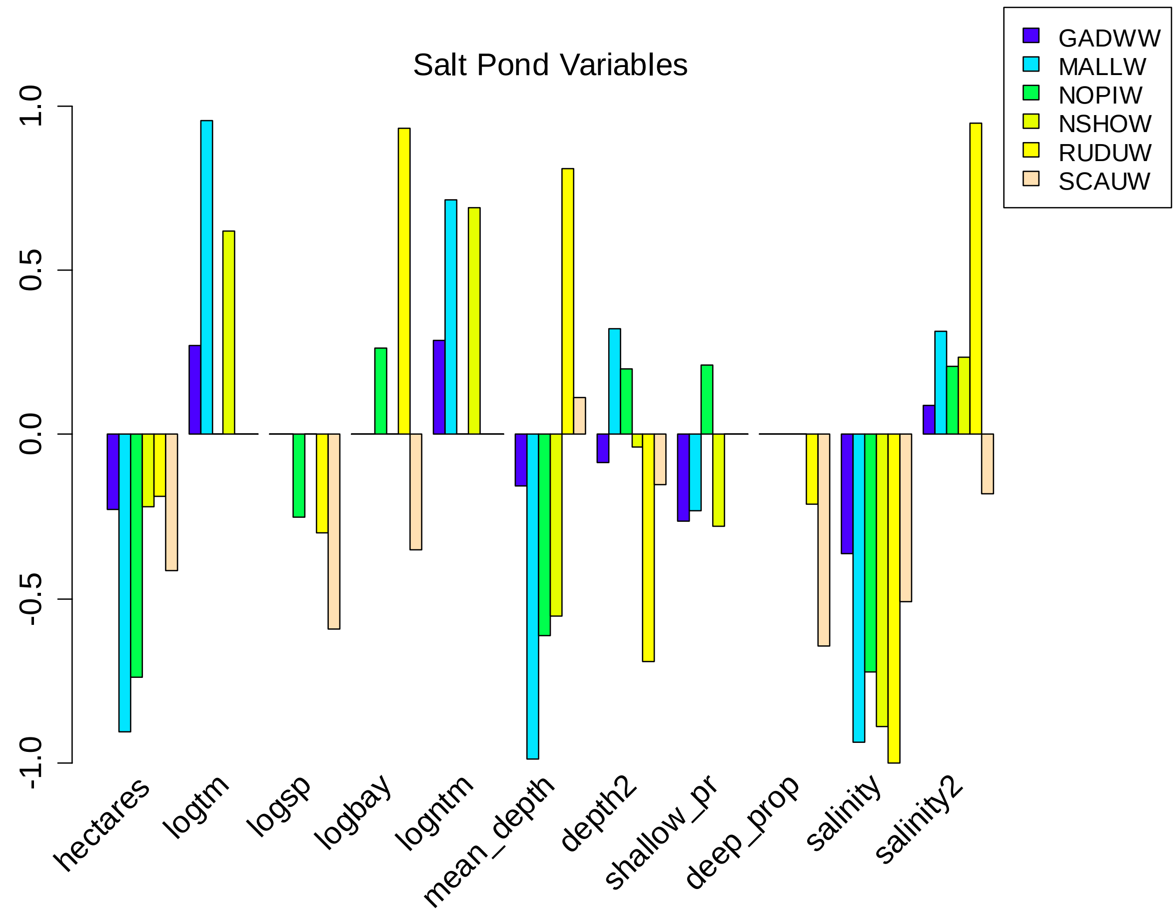

Figure 6. Variable support for waterfowl salt pond models.
Variable support represents the proportion of top models, weighted by AICc weight, in which a variable occurred. Variables with a support of 1.0 were included in all "top" (AICc weight $\geq$ 0.01) models. Figure legend = 4-letter species code (for abbreviations see Table 6) + season (S = spring, F = fall, W = winter). Data collected 1999-2004 at south San Francisco Bay salt ponds.

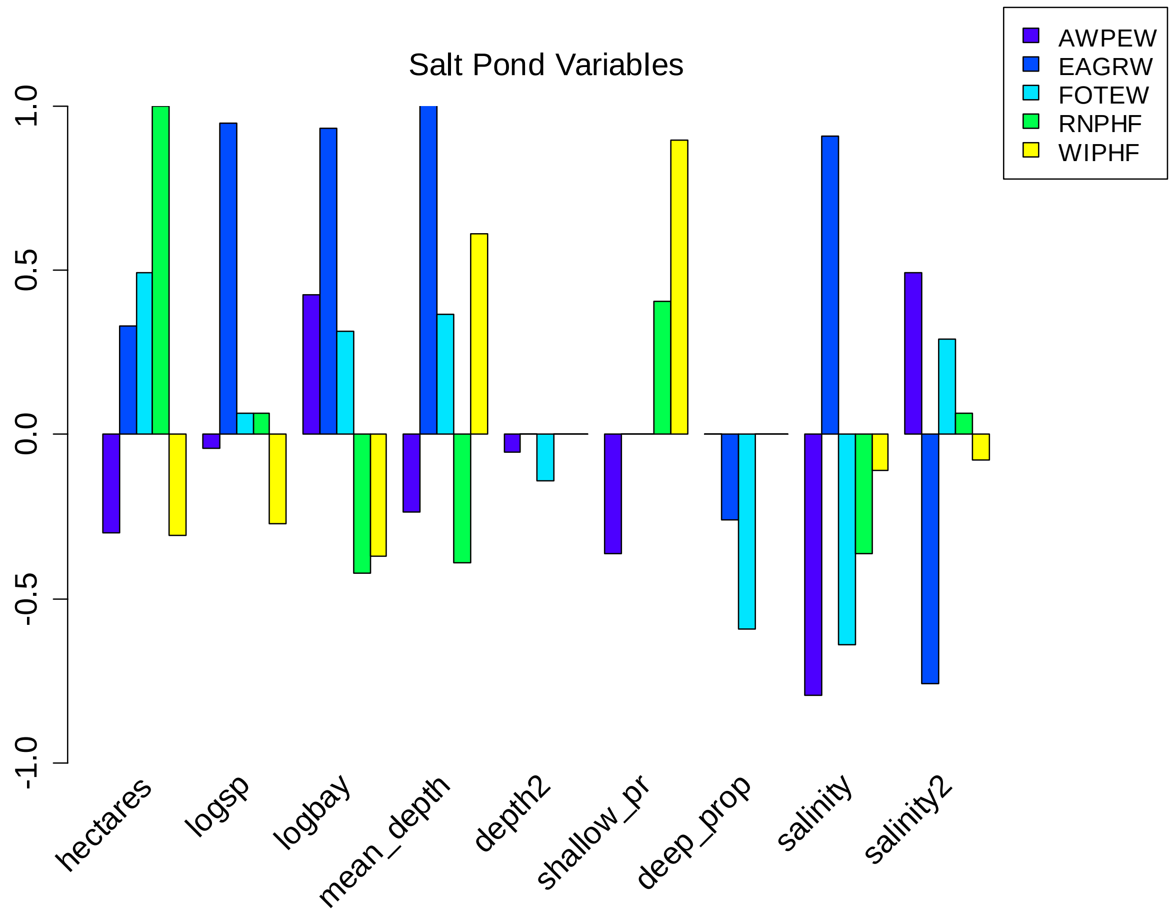

Figure 7. Variable support for other waterbird salt pond models.
Variable support represents the proportion of top models, weighted by AICc weight, in which a variable occurred. Variables with a support of 1.0 were included in all "top" (AICc weight ≥ 0.01) models. Figure legend = 4-letter species code (for abbreviations see Table 6) + season (S = spring, F = fall, W = winter). Data collected 1999-2004 at south San Francisco Bay salt ponds.

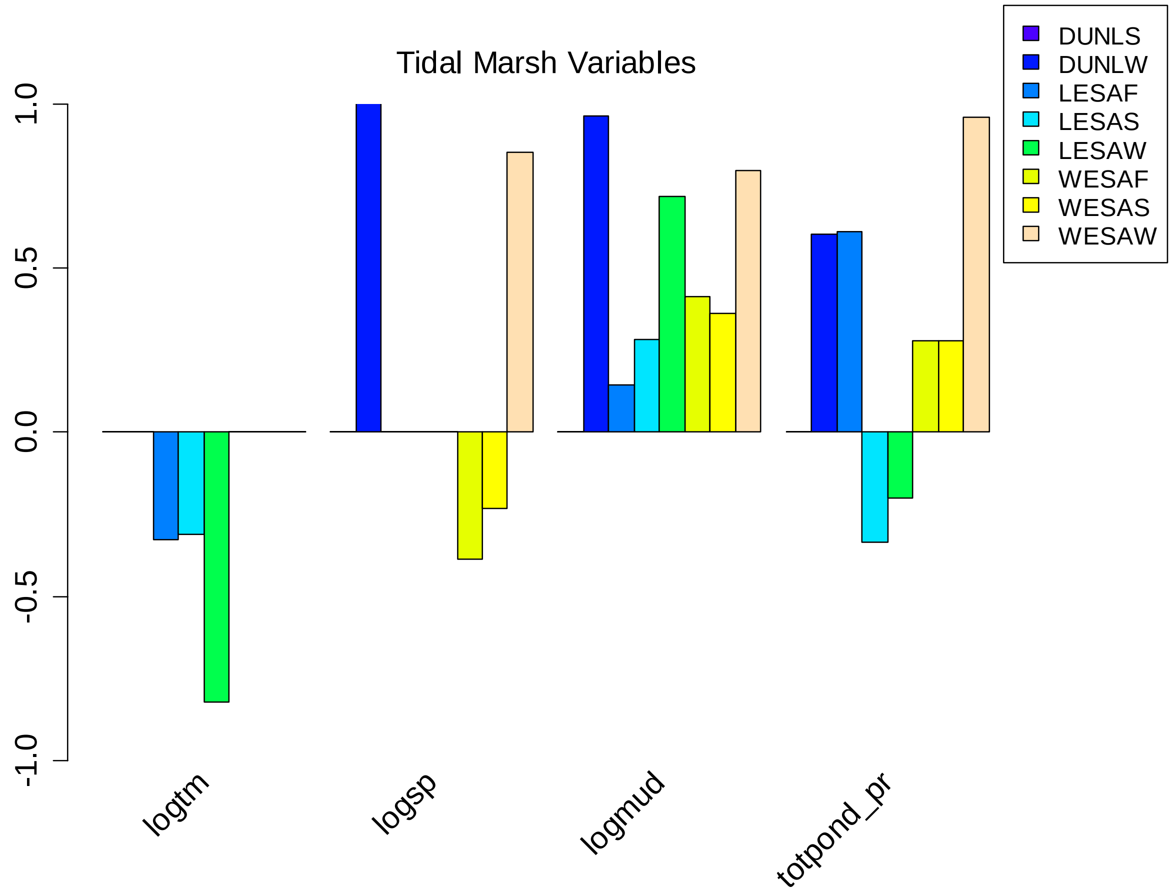

Figure 8. Variable support for small shorebird tidal marsh models.
Variable support represents the proportion of top models, weighted by AICc weight, in which a variable occurred. Variables with a support of 1.0 were included in all "top" (AICc weight ≥ 0.01) models. Figure legend = 4-letter species code (for abbreviations see Table 6) + season (S = spring, F = fall, W = winter). Data collected 1999-2001 at south San Francisco Bay tidal marshes.

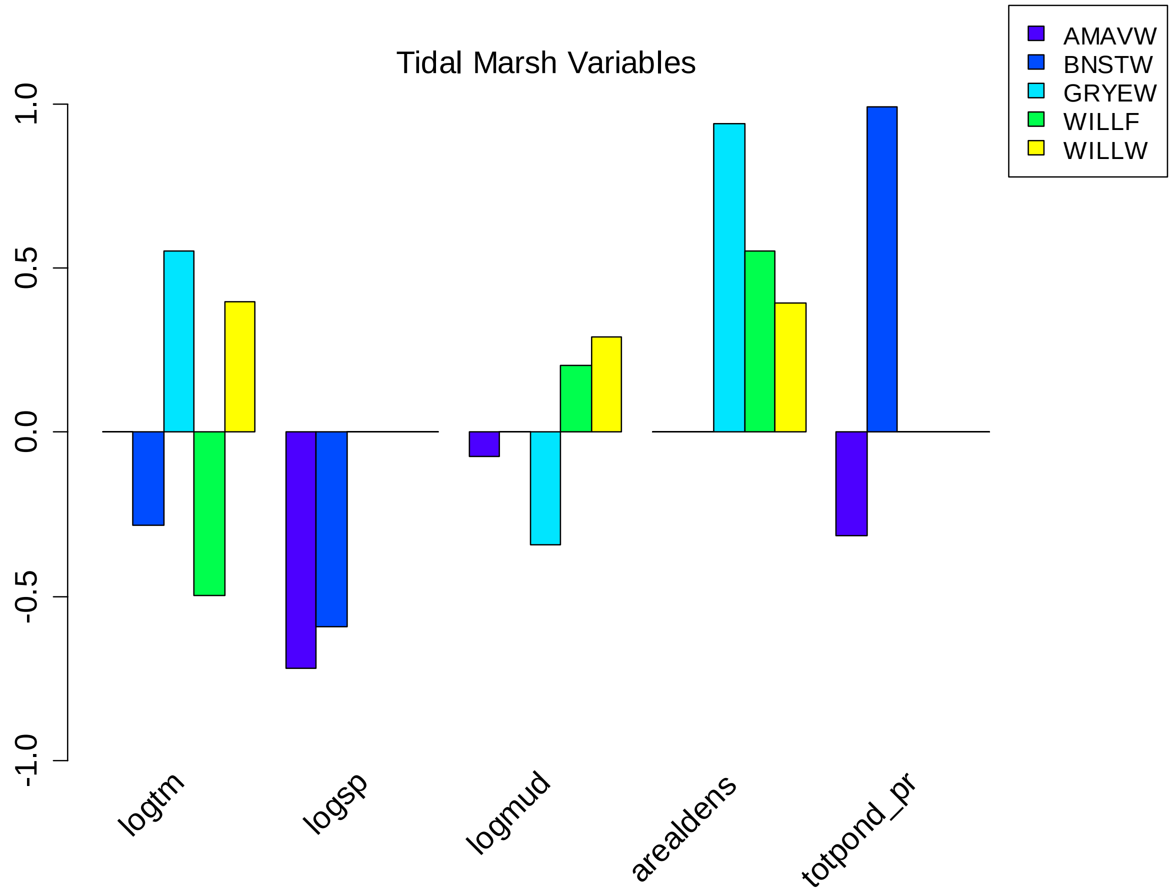

Figure 9. Variable support for large shorebird tidal marsh models.
Variable support represents the proportion of top models, weighted by AICc weight, in which a variable occurred. Variables with a support of 1.0 were included in all "top" (AICc weight ≥ 0.01) models. Figure legend = 4-letter species code (for abbreviations see Table 6) + season (S = spring, F = fall, W = winter). Data collected 1999-2001 at south San Francisco Bay tidal marshes.

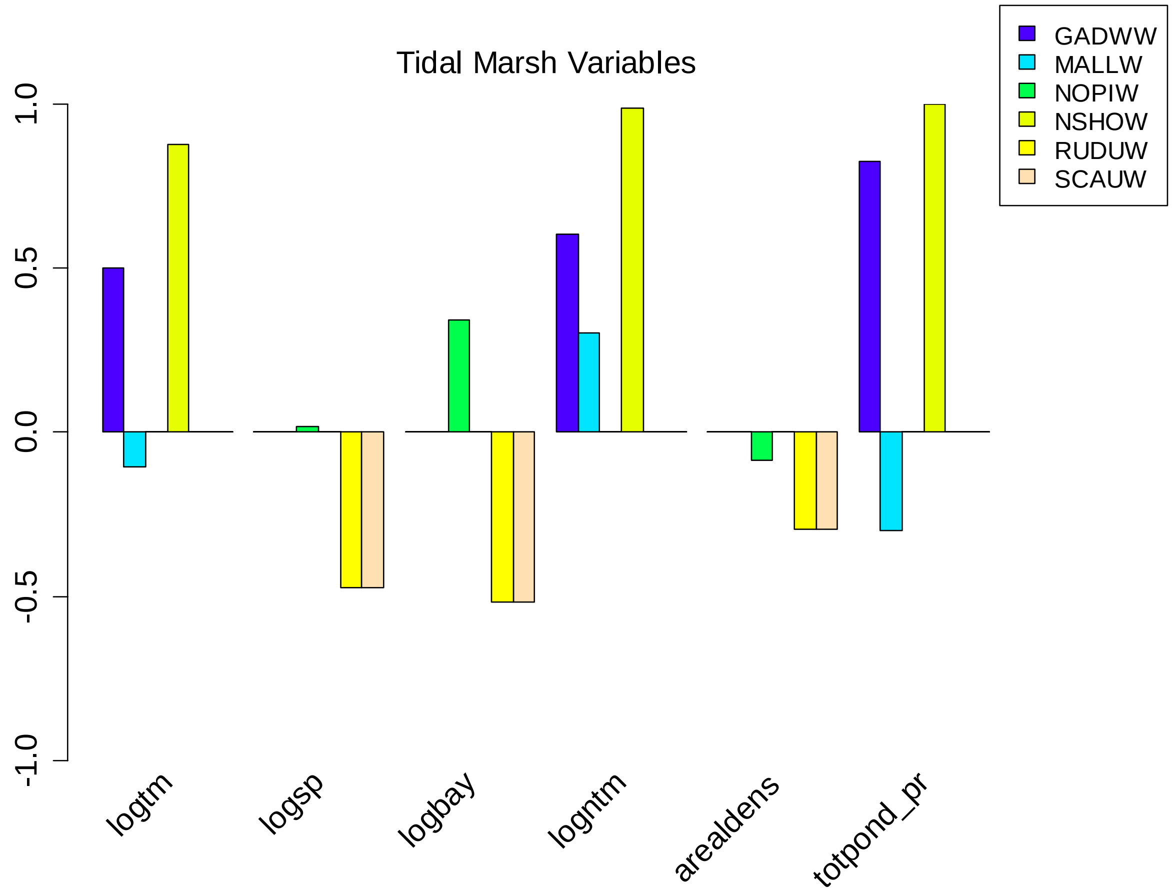

Figure 10. Variable support for waterfowl tidal marsh models.
Variable support represents the proportion of top models, weighted by AICc weight, in which a variable occurred. Variables with a support of 1.0 were included in all "top" (AICc weight ≥ 0.01) models. Figure legend = 4-letter species code (for abbreviations see Table 6) + season (S = spring, F = fall, W = winter). Data collected 1999-2001 at south San Francisco Bay salt ponds.

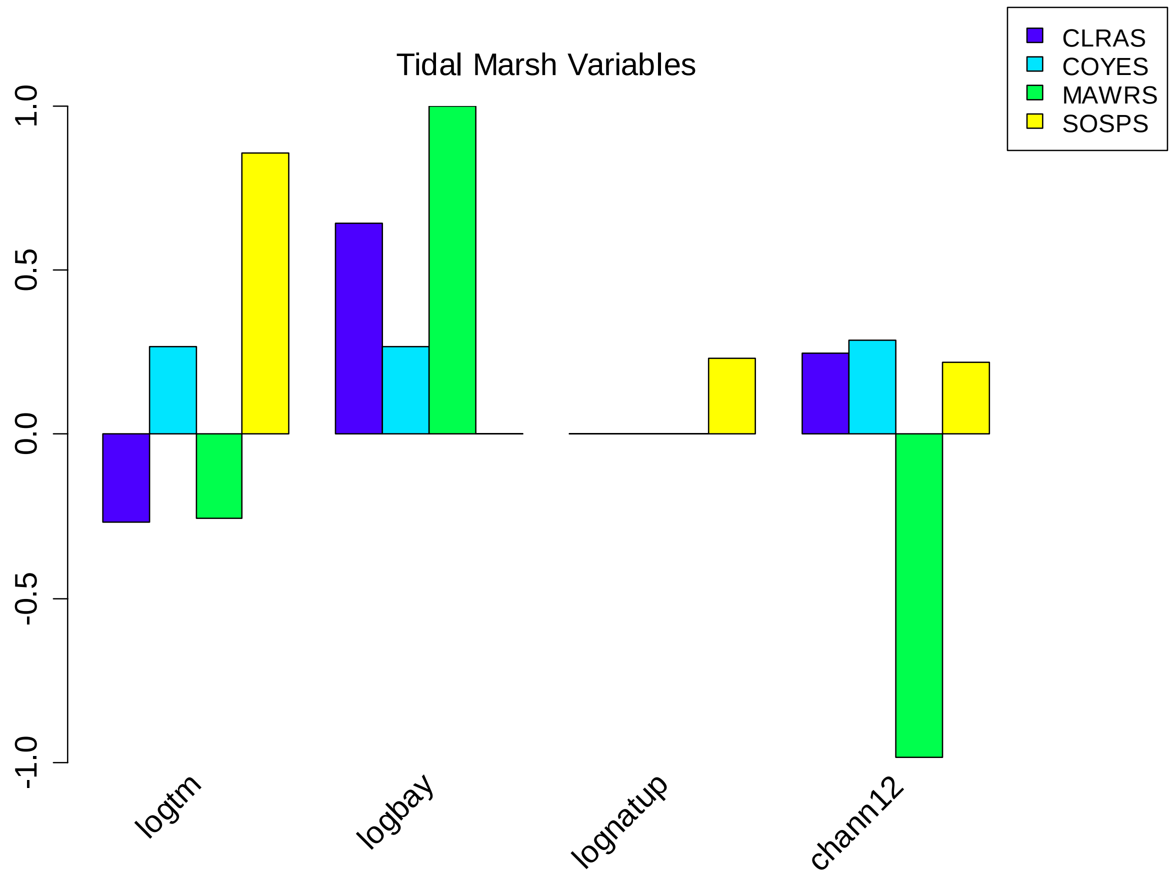

Figure 11. Variable support for landbird tidal marsh models.
Variable support represents the proportion of top models, weighted by AICc weight, in which a variable occurred. Variables with a support of 1.0 were included in all "top" (AICc weight ≥ 0.01) models. Figure legend = 4-letter species code (for abbreviations see Table 6) + season (S = spring, F = fall, W = winter). Data collected 1999-2004 at south San Francisco Bay tidal marshes.

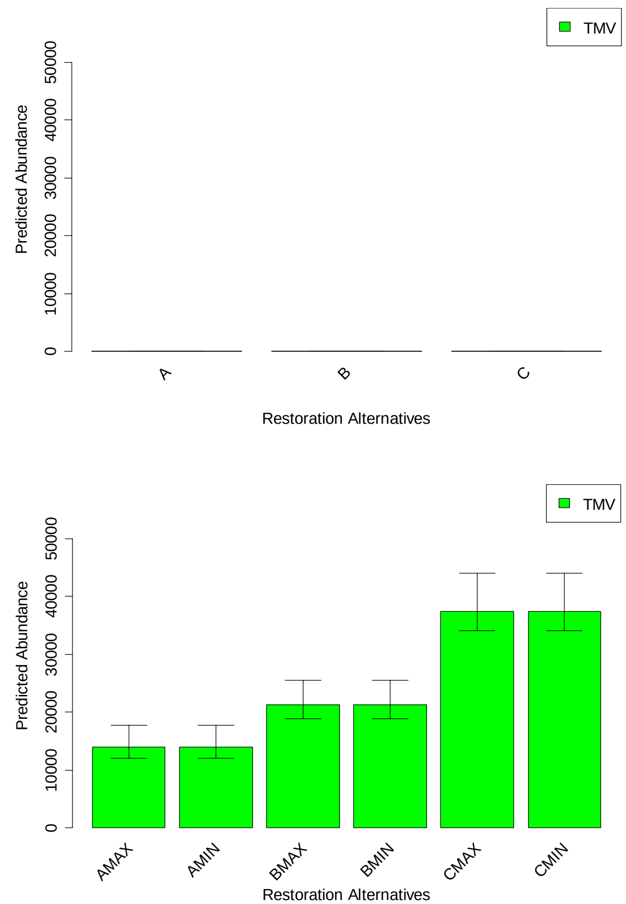

Figure 12. Breeding landbird project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. “Max” and “min” alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

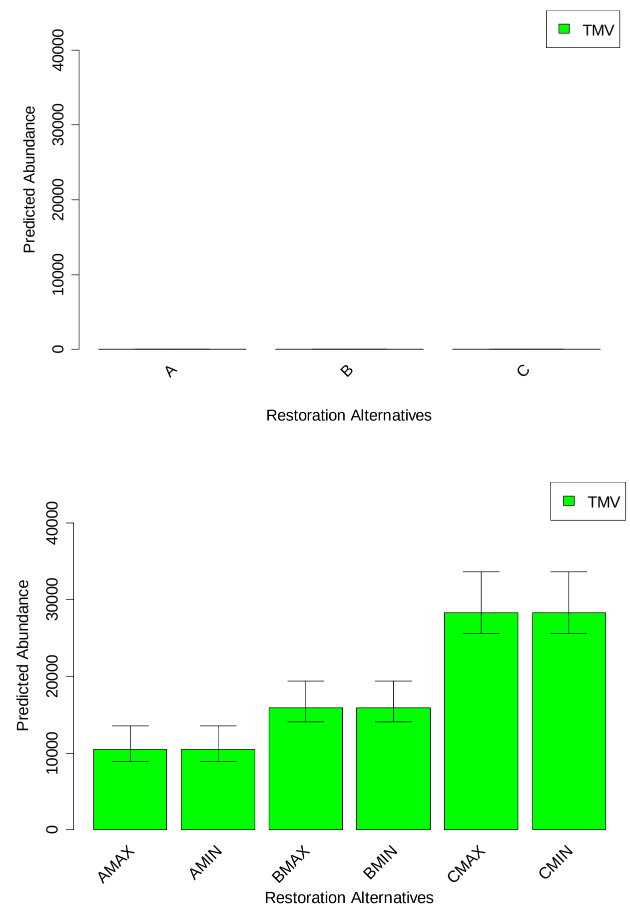

Figure 13. Breeding Song Sparrow project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. “Max” and “min” alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

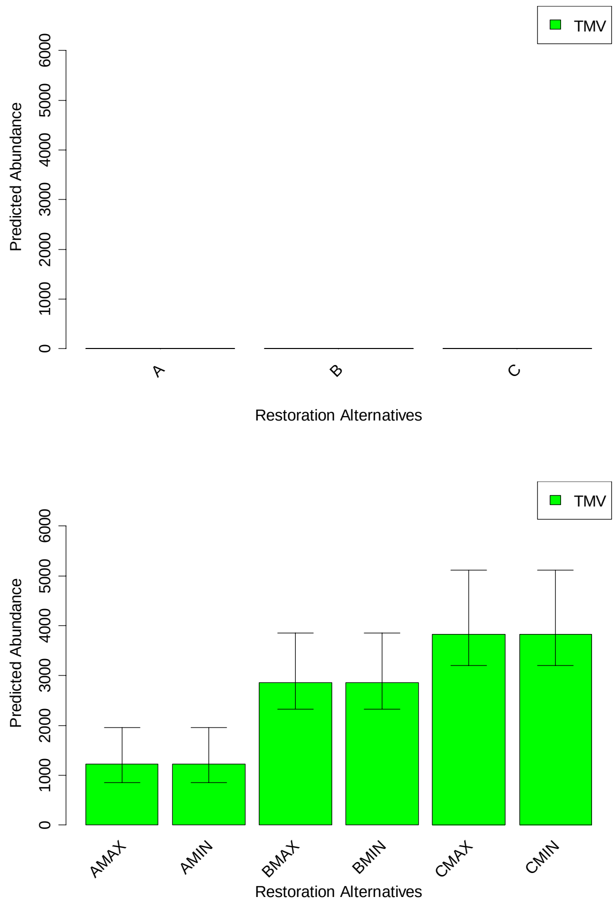

Figure 14. Breeding Marsh Wren project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

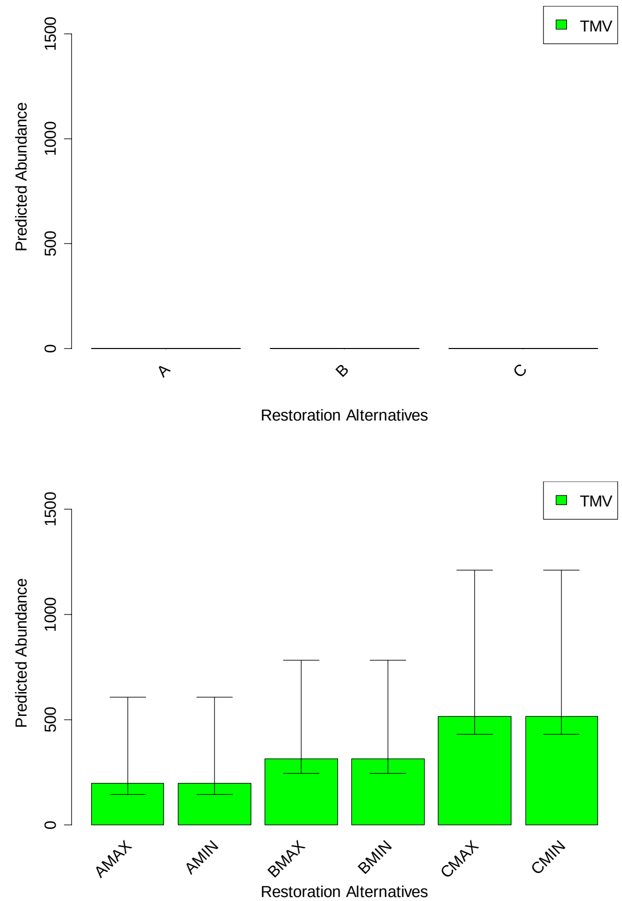

Figure 15. Breeding Common Yellowthroat project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

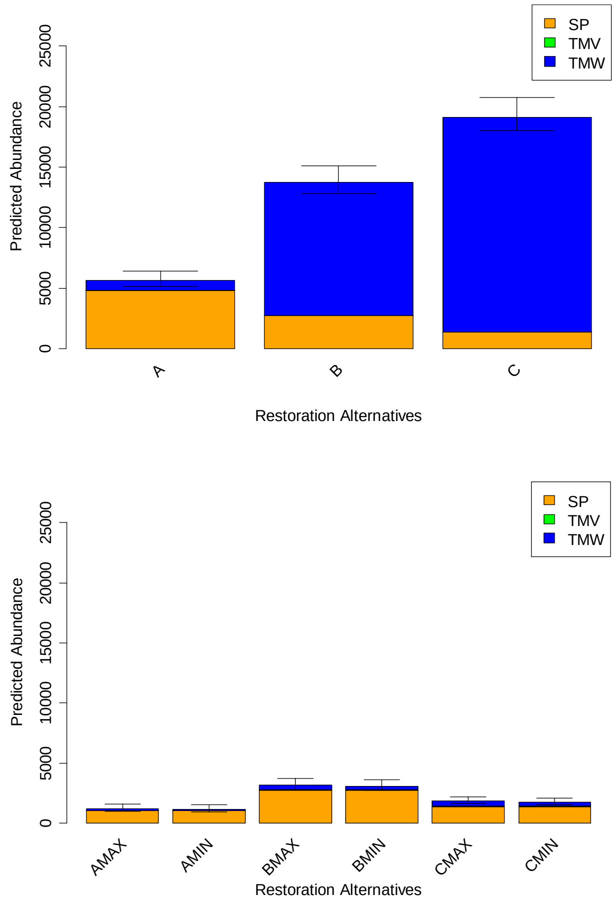

Figure 16. Winter small shorebird project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

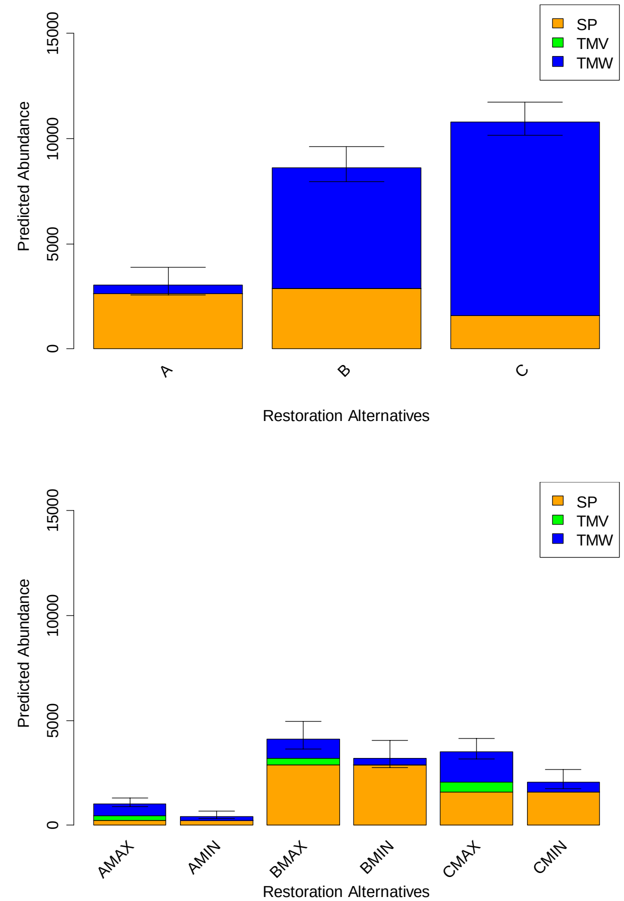

Figure 17. Fall small shorebird project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

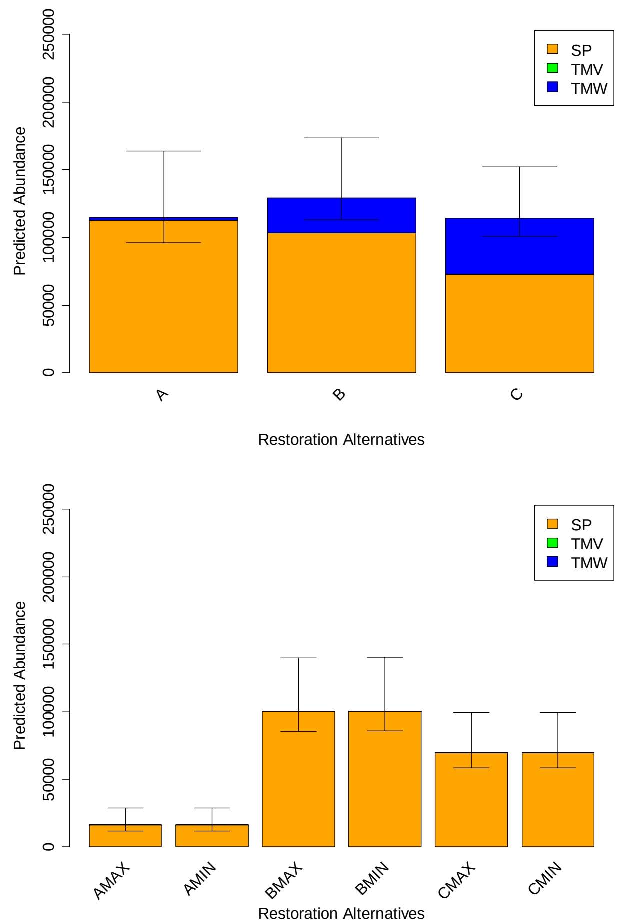

Figure 18. Spring small shorebirds project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

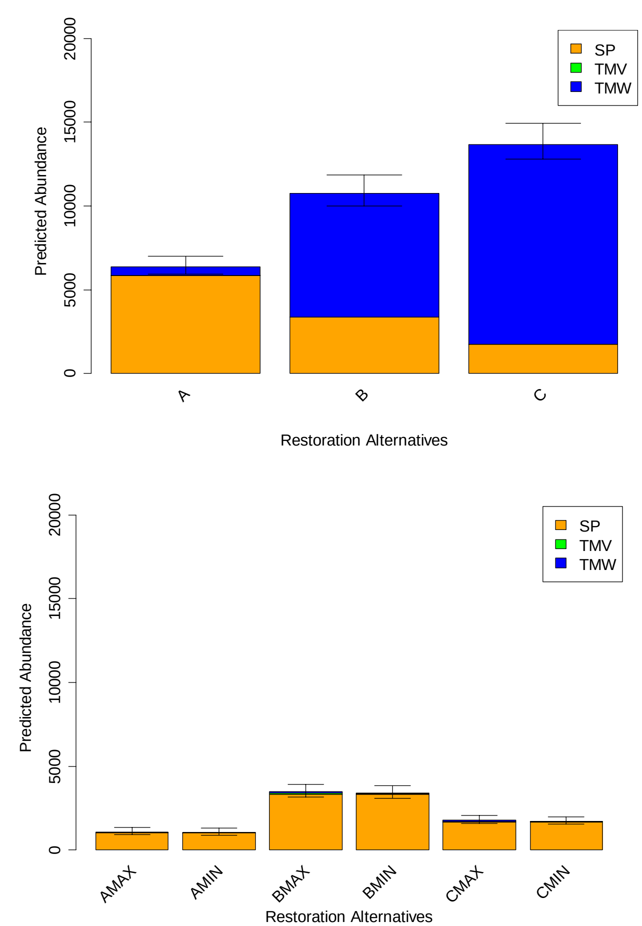

Figure 19. Winter Dunlin project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

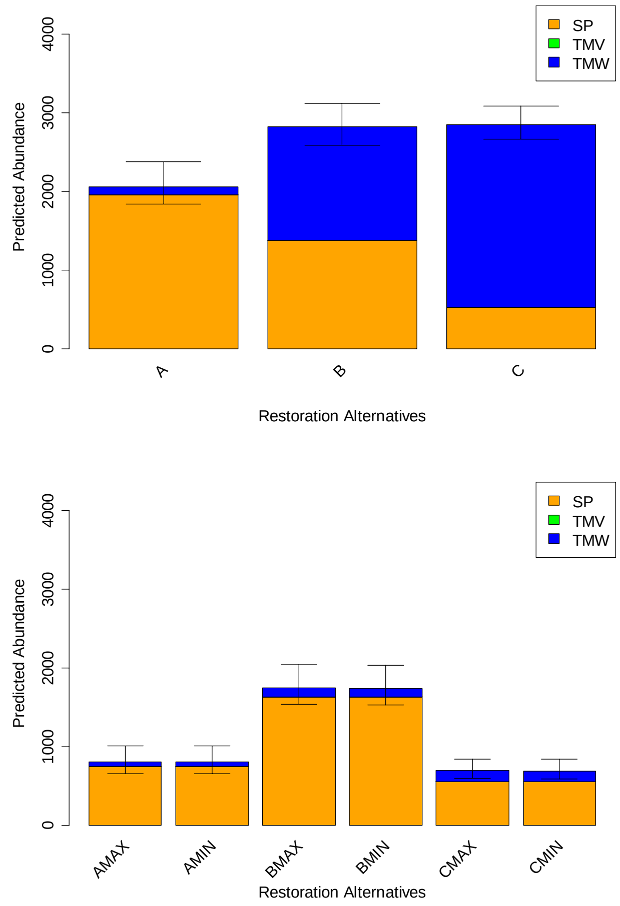

Figure 20. Winter Least Sandpiper project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. “Max” and “min” alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

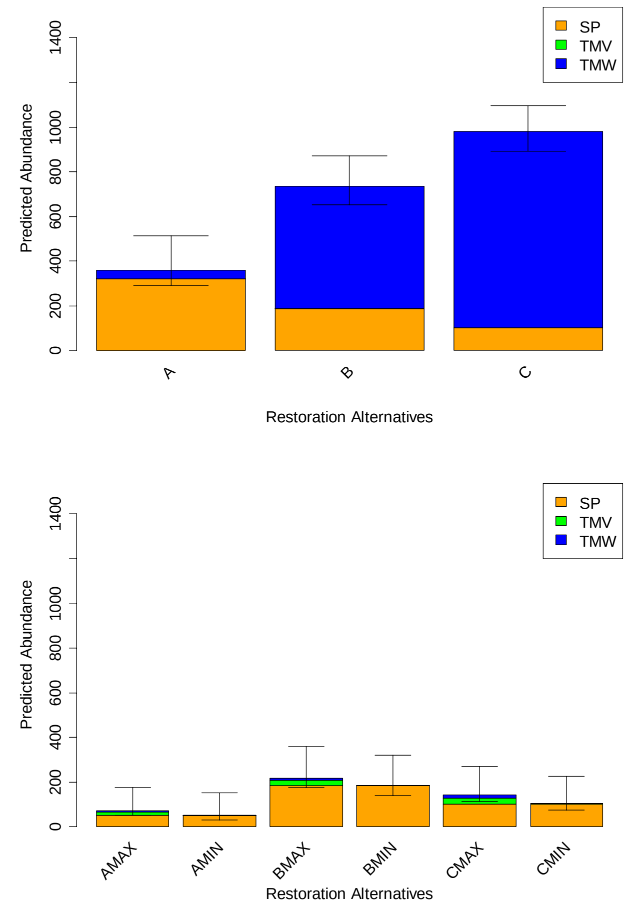

Figure 21. Winter Semipalmated Plover project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. “Max” and “min” alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

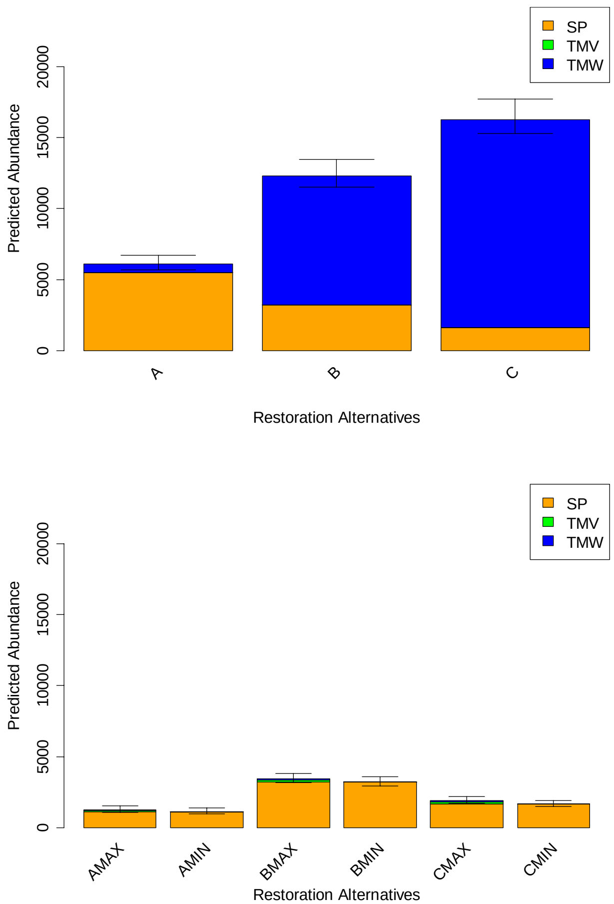

Figure 22. Winter Western Sandpiper project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. “Max” and “min” alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

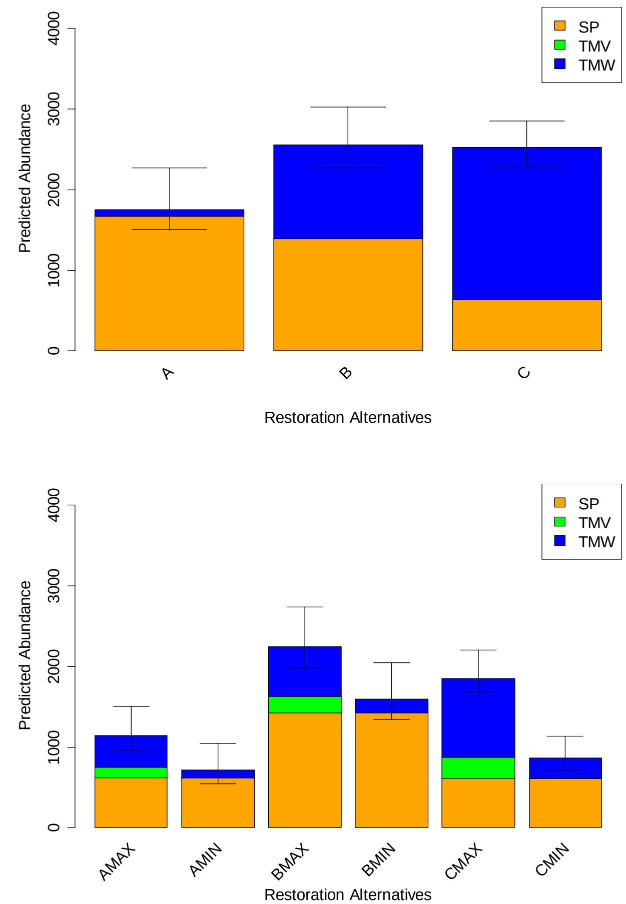

Figure 23. Fall Least Sandpiper project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. “Max” and “min” alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

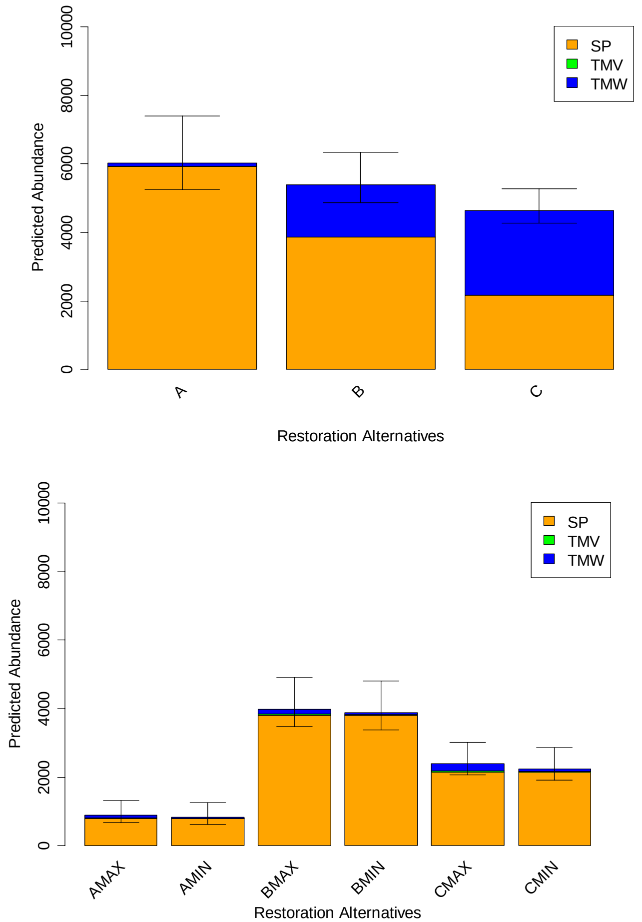

Figure 24. Fall Western Sandpiper project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

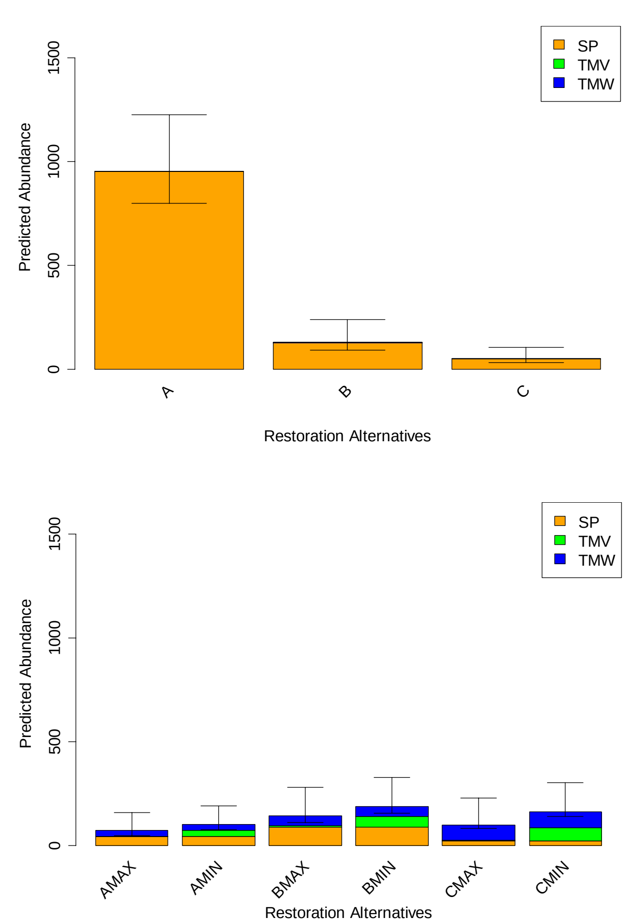

Figure 25. Spring Least Sandpiper project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

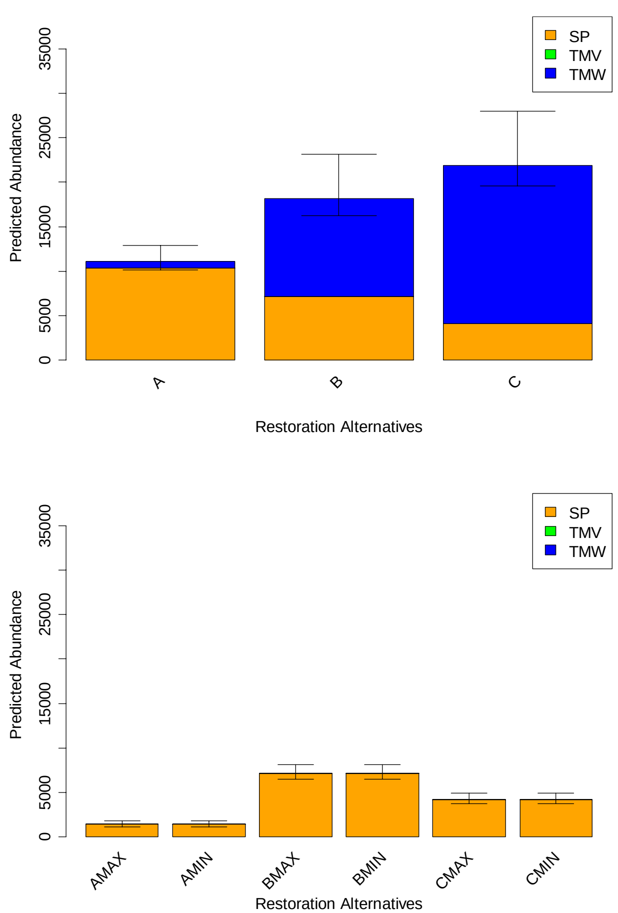

Figure 26. Spring Dunlin project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

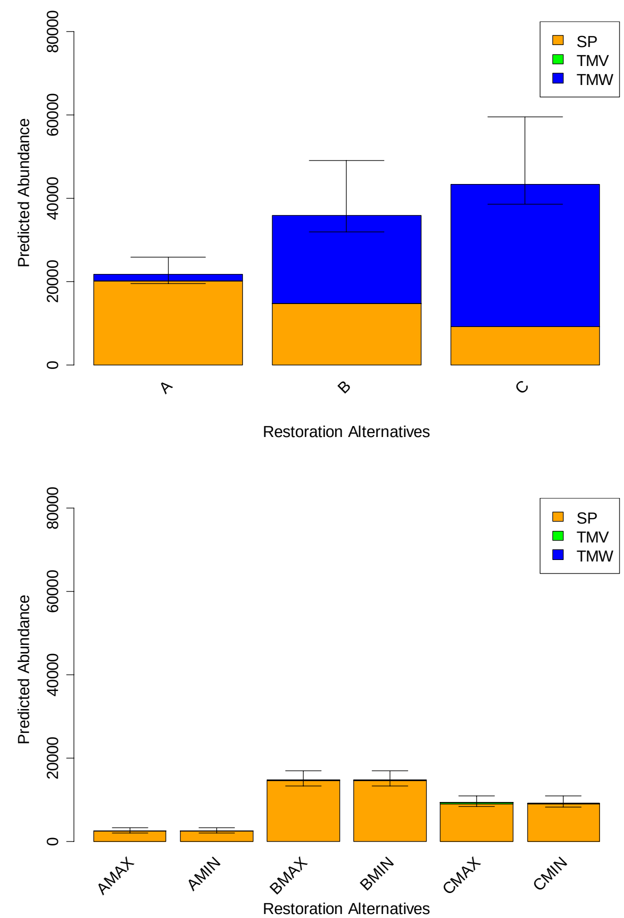

Figure 27. Spring Western Sandpiper project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

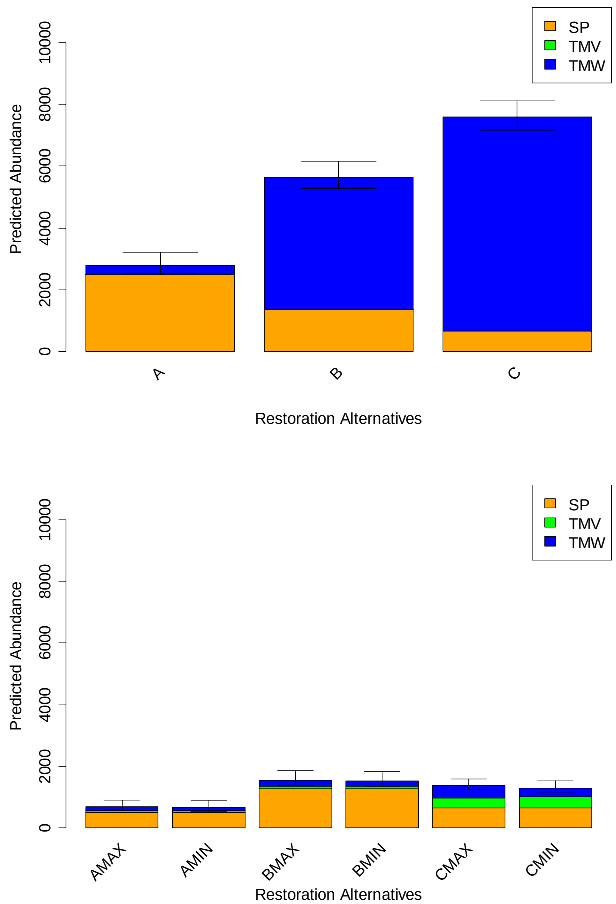

Figure 28. Winter medium shorebird project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

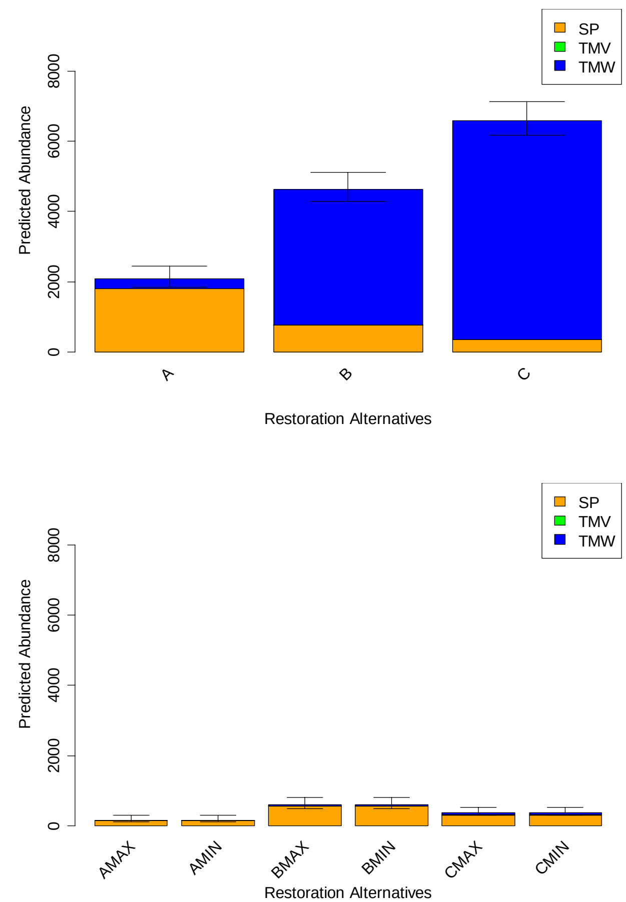

Figure 29. Winter American Avocet project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

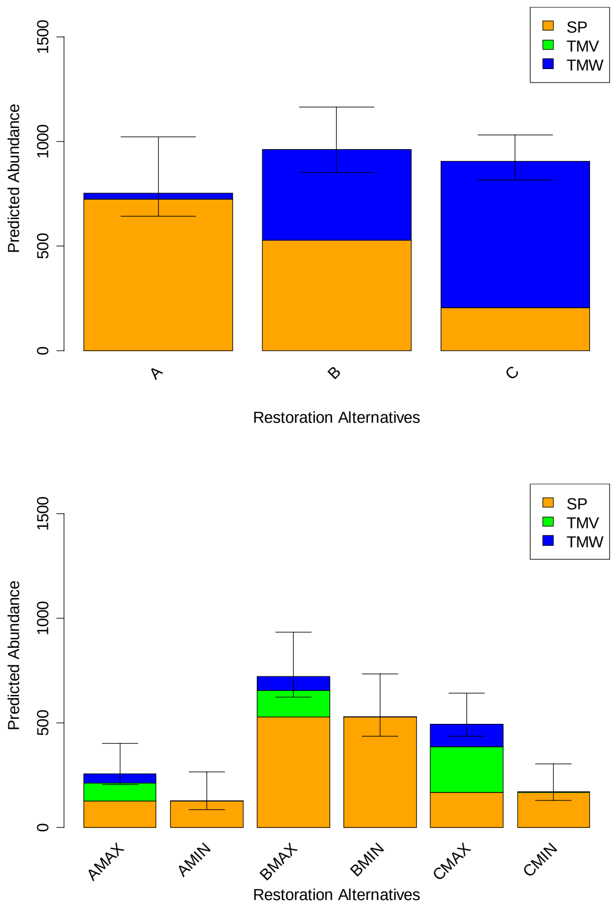

Figure 30. Winter Black-necked Stilt project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

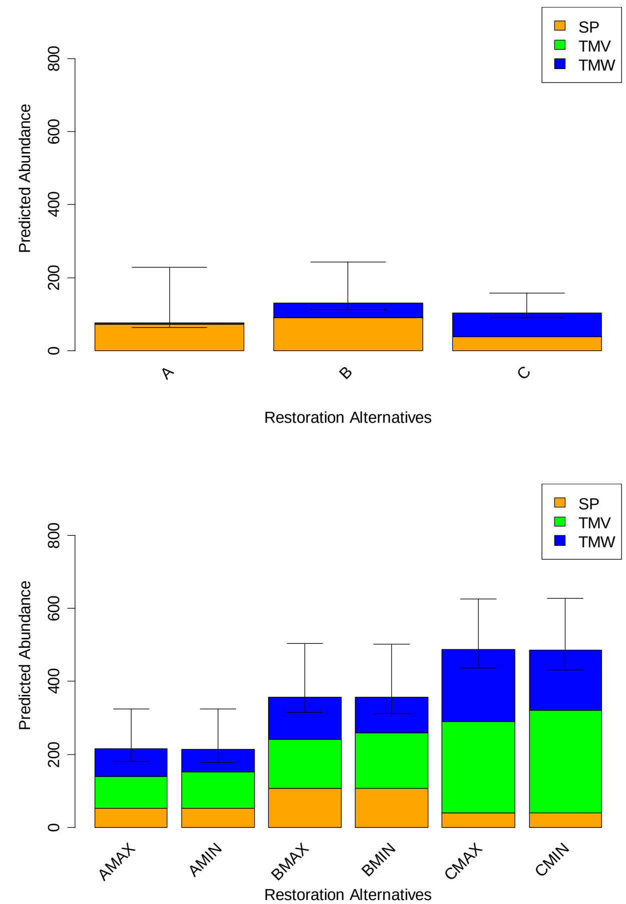

Figure 31. Winter Greater Yellowlegs scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

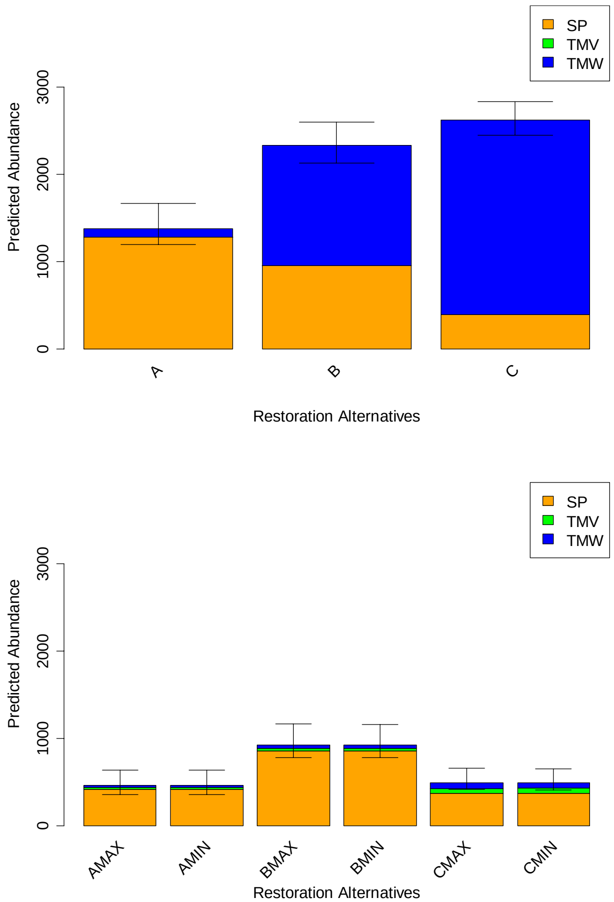

Figure 32. Winter Willet project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

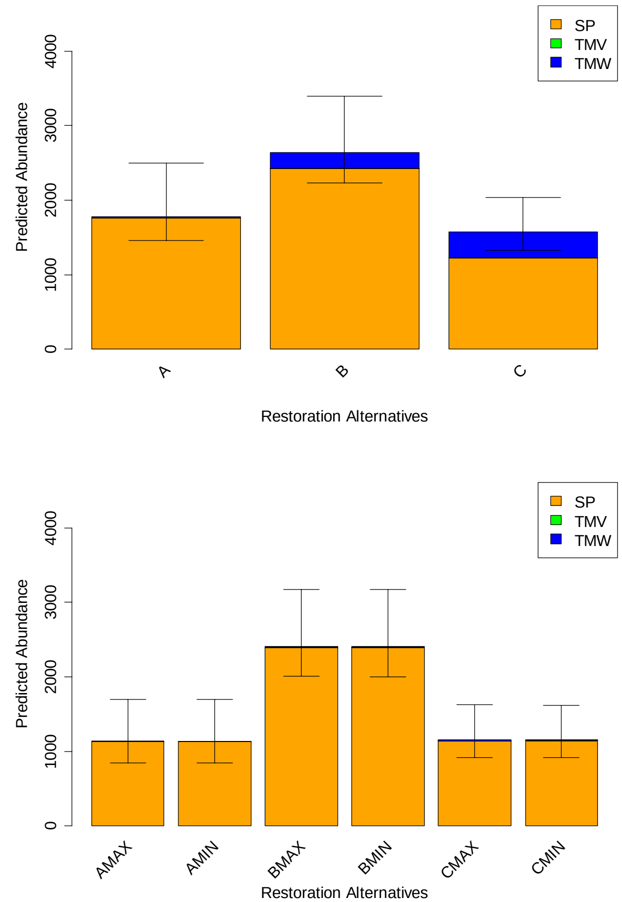

Figure 33. Fall Willet project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

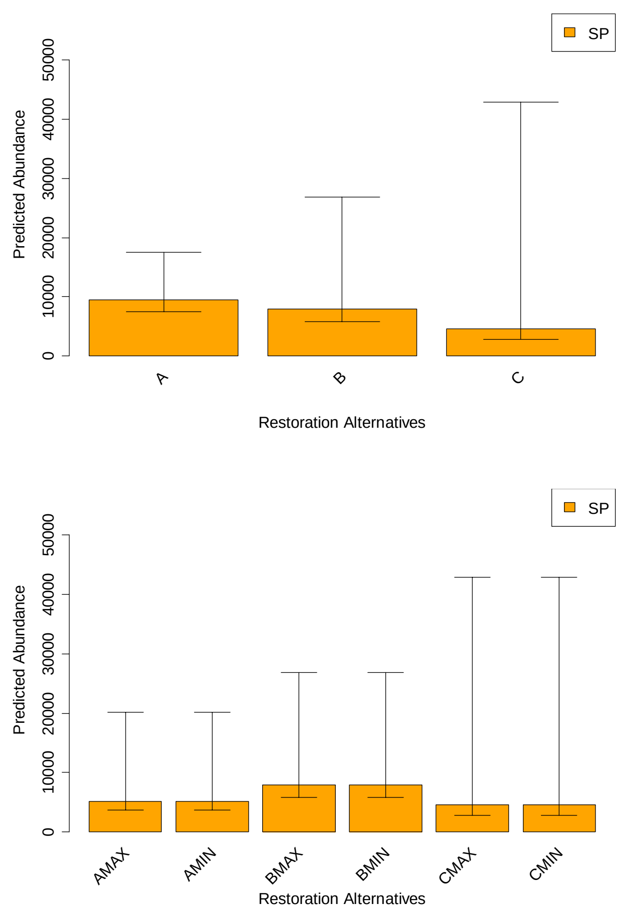

Figure 34. Fall phalarope project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

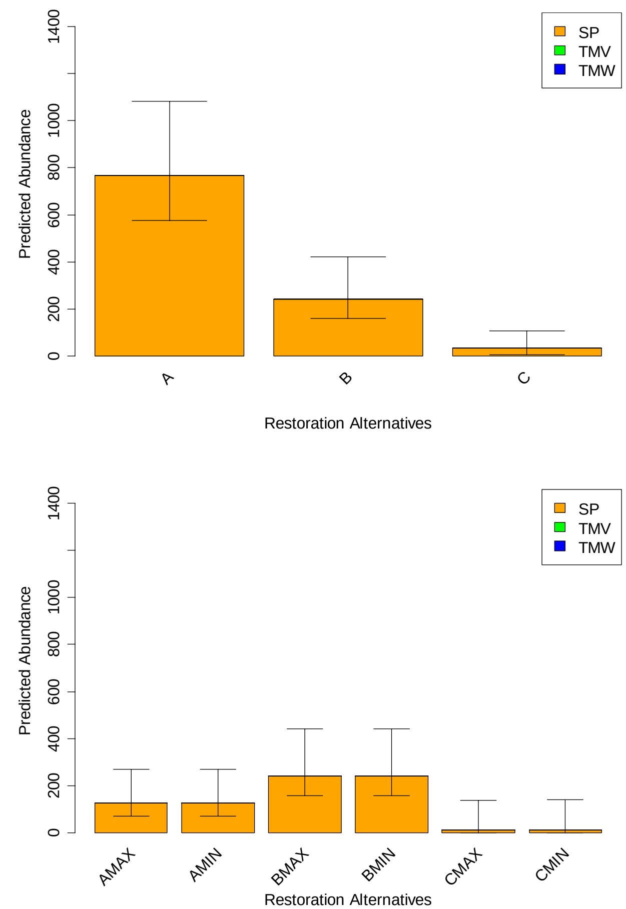

Figure 35. Fall Red-necked Phalarope project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

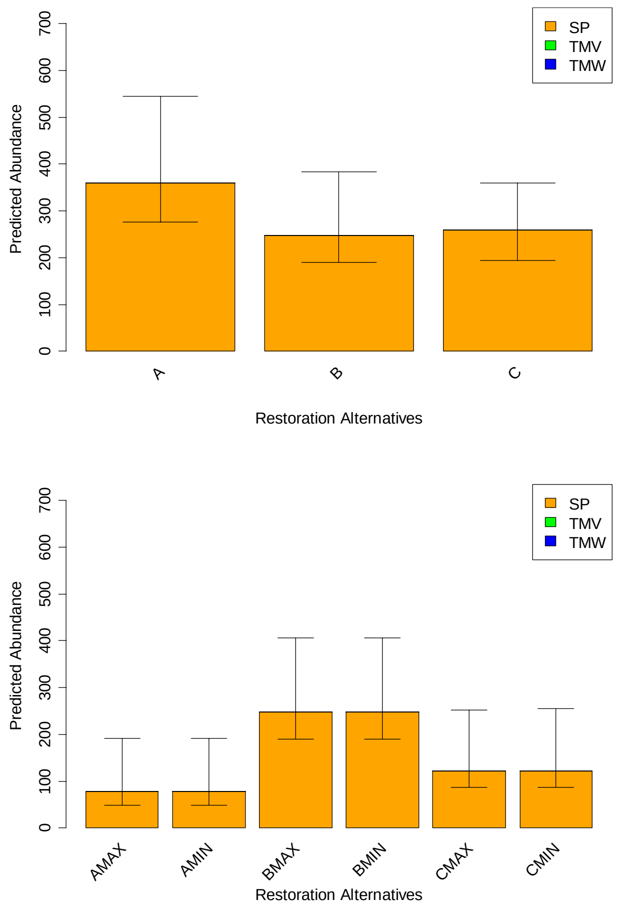

Figure 36. Fall Wilson's Phalarope project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

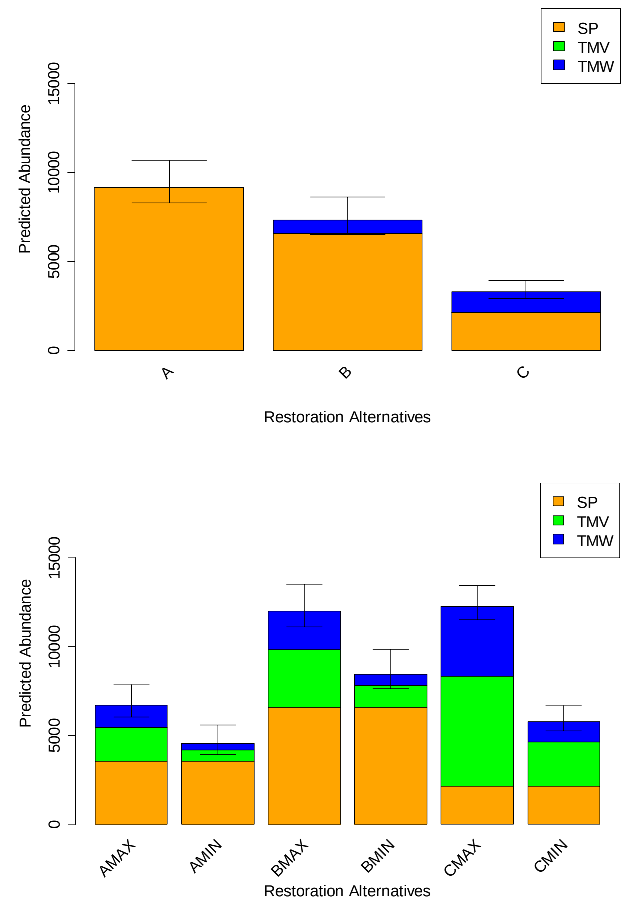

Figure 37. Winter dabbling duck project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

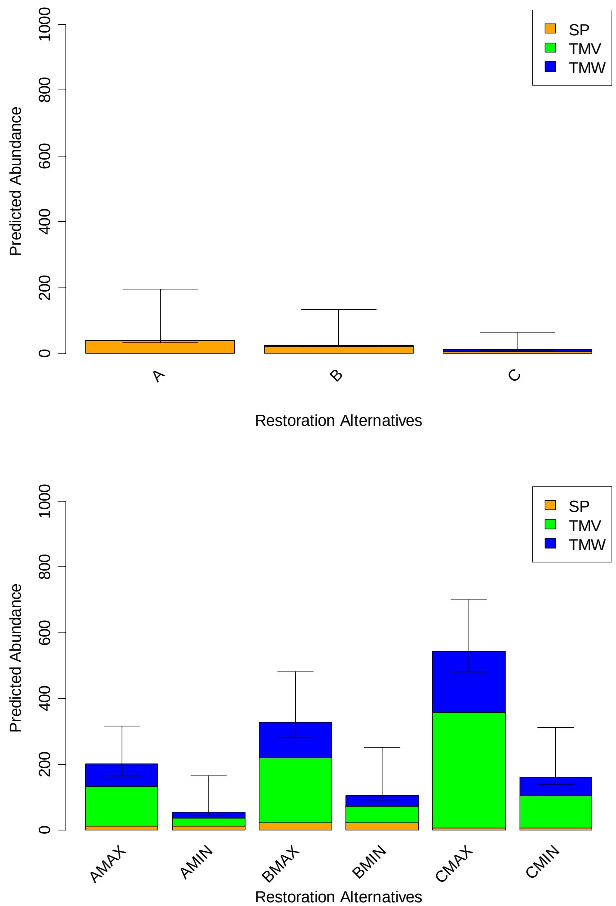

Figure 38. Winter Gadwall project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

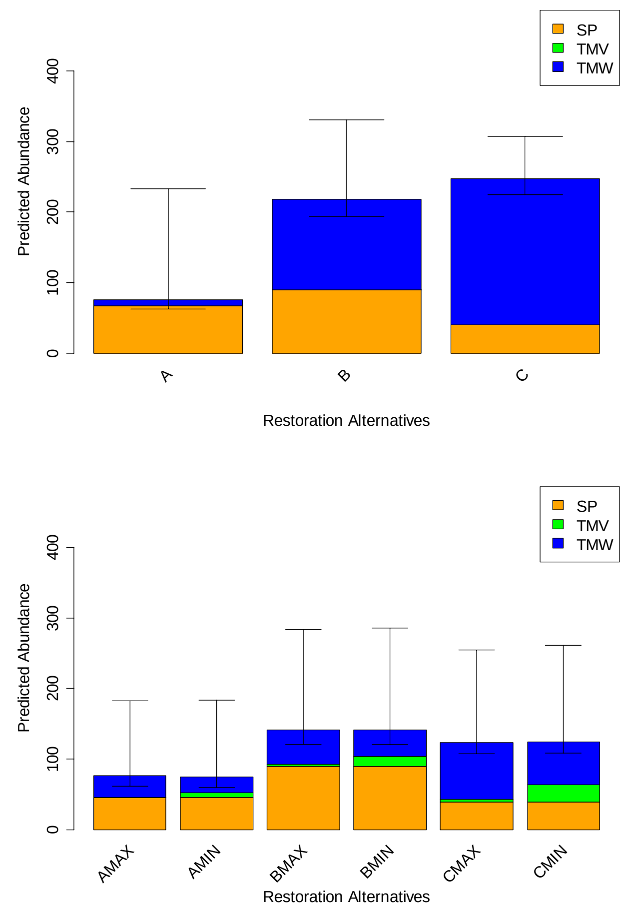

Figure 39. Winter Mallard project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

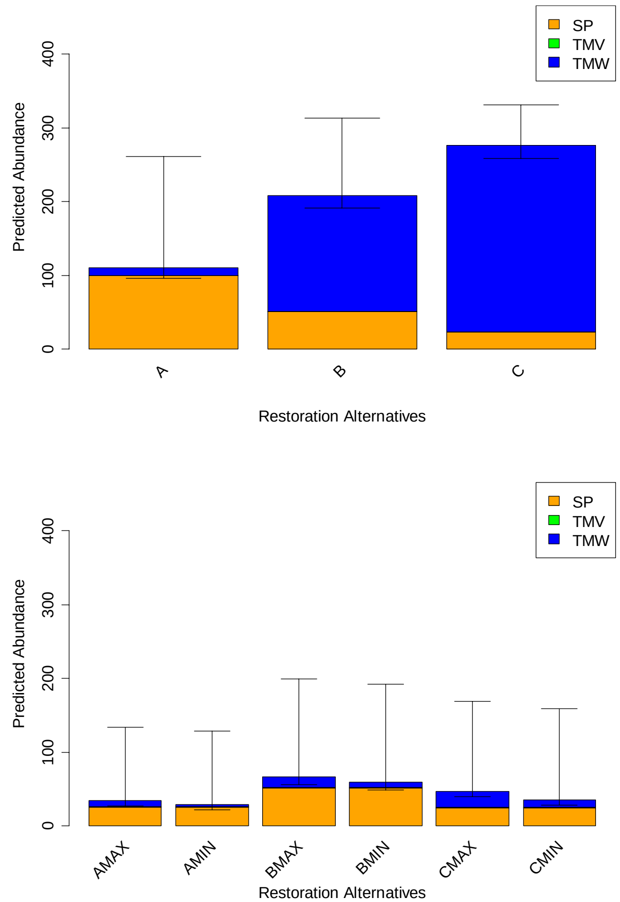

Figure 40. Winter Northern Pintail project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

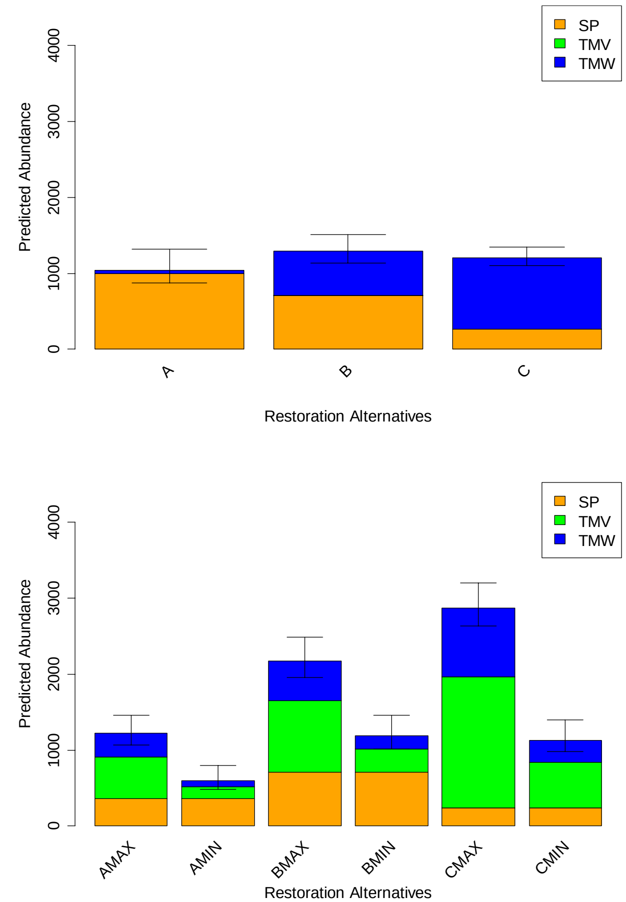

Figure 41. Winter Northern Shoveler scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

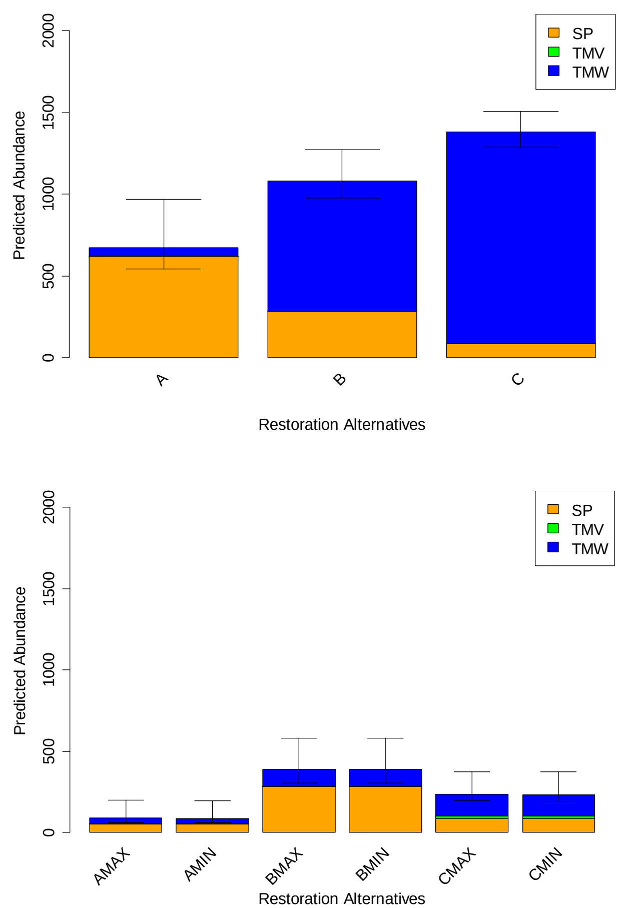

Figure 42. Winter diving duck project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

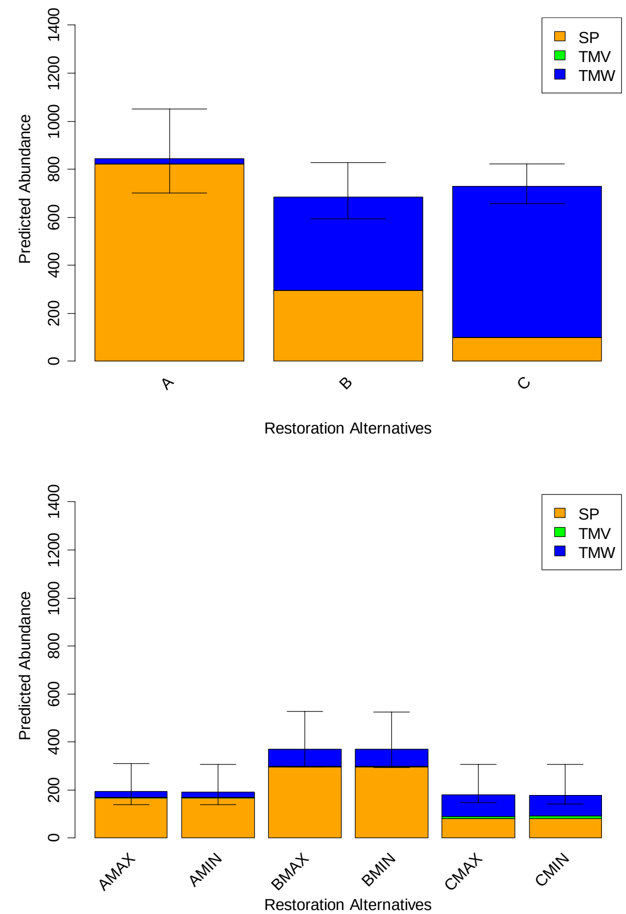

Figure 43. Winter Ruddy Duck scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

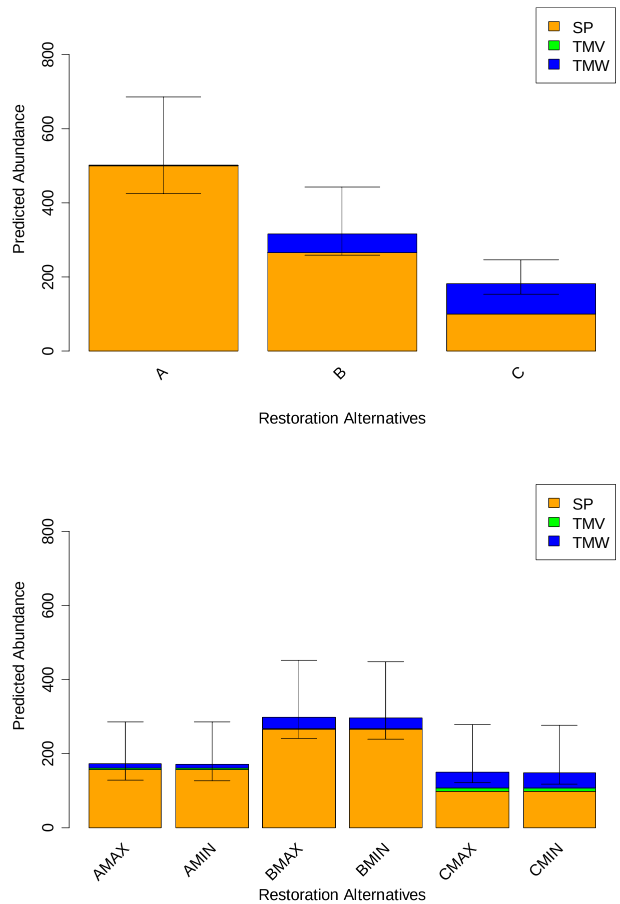

Figure 44. Winter scaup spp. scenario evaluation.
Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

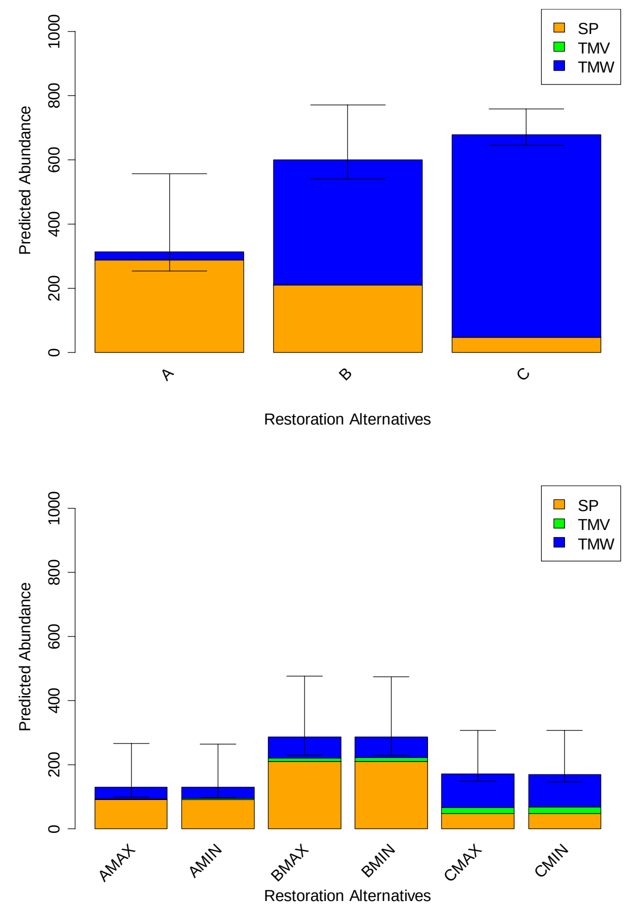

Figure 45. Winter heron and egret scenario evaluation.
Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

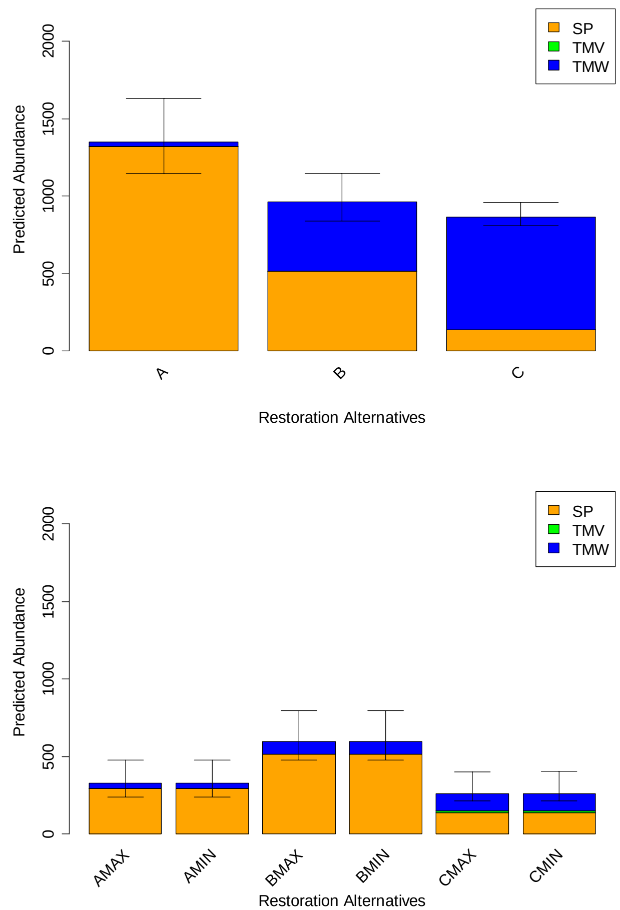

Figure 46. Winter fish-eater project area scenario evaluation.
Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. “Max” and “min” alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

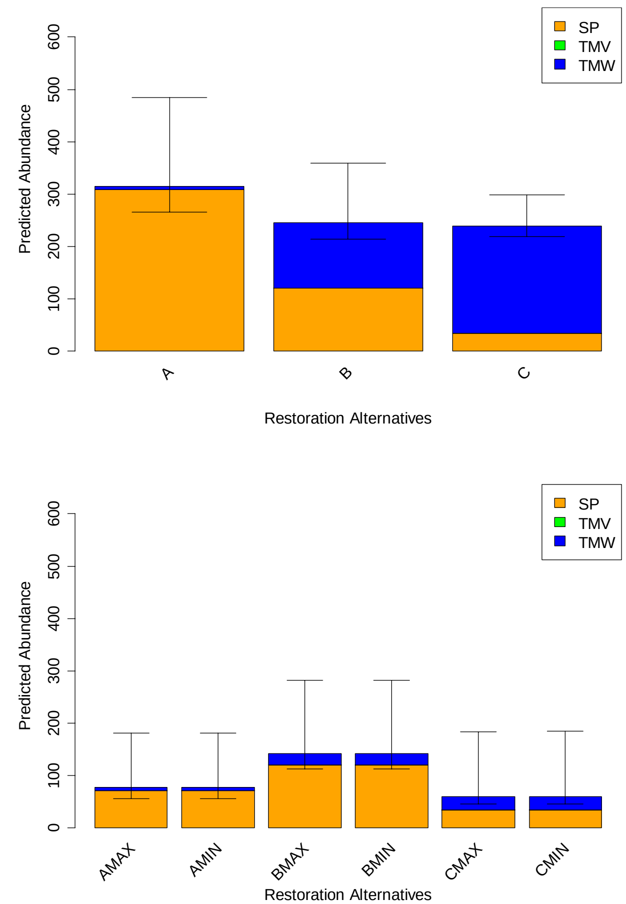

Figure 47. Winter American White Pelican project area scenario evaluation.
Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. “Max” and “min” alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

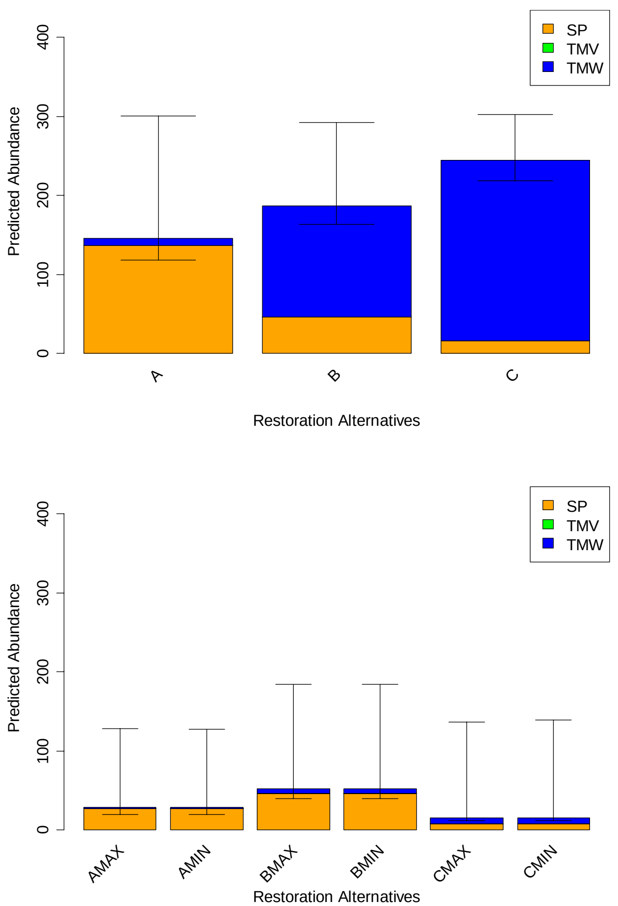

Figure 48. Winter Forster's Tern project area evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

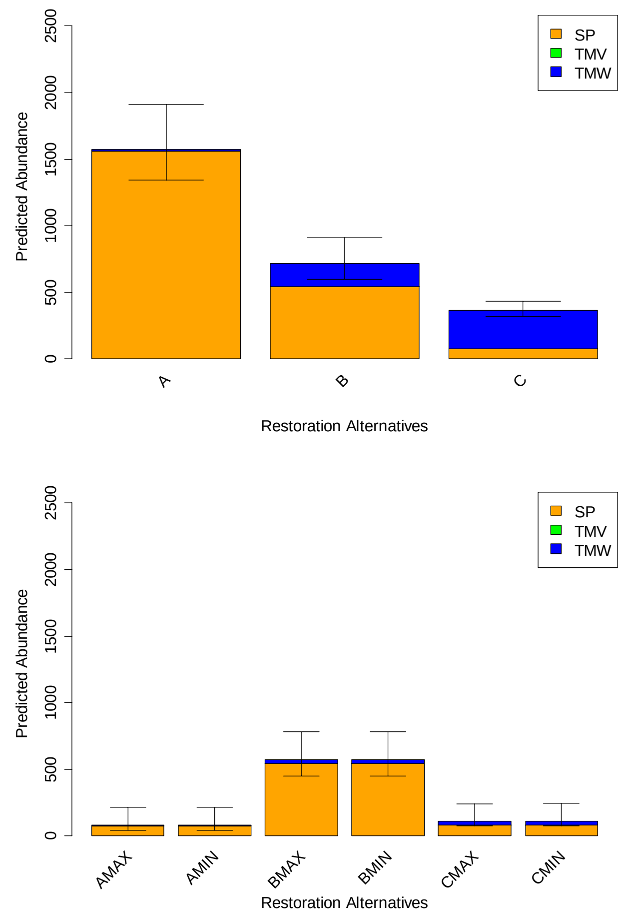

Figure 49. Winter Eared Grebe project area scenario evaluation. Predicted abundance index for each restoration alternative within the project area, by habitat type. TMV = tidal marsh vegetation; TMW = tidal marsh open water; SP = managed pond. "Max" and "min" alternatives refer to maximum and minimum assumptions about the expected tidal marsh pond/panne area. Error bars represent one standard error. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

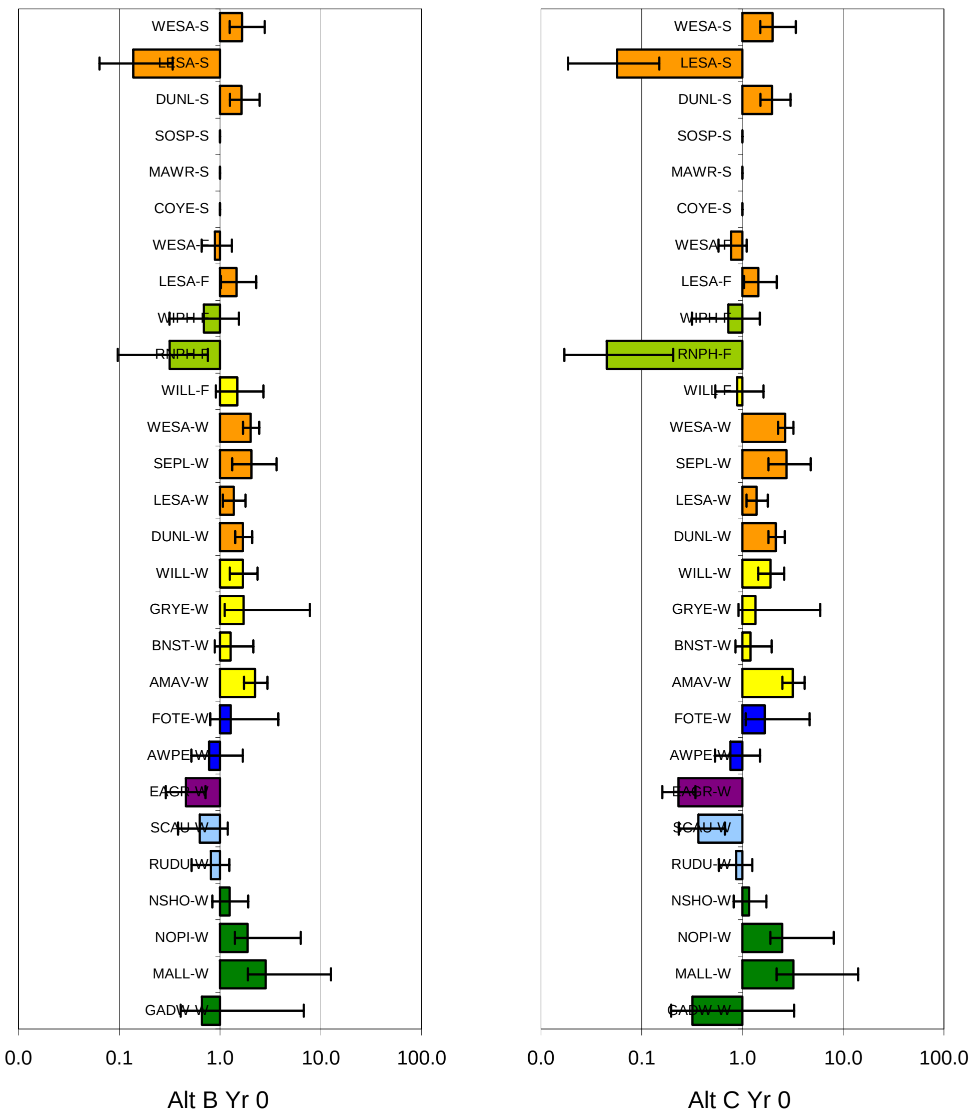

Figure 50. Project area scenario evaluation, year 0, "max" marsh pond/pannes.
Model-predicted proportion of baseline (alternative A, year 0) focal species' abundance indices across all restoration area managed ponds and restored marshes. Error bars represent 90% confidence intervals. Species 4-letter codes defined in Table 6. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data. S = spring, F = fall, W = winter.

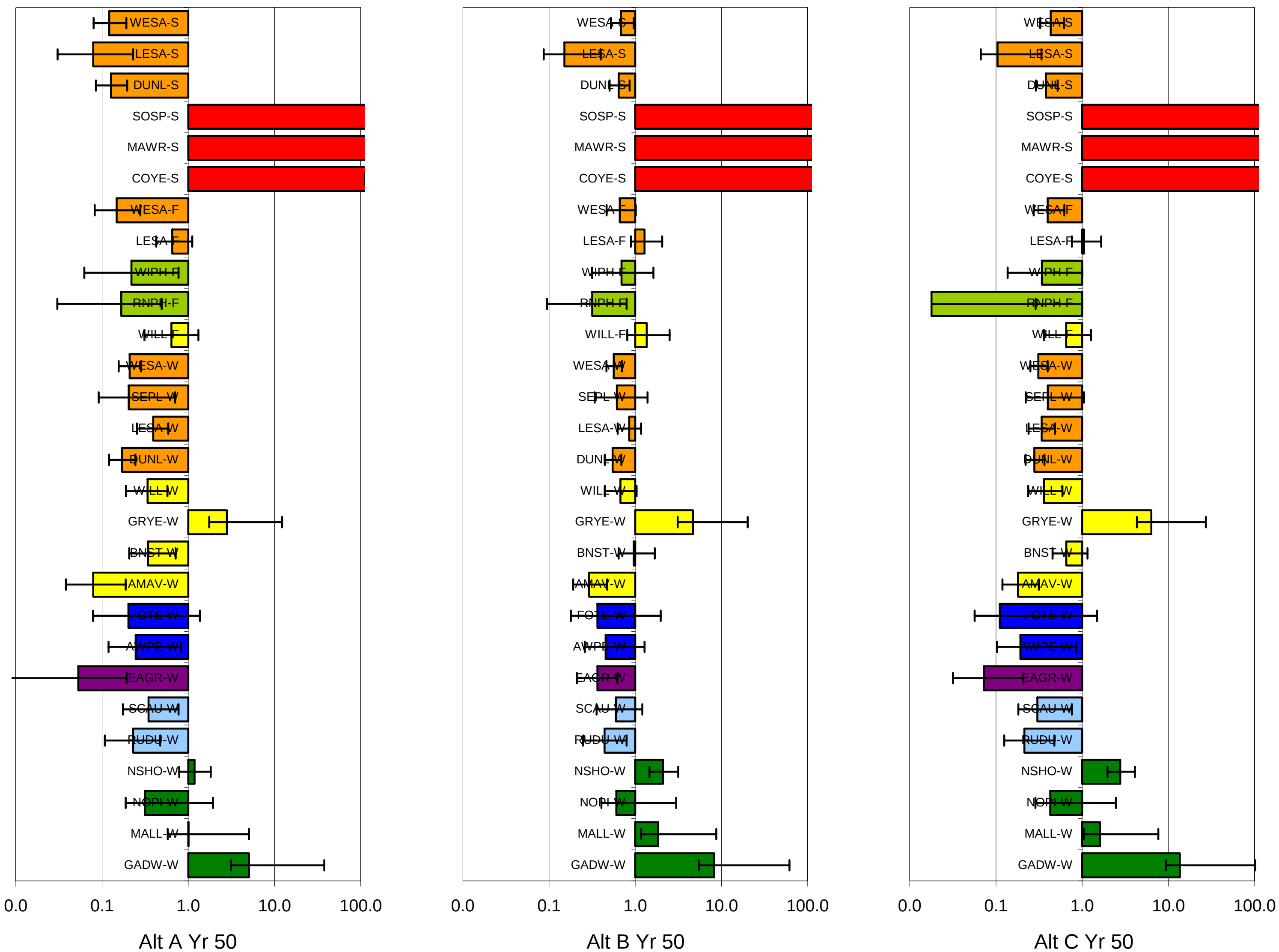

Figure 51. Project area scenario evaluation, year 50, "max" marsh pond/pannes.
Model-predicted proportion of baseline (alternative A, year 0) focal species' abundance indices across all restoration area managed ponds and restored marshes. Error bars represent 90% confidence intervals. Species 4-letter codes defined in Table 6. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data. S = spring, F = fall, W = winter.

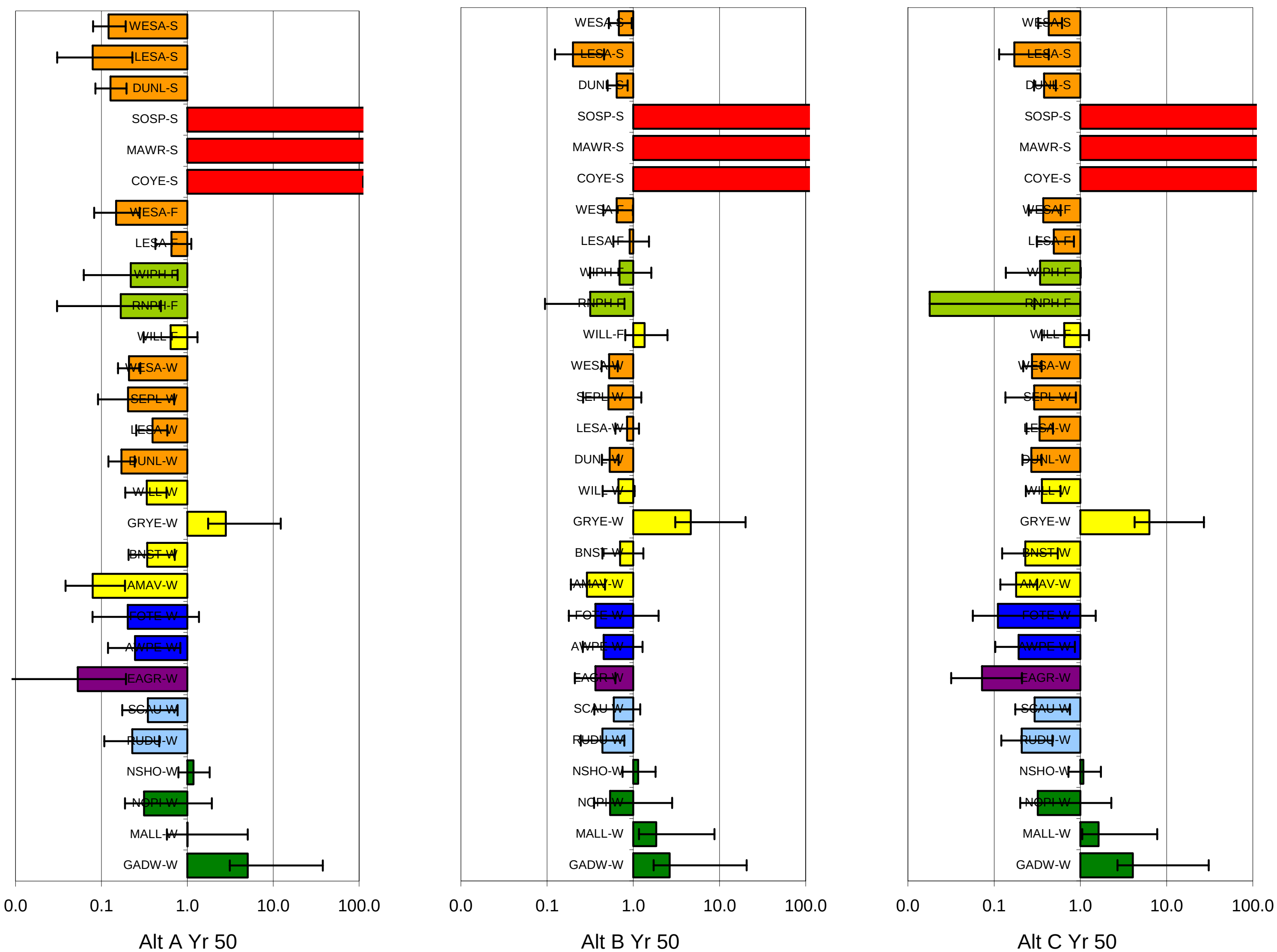

Figure 52. Project area scenario evaluation, year 50, "min" marsh pond/pannes.
Model-predicted proportion of baseline (alternative A, year 0) in focal species' abundance indices across all restoration area managed ponds and restored marshes. Error bars represent 90% confidence intervals. Species 4-letter codes defined in Table 6. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data. S = spring, F = fall, W = winter.

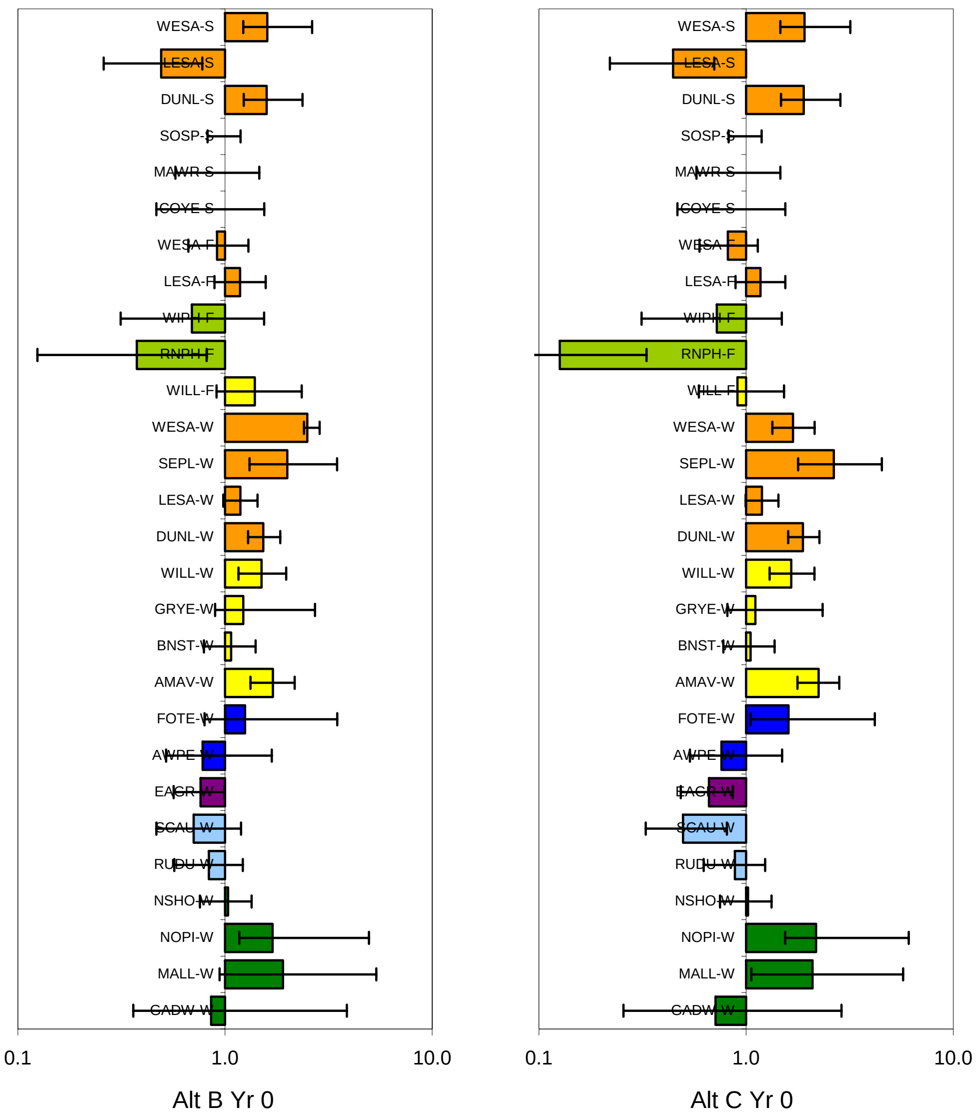

Figure 53. South Bay scenario evaluation, year 0.
Model-predicted proportion of baseline (alternative A, year 0) focal species' abundance indices across all South Bay habitats. Error bars represent 90% confidence intervals. Species 4-letter codes defined in Table 6. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data. S = spring, F = fall, W = winter.

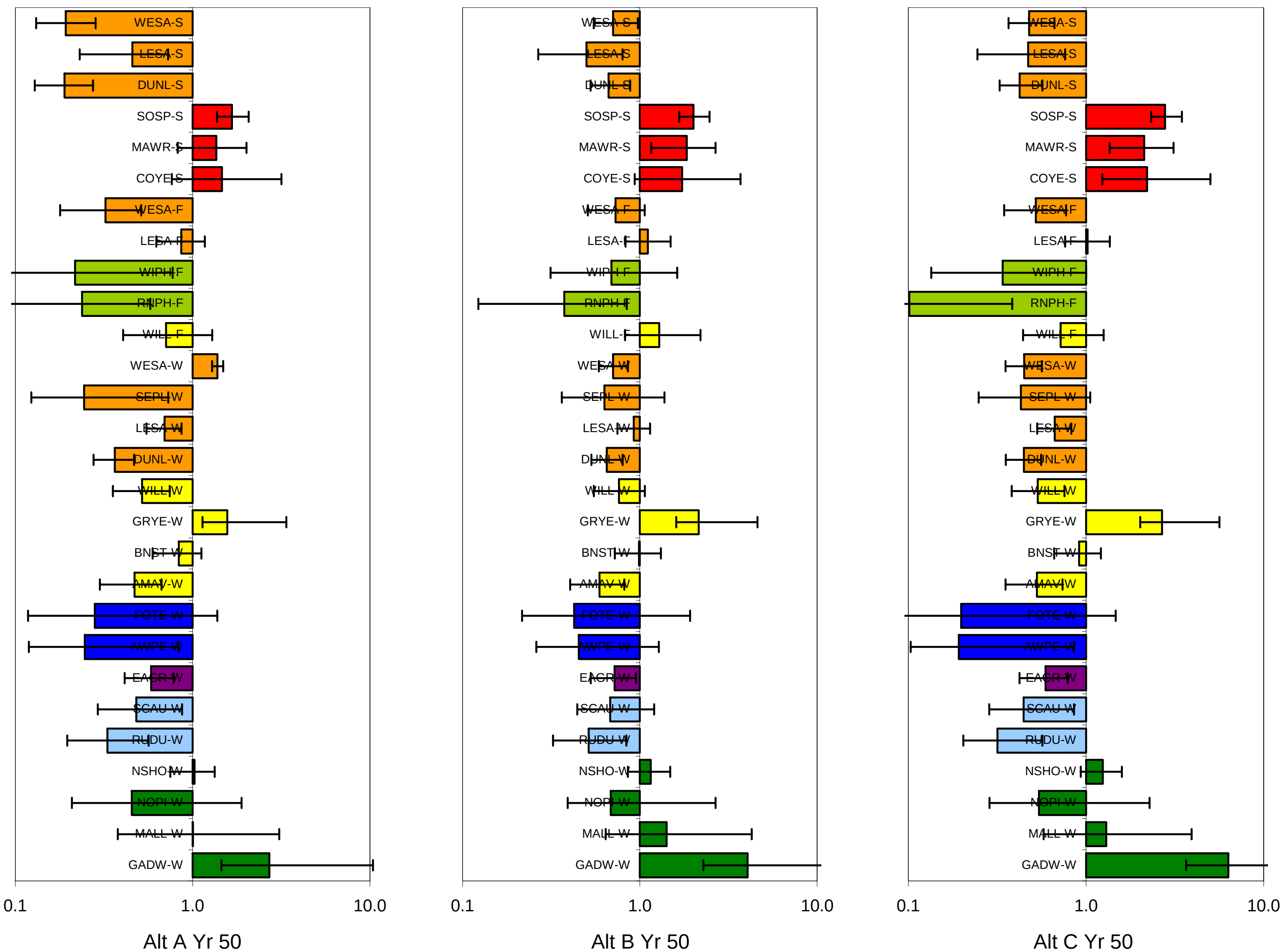

Figure 54. South Bay scenario evaluation, year 50, "max" marsh pond/pannes.
Model-predicted proportion of baseline (alternative A, year 0) focal species' abundance indices across all South Bay habitats. Error bars represent 90% confidence intervals. Species 4-letter codes defined in Table 6. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data. S = spring, F = fall, W = winter.

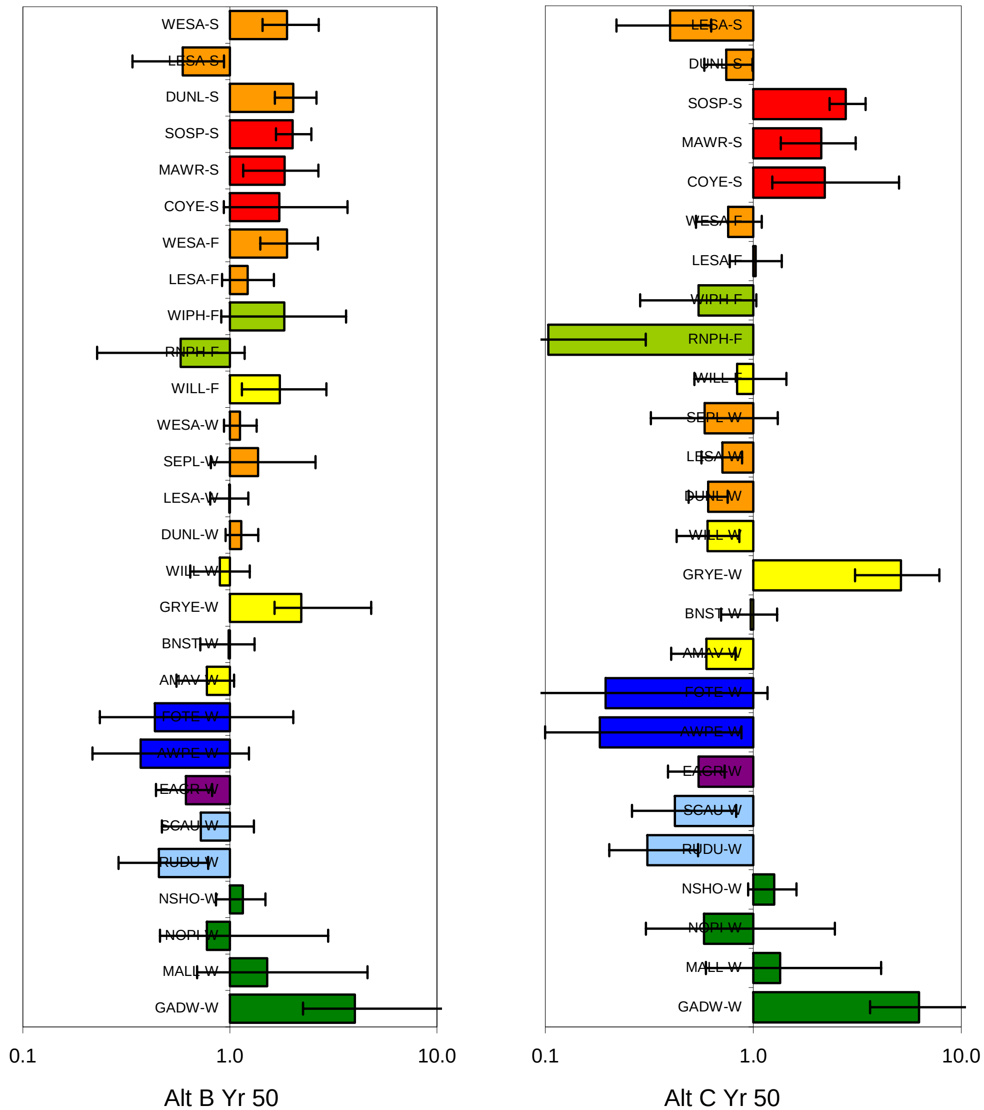

Figure 55. South Bay scenario evaluation, "shallow" alternatives, year 50.
Model-predicted proportion of baseline (alternative A, year 0) focal species' abundance indices across all South Bay habitats. Error bars represent 90% confidence intervals. Species 4-letter codes defined in Table 6. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data. S = spring, F = fall, W = winter.

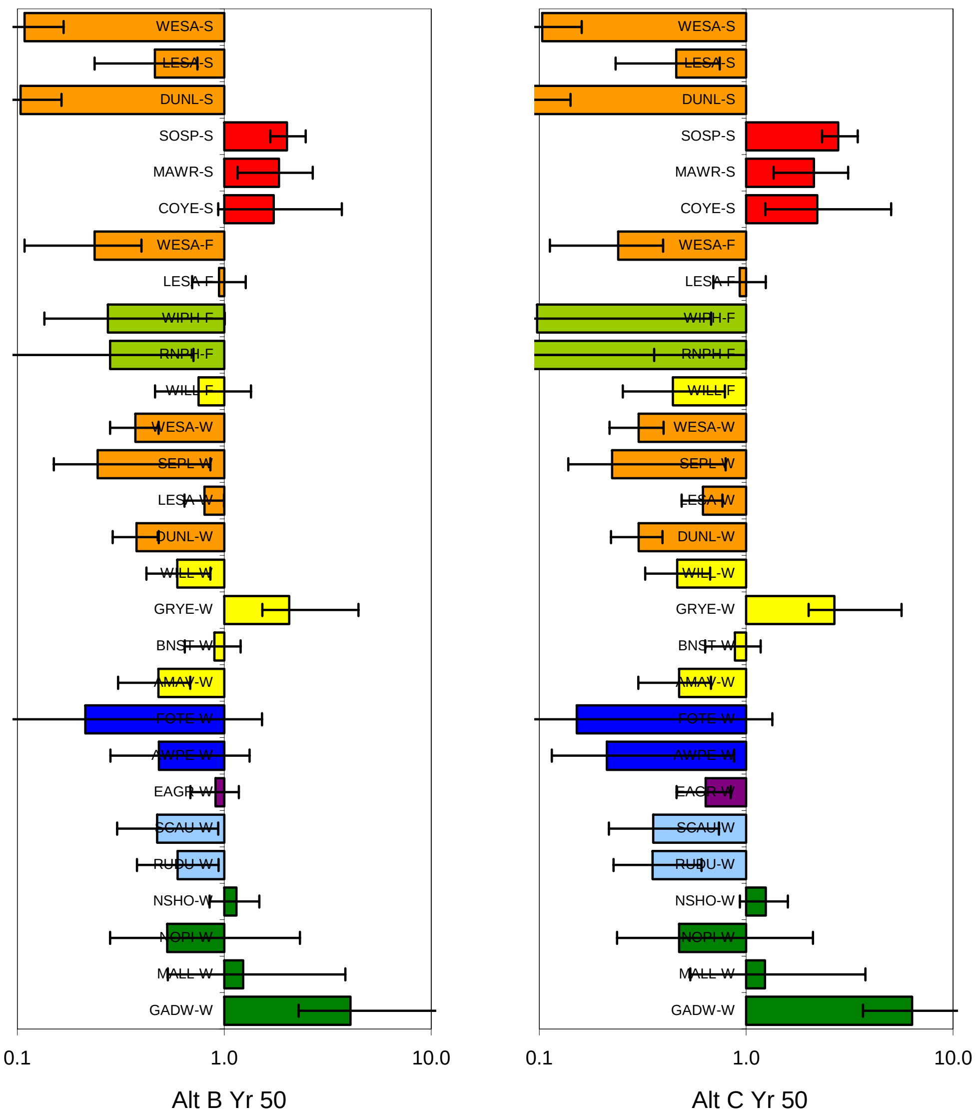

Figure 56. South Bay scenario evaluation, "deep" alternatives, year 50.
Model-predicted proportion of baseline (alternative A, year 0) focal species' abundance indices across all South Bay habitats. Error bars represent 90% confidence intervals. Species 4-letter codes defined in Table 6. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data. S = spring, F = fall, W = winter.

TABLES

Table 1. Surveyed salt pond characteristics.
South San Francisco Bay salt ponds surveyed between October 1999 and January 2004. Years surveyed indicated with 1 (surveyed) or 0 (not surveyed).

Pond ID	Complex	1999/ 2000	2000/ 2001	2002/ 2003	2003/ 2004	Pond Size (ha)	Mean Salinity (ppt)	Min. Salinity (ppt)	Max. Salinity (ppt)	Bathymetry Source
A1	Alviso	0	0	1	1	115.3	25	19	31	Boat
A11	Alviso	0	0	1	1	108.5	62	50	72	Boat
A11	Alviso	0	1	0	0	108.5	59	51	71	Boat
A13	Alviso	0	0	1	1	114.4	73	59	80	Boat
A14	Alviso	0	1	0	0	142.0	79	67	100	Boat
A14	Alviso	0	0	1	1	142.0	89	73	107	Boat
A15	Alviso	0	0	1	1	101.9	83	76	94	Boat
A16	Alviso	0	1	0	0	97.3	70	63	81	Boat
A16	Alviso	0	0	1	1	97.3	95	89	106	Boat
A17	Alviso	0	0	1	1	55.2	97	88	109	Boat
A19	Alviso	0	0	0	1	111.8	176	158	207	Boat
A20	Alviso	0	0	1	1	27.0	218	150	372	Boat
A21	Alviso	0	0	1	1	57.6	177	74	231	Boat
A22	Alviso	0	0	1	1	110.6	163	77	286	LiDAR
A23	Alviso	0	0	1	1	183.9	247	215	282	LiDAR
A5	Alviso	0	0	1	1	267.6	49	35	58	LiDAR
AB1	Alviso	0	0	1	0	63.9	21	21	21	Boat
AB2	Alviso	0	0	0	1	75.4	32	27	39	Boat
B1	EdenLanding	0	0	1	0	120.3	32	29	35	Boat
B10	EdenLanding	0	1	0	0	108.8	32	27	37	Boat
B10	EdenLanding	0	0	1	1	108.8	32	23	41	Boat
B11	EdenLanding	0	1	0	0	51.7	38	25	53	Boat
B11	EdenLanding	0	0	1	1	51.7	41	22	58	Boat
B12	EdenLanding	0	0	1	0	47.5	129	129	129	LiDAR
B12/13	EdenLanding	0	1	0	0	101.9	49	38	60	LiDAR
B13	EdenLanding	0	0	1	0	54.3	129	129	129	LiDAR
B14	EdenLanding	0	1	0	0	69.5	65	48	80	Boat
B14	EdenLanding	0	0	1	1	69.5	128	57	260	Boat
B1C	EdenLanding	0	0	1	1	26.5	56	32	97	Boat
B2	EdenLanding	0	0	1	1	280.0	45	40	49	Boat
B2C	EdenLanding	0	0	1	1	13.0	89	31	264	Boat
B4	EdenLanding	0	0	1	1	81.8	48	33	105	Boat
B5	EdenLanding	0	0	1	1	69.5	71	35	111	Boat
B5C	EdenLanding	0	0	1	1	38.8	58	30	106	Boat
B6A	EdenLanding	0	1	0	0	133.1	68	62	77	LiDAR
B6A	EdenLanding	0	0	1	1	133.1	60	40	141	LiDAR
B6B	EdenLanding	0	0	1	1	118.7	70	47	127	LiDAR
B6C	EdenLanding	0	0	1	1	34.3	67	38	127	Boat
B7	EdenLanding	0	0	1	1	87.9	47	38	57	Boat
B8A	EdenLanding	0	1	0	0	125.4	125	93	185	LiDAR
B8A	EdenLanding	0	0	1	1	125.4	151	72	277	LiDAR
B9	EdenLanding	0	1	0	0	156.2	99	77	139	Boat
B9	EdenLanding	0	0	1	1	156.2	90	53	135	Boat
R2	Ravenswood	1	0	0	0	57.2	228	176	263	LiDAR
R2	Ravenswood	0	0	0	1	57.2	63	53	74	LiDAR
R3	Ravenswood	0	0	0	1	120.0	132	132	132	LiDAR
R5	Ravenswood	0	0	1	1	14.3	138	65	235	LiDAR
RSF2	Ravenswood	0	0	1	1	96.9	103	52	191	LiDAR
RSF2	Ravenswood	1	1	0	0	96.9	236	156	264	LiDAR

Table 1, cont.

Pond ID	Mean Depth (m)	Minimum Depth (m)	Maximum Depth (m)	Mean Shallow Prop.	Min. Shallow Prop.	Max. Shallow Prop.	Mean Deep Prop.	Min. Deep Prop.	Max. Deep Prop.
A1	0.70	0.52	0.96	0.00	0.00	0.01	0.03	0.65	19.38
A11	1.29	1.16	1.60	0.00	0.00	0.01	0.86	68.66	98.47
A11	1.28	1.04	1.44	0.00	0.00	0.28	0.82	45.21	97.33
A13	1.63	1.33	2.16	0.00	0.00	0.00	0.94	85.15	99.98
A14	1.01	0.73	1.30	0.00	0.00	0.04	0.43	14.90	87.49
A14	1.09	0.90	1.36	0.00	0.00	0.00	0.51	21.93	92.49
A15	1.78	1.57	1.88	0.00	0.00	0.00	0.97	84.95	99.85
A16	1.61	1.31	1.77	0.00	0.00	0.00	0.83	58.19	93.30
A16	1.73	1.56	1.86	0.00	0.00	0.00	0.92	83.26	97.59
A17	2.32	2.10	3.01	0.00	0.00	0.00	1.00	100.00	100.00
A19	0.60	0.05	0.94	0.01	0.00	3.39	0.22	8.48	30.71
A20	0.91	0.74	1.08	0.01	0.00	4.80	0.35	30.15	40.57
A21	0.63	0.53	0.85	0.01	0.00	3.93	0.17	15.73	21.75
A22	0.05	-0.18	0.92	0.20	0.62	27.55	0.04	0.00	41.85
A23	0.11	0.06	0.15	0.25	19.73	31.30	0.00	0.00	0.00
A5	-0.02	-0.11	0.18	0.46	4.28	61.81	0.00	0.00	0.00
AB1	0.50	0.50	0.50	0.01	0.84	0.84	0.04	3.59	3.59
AB2	0.50	0.25	0.80	0.06	0.00	16.75	0.01	0.04	1.96
B1	0.95	0.90	1.00	0.00	0.00	0.00	0.35	14.00	55.39
B10	0.45	0.26	0.57	0.03	0.03	14.84	0.00	0.01	0.79
B10	0.63	0.36	0.98	0.01	0.00	5.16	0.05	0.03	35.42
B11	0.16	0.00	0.27	0.51	3.94	89.40	0.00	0.00	0.22
B11	0.37	0.12	0.55	0.15	0.00	70.60	0.00	0.03	0.08
B12	-0.24	-0.24	-0.24	0.00	0.23	0.23	0.00	0.00	0.00
B12/13	-0.39	-0.50	-0.32	0.01	0.09	0.90	0.00	0.00	0.28
B13	-0.18	-0.18	-0.18	0.09	8.53	8.53	0.00	0.00	0.00
B14	0.51	0.40	0.61	0.00	0.00	0.02	0.01	0.18	1.32
B14	0.50	0.17	0.93	0.09	0.00	57.76	0.04	0.00	21.71
B1C	0.31	0.20	0.51	0.04	0.00	19.89	0.00	0.00	0.00
B2	0.82	0.76	0.88	0.00	0.00	0.00	0.00	0.01	0.80
B2C	0.37	0.22	0.68	0.01	0.00	6.06	0.00	0.00	0.04
B4	0.67	0.59	0.82	0.00	0.00	0.02	0.00	0.01	0.25
B5	0.65	0.41	0.79	0.00	0.00	1.95	0.01	0.06	1.89
B5C	0.39	0.26	0.57	0.00	0.00	4.12	0.00	0.00	0.01
B6A	0.41	0.38	0.50	0.06	4.00	6.89	0.01	0.15	2.89
B6A	0.19	-0.35	0.54	0.10	2.90	18.63	0.02	0.00	5.17
B6B	0.34	0.13	0.54	0.13	0.69	27.95	0.00	0.00	0.30
B6C	0.53	-0.08	0.71	0.01	0.00	11.58	0.03	0.07	5.70
B7	0.89	0.80	0.99	0.00	0.00	0.00	0.20	5.15	47.70
B8A	0.40	0.28	0.49	0.04	1.35	14.38	0.08	6.18	9.60
B8A	0.31	0.09	0.83	0.18	0.85	30.70	0.09	5.19	31.00
B9	0.65	0.53	0.74	0.00	0.00	0.04	0.02	0.33	3.98
B9	0.63	0.37	0.95	0.00	0.00	1.05	0.05	0.08	33.63
R2	0.01	-0.15	0.49	0.29	0.00	59.50	0.10	0.00	59.50
R2	-0.32	-0.36	-0.27	0.00	0.00	0.43	0.00	0.00	0.00
R3	-0.45	-0.45	-0.45	0.00	0.00	0.00	0.01	0.68	0.68
R5	-0.44	-0.65	-0.18	0.06	0.00	18.53	0.00	0.00	0.00
RSF2	0.06	0.02	0.11	0.47	43.63	48.20	0.00	0.00	0.00
RSF2	0.23	0.13	0.43	0.22	0.00	44.24	0.00	0.00	7.71

Table 2. Surveyed tidal marsh site characteristics.
South San Francisco Bay salt ponds used for model development, surveyed between October 1999 and January 2004. hectares = size of area surveyed in hectares. arealdens = channel area (proportion of area surveyed that contained channels); lindens = linear channel density within survey area (m/ha); chann5mha = linear channel density of channels greater than 8 m in width; chann4mha = linear channel density of channels greater than 4 m but less than 8 m in width; chann3mha = linear channel density of channels greater than 2 m but less than 4 m in width; chann2mha = linear channel density of channels greater than 0.6 m but less than 2 m in width; chann1mha = linear channel density of channels less than 0.6 m in with; totpond_pr = proportion of area surveyed that contained ponds/pannes; avgpondsize = average pond/panne size within survey area (ha); pondnumha: average number of ponds per ha. For site locations see Figure 1. For site names, see Table 4.

Site Code	Complex	hectares	arealdens	lindens	chann5mha	chann4mha
ALA	EdenLanding	17.20	0.15	151.07	151.07	0.00
AUDE	Newark	9.52	0.14	347.24	118.50	6.82
AUDW	Newark	10.42	0.07	419.87	5.13	22.51
DUME	Newark	14.02	0.04	227.62	0.00	30.54
DUMW	Newark	85.34	0.05	363.77	4.41	30.83
EPA	Ravenswood	29.40	0.10	623.27	0.00	111.98
HAM	EdenLanding	30.60	0.04	279.05	8.68	19.08
NCH	Alviso	33.09	0.12	284.67	88.14	22.31
NEW	Newark	19.96	0.10	324.81	81.10	0.00
PAB	Ravenswood	20.59	0.04	188.27	23.05	4.20
RAV	Ravenswood	30.95	0.12	590.30	64.13	17.94
WTL	EdenLanding	21.65	0.02	169.86	0.00	1.52

Site Code	chann3mha	chann2mha	chann1mha	totpond_pr	avgpondsize	pondnumha
ALA	0.00	0.00	0.00	0.00	0.00	0.00
AUDE	9.82	30.28	181.82	0.01	0.00	3.99
AUDW	105.47	79.14	207.62	0.00	0.00	0.86
DUME	31.08	63.51	102.50	0.03	0.01	4.28
DUMW	36.44	66.09	225.99	0.02	0.01	1.71
EPA	17.55	19.43	474.30	0.01	0.03	0.24
HAM	25.39	95.54	130.36	0.02	0.01	1.96
NCH	44.82	70.09	59.31	0.13	0.06	2.21
NEW	1.20	90.94	151.57	0.00	0.00	1.10
PAB	19.36	44.31	97.34	0.08	0.05	1.80
RAV	45.89	206.43	255.90	0.01	0.02	0.48
WTL	26.86	61.33	80.15	0.10	0.07	1.39

Table 3. Salt pond landscape characteristics with a 1-km radius.
South San Francisco Bay salt ponds used for model development, surveyed between October 1999 and January 2004. tm1kmp = proportion of area within a 1-km radius containing tidal marsh; ntm1kmp = proportion of area with a 1-km radius containing non-tidal marsh; sp1kmp = proportion of area within a 1-km radius containing salt ponds; mud1kmp = proportion of area within 1-km radius containing tidal flats; bay1kmp = proportion of area within 1-km radius containing bay open water and tidal flats; natup1kmp = proportion of area within 1-km radius containing natural uplands. For pond locations see Figure 1.

Pond ID	Complex	tm1kmp	ntm1kmp	sp1kmp	mud1kmp	bay1kmp	natup1kmp
A1	Alviso	0.10	0.18	0.28	0.17	0.20	0.00
A11	Alviso	0.07	0.00	0.90	0.03	0.03	0.00
A11	Alviso	0.07	0.00	0.90	0.03	0.03	0.00
A13	Alviso	0.11	0.11	0.75	0.02	0.02	0.00
A14	Alviso	0.14	0.00	0.73	0.10	0.13	0.00
A14	Alviso	0.14	0.00	0.73	0.10	0.13	0.00
A15	Alviso	0.17	0.03	0.71	0.07	0.10	0.00
A16	Alviso	0.11	0.21	0.59	0.01	0.01	0.00
A16	Alviso	0.11	0.21	0.59	0.01	0.01	0.00
A17	Alviso	0.21	0.01	0.73	0.02	0.04	0.00
A19	Alviso	0.21	0.02	0.53	0.09	0.11	0.00
A20	Alviso	0.27	0.00	0.68	0.02	0.05	0.00
A21	Alviso	0.25	0.00	0.64	0.07	0.10	0.00
A22	Alviso	0.04	0.11	0.36	0.03	0.03	0.03
A23	Alviso	0.12	0.09	0.50	0.05	0.06	0.01
A5	Alviso	0.09	0.00	0.67	0.05	0.19	0.00
AB1	Alviso	0.04	0.07	0.57	0.28	0.29	0.00
AB2	Alviso	0.03	0.05	0.65	0.21	0.24	0.00
B1	EdenLanding	0.15	0.00	0.78	0.05	0.07	0.00
B10	EdenLanding	0.10	0.04	0.53	0.19	0.30	0.00
B10	EdenLanding	0.10	0.04	0.53	0.19	0.30	0.00
B11	EdenLanding	0.10	0.10	0.59	0.08	0.12	0.00
B11	EdenLanding	0.10	0.10	0.59	0.08	0.12	0.00
B12	EdenLanding	0.02	0.05	0.78	0.00	0.00	0.00
B12/13	EdenLanding	0.03	0.05	0.84	0.00	0.00	0.00
B13	EdenLanding	0.03	0.05	0.90	0.00	0.00	0.00
B14	EdenLanding	0.05	0.02	0.93	0.00	0.00	0.00
B14	EdenLanding	0.05	0.02	0.93	0.00	0.00	0.00
B1C	EdenLanding	0.10	0.16	0.64	0.00	0.04	0.04
B2	EdenLanding	0.16	0.02	0.53	0.26	0.30	0.00
B2C	EdenLanding	0.12	0.18	0.53	0.00	0.04	0.10
B4	EdenLanding	0.11	0.09	0.76	0.00	0.03	0.01
B5	EdenLanding	0.04	0.18	0.65	0.00	0.01	0.01
B5C	EdenLanding	0.07	0.22	0.53	0.00	0.02	0.04
B6A	EdenLanding	0.06	0.07	0.56	0.00	0.02	0.01
B6A	EdenLanding	0.06	0.07	0.56	0.00	0.02	0.01
B6B	EdenLanding	0.05	0.04	0.84	0.00	0.01	0.00
B6C	EdenLanding	0.03	0.20	0.67	0.00	0.02	0.02
B7	EdenLanding	0.07	0.06	0.86	0.00	0.01	0.00
B8A	EdenLanding	0.15	0.01	0.74	0.08	0.10	0.00
B8A	EdenLanding	0.15	0.01	0.74	0.08	0.10	0.00
B9	EdenLanding	0.13	0.01	0.72	0.12	0.14	0.00
B9	EdenLanding	0.13	0.01	0.72	0.12	0.14	0.00
R2	Ravenswood	0.09	0.08	0.58	0.11	0.13	0.00
R2	Ravenswood	0.09	0.08	0.58	0.11	0.13	0.00
R3	Ravenswood	0.05	0.04	0.46	0.01	0.02	0.00
R5	Ravenswood	0.06	0.01	0.46	0.02	0.03	0.00
RSF2	Ravenswood	0.07	0.09	0.29	0.26	0.30	0.00
RSF2	Ravenswood	0.07	0.09	0.29	0.26	0.30	0.00

Table 4. Tidal marsh landscape characteristics.
South San Francisco Bay tidal marshes used for model development, surveyed between October 1999 and April 2001 (waterbirds) or May 2004 (landbirds). tm1kmp = proportion of area within a 1-km radius containing tidal marsh; Ntm1kmp = proportion of area with a 1-km radius containing non-tidal marsh; sp1kmp = proportion of area within a 1-km radius containing salt ponds; mud1kmp = proportion of area within 1-km radius containing tidal flats; bay1kmp = proportion of area within 1-km radius containing bay open water and tidal flats; natup1kmp = proportion of area within 1-km radius containing natural uplands. For site locations, see Figure 1.

Site Code	Site Name	Complex	tm1kmp	ntm1kmp	mud1kmp	bay1kmp	sp1kmp	natup1kmp
ALA	Alameda Creek	Eden Landing	0.20	0.00	0.18	0.31	0.49	0.00
AUDE	Audubon Marsh East	Newark	0.43	0.00	0.02	0.04	0.53	0.00
AUDW	Audubon Marsh West	Newark	0.35	0.00	0.11	0.35	0.29	0.00
DUME	Dumbarton Marsh East	Newark	0.22	0.04	0.01	0.02	0.50	0.02
DUMW	Dumbarton Marsh West	Newark	0.34	0.00	0.14	0.37	0.28	0.00
EPA	East Palo Alto	Ravenswood	0.15	0.03	0.25	0.27	0.00	0.00
HAM	Hayward Area Marsh	Eden Landing	0.18	0.02	0.32	0.58	0.05	0.01
NCH	New Chicago Marsh	Alviso	0.06	0.28	0.00	0.00	0.47	0.00
NEW	Newark Slough	Newark	0.22	0.05	0.01	0.01	0.39	0.10
PAB	Palo Alto Baylands	Ravenswood	0.19	0.06	0.42	0.48	0.00	0.00
RAV	Ravenswood Slough	Ravenswood	0.08	0.04	0.22	0.24	0.53	0.00
WTL	Whaletail Marsh	Eden Landing	0.21	0.00	0.24	0.32	0.48	0.00

Table 5. Candidate focal species.
Bird species of numerical or ecological significance in either salt pond or tidal marsh habitats in the South San Francisco Bay. For the winter season, numerical significance meant that species were required to be one of the 40 most abundant species in either tidal marsh or salt pond habitats (Stralberg et al. 2003), or of special conservation status.

Shorebirds	**Other Waterbirds**
Western Sandpiper	Eared Grebe
Least Sandpiper	Western/Clarks Grebe
Dunlin	Pied-billed Grebe
Semipalmated Plover	Double-crested Cormorant
Killdeer	Brown Pelican
Sanderling	American White Pelican
Marbled Godwit	Forster's Tern
Willet	California Gull
Long-billed Curlew	Bonaparte's Gull
Black-bellied Plover	Mew Gull
American Avocet	Herring Gull
Red Knot	Western Gull
Dowitchers	Ring-billed Gull
Greater Yellowlegs	
Black-necked Stilt	**Herons and Egrets**
Snowy Plover	Snowy Egret
	Great Egret
Waterfowl and Coot	Great Blue Heron
Northern Pintail	
Green-winged Teal	**Rails**
Northern Shoveler	Clapper Rail
American Wigeon	
Mallard	**Raptors and Corvids**
Cinnamon Teal	Northern Harrier
Gadwall	Red-tailed Hawk
Canvasback	White-tailed Kite
Common Goldeneye	Short-eared Owl
Greater and Lesser Scaup	Common Raven
Ruddy Duck	
Bufflehead	**Landbirds**
Red-breasted Merganser	Common Yellowthroat
American Coot	Savannah Sparrow
	Song Sparrow
	Marsh Wren

Table 6. Final focal species and groups.
Focal species used to develop South San Francisco Bay habitat models, corresponding taxonomic groups, and represented species, habitats, and seasons. Species represented are based on TWINSPAN analysis of shorebirds, waterfowl, and other waterbird groups. Within each group, focal species were selected based on highest explanatory power (R^2) for either salt ponds (*), tidal marshes (**), or both habitats (***), except for italicized species of conservation concern, which were added *post hoc*. SP = salt pond; TM = tidal marsh; TF = tidal flat.

Group	4-letter Code	Focal Species	Species Represented	Habitats	Fall	Winter	Spring	Breeding
Large Shorebirds	AMAV	American Avocet*	Long-billed Curlew, Dowitchers	SP, TM, TF		X		
Large Shorebirds	BNST	Black-necked Stilt**		SP, TM, TF		X		
Small Shorebirds	DUNL	*Dunlin*		SP, TM, TF		X	X	
Small Shorebirds	WESA	*Western Sandpiper*		SP, TM, TF	X	X	X	
Large Shorebirds	GRYE	Greater Yellowlegs**	Killdeer	TM		X	.	
Small Shorebirds	LESA	Least Sandpiper*		SP, TM, TF	X	X	X	
Large Shorebirds	WILL	Willet***		SP, TM, TF	X	X		
Small Shorebirds	SEPL	Semipalmated Plover**	Black-bellied Plover, Sanderling, Marbled Godwit, Red Knot	SP, TM, TF		X		
Small Shorebirds	SNPL	*Snowy Plover*		SP				X
Phalaropes	WIPH	*Wilson's Phalarope*		SP	X			
Phalaropes	RNPH	*Red-necked Phalarope*		SP	X			
Dabbling Ducks	GADW	Gadwall**	Green-winged Teal, Cinnamon Teal, Canvasback	SP, TM		X		
Dabbling Ducks	MALL	Mallard**	American Wigeon, American Coot	SP, TM		X		
Dabbling Ducks	NOPI	*Northern Pintail*		SP, TM		X		
Dabbling Ducks	NSHO	Northern Shoveler***	Bufflehead	SP, TM		X		
Diving Ducks	RUDU	Ruddy Duck***	Common Goldeneye	SP, TM		X		
Diving Ducks	SCAU	*Scaup*		SP, TM		X		
Fish-eaters	AWPE	*American White Pelican*		SP		X		
Fish-eaters	FOTE	Forster's Tern*	Western Gull, Red-breasted Merganser	SP		X		
Eared Grebe	EAGR	Eared Grebe*	Bonaparte's Gull, California Gull, Herring Gull, Mew Gull	SP		X		
Rails	CLRA	*Clapper Rail*		TM				X
Landbirds	COYE	*Common Yellowthroat*		TM				X
Landbirds	MAWR	*Marsh Wren*		TM				X
Landbirds	SOSP	*Song Sparrow*		TM				X

Table 7. Candidate salt pond variables for each focal species and group by season.
DABBLER = dabbling ducks; DIVER = diving ducks; LAND = landbirds; LGSHORE = large shorebirds; PHAL = phalaropes; SMSHORE = small shorebirds; WADER = herons and egrets. Species 4-letter codes are defined in Table 6. W = winter (Nov-Feb), S = spring (Mar-May), F = fall (Aug-Oct). year = survey year (1999/2000, 2000/2001, 2002/2003, 2003/2004, August–May), hectares = pond size (ha); mean_depth = mean pond depth (m) during survey; depth2 = (mean_depth)^2; shallow_pr = pond shallow proportion (<15 cm) during survey; deep_prop = pond deep proportion (>1 m) during survey; salinity = pond salinity during survey (ppt); salinity2 = salinity^2; logbay = ln(bay1kmp+0.01); logntm = ln(ntm1kmp+0.01); logtm = ln(tm1kmp+0.01); logsp = ln(sp1kmp+0.01); logmud = ln(mud1kmp+0.01). Other variables defined in Table 1.

Group	Species	Season	year	hectares	mean_depth	depth2	shallow_pr	deep_prop	salinity	salinity2	logbay	logntm	logtm	logsp	logmud
DABBLER	Group	W	X	X	X	X	X		X	X		X	X		
DABBLER	GADW	W	X	X	X	X	X		X	X		X	X		
DABBLER	MALL	W	X	X	X	X	X		X	X		X	X		
DABBLER	NOPI	W	X	X	X	X	X		X	X	X		X		
DABBLER	NSHO	W	X	X	X	X	X		X	X		X	X		
DIVER	Group	W	X	X	X	X		X	X	X	X			X	
DIVER	RUDU	W	X	X	X			X	X	X	X			X	
DIVER	SCAU	W	X	X	X			X	X	X	X			X	
EAREDGR	EAGR	W	X	X	X			X	X	X	X			X	
FISHEAT	Group	W	X	X	X	X		X	X	X	X			X	
FISHEAT	AWPE	W	X	X	X	X	X		X	X	X			X	
FISHEAT	FOTE	W	X	X	X	X		X	X	X	X			X	
LGSHORE	Group	W	X	X	X		X		X	X				X	X
LGSHORE	AMAV	W	X	X	X		X		X	X				X	X
LGSHORE	BNST	W	X	X	X		X		X	X				X	X
LGSHORE	WILL	F	X	X	X		X		X	X			X		X
LGSHORE	WILL	W	X	X	X		X		X	X			X		X
PHAL	Group	F	X	X	X	X			X	X	X			X	
PHAL	RNPH	F	X	X	X		X		X	X	X			X	
PHAL	WIPH	F	X	X	X		X		X	X	X			X	
SMSHORE	Group	F		X	X		X		X	X				X	X
SMSHORE	Group	S		X	X		X		X	X				X	X
SMSHORE	Group	W		X	X		X		X	X				X	X
SMSHORE	DUNL	S	X	X	X		X		X	X				X	X
SMSHORE	DUNL	W	X	X	X		X		X	X				X	X
SMSHORE	LESA	F	X	X	X		X		X	X				X	X
SMSHORE	LESA	S	X	X	X		X		X	X				X	X
SMSHORE	LESA	W	X	X	X		X		X	X				X	X
SMSHORE	SEPL	W	X	X	X		X		X	X				X	X
SMSHORE	WESA	F	X	X	X		X		X	X				X	X
SMSHORE	WESA	S	X	X	X		X		X	X				X	X
SMSHORE	WESA	W	X	X	X		X		X	X				X	X
WADER	Group	W	X	X	X	X			X	X			X		

Table 8. Candidate tidal marsh variables for each species/season.
DABBLER = dabbling ducks; DIVER = diving ducks; LAND = landbirds; LGSHORE = large shorebirds; PHAL = phalaropes; SMSHORE = small shorebirds. Species 4-letter codes are defined in Table 6. W = winter (Nov-Feb), S = spring (Mar-May), F = fall (Aug-Oct). AS = area survey; PC = point count. HABCAT = microhabitat category (channels, ponds/pannes, vegetation); chann12 = chann1mha + chann2mha; logbay = ln(bay1kmp+0.01); logntm = ln(Ntm1kmp+0.01); logtm = ln(tm1kmp+0.01); logsp = ln(sp1kmp+0.01); logmud = ln(mud1kmp+0.01). Other variables defined in Tables 2 and 4.

Group	Species	Season	Survey Method	habcat	chann12	arealdens	lindens	totpond_pr	logbay	logntm	logtm	logsp	logmud	lognatup
DABBLER	Group	W	AS	X				X						
DABBLER	GADW	W	AS	X				X						
DABBLER	MALL	W	AS	X				X						
DABBLER	NOPI	W	AS	X		X								
DABBLER	NSHO	W	AS	X				X						
DIVER	Group	W	AS	X		X								
DIVER	RUDU	W	AS	X		X								
DIVER	SCAU	W	AS	X		X								
LAND	Group	S	PC	X	X						X			X
LAND	CLRA	S	PC	X	X				X		X			
LAND	COYE	S	PC	X	X				X		X			
LAND	MAWR	S	PC	X	X				X		X			
LAND	SOSP	S	PC	X	X						X			X
LGSHORE	Group	W	AS	X				X				X	X	
LGSHORE	AMAV	W	AS	X				X				X	X	
LGSHORE	BNST	W	AS	X				X				X	X	
LGSHORE	GRYE	W	AS	X		X					X		X	
LGSHORE	WILL	F	AS	X		X					X		X	
LGSHORE	WILL	W	AS	X		X					X		X	
SMSHORE	Group	F	AS	X				X			X	X		
SMSHORE	Group	S	AS	X				X			X	X		
SMSHORE	Group	W	AS	X				X			X	X		
SMSHORE	DUNL	S	AS	X				X			X	X		
SMSHORE	DUNL	W	AS	X				X			X	X		
SMSHORE	LESA	F	AS	X				X			X	X		
SMSHORE	LESA	S	AS	X				X			X	X		
SMSHORE	LESA	W	AS	X				X			X	X		
SMSHORE	SEPL	W	AS	X				X			X	X		
SMSHORE	WESA	F	AS	X				X			X	X		
SMSHORE	WESA	S	AS	X				X			X	X		
SMSHORE	WESA	W	AS	X				X			X	X		

Table 9. Summary of restoration alternatives.
Restoration alternatives provided by PWA / HT Harvey consultant team for South San Francisco Bay salt pond restoration project. SP = number of managed ponds; TM = number of existing ponds restored to tidal action. Ha = hectares covered by scenario. Mean salinity measured in ppt. Mean depth measured in m. Shallow and deep pond areas are <15 cm and >1 m deep on, respectively. The number of "summer dry" ponds refers to the number of ponds to be managed as dry during the summer season, primarily for nesting Snowy Plover.

Alternative	Year	SP	TM	Ha	Mean Salinity	Mean Depth	Mean Shallow %	Mean Deep %	Summer Dry
A	0	64	3	5323	43.4	0.41	14.5	12.0	24
Amax	50	14	24	3023	14.7	0.16	5.5	6.9	4
Amin	50	14	24	3023	14.7	0.16	5.5	6.9	4
B	0	35	32	5323	32.3	0.18	14.1	8.7	6
Bmax	50	35	32	5323	32.3	0.18	14.1	8.7	6
Bmin	50	35	32	5323	32.3	0.18	14.1	8.7	6
C	0	19	48	5323	18.5	0.05	10.3	2.0	0
Cmax	50	19	48	5323	18.5	0.05	10.3	2.0	0
Cmin	50	19	48	5323	18.5	0.05	10.3	2.0	0

Alternative	Year	SP	TM	Ha	Subtidal Area (ha)	Mudflat Area (ha)	Marsh Area (ha)	Vegetated Area (ha)	Channel Area (ha)	Pond/ Panne Area (ha)	Mean Pond Prop	Mean Channel Prop.	Mean Channel Density	Mean Small Channel Density
A	0	64	3	5323	4	189	0	0	0	0	0.000	0.000	0.0	0.0
Amax	50	14	24	3023	36	0	1794	1372	225	192	0.070	0.069	118.9	69.5
Amin	50	14	24	3023	36	0	1794	1557	225	12	0.007	0.069	118.9	69.5
B	0	35	32	5323	217	2676	0	0	0	0	0.000	0.000	0.0	0.0
Bmax	50	35	32	5323	142	0	2799	2149	356	291	0.050	0.058	90.3	52.3
Bmin	50	35	32	5323	142	0	2799	2420	356	24	0.004	0.058	90.3	52.3
C	0	19	48	5323	363	4312	0	0	0	0	0.000	0.000	0.0	0.0
Cmax	50	19	48	5323	159	0	4566	3507	587	470	0.074	0.090	134.9	78.1
Cmin	50	19	48	5323	159	0	4566	3949	587	29	0.005	0.090	134.9	78.1

Table 10. Mean salt pond bird densities by depth and salinity category.
Mean bird densities calculated from the same dataset used for model development (1999-2004 South San Francisco Bay avian survey data). Species 4-letter codes are defined in Table 6. W = winter (Nov-Feb), S = spring (Mar-May), F = fall (Aug-Oct). Overall = across all ponds used for model development (see Table 1), high tide surveys only; Cargill = PRBO-surveyed Cargill ponds only (see Figure 1), all tides. Salinity categories are defined as follows: "V.Low" = 20-40 ppt; "Low" = 40-60 ppt; "Med" = 60-120 ppt; "High" = 120-180 ppt.; "V.High" = >180 ppt. Depth categories are defined as follows: "Shallow" = At least 10% <15 cm deep AND not more than 10% >1 m deep) OR (mean depth < 0.5 m); "Med" = At least 50% >1 m deep AND no more than 10% <15 cm deep; "Med" = NOT "Shallow" or "Deep."

Species	Season	Overall	Cargill	V.Low / Deep	V. Low / Med	V. Low / Shallow	Low / Deep	Low / Med	Low / Shallow	Med / Deep	Med / Med	Med / Shallow	High / Med	High / Shallow	V.High / Med	V.High / Shallow
AMAV	W	0.23	0.36	0.00	0.16	1.43	0.00	0.00	0.08	0.03	0.02	0.41	0.18	0.53	0.00	0.37
AWPE	W	0.04	0.00	0.00	0.16	0.03	0.00	0.26	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BNST	W	0.23	0.53	0.00	0.00	0.16	0.00	0.02	0.12	0.01	0.02	1.64	0.07	0.93	0.00	0.43
DUNL	W	0.47	0.42	0.00	0.01	2.76	0.00	0.02	0.54	0.01	0.06	1.87	0.02	2.79	0.09	0.33
DUNL	S	0.49	0.20		0.01	4.12	0.00		1.29	0.00	0.11	2.80	0.01	2.79	0.00	0.28
EAGR	W	0.55	0.48	0.02	0.20	0.05	0.32	0.49	0.31	1.75	1.43	0.28	1.39	0.13	0.48	0.00
FOTE	W	0.02	0.00	0.00	0.05	0.05	0.00	0.12	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
GADW	W	0.01	0.00	0.00	0.00	0.00	0.00	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
GRYE	W	0.01	0.01	0.00	0.00	0.01	0.00	0.00	0.06	0.00	0.01	0.00	0.00	0.00	0.00	0.00
LESA	W	0.27	0.31	0.00	0.12	0.53	0.05	0.10	0.63	0.06	0.07	0.50	0.04	1.19	0.35	0.38
LESA	S	0.11	0.10		0.01	0.00	0.00		0.09	0.01	0.08	0.88	0.01	0.56	0.00	0.20
LESA	F	0.39	0.33		0.01	0.44	0.00	0.03	1.10	0.17	0.16	1.03	2.63	1.64		0.46
MALL	W	0.01	0.00	0.00	0.01	0.05	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00
NOPI	W	0.01	0.00	0.00	0.01	0.06	0.00	0.00	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.00
NSHO	W	0.16	1.47	0.00	0.22	0.20	0.12	0.09	0.54	0.11	0.20	0.11	0.00	0.13	0.00	0.00
RNPH	F	0.11	0.02		0.00	0.00	0.09	0.01	0.71	0.02	0.10	0.13	0.02	0.00		0.00
RUDU	W	0.08	0.03	1.01	0.43	0.11	0.00	0.07	0.01	0.03	0.02	0.01	0.00	0.00	0.00	0.00
SCAU	W	0.07	0.03	0.37	0.14	0.01	0.02	0.41	0.09	0.00	0.01	0.10	0.00	0.19	0.00	0.00
SEPL	W	0.02	0.00	0.00	0.00	0.16	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00
WESA	W	0.43	0.32	0.00	0.01	3.39	0.00	0.00	0.46	0.01	0.05	1.70	0.23	1.18	0.00	0.21
WESA	S	0.64	0.36		0.00	7.90	0.00		1.53	0.00	0.10	2.80	0.00	4.57	0.00	1.45
WESA	F	0.89	0.30		0.15	0.56	0.07	0.02	7.73	0.03	0.04	1.88	64.77	6.82		1.16
WILL	W	0.14	0.11	0.00	0.03	0.51	0.00	0.00	0.11	0.01	0.02	0.53	0.01	0.40	0.00	0.14
WILL	F	0.24	0.10		0.00	0.08	0.00	0.02	1.16	0.00	0.11	0.27	9.27	0.98		0.10
WIPH	F	0.04	0.00		0.00	0.00	0.00	0.00	0.16	0.12	0.00	0.00	0.00	0.00		0.00

Table 11. Tidal marsh bird densities by microhabitat.
Mean bird densities were calculated from the same dataset used for model development (1999-2004 South San Francisco Bay avian survey data). Species 4-letter codes are defined in Table 6. W = winter (Nov-Feb), S = spring (Mar-May), F = fall (Aug-Oct). Overall = across all tidal marshes used for model development (see Table 2), high tide surveys only. TMC = tidal marsh channels; TMP = tidal marsh ponds/pannes; TMV = tidal marsh vegetation.

Species	Season	Overall	TMC	TMP	TMV
AMAV	W	0.0071	0.030	0.039	0
BNST	W	0.015	0.0083	0.075	0
COYE	S	0.11			
DUNL	W	0.050	0.12	0.081	0
DUNL	S	0.00094	0	0	0
GADW	W	0.014	0.065	0.069	0
GRYE	W	0.028	0.18	0	0
GRYE	S	0.015	0.093	0.047	0
GRYE	F	0.028	0.080	0.089	0
LESA	W	0.19	0.49	0.25	0.0063
LESA	S	0.064	0.15	0.069	0.034
LESA	F	0.33	0.59	1.03	0
MALL	W	0.020	0.095	0.062	0.0038
MAWR	S	0.87			
NOPI	W	0.0066	0.014	0.029	0
NSHO	W	0.078	0.22	0.23	0
RUDU	W	0.0036	0.028	0.0089	0
SCAU	W	0	0.028	0.0089	0
SEPL	W	0.0021	0	0.014	0
SOSP	S	4.05			
WESA	W	0.051	0.0047	0.086	0
WESA	S	0.11	0.0085	0.072	0.052
WESA	F	0.082	0.12	0.32	0
WILL	W	0.015	0.076	0.010	0.0017
WILL	F	0.0068	0.030	0.013	0

Table 12. Predicted future tidal flat habitat in South San Francisco Bay.
Predicted hectares of tidal flat habitat according to PWA geomorphic models (PWA 2006). Tract ID = Pacific Flyway tidal tracts delineated in Figure 1.

Region	Tract ID	Baseline	AltA Yr50	AltB Yr50	AltC Yr50
CoyoteSlough	S3	200	0	0	0
FarSouthBay	S3	2300	2500	2200	1500
NorthofDumbarton	S1	2500	1000	1000	1000
NorthofDumbarton	S2	2500	1000	1000	1000

Table 13. Tidal flat species/taxon densities in South San Francisco Bay by tract and season. Means and standard deviations for each species/taxon were based on complete tidal flat surveys conducted from 1988 to 1993 as part of PRBO's Pacific Flyway Project (Stenzel et al. 2002). Tidal tracts correspond with areas delineated in Figure 1. Species 4-letter codes are defined in Table 6; other codes are defined as follows: PHAL = Red-necked Phalarope, Red Phalarope, Wilson's Phalarope; WLDU = Western Sandpiper, Least Sandpiper, Dunlin; YELL = Greater Yellowlegs, Lesser Yellowlegs.

Season	Taxa	Tract	Name	Density Mean	Density SD
Fall	AMAV	C1	W Bay north of San Mateo Bridge	0.314	0.198
Fall	AMAV	C2	Alameda	0.099	0.091
Fall	AMAV	C3	Hayward Shoreline	0.928	0.818
Fall	AMAV	S1	SE Bay between bridges	0.006	0.003
Fall	AMAV	S2	SW Bay between bridges	0.010	0.008
Fall	AMAV	S3	South of Dumbarton Bridge	1.802	1.212
Spring	AMAV	C1	W Bay north of San Mateo Bridge	0.048	0.099
Spring	AMAV	C2	Alameda	0.155	0.129
Spring	AMAV	C3	Hayward Shoreline	0.187	0.191
Spring	AMAV	S1	SE Bay between bridges	0.047	0.095
Spring	AMAV	S2	SW Bay between bridges	0.012	0.014
Spring	AMAV	S3	South of Dumbarton Bridge	0.246	0.204
Fall	BNST	C1	W Bay north of San Mateo Bridge	0.027	0.012
Fall	BNST	C2	Alameda	0.121	0.047
Fall	BNST	C3	Hayward Shoreline	0.079	0.023
Fall	BNST	S1	SE Bay between bridges	0.002	0.003
Fall	BNST	S2	SW Bay between bridges	0.003	0.001
Fall	BNST	S3	South of Dumbarton Bridge	0.031	0.010
Spring	BNST	C1	W Bay north of San Mateo Bridge	0.010	0.007
Spring	BNST	C2	Alameda	0.005	0.007
Spring	BNST	C3	Hayward Shoreline	0.045	0.040
Spring	BNST	S1	SE Bay between bridges	0.001	0.002
Spring	BNST	S2	SW Bay between bridges	0.011	0.014
Spring	BNST	S3	South of Dumbarton Bridge	0.008	0.008
Fall	PHAL	C1	W Bay north of San Mateo Bridge	0.000	0.000
Fall	PHAL	C2	Alameda	0.000	0.000
Fall	PHAL	C3	Hayward Shoreline	0.009	0.016
Fall	PHAL	S1	SE Bay between bridges	0.085	0.148
Fall	PHAL	S2	SW Bay between bridges	0.000	0.000
Fall	PHAL	S3	South of Dumbarton Bridge	0.009	0.009
Spring	PHAL	C1	W Bay north of San Mateo Bridge	0.000	0.000
Spring	PHAL	C2	Alameda	0.000	0.000
Spring	PHAL	C3	Hayward Shoreline	0.001	0.001
Spring	PHAL	S1	SE Bay between bridges	0.000	0.000
Spring	PHAL	S2	SW Bay between bridges	0.000	0.000
Spring	PHAL	S3	South of Dumbarton Bridge	0.000	0.000
Fall	SEPL	C1	W Bay north of San Mateo Bridge	0.606	0.542
Fall	SEPL	C2	Alameda	0.272	0.232
Fall	SEPL	C3	Hayward Shoreline	0.104	0.015
Fall	SEPL	S1	SE Bay between bridges	0.192	0.153
Fall	SEPL	S2	SW Bay between bridges	0.211	0.097
Fall	SEPL	S3	South of Dumbarton Bridge	0.257	0.053
Spring	SEPL	C1	W Bay north of San Mateo Bridge	0.139	0.112
Spring	SEPL	C2	Alameda	0.287	0.260
Spring	SEPL	C3	Hayward Shoreline	0.166	0.141
Spring	SEPL	S1	SE Bay between bridges	0.109	0.079
Spring	SEPL	S2	SW Bay between bridges	0.168	0.131
Spring	SEPL	S3	South of Dumbarton Bridge	0.055	0.056

Table 13, cont.

Season	Taxa	Tract	Name	Density Mean	Density SD
Fall	WILL	C1	W Bay north of San Mateo Bridge	1.179	0.383
Fall	WILL	C2	Alameda	4.391	1.757
Fall	WILL	C3	Hayward Shoreline	1.254	0.454
Fall	WILL	S1	SE Bay between bridges	6.398	1.181
Fall	WILL	S2	SW Bay between bridges	0.879	0.565
Fall	WILL	S3	South of Dumbarton Bridge	2.182	0.534
Spring	WILL	C1	W Bay north of San Mateo Bridge	0.245	0.210
Spring	WILL	C2	Alameda	0.594	0.501
Spring	WILL	C3	Hayward Shoreline	0.616	0.943
Spring	WILL	S1	SE Bay between bridges	0.701	0.583
Spring	WILL	S2	SW Bay between bridges	0.267	0.310
Spring	WILL	S3	South of Dumbarton Bridge	0.129	0.101
Fall	WLDU	C1	W Bay north of San Mateo Bridge	15.293	5.469
Fall	WLDU	C2	Alameda	19.027	13.223
Fall	WLDU	C3	Hayward Shoreline	33.143	7.198
Fall	WLDU	S1	SE Bay between bridges	34.311	7.848
Fall	WLDU	S2	SW Bay between bridges	9.637	5.339
Fall	WLDU	S3	South of Dumbarton Bridge	32.367	8.375
Spring	WLDU	C1	W Bay north of San Mateo Bridge	46.839	22.717
Spring	WLDU	C2	Alameda	47.370	28.040
Spring	WLDU	C3	Hayward Shoreline	48.686	35.719
Spring	WLDU	S1	SE Bay between bridges	130.795	61.032
Spring	WLDU	S2	SW Bay between bridges	38.679	18.250
Spring	WLDU	S3	South of Dumbarton Bridge	82.934	20.085
Fall	YELL	C1	W Bay north of San Mateo Bridge	0.013	0.007
Fall	YELL	C2	Alameda	0.024	0.005
Fall	YELL	C3	Hayward Shoreline	0.052	0.021
Fall	YELL	S1	SE Bay between bridges	0.037	0.036
Fall	YELL	S2	SW Bay between bridges	0.002	0.002
Fall	YELL	S3	South of Dumbarton Bridge	0.025	0.035
Spring	YELL	C1	W Bay north of San Mateo Bridge	0.005	0.005
Spring	YELL	C2	Alameda	0.046	0.042
Spring	YELL	C3	Hayward Shoreline	0.013	0.006
Spring	YELL	S1	SE Bay between bridges	0.007	0.013
Spring	YELL	S2	SW Bay between bridges	0.001	0.001
Spring	YELL	S3	South of Dumbarton Bridge	0.001	0.001

Table 14. Salt pond focal species model diagnostics.
Full model and variable-specific partial R^2 values for top focal species models (based on the lowest AICc value) from each species / season combination. Variables are defined in Table 7. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

Species / Season	Sample size	Model R^2	Model Adj R^2	Partial R^2 values				
				year	hectares	mean_depth	depth2	shallow_pr
AMAVW	218	0.229	0.200	0.064	0.065	0.034		0.021
AWPEW	218	0.067	0.058					
BNSTW	218	0.218	0.204			0.115		0.020
DUNLS	88	0.558	0.531		0.009	0.167		0.359
DUNLW	218	0.316	0.290	0.167	0.033	0.051		0.040
EAGRW	218	0.430	0.416			0.288		
FOTEW	218	0.007	0.000		0.002	0.006		
GADWW	208	0.000	0.000					
GRYEW	208	0.087	0.069		0.028	0.041		
LESAF	67	0.135	0.094		0.099	0.105		
LESAS	88	0.281	0.227		0.007			0.078
LESAW	218	0.153	0.125	0.044	0.006	0.077		
MALLW	218	0.136	0.116		0.022	0.054		
NOPIW	218	0.059	0.046		0.014	0.025		
NSHOW	218	0.095	0.078			0.019		
RNPHF	67	0.224	0.212		0.224			
RUDUW	218	0.309	0.283	0.016		0.080	0.075	
SCAUW	218	0.152	0.132	0.072				
SEPLW	218	0.294	0.274	0.170	0.037			0.143
WESAF	67	0.391	0.372			0.079		0.156
WESAS	88	0.628	0.606		0.021	0.118		0.490
WESAW	218	0.334	0.312	0.162	0.029	0.045		0.068
WILLF	67	0.272	0.238		0.100	0.260		
WILLW	218	0.251	0.219	0.064	0.014	0.077		
WIPHF	67	0.109	0.082			0.054		0.108

Table 14, cont.

Species / Season	deep_prop	salinity	salinity2	logntm	logtm	logsp	logmud	logbay
				Partial R^2 values				
AMAVW						0.019	0.072	
AWPEW		0.050	0.032					
BNSTW		0.129	0.102					
DUNLS		0.178	0.230					
DUNLW		0.015	0.023					
EAGRW		0.099	0.079			0.018		0.052
FOTEW		0.021	0.011					
GADWW								
GRYE					0.024		0.022	
LESAF					0.055			
LESAS		0.128	0.125		0.126		0.005	
LESAW					0.006		0.030	
MALLW		0.050		0.028	0.050			
NOPIW		0.034						
NSHOW		0.069		0.022	0.028			
RNPHF								
RUDUW		0.116	0.071					0.055
SCAUW		0.051				0.062		
SEPLW		0.004						
WESAF								
WESAS		0.145	0.173					
WESAW		0.020						
WILLF					0.143			
WILLW		0.011	0.018		0.013		0.028	
WIPHF								

Table 15. Salt pond group model diagnostics.
Full model and variable-specific partial R^2 values for top group models (based on the lowest AICc value) from each species / season combination. Variables are defined in Table 7. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

Group / Season	Model R^2	Model Adj R^2	Partial R^2 values year	hectares	mean_depth	depth2	shallow_pr	deep_prop
DABBLERW	0.09	0.06			0.02			
DIVERW	0.27	0.21	0.15	0.03	0.04	0.08		
FISHEATW	0.08	0.06						
LGSHOREW	0.18	0.16		0.08	0.11			
SMSHOREF	0.55	0.50			0.50			
SMSHORES	0.81	0.76		0.50	0.33		0.51	
SMSHOREW	0.22	0.19		0.13	0.13			
WADERW	0.17	0.15						

Group / Season	Partial R^2 values salinity	salinity2	logbay	logntm	logtm	logsp	logmud	lognatup
DABBLERW	0.42				0.02			
DIVERW	0.13		0.12					
FISHEATW	0.06	0.03						
LGSHOREW	0.39							
SMSHOREF	0.38	0.31						
SMSHORES	0.29	0.33					0.15	
SMSHOREW	0.29							
WADERW					0.04			0.16

Table 16. Tidal marsh focal species model diagnostics.
Full model and variable-specific partial R^2 values for top focal species models (based on the lowest AICc value) from each species / season combination. Variables are defined in Table 8. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

Species /	Sample	Model	Model	Partial R^2 values									
Season	size	R^2	Adj R^2	habitat	chann12	arealdens	lindens	totpond_pr	logntm	logtm	logsp	logmud	lognatup
AMAVW	350	0.018	0.009	0.015							0.003		
BNSTW	350	0.079	0.069	0.031				0.041			0.005		
DUNLW	350	0.058	0.044	0.018				0.006			0.037	0.023	
GADWW	350	0.071	0.057	0.014				0.026	0.018	0.018			
GRYEW	350	0.147	0.137	0.121		0.029				0.009			
LESAF	101	0.139	0.112	0.107				0.035					
LESAS	143	0.022	0.008	0.022									
LESAW	350	0.078	0.067	0.056						0.012		0.011	
MALLW	350	0.019	0.014	0.019									
NOPIW	350	0.008	0.003	0.008									
NSHOW	350	0.141	0.128	0.039				0.043	0.035	0.017			
RUDUW	350	0.016	0.011	0.016									
SCAUW	350	0.016	0.011	0.016									
SEPLW	350	0.044	0.030	0.011				0.021			0.017	0.013	
WESAF	101	0.039	0.019	0.039									
WESAS	143	0.004	-0.011	0.004									
WESAW	350	0.065	0.051	0.015				0.031			0.025	0.021	
WILLF	101	0.052	0.022	0.017		0.035							
WILLW	350	0.037	0.031	0.037									
CLRAS	41	0.082	0.059										
COYES	41	0.000	0.000										
MAWRS	41	0.503	0.477		0.226								
SOSPS	41	0.133	0.111							0.133			

Table 17. Tidal marsh group model diagnostics.
Full model and variable-specific partial R^2 values for top group models (based on the lowest AICc value) from each species / season combination. Variables are defined in Table 8. Models are based on 1999-2001 South San Francisco Bay avian survey data.

Group / Season	Sample size	Model R^2	Model Adj R^2	Partial R^2 values: habitat	chann12	arealdens	lindens	totpond_pr	logbay	logntm	logtm	logsp	logmud	lognatup
DABBLERW	350	0.18	0.16	0.08	0.01		0.02	0.01		0.00	0.00			
DIVERW	350	0.07	0.05	0.05	0.00		0.00					0.01		
FISHEATW	350	0.10	0.09	0.07	0.00		0.00					0.01		
MEDSHOREF	101	0.24	0.19	0.11	0.03		0.02	0.04					0.00	
MEDSHORES	143	0.20	0.17	0.13	0.00		0.00	0.04						
MEDSHOREW	350	0.11	0.10	0.09	0.01		0.01					0.02	0.00	
SMSHOREF	101	0.16	0.12	0.11	0.00		0.00	0.00						
SMSHORES	143	0.03	0.00	0.02	0.01		0.01							
SMSHOREW	350	0.11	0.10	0.08	0.01		0.01					0.02	0.03	
WADERW	350	0.14	0.13	0.10	0.00		0.00							0.04

Table 18. Group scenario evaluation summary.
Best-performing scenarios / years, in terms of predicted abundance, for each group and season, are indicated with ones. Differences between alternatives are not necessarily statistically significant for all groups. Sums indicate the total number of groups for which a scenario / year performed best. A0 = alternative A, year 0; B0 = alternative B, year 0; C0 = alternative C, year 0; Amax50 = alternative A, year 50, maximum tidal marsh pond/panne assumptions; Bmax50 = alternative B, year 50, maximum tidal marsh pond/panne assumptions; Cmas50 = alternative C, year 50, maximum tidal marsh pond/panne assumptions. DABBLER = dabbling ducks; DIVER = diving ducks; LAND = landbirds; LGSHORE = large shorebirds; PHAL = phalaropes; SMSHORE = small shorebirds. W = winter (Nov-Feb), S = spring (Mar-May), F = fall (Aug-Oct). Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

Season	Group	A0	B0	C0	Amax50	Bmax50	Cmax50
W	DABBLER	1					1
W	DIVER			1		1	
W	FISHEAT	1				1	
W	WADER			1		1	
W	MEDSHORE			1		1	
W	SMSHORE			1		1	
S	LAND						1
S	SMSHORE		1			1	
F	PHAL	1				1	
F	SMSHORE			1		1	
Total # of groups / seasons		3	1	5	0	8	2

Table 19. Focal species scenario evaluation summary.
Best-performing scenarios / years, in terms of predicted abundance, for each focal species and season, are indicated with ones. Differences between alternatives are not necessarily statistically significant for all species. Sums indicate the total number of groups for which a scenario / year performed best. A0 = alternative A, year 0; B0 = alternative B, year 0; C0 = alternative C, year 0; Amax50 = alternative A, year 50, maximum tidal marsh pond/panne assumptions; Bmax50 = alternative B, year 50, maximum tidal marsh pond/panne assumptions; Cmas50 = alternative C, year 50, maximum tidal marsh pond/panne assumptions. DABBLER = dabbling ducks; DIVER = diving ducks; LAND = landbirds; LGSHORE = large shorebirds; PHAL = phalaropes; SMSHORE = small shorebirds. W = winter (Nov-Feb), S = spring (Mar-May), F = fall (Aug-Oct). Species 4-letter codes are defined in Table 6. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data.

Season	Group	Species	A0	B0	C0	Amax50	Bmax50	Cmax50
W	DABBLER	GADW	1					1
W	DABBLER	MALL			1		1	
W	DABBLER	NOPI			1		1	
W	DABBLER	NSHO		1				1
W	DIVER	RUDU	1				1	
W	DIVER	SCAU	1				1	
W	EAREDGR	EAGR	1				1	
W	FISHEAT	AWPE	1				1	
W	FISHEAT	FOTE			1		1	
W	MEDSHORE	AMAV			1		1	
W	MEDSHORE	BNST		1			1	
W	MEDSHORE	GRYE		1				1
W	MEDSHORE	WILL			1		1	
W	SMSHORE	DUNL			1		1	
W	SMSHORE	LESA			1		1	
W	SMSHORE	SEPL			1		1	
W	SMSHORE	WESA			1		1	
S	LAND	COYE						1
S	LAND	MAWR						1
S	LAND	SOSP						1
S	SMSHORE	DUNL			1		1	
S	SMSHORE	LESA	1				1	
S	SMSHORE	WESA			1		1	
F	MEDSHORE	WILL		1			1	
F	PHAL	RNPH	1				1	
F	PHAL	WIPH	1				1	
F	SMSHORE	LESA		1			1	
F	SMSHORE	WESA	1				1	
Total # of species / seasons			9	5	11	0	22	6

Table 20. Group density predictions by microhabitat.
Minimum, mean, and maximum density predictions, across ponds, for each group / season and alternative / year. A0 = alternative A, year 0; B0 = alternative B, year 0; C0 = alternative C, year 0; Amax50 = alternative A, year 50, maximum tidal marsh pond/panne assumptions; Bmax50 = alternative B, year 50, maximum tidal marsh pond/panne assumptions; Cmas50 = alternative C, year 50, maximum tidal marsh pond/panne assumptions. DABBLER = dabbling ducks; DIVER = diving ducks; LAND = landbirds; LGSHORE = large shorebirds; PHAL = phalaropes; SMSHORE = small shorebirds. W = winter (Nov-Feb), S = spring (Mar-May), F = fall (Aug-Oct). SP = managed ponds; SUBT = subtidal habitat within restored marshes; MUD = intertidal mudflat habitat within restored marshes; TMC = tidal marsh channels; TMP = tidal marsh ponds/pannes; TMV = tidal marsh vegetation. Model-predicted densities are based on 1999-2004 South San Francisco Bay avian survey data.

Group	Alternative	Min	SP	Max	Min	SubT	Max	Min	Mud	Max	Min	TMC	Max	Min	TMP	Max	Min	TMV	Max
DABBLERW	Amax50	1.574	2.896	4.741	0.402	0.402	0.402	0.233	0.233	0.233	1.295	3.424	6.429	1.201	3.243	6.126	0.362	1.626	3.410
DABBLERW	Amin50	1.574	2.896	4.741	0.402	0.402	0.402	0.233	0.233	0.233	0.337	1.551	3.329	0.932	2.151	3.153	0.000	0.549	1.570
DABBLERW	Bmax50	0.872	2.588	4.552	0.402	0.402	0.402	0.233	0.233	0.233	1.288	3.570	7.840	1.195	3.383	7.480	0.358	1.713	4.248
DABBLERW	Bmin50	0.872	2.588	4.552	0.402	0.402	0.402	0.233	0.233	0.233	0.333	1.698	4.152	0.385	2.134	3.942	0.000	0.628	2.059
DABBLERW	Cmax50	1.749	3.020	4.076	0.402	0.402	0.402	0.233	0.233	0.233	1.620	3.859	7.840	1.513	3.661	7.480	0.555	1.885	4.248
DABBLERW	Cmin50	1.749	3.020	4.076	0.402	0.402	0.402	0.233	0.233	0.233	0.527	1.857	4.152	0.465	2.257	3.942	0.000	0.701	2.059
DIVERW	Amax50	0.000	0.069	0.336	0.579	0.579	0.579	0.252	0.252	0.252	0.049	0.058	0.071	0.000	0.008	0.020	0.000	0.002	0.012
DIVERW	Amin50	0.000	0.069	0.336	0.579	0.579	0.579	0.252	0.252	0.252	0.049	0.058	0.071	0.002	0.009	0.020	0.000	0.002	0.012
DIVERW	Bmax50	0.000	0.105	0.677	0.579	0.579	0.579	0.252	0.252	0.252	0.043	0.055	0.073	0.000	0.005	0.023	0.000	0.002	0.015
DIVERW	Bmin50	0.000	0.105	0.677	0.579	0.579	0.579	0.252	0.252	0.252	0.043	0.055	0.073	0.000	0.007	0.023	0.000	0.002	0.015
DIVERW	Cmax50	0.000	0.101	0.677	0.579	0.579	0.579	0.252	0.252	0.252	0.047	0.061	0.080	0.000	0.011	0.029	0.000	0.004	0.021
DIVERW	Cmin50	0.000	0.101	0.677	0.579	0.579	0.579	0.252	0.252	0.252	0.047	0.061	0.080	0.000	0.012	0.029	0.000	0.004	0.021
FISHEATW	Amax50	0.131	0.244	0.297	0.347	0.347	0.347	0.140	0.140	0.140	0.089	0.092	0.100	0.000	0.001	0.007	0.000	0.001	0.007
FISHEATW	Amin50	0.131	0.244	0.297	0.347	0.347	0.347	0.140	0.140	0.140	0.089	0.092	0.100	0.000	0.001	0.007	0.000	0.001	0.007
FISHEATW	Bmax50	0.000	0.188	0.311	0.347	0.347	0.347	0.140	0.140	0.140	0.086	0.092	0.101	0.000	0.001	0.007	0.000	0.001	0.008
FISHEATW	Bmin50	0.000	0.188	0.311	0.347	0.347	0.347	0.140	0.140	0.140	0.086	0.092	0.101	0.000	0.001	0.007	0.000	0.001	0.008
FISHEATW	Cmax50	0.000	0.171	0.285	0.347	0.347	0.347	0.140	0.140	0.140	0.088	0.095	0.105	0.000	0.002	0.011	0.000	0.003	0.012
FISHEATW	Cmin50	0.000	0.171	0.285	0.347	0.347	0.347	0.140	0.140	0.140	0.088	0.095	0.105	0.000	0.003	0.011	0.000	0.003	0.012
LANDS	Amax50																6.664	7.624	8.903
LANDS	Amin50																6.664	7.624	8.903
LANDS	Bmax50																5.689	7.473	8.904
LANDS	Bmin50																5.689	7.473	8.904
LANDS	Cmax50																6.338	8.118	9.523
LANDS	Cmin50																6.338	8.118	9.523
MEDSHORW	Amax50	0.118	0.569	1.178	0.189	0.189	0.189	1.587	1.587	1.587	0.272	0.385	0.542	0.112	0.211	0.347	0.000	0.058	0.174
MEDSHORW	Amin50	0.118	0.569	1.178	0.189	0.189	0.189	1.587	1.587	1.587	0.272	0.385	0.541	0.175	0.225	0.347	0.000	0.058	0.174
MEDSHORW	Bmax50	0.017	0.819	1.569	0.189	0.189	0.189	1.587	1.587	1.587	0.191	0.360	0.542	0.041	0.189	0.347	0.000	0.048	0.174
MEDSHORW	Bmin50	0.017	0.819	1.569	0.189	0.189	0.189	1.587	1.587	1.587	0.191	0.360	0.541	0.041	0.198	0.347	0.000	0.048	0.174
MEDSHORW	Cmax50	0.119	1.152	1.547	0.189	0.189	0.189	1.587	1.587	1.587	0.225	0.427	0.630	0.071	0.247	0.425	0.000	0.091	0.242
MEDSHORW	Cmin50	0.119	1.152	1.547	0.189	0.189	0.189	1.587	1.587	1.587	0.225	0.427	0.630	0.071	0.242	0.425	0.000	0.091	0.241

Table 20, cont.

Group	Alternative	Min	SP	Max	Min	SubT	Max	Min	Mud	Max	Min	TMC	Max	Min	TMP	Max	Min	TMV	Max
PHALAROF	Amax50	1.671	6.249	19.899	0.000	0.000	0.000	0.000	0.000	0.000									
PHALAROF	Amin50	1.671	6.249	19.899	0.000	0.000	0.000	0.000	0.000	0.000									
PHALAROF	Bmax50	0.000	3.267	23.048	0.000	0.000	0.000	0.000	0.000	0.000									
PHALAROF	Bmin50	0.000	3.267	23.048	0.000	0.000	0.000	0.000	0.000	0.000									
PHALAROF	Cmax50	0.406	6.729	35.103	0.000	0.000	0.000	0.000	0.000	0.000									
PHALAROF	Cmin50	0.406	6.729	35.103	0.000	0.000	0.000	0.000	0.000	0.000									
SMSHOREF	Amax50	0.000	0.286	0.746	0.155	0.155	0.155	2.125	2.125	2.125	1.038	1.110	1.265	1.624	1.717	1.916	0.124	0.163	0.249
SMSHOREF	Amin50	0.000	0.286	0.746	0.155	0.155	0.155	2.125	2.125	2.125	0.690	0.743	0.856	1.202	1.288	1.390	0.000	0.003	0.023
SMSHOREF	Bmax50	0.000	2.004	8.791	0.155	0.155	0.155	2.125	2.125	2.125	0.766	1.077	1.259	1.273	1.674	1.908	0.000	0.146	0.246
SMSHOREF	Bmin50	0.000	2.004	8.791	0.155	0.155	0.155	2.125	2.125	2.125	0.684	0.730	0.898	1.187	1.251	1.444	0.000	0.001	0.047
SMSHOREF	Cmax50	0.000	3.099	8.980	0.155	0.155	0.155	2.125	2.125	2.125	0.761	1.067	1.247	1.267	1.661	1.893	0.000	0.140	0.239
SMSHOREF	Cmin50	0.000	3.099	8.980	0.155	0.155	0.155	2.125	2.125	2.125	0.684	0.719	0.895	1.181	1.239	1.440	0.000	0.001	0.045
SMSHORES	Amax50	0.000	28.28	115.67	0.023	0.023	0.023	9.609	9.609	9.609	0.264	0.280	0.291	0.084	0.097	0.106	0.018	0.031	0.039
SMSHORES	Amin50	0.000	28.28	115.67	0.023	0.023	0.023	9.609	9.609	9.609	0.311	0.322	0.332	0.124	0.129	0.138	0.056	0.065	0.073
SMSHORES	Bmax50	0.000	96.44	323.77	0.023	0.023	0.023	9.609	9.609	9.609	0.264	0.283	0.321	0.084	0.100	0.133	0.018	0.033	0.064
SMSHORES	Bmin50	0.000	96.44	323.77	0.023	0.023	0.023	9.609	9.609	9.609	0.311	0.324	0.334	0.124	0.132	0.141	0.056	0.066	0.075
SMSHORES	Cmax50	0.443	147.02	249.97	0.023	0.023	0.023	9.609	9.609	9.609	0.259	0.280	0.318	0.079	0.097	0.130	0.014	0.031	0.062
SMSHORES	Cmin50	0.443	147.02	249.97	0.023	0.023	0.023	9.609	9.609	9.609	0.306	0.321	0.332	0.120	0.130	0.140	0.052	0.063	0.073
SMSHOREW	Amax50	0.319	1.134	1.999	0.125	0.125	0.125	4.102	4.102	4.102	0.337	0.541	1.009	0.000	0.147	0.496	0.000	0.005	0.096
SMSHOREW	Amin50	0.319	1.134	1.999	0.125	0.125	0.125	4.102	4.102	4.102	0.309	0.508	0.967	0.000	0.080	0.330	0.000	0.003	0.073
SMSHOREW	Bmax50	0.152	1.823	3.686	0.125	0.125	0.125	4.102	4.102	4.102	0.337	0.668	1.206	0.000	0.242	0.642	0.000	0.030	0.203
SMSHOREW	Bmin50	0.152	1.823	3.686	0.125	0.125	0.125	4.102	4.102	4.102	0.309	0.634	1.171	0.000	0.226	0.616	0.000	0.024	0.184
SMSHOREW	Cmax50	0.286	2.554	3.598	0.125	0.125	0.125	4.102	4.102	4.102	0.275	0.553	1.129	0.000	0.159	0.585	0.000	0.012	0.161
SMSHOREW	Cmin50	0.286	2.554	3.598	0.125	0.125	0.125	4.102	4.102	4.102	0.248	0.521	1.095	0.000	0.170	0.560	0.000	0.009	0.143
WADERW	Amax50	0.012	0.067	0.138	0.092	0.092	0.092	0.139	0.139	0.139	0.123	0.138	0.191	0.000	0.008	0.054	0.000	0.004	0.042
WADERW	Amin50	0.012	0.067	0.138	0.092	0.092	0.092	0.139	0.139	0.139	0.123	0.138	0.191	0.000	0.013	0.054	0.000	0.004	0.042
WADERW	Bmax50	0.004	0.069	0.258	0.092	0.092	0.092	0.139	0.139	0.139	0.124	0.149	0.256	0.000	0.018	0.111	0.000	0.012	0.098
WADERW	Bmin50	0.004	0.069	0.258	0.092	0.092	0.092	0.139	0.139	0.139	0.124	0.149	0.256	0.000	0.028	0.111	0.000	0.012	0.098
WADERW	Cmax50	0.033	0.066	0.088	0.092	0.092	0.092	0.139	0.139	0.139	0.126	0.148	0.256	0.000	0.016	0.111	0.000	0.010	0.098
WADERW	Cmin50	0.033	0.066	0.088	0.092	0.092	0.092	0.139	0.139	0.139	0.126	0.148	0.256	0.000	0.027	0.111	0.000	0.010	0.098

Table 21. Focal species density predictions by microhabitat.
Minimum, mean, and maximum density predictions, across ponds, for each focal species / season and alternative / year. A0 = alternative A, year 0; B0 = alternative B, year 0; C0 = alternative C, year 0; Amax50 = alternative A, year 50, maximum tidal marsh pond/panne assumptions; Bmax50 = alternative B, year 50, maximum tidal marsh pond/panne assumptions; Cmas50 = alternative C, year 50, maximum tidal marsh pond/panne assumptions. DABBLER = dabbling ducks; DIVER = diving ducks; LAND = landbirds; LGSHORE = large shorebirds; PHAL = phalaropes; SMSHORE = small shorebirds. Species 4-letter codes are defined in Table 6. W = winter (Nov-Feb), S = spring (Mar-May), F = fall (Aug-Oct). SP = managed ponds; SUBT = subtidal habitat within restored marshes; MUD = intertidal mudflat habitat within restored marshes; TMC = tidal marsh channels; TMP = tidal marsh ponds/pannes; TMV = tidal marsh vegetation. Model-predicted densities are based on 1999-2004 South San Francisco Bay avian survey data.

Group	Species / Season	Alt.	Min	SubT	Max	Min	Mud	Max	Min	SP	Max	Min	TMC	Max	Min	TMP	Max	Min	TMV	Max
DABBLER	GADWW	Amax50	0.000	0.000	0.000	0.001	0.001	0.001	0.008	0.010	0.013	0.114	0.169	0.220	0.115	0.170	0.221	0.046	0.098	0.146
DABBLER	GADWW	Amin50	0.000	0.000	0.000	0.001	0.001	0.001	0.008	0.010	0.013	0.034	0.084	0.133	0.069	0.112	0.134	0.000	0.024	0.064
DABBLER	GADWW	Bmax50	0.000	0.000	0.000	0.001	0.001	0.001	0.005	0.009	0.012	0.114	0.168	0.236	0.114	0.168	0.237	0.046	0.097	0.161
DABBLER	GADWW	Bmin50	0.000	0.000	0.000	0.001	0.001	0.001	0.005	0.009	0.012	0.034	0.086	0.147	0.044	0.106	0.148	0.000	0.026	0.078
DABBLER	GADWW	Cmax50	0.000	0.000	0.000	0.001	0.001	0.001	0.006	0.010	0.013	0.125	0.175	0.236	0.125	0.175	0.237	0.056	0.103	0.161
DABBLER	GADWW	Cmin50	0.000	0.000	0.000	0.001	0.001	0.001	0.006	0.010	0.013	0.047	0.092	0.147	0.048	0.110	0.148	0.000	0.028	0.078
DABBLER	MALLW	Amax50	0.008	0.008	0.008	0.047	0.047	0.047	0.017	0.041	0.072	0.079	0.088	0.096	0.047	0.056	0.063	0.000	0.001	0.005
DABBLER	MALLW	Amin50	0.008	0.008	0.008	0.047	0.047	0.047	0.017	0.041	0.072	0.092	0.096	0.104	0.062	0.067	0.072	0.002	0.006	0.013
DABBLER	MALLW	Bmax50	0.008	0.008	0.008	0.047	0.047	0.047	0.009	0.041	0.077	0.079	0.089	0.100	0.047	0.057	0.067	0.000	0.002	0.009
DABBLER	MALLW	Bmin50	0.008	0.008	0.008	0.047	0.047	0.047	0.009	0.041	0.077	0.091	0.098	0.105	0.059	0.067	0.072	0.000	0.007	0.013
DABBLER	MALLW	Cmax50	0.008	0.008	0.008	0.047	0.047	0.047	0.009	0.060	0.082	0.079	0.089	0.100	0.047	0.057	0.067	0.000	0.001	0.009
DABBLER	MALLW	Cmin50	0.008	0.008	0.008	0.047	0.047	0.047	0.009	0.060	0.082	0.091	0.098	0.105	0.059	0.067	0.072	0.000	0.007	0.013
DABBLER	NOPIW	Amax50	0.009	0.009	0.009	0.058	0.058	0.058	0.011	0.027	0.050	0.008	0.011	0.016	0.022	0.025	0.030	0.000	0.000	0.001
DABBLER	NOPIW	Amin50	0.009	0.009	0.009	0.058	0.058	0.058	0.011	0.027	0.050	0.008	0.011	0.016	0.022	0.023	0.028	0.000	0.000	0.001
DABBLER	NOPIW	Bmax50	0.009	0.009	0.009	0.058	0.058	0.058	0.000	0.028	0.051	0.008	0.013	0.017	0.022	0.027	0.031	0.000	0.000	0.002
DABBLER	NOPIW	Bmin50	0.009	0.009	0.009	0.058	0.058	0.058	0.000	0.028	0.051	0.008	0.013	0.017	0.022	0.026	0.030	0.000	0.000	0.002
DABBLER	NOPIW	Cmax50	0.009	0.009	0.009	0.058	0.058	0.058	0.022	0.038	0.052	0.008	0.012	0.017	0.022	0.026	0.031	0.000	0.000	0.002
DABBLER	NOPIW	Cmin50	0.009	0.009	0.009	0.058	0.058	0.058	0.022	0.038	0.052	0.008	0.012	0.017	0.022	0.026	0.030	0.000	0.000	0.002
DABBLER	NSHOW	Amax50	0.225	0.225	0.225	0.200	0.200	0.200	0.196	0.303	0.417	0.393	0.785	1.229	0.393	0.785	1.229	0.146	0.468	0.834
DABBLER	NSHOW	Amin50	0.225	0.225	0.225	0.200	0.200	0.200	0.196	0.303	0.417	0.066	0.358	0.706	0.259	0.528	0.706	0.000	0.143	0.403
DABBLER	NSHOW	Bmax50	0.225	0.225	0.225	0.200	0.200	0.200	0.087	0.275	0.432	0.391	0.797	1.372	0.391	0.797	1.373	0.144	0.478	0.951
DABBLER	NSHOW	Bmin50	0.225	0.225	0.225	0.200	0.200	0.200	0.087	0.275	0.432	0.065	0.385	0.816	0.102	0.507	0.816	0.000	0.160	0.493
DABBLER	NSHOW	Cmax50	0.225	0.225	0.225	0.200	0.200	0.200	0.151	0.325	0.495	0.468	0.841	1.372	0.468	0.841	1.373	0.207	0.514	0.951
DABBLER	NSHOW	Cmin50	0.225	0.225	0.225	0.200	0.200	0.200	0.151	0.325	0.495	0.123	0.416	0.816	0.123	0.531	0.816	0.000	0.172	0.493
DIVER	RUDUW	Amax50	0.431	0.431	0.431	0.110	0.110	0.110	0.061	0.148	0.263	0.025	0.032	0.035	0.006	0.012	0.016	0.000	0.004	0.007
DIVER	RUDUW	Amin50	0.431	0.431	0.431	0.110	0.110	0.110	0.061	0.148	0.263	0.025	0.032	0.035	0.008	0.014	0.016	0.000	0.004	0.007
DIVER	RUDUW	Bmax50	0.431	0.431	0.431	0.110	0.110	0.110	0.000	0.095	0.258	0.023	0.029	0.035	0.003	0.009	0.015	0.000	0.002	0.006
DIVER	RUDUW	Bmin50	0.431	0.431	0.431	0.110	0.110	0.110	0.000	0.095	0.258	0.023	0.029	0.035	0.003	0.010	0.015	0.000	0.002	0.006
DIVER	RUDUW	Cmax50	0.431	0.431	0.431	0.110	0.110	0.110	0.000	0.081	0.258	0.024	0.030	0.037	0.004	0.011	0.017	0.000	0.003	0.009
DIVER	RUDUW	Cmin50	0.431	0.431	0.431	0.110	0.110	0.110	0.000	0.081	0.258	0.024	0.030	0.037	0.004	0.011	0.017	0.000	0.003	0.009

Table 21, cont.

Group	Species / Season	Alt.	Min	SubT	Max	Min	Mud	Max	Min	SP	Max	Min	TMC	Max	Min	TMP	Max	Min	TMV	Max
DIVER	SCAUW	Amax50	0.139	0.139	0.139	0.008	0.008	0.008	0.073	0.143	0.217									
DIVER	SCAUW	Amin50	0.139	0.139	0.139	0.008	0.008	0.008	0.073	0.143	0.217									
DIVER	SCAUW	Bmax50	0.139	0.139	0.139	0.008	0.008	0.008	0.021	0.118	0.221									
DIVER	SCAUW	Bmin50	0.139	0.139	0.139	0.008	0.008	0.008	0.021	0.118	0.221									
DIVER	SCAUW	Cmax50	0.139	0.139	0.139	0.008	0.008	0.008	0.067	0.151	0.239									
DIVER	SCAUW	Cmin50	0.139	0.139	0.139	0.008	0.008	0.008	0.067	0.151	0.239									
EAREDGR	EAGRW	Amax50	0.205	0.205	0.205	0.049	0.049	0.049	0.000	0.045	0.260									
EAREDGR	EAGRW	Amin50	0.205	0.205	0.205	0.049	0.049	0.049	0.000	0.045	0.260									
EAREDGR	EAGRW	Bmax50	0.205	0.205	0.205	0.049	0.049	0.049	0.000	0.156	0.943									
EAREDGR	EAGRW	Bmin50	0.205	0.205	0.205	0.049	0.049	0.049	0.000	0.156	0.943									
EAREDGR	EAGRW	Cmax50	0.205	0.205	0.205	0.049	0.049	0.049	0.000	0.070	0.943									
EAREDGR	EAGRW	Cmin50	0.205	0.205	0.205	0.049	0.049	0.049	0.000	0.070	0.943									
FISHEAT	AWPEW	Amax50	0.160	0.160	0.160	0.034	0.034	0.034	0.040	0.063	0.086									
FISHEAT	AWPEW	Amin50	0.160	0.160	0.160	0.034	0.034	0.034	0.040	0.063	0.086									
FISHEAT	AWPEW	Bmax50	0.160	0.160	0.160	0.034	0.034	0.034	0.000	0.042	0.084									
FISHEAT	AWPEW	Bmin50	0.160	0.160	0.160	0.034	0.034	0.034	0.000	0.042	0.084									
FISHEAT	AWPEW	Cmax50	0.160	0.160	0.160	0.034	0.034	0.034	0.000	0.039	0.084									
FISHEAT	AWPEW	Cmin50	0.160	0.160	0.160	0.034	0.034	0.034	0.000	0.039	0.084									
FISHEAT	FOTEW	Amax50	0.048	0.048	0.048	0.049	0.049	0.049	0.000	0.020	0.037									
FISHEAT	FOTEW	Amin50	0.048	0.048	0.048	0.049	0.049	0.049	0.000	0.020	0.037									
FISHEAT	FOTEW	Bmax50	0.048	0.048	0.048	0.049	0.049	0.049	0.000	0.014	0.040									
FISHEAT	FOTEW	Bmin50	0.048	0.048	0.048	0.049	0.049	0.049	0.000	0.014	0.040									
FISHEAT	FOTEW	Cmax50	0.048	0.048	0.048	0.049	0.049	0.049	0.000	0.008	0.019									
FISHEAT	FOTEW	Cmin50	0.048	0.048	0.048	0.049	0.049	0.049	0.000	0.008	0.019									
LAND	COYES	Amax50																0.101	0.108	0.116
LAND	COYES	Amin50																0.101	0.108	0.116
LAND	COYES	Bmax50																0.095	0.110	0.118
LAND	COYES	Bmin50																0.095	0.110	0.118
LAND	COYES	Cmax50																0.103	0.112	0.118
LAND	COYES	Cmin50																0.103	0.112	0.118
LAND	MAWRS	Amax50																0.000	0.413	1.481
LAND	MAWRS	Amin50																0.000	0.413	1.481
LAND	MAWRS	Bmax50																0.000	0.871	1.793
LAND	MAWRS	Bmin50																0.000	0.871	1.793
LAND	MAWRS	Cmax50																0.000	0.746	1.793
LAND	MAWRS	Cmin50																0.000	0.746	1.793

Table 21, cont.

Group	Species/ Season	Alt.	Min	SubT	Max	Min	Mud	Max	Min	SP	Max	Min	TMC	Max	Min	TMP	Max	Min	TMV	Max
LAND	SOSPS	Amax50																4.916	5.709	6.779
LAND	SOSPS	Amin50																4.916	5.709	6.779
LAND	SOSPS	Bmax50																4.112	5.590	6.780
LAND	SOSPS	Bmin50																4.112	5.590	6.780
LAND	SOSPS	Cmax50																4.647	6.125	7.302
LAND	SOSPS	Cmin50																4.647	6.125	7.302
MEDSHORE	AMAVW	Amax50	0.155	0.155	0.155	1.430	1.430	1.430	0.000	0.229	0.790	0.010	0.019	0.032	0.019	0.028	0.041	0.000	0.000	0.002
MEDSHORE	AMAVW	Amin50	0.155	0.155	0.155	1.430	1.430	1.430	0.000	0.229	0.790	0.016	0.025	0.037	0.025	0.032	0.046	0.000	0.001	0.007
MEDSHORE	AMAVW	Bmax50	0.155	0.155	0.155	1.430	1.430	1.430	0.000	0.421	1.045	0.009	0.022	0.042	0.017	0.030	0.051	0.000	0.000	0.012
MEDSHORE	AMAVW	Bmin50	0.155	0.155	0.155	1.430	1.430	1.430	0.000	0.421	1.045	0.014	0.027	0.047	0.023	0.037	0.056	0.000	0.001	0.017
MEDSHORE	AMAVW	Cmax50	0.155	0.155	0.155	1.430	1.430	1.430	0.000	0.546	0.999	0.012	0.026	0.042	0.021	0.035	0.051	0.000	0.001	0.012
MEDSHORE	AMAVW	Cmin50	0.155	0.155	0.155	1.430	1.430	1.430	0.000	0.546	0.999	0.018	0.032	0.047	0.027	0.041	0.056	0.000	0.004	0.017
MEDSHORE	BNSTW	Amax50	0.001	0.001	0.001	0.162	0.162	0.162	0.000	0.102	0.226	0.056	0.073	0.110	0.123	0.141	0.180	0.047	0.064	0.101
MEDSHORE	BNSTW	Amin50	0.001	0.001	0.001	0.162	0.162	0.162	0.000	0.102	0.226	0.000	0.001	0.007	0.038	0.056	0.071	0.000	0.000	0.000
MEDSHORE	BNSTW	Bmax50	0.001	0.001	0.001	0.162	0.162	0.162	0.000	0.264	0.762	0.000	0.068	0.110	0.059	0.136	0.180	0.000	0.059	0.101
MEDSHORE	BNSTW	Bmin50	0.001	0.001	0.001	0.162	0.162	0.162	0.000	0.264	0.762	0.000	0.001	0.024	0.038	0.054	0.089	0.000	0.000	0.016
MEDSHORE	BNSTW	Cmax50	0.001	0.001	0.001	0.162	0.162	0.162	0.000	0.319	0.771	0.000	0.070	0.116	0.062	0.138	0.187	0.000	0.061	0.107
MEDSHORE	BNSTW	Cmin50	0.001	0.001	0.001	0.162	0.162	0.162	0.000	0.319	0.771	0.000	0.001	0.025	0.039	0.055	0.091	0.000	0.000	0.017
MEDSHORE	GRYEW	Amax50	0.003	0.003	0.003	0.015	0.015	0.015	0.010	0.043	0.084	0.166	0.244	0.299	0.000	0.056	0.102	0.000	0.053	0.098
MEDSHORE	GRYEW	Amin50	0.003	0.003	0.003	0.015	0.015	0.015	0.010	0.043	0.084	0.166	0.244	0.299	0.000	0.060	0.102	0.000	0.053	0.098
MEDSHORE	GRYEW	Bmax50	0.003	0.003	0.003	0.015	0.015	0.015	0.023	0.050	0.082	0.179	0.253	0.302	0.000	0.063	0.104	0.000	0.059	0.100
MEDSHORE	GRYEW	Bmin50	0.003	0.003	0.003	0.015	0.015	0.015	0.023	0.050	0.082	0.179	0.253	0.302	0.024	0.075	0.104	0.000	0.059	0.100
MEDSHORE	GRYEW	Cmax50	0.003	0.003	0.003	0.015	0.015	0.015	0.027	0.067	0.087	0.192	0.264	0.327	0.011	0.072	0.125	0.008	0.068	0.121
MEDSHORE	GRYEW	Cmin50	0.003	0.003	0.003	0.015	0.015	0.015	0.027	0.067	0.087	0.192	0.264	0.327	0.024	0.077	0.104	0.008	0.068	0.121
MEDSHORE	WILLF	Amax50	0.004	0.004	0.004	0.079	0.079	0.079	0.374	1.000	2.539	0.000	0.018	0.030	0.000	0.006	0.013	0.000	0.000	0.000
MEDSHORE	WILLF	Amin50	0.004	0.004	0.004	0.079	0.079	0.079	0.374	1.000	2.539	0.000	0.018	0.030	0.000	0.006	0.013	0.000	0.000	0.000
MEDSHORE	WILLF	Bmax50	0.004	0.004	0.004	0.079	0.079	0.079	0.113	1.158	2.405	0.004	0.023	0.037	0.000	0.008	0.021	0.000	0.001	0.007
MEDSHORE	WILLF	Bmin50	0.004	0.004	0.004	0.079	0.079	0.079	0.113	1.158	2.405	0.004	0.023	0.037	0.002	0.011	0.021	0.000	0.001	0.007
MEDSHORE	WILLF	Cmax50	0.004	0.004	0.004	0.079	0.079	0.079	0.113	1.826	2.338	0.000	0.020	0.033	0.000	0.006	0.017	0.000	0.000	0.003
MEDSHORE	WILLF	Cmin50	0.004	0.004	0.004	0.079	0.079	0.079	0.113	1.826	2.338	0.000	0.020	0.033	0.000	0.007	0.017	0.000	0.000	0.003
MEDSHORE	WILLW	Amax50	0.025	0.025	0.025	0.514	0.514	0.514	0.177	0.396	0.614	0.076	0.086	0.097	0.011	0.020	0.030	0.001	0.011	0.021
MEDSHORE	WILLW	Amin50	0.025	0.025	0.025	0.514	0.514	0.514	0.177	0.396	0.614	0.076	0.086	0.097	0.011	0.020	0.030	0.001	0.011	0.021
MEDSHORE	WILLW	Bmax50	0.025	0.025	0.025	0.514	0.514	0.514	0.097	0.414	0.805	0.076	0.088	0.097	0.011	0.022	0.030	0.002	0.013	0.021
MEDSHORE	WILLW	Bmin50	0.025	0.025	0.025	0.514	0.514	0.514	0.097	0.414	0.805	0.076	0.088	0.097	0.016	0.024	0.030	0.002	0.013	0.021
MEDSHORE	WILLW	Cmax50	0.025	0.025	0.025	0.514	0.514	0.514	0.097	0.576	0.815	0.079	0.090	0.099	0.014	0.024	0.032	0.004	0.015	0.022
MEDSHORE	WILLW	Cmin50	0.025	0.025	0.025	0.514	0.514	0.514	0.097	0.576	0.815	0.079	0.090	0.099	0.015	0.025	0.030	0.004	0.015	0.022

Table 21, cont.

Group	Species/ Season	Alt.	Min	SubT	Max	Min	Mud	Max	Min	SP	Max	Min	TMC	Max	Min	TMP	Max	Min	TMV	Max
PHAL	RNPHF	Amax50	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.072	0.215									
PHAL	RNPHF	Amin50	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.072	0.215									
PHAL	RNPHF	Bmax50	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.052	0.389									
PHAL	RNPHF	Bmin50	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.052	0.389									
PHAL	RNPHF	Cmax50	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.007	0.130									
PHAL	RNPHF	Cmin50	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.007	0.130									
PHAL	WIPHF	Amax50	0.000	0.000	0.000	0.000	0.000	0.000	0.021	0.086	0.236									
PHAL	WIPHF	Amin50	0.000	0.000	0.000	0.000	0.000	0.000	0.021	0.086	0.236									
PHAL	WIPHF	Bmax50	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.150	0.350									
PHAL	WIPHF	Bmin50	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.150	0.350									
PHAL	WIPHF	Cmax50	0.000	0.000	0.000	0.000	0.000	0.000	0.025	0.205	0.249									
PHAL	WIPHF	Cmin50	0.000	0.000	0.000	0.000	0.000	0.000	0.025	0.205	0.249									
SMSHORE	DUNLS	Amax50	0.012	0.012	0.012	4.125	4.125	4.125	0.000	1.877	5.744	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
SMSHORE	DUNLS	Amin50	0.012	0.012	0.012	4.125	4.125	4.125	0.000	1.877	5.744	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
SMSHORE	DUNLS	Bmax50	0.012	0.012	0.012	4.125	4.125	4.125	0.000	5.180	12.054	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
SMSHORE	DUNLS	Bmin50	0.012	0.012	0.012	4.125	4.125	4.125	0.000	5.180	12.054	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
SMSHORE	DUNLS	Cmax50	0.012	0.012	0.012	4.125	4.125	4.125	0.000	7.670	12.058	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
SMSHORE	DUNLS	Cmin50	0.012	0.012	0.012	4.125	4.125	4.125	0.000	7.670	12.058	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
SMSHORE	DUNLW	Amax50	0.008	0.008	0.008	2.763	2.763	2.763	0.300	1.168	2.645	0.000	0.085	0.203	0.000	0.053	0.164	0.000	0.009	0.072
SMSHORE	DUNLW	Amin50	0.008	0.008	0.008	2.763	2.763	2.763	0.300	1.168	2.645	0.000	0.040	0.147	0.000	0.011	0.056	0.000	0.001	0.022
SMSHORE	DUNLW	Bmax50	0.008	0.008	0.008	2.763	2.763	2.763	0.129	2.214	4.942	0.000	0.113	0.303	0.000	0.080	0.261	0.000	0.027	0.161
SMSHORE	DUNLW	Bmin50	0.008	0.008	0.008	2.763	2.763	2.763	0.129	2.214	4.942	0.000	0.069	0.258	0.000	0.051	0.217	0.000	0.011	0.121
SMSHORE	DUNLW	Cmax50	0.008	0.008	0.008	2.763	2.763	2.763	0.494	2.981	4.037	0.000	0.066	0.268	0.000	0.042	0.227	0.000	0.010	0.130
SMSHORE	DUNLW	Cmin50	0.008	0.008	0.008	2.763	2.763	2.763	0.494	2.981	4.037	0.000	0.035	0.224	0.000	0.030	0.184	0.000	0.004	0.091
SMSHORE	LESAF	Amax50	0.005	0.005	0.005	0.437	0.437	0.437	0.288	0.563	1.082	0.678	0.755	0.887	1.117	1.214	1.381	0.054	0.103	0.186
SMSHORE	LESAF	Amin50	0.005	0.005	0.005	0.437	0.437	0.437	0.288	0.563	1.082	0.359	0.417	0.511	0.718	0.823	0.907	0.000	0.000	0.000
SMSHORE	LESAF	Bmax50	0.005	0.005	0.005	0.437	0.437	0.437	0.169	0.717	1.155	0.464	0.742	0.921	0.847	1.197	1.423	0.000	0.097	0.207
SMSHORE	LESAF	Bmin50	0.005	0.005	0.005	0.437	0.437	0.437	0.169	0.717	1.155	0.358	0.420	0.641	0.713	0.820	1.071	0.000	0.001	0.031
SMSHORE	LESAF	Cmax50	0.005	0.005	0.005	0.437	0.437	0.437	0.169	0.994	1.243	0.454	0.713	0.880	0.834	1.161	1.372	0.000	0.078	0.181
SMSHORE	LESAF	Cmin50	0.005	0.005	0.005	0.437	0.437	0.437	0.169	0.994	1.243	0.333	0.394	0.606	0.702	0.794	1.026	0.000	0.000	0.009

Table 21, cont.

Group	Species/ Season	Alt.	Min	SubT	Max	Min	Mud	Max	Min	SP	Max	Min	TMC	Max	Min	TMP	Max	Min	TMV	Max
SMSHORE	LESAS	Amax50	0.008	0.008	0.008	0.000	0.000	0.000	0.000	0.103	0.696	0.087	0.100	0.117	0.015	0.026	0.043	0.000	0.000	0.008
SMSHORE	LESAS	Amin50	0.008	0.008	0.008	0.000	0.000	0.000	0.000	0.103	0.696	0.115	0.126	0.144	0.040	0.048	0.053	0.006	0.016	0.032
SMSHORE	LESAS	Bmax50	0.008	0.008	0.008	0.000	0.000	0.000	0.000	0.099	0.871	0.087	0.106	0.132	0.015	0.032	0.057	0.000	0.003	0.022
SMSHORE	LESAS	Bmin50	0.008	0.008	0.008	0.000	0.000	0.000	0.000	0.099	0.871	0.118	0.131	0.152	0.044	0.055	0.066	0.009	0.020	0.039
SMSHORE	LESAS	Cmax50	0.008	0.008	0.008	0.000	0.000	0.000	0.000	0.051	0.309	0.085	0.100	0.130	0.013	0.027	0.055	0.000	0.001	0.019
SMSHORE	LESAS	Cmin50	0.008	0.008	0.008	0.000	0.000	0.000	0.000	0.051	0.309	0.115	0.126	0.151	0.041	0.052	0.066	0.006	0.016	0.038
SMSHORE	LESAW	Amax50	0.115	0.115	0.115	0.528	0.528	0.528	0.321	0.615	1.053	0.103	0.192	0.357	0.000	0.017	0.142	0.000	0.000	0.000
SMSHORE	LESAW	Amin50	0.115	0.115	0.115	0.528	0.528	0.528	0.321	0.615	1.053	0.118	0.209	0.377	0.000	0.015	0.065	0.000	0.000	0.000
SMSHORE	LESAW	Bmax50	0.115	0.115	0.115	0.528	0.528	0.528	0.371	0.748	1.080	0.098	0.242	0.454	0.000	0.059	0.223	0.000	0.000	0.000
SMSHORE	LESAW	Bmin50	0.115	0.115	0.115	0.528	0.528	0.528	0.371	0.748	1.080	0.114	0.259	0.474	0.000	0.080	0.236	0.000	0.000	0.000
SMSHORE	LESAW	Cmax50	0.115	0.115	0.115	0.528	0.528	0.528	0.371	0.904	1.133	0.066	0.184	0.437	0.000	0.031	0.209	0.000	0.000	0.000
SMSHORE	LESAW	Cmin50	0.115	0.115	0.115	0.528	0.528	0.528	0.371	0.904	1.133	0.081	0.201	0.457	0.000	0.052	0.165	0.000	0.000	0.000
SMSHORE	SEPLW	Amax50	0.000	0.000	0.000	0.204	0.204	0.204	0.000	0.070	0.193	0.003	0.010	0.020	0.017	0.024	0.034	0.003	0.010	0.020
SMSHORE	SEPLW	Amin50	0.000	0.000	0.000	0.204	0.204	0.204	0.000	0.070	0.193	0.000	0.000	0.000	0.000	0.006	0.012	0.000	0.000	0.000
SMSHORE	SEPLW	Bmax50	0.000	0.000	0.000	0.204	0.204	0.204	0.000	0.121	0.274	0.000	0.010	0.025	0.014	0.025	0.040	0.000	0.010	0.025
SMSHORE	SEPLW	Bmin50	0.000	0.000	0.000	0.204	0.204	0.204	0.000	0.121	0.274	0.000	0.000	0.012	0.000	0.008	0.026	0.000	0.000	0.012
SMSHORE	SEPLW	Cmax50	0.000	0.000	0.000	0.204	0.204	0.204	0.038	0.172	0.217	0.000	0.007	0.023	0.012	0.021	0.038	0.000	0.007	0.023
SMSHORE	SEPLW	Cmin50	0.000	0.000	0.000	0.204	0.204	0.204	0.038	0.172	0.217	0.000	0.000	0.010	0.000	0.006	0.024	0.000	0.000	0.010
SMSHORE	WESAF	Amax50	0.151	0.151	0.151	0.560	0.560	0.560	0.000	1.030	3.186	0.037	0.069	0.142	0.219	0.257	0.342	0.000	0.002	0.021
SMSHORE	WESAF	Amin50	0.151	0.151	0.151	0.560	0.560	0.560	0.000	1.030	3.186	0.016	0.047	0.118	0.194	0.221	0.314	0.000	0.000	0.000
SMSHORE	WESAF	Bmax50	0.151	0.151	0.151	0.560	0.560	0.560	0.000	2.564	5.249	0.032	0.091	0.180	0.213	0.282	0.387	0.000	0.008	0.055
SMSHORE	WESAF	Bmin50	0.151	0.151	0.151	0.560	0.560	0.560	0.000	2.564	5.249	0.011	0.069	0.156	0.188	0.266	0.359	0.000	0.003	0.033
SMSHORE	WESAF	Cmax50	0.151	0.151	0.151	0.560	0.560	0.560	0.000	3.708	5.234	0.038	0.091	0.184	0.220	0.283	0.392	0.000	0.007	0.058
SMSHORE	WESAF	Cmin50	0.151	0.151	0.151	0.560	0.560	0.560	0.000	3.708	5.234	0.016	0.069	0.159	0.199	0.268	0.363	0.000	0.003	0.036
SMSHORE	WESAS	Amax50	0.002	0.002	0.002	7.901	7.901	7.901	0.000	3.755	11.865	0.000	0.003	0.020	0.054	0.062	0.084	0.035	0.043	0.064
SMSHORE	WESAS	Amin50	0.002	0.002	0.002	7.901	7.901	7.901	0.000	3.755	11.865	0.000	0.001	0.010	0.044	0.050	0.073	0.025	0.033	0.054
SMSHORE	WESAS	Bmax50	0.002	0.002	0.002	7.901	7.901	7.901	0.000	11.501	29.446	0.000	0.008	0.026	0.053	0.068	0.090	0.034	0.049	0.070
SMSHORE	WESAS	Bmin50	0.002	0.002	0.002	7.901	7.901	7.901	0.000	11.501	29.446	0.000	0.004	0.017	0.044	0.061	0.081	0.024	0.039	0.061
SMSHORE	WESAS	Cmax50	0.002	0.002	0.002	7.901	7.901	7.901	0.000	17.354	29.604	0.000	0.006	0.027	0.054	0.067	0.092	0.034	0.047	0.071
SMSHORE	WESAS	Cmin50	0.002	0.002	0.002	7.901	7.901	7.901	0.000	17.354	29.604	0.000	0.003	0.017	0.045	0.060	0.081	0.025	0.037	0.061
SMSHORE	WESAW	Amax50	0.006	0.006	0.006	3.389	3.389	3.389	0.373	1.335	2.964	0.000	0.060	0.146	0.081	0.147	0.239	0.000	0.055	0.140
SMSHORE	WESAW	Amin50	0.006	0.006	0.006	3.389	3.389	3.389	0.373	1.335	2.964	0.000	0.000	0.000	0.000	0.023	0.062	0.000	0.000	0.000
SMSHORE	WESAW	Bmax50	0.006	0.006	0.006	3.389	3.389	3.389	0.000	2.051	4.485	0.000	0.070	0.209	0.081	0.157	0.307	0.000	0.065	0.203
SMSHORE	WESAW	Bmin50	0.006	0.006	0.006	3.389	3.389	3.389	0.000	2.051	4.485	0.000	0.005	0.102	0.000	0.042	0.192	0.000	0.004	0.097
SMSHORE	WESAW	Cmax50	0.006	0.006	0.006	3.389	3.389	3.389	0.555	2.868	3.765	0.000	0.043	0.190	0.052	0.124	0.288	0.000	0.039	0.185
SMSHORE	WESAW	Cmin50	0.006	0.006	0.006	3.389	3.389	3.389	0.555	2.868	3.765	0.000	0.002	0.085	0.000	0.027	0.173	0.000	0.002	0.080

Table 22. Clapper Rail predictions.
Estimated tidal marsh area (ha) and predicted Clapper Rail abundance, based on lower, mean, and upper density estimates, for each alternative, and for the rest of the South Bay. Density estimates were based on data collected from 2005-2006 breeding season Clapper Rail surveys in South San Francisco Bay. "Rest of South Bay" tidal marsh ha was derived from EcoAtlas "modern baylands" layer (SFEI 1998).

Alternative	Tidal Marsh ha	Upper Abundance	Mean Abundance	Lower Abundance
A	1794	2063	969	108
B	2802	3222	1513	168
C	4570	5256	2468	274
Rest of South Bay	3902	4487	2107	234

Table 23. Snowy Plover predictions.
Estimated managed dry pond area (ha) and predicted Snowy Plover abundance, based on lower and upper density estimates, for each alternative. Mean nesting densities from Feeney (1991) were used as an upper density estimate, and mean nesting estimates across multiple data sources were used as a lower density estimate (see Appendix 2).

Alternative	Available Dry Ponds (ha)	Lower	Upper
A	370	74	518
B	675	135	945
C	0	0	0

Table 24. Predicted percent change within project area.
Model-predicted percent change from baseline (alternative A, year 0) abundance indices for each alternative in year 50 across all restoration area managed ponds and restored marshes. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data. DABBLER = dabbling ducks; DIVER = diving ducks; LAND = landbirds; LGSHORE = large shorebirds; PHAL = phalaropes; SMSHORE = small shorebirds. Species 4-letter codes are defined in Table 6. W = winter (Nov-Feb), S = spring (Mar-May), F = fall (Aug-Oct).

Group	Species-Season	A	B	C
DABBLER	GADW-W	404%	722%	1260%
DABBLER	MALL-W	1%	84%	61%
DABBLER	NOPI-W	-69%	-40%	-57%
DABBLER	NSHO-W	18%	110%	176%
DIVER	RUDU-W	-77%	-56%	-79%
DIVER	SCAU-W	-65%	-40%	-70%
EAREDGR	EAGR-W	-95%	-64%	-93%
FISHEAT	AWPE-W	-75%	-55%	-81%
FISHEAT	FOTE-W	-80%	-64%	-89%
LAND	COYE-S	19843%	31382%	51723%
LAND	MAWR-S	123144%	286351%	383493%
LAND	SOSP-S	1050442%	1592222%	2824107%
MEDSHORE	AMAV-W	-92%	-71%	-82%
MEDSHORE	BNST-W	-66%	-4%	-34%
MEDSHORE	GRYE-W	181%	365%	534%
MEDSHORE	WILL-F	-36%	36%	-35%
MEDSHORE	WILL-W	-66%	-33%	-64%
PHAL	RNPH-F	-83%	-68%	-98%
PHAL	WIPH-F	-78%	-31%	-66%
SMSHORE	DUNL-S	-87%	-36%	-62%
SMSHORE	DUNL-W	-83%	-45%	-72%
SMSHORE	LESA-F	-35%	28%	5%
SMSHORE	LESA-S	-92%	-85%	-90%
SMSHORE	LESA-W	-61%	-15%	-66%
SMSHORE	SEPL-W	-80%	-39%	-60%
SMSHORE	WESA-F	-85%	-34%	-60%
SMSHORE	WESA-S	-88%	-32%	-57%
SMSHORE	WESA-W	-79%	-44%	-69%

Table 25. Predicted percent change across South Bay habitats.
Model-predicted percent change from baseline (alternative A, year 0) abundance indices for each alternative in year 50 across all South Bay habitats. Model predictions are based on 1999-2004 South San Francisco Bay avian survey data. DABBLER = dabbling ducks; DIVER = diving ducks; LAND = landbirds; LGSHORE = large shorebirds; PHAL = phalaropes; SMSHORE = small shorebirds. Species 4-letter codes are defined in Table 6. W = winter (Nov-Feb), S = spring (Mar-May), F = fall (Aug-Oct).

Group	Species-Season	A	B	C
DABBLER	GADW-W	171%	305%	533%
DABBLER	MALL-W	0%	42%	30%
DABBLER	NOPI-W	-55%	-31%	-46%
DABBLER	NSHO-W	3%	15%	25%
DIVER	RUDU-W	-67%	-49%	-68%
DIVER	SCAU-W	-52%	-32%	-55%
EAREDGR	EAGR-W	-42%	-28%	-41%
FISHEAT	AWPE-W	-75%	-55%	-81%
FISHEAT	FOTE-W	-72%	-57%	-80%
LAND	COYE-S	46%	73%	121%
LAND	MAWR-S	36%	84%	113%
LAND	SOSP-S	66%	101%	179%
MEDSHORE	AMAV-W	-53%	-41%	-47%
MEDSHORE	BNST-W	-16%	-1%	-9%
MEDSHORE	GRYE-W	57%	115%	168%
MEDSHORE	WILL-F	-29%	29%	-28%
MEDSHORE	WILL-W	-48%	-24%	-46%
PHAL	RNPH-F	-76%	-63%	-90%
PHAL	WIPH-F	-78%	-31%	-66%
SMSHORE	DUNL-S	-81%	-33%	-58%
SMSHORE	DUNL-W	-64%	-35%	-55%
SMSHORE	LESA-F	-14%	11%	2%
SMSHORE	LESA-S	-54%	-50%	-53%
SMSHORE	LESA-W	-31%	-8%	-33%
SMSHORE	SEPL-W	-76%	-37%	-57%
SMSHORE	WESA-F	-68%	-27%	-48%
SMSHORE	WESA-S	-81%	-29%	-52%
SMSHORE	WESA-W	-63%	-35%	-55%

Table 26. Predicted proportional change in tidal flat shorebird numbers.
Predicted proportions of baseline (alternative A, year 0) abundance indices for each alternative in year 50 based on tidal flat projections provided by PWA (Table 12) and tidal flat bird densities calculated from 1988-1993 South San Francisco Bay avian survey data (Stenzel et al. 2002) (Table 13). Species 4-letter codes are defined in Table 6; other codes are defined as follows: PHAL = Red-necked Phalarope, Red Phalarope, Wilson's Phalarope; WLDU = Western Sandpiper, Least Sandpiper, Dunlin; YELL = Greater Yellowlegs, Lesser Yellowlegs.

Taxon	Season	A	B	C
AMAV	Fall	0.99	0.88	0.60
BNST	Fall	0.92	0.81	0.57
PHAL	Fall	0.46	0.45	0.42
SEPL	Fall	0.63	0.59	0.48
WILL	Fall	0.54	0.51	0.45
WLDU	Fall	0.65	0.60	0.48
YELL	Fall	0.63	0.59	0.48
AMAV	Spring	0.88	0.79	0.56
BNST	Spring	0.64	0.59	0.48
SEPL	Spring	0.50	0.48	0.43
WILL	Spring	0.47	0.46	0.42
WLDU	Spring	0.60	0.56	0.47
YELL	Spring	0.46	0.44	0.42

Table 27. Predicted tidal flat abundance index.

Predicted year 0 and year 50 abundance indices, and year 50 decrease (year 0 – year 50), for tidal flat shorebirds under each alternative, based on tidal flat projections provided by PWA (Table 12) and tidal flat bird densities calculated from 1988-1993 South San Francisco Bay avian survey data (Stenzel et al. 2002) (Table 13). Species 4-letter codes are defined in Table 6; other codes are defined as follows: PHAL = Red-necked Phalarope, Red Phalarope, Wilson's Phalarope; WLDU = Western Sandpiper, Least Sandpiper, Dunlin; YELL = Greater Yellowlegs, Lesser Yellowlegs.

Species	Alt	Season	Yr 0 Abund	Yr 50 Abund	Yr 50 Decrease
AMAV	A	Fall	4,543	4,520	23
AMAV	A	Spring	761	673	88
AMAV	B	Fall	4,543	3,979	564
AMAV	B	Spring	761	599	162
AMAV	C	Fall	4,543	2,718	1,825
AMAV	C	Spring	761	427	334
BNST	A	Fall	90	83	7
BNST	A	Spring	52	33	19
BNST	B	Fall	90	74	17
BNST	B	Spring	52	31	21
BNST	C	Fall	90	52	39
BNST	C	Spring	52	25	27
PHAL	A	Fall	237	109	128
PHAL	B	Fall	237	106	131
PHAL	C	Fall	237	100	137
SEPL	A	Fall	1,650	1,046	604
SEPL	A	Spring	829	415	414
SEPL	B	Fall	1,650	969	681
SEPL	B	Spring	829	398	431
SEPL	C	Fall	1,650	789	861
SEPL	C	Spring	829	359	470
SNPL	A	Fall	13	6	8
SNPL	A	Spring	16	6	10
SNPL	B	Fall	13	5	8
SNPL	B	Spring	16	6	10
SNPL	C	Fall	13	5	8
SNPL	C	Spring	16	6	10
WLDU	A	Fall	190,790	124,867	65,923
WLDU	A	Spring	631,019	376,808	254,211
WLDU	B	Fall	190,790	115,157	75,633
WLDU	B	Spring	631,019	351,928	279,091
WLDU	C	Fall	190,790	92,500	98,290
WLDU	C	Spring	631,019	293,874	337,144
YELL	A	Fall	158	100	58
YELL	A	Spring	21	10	12
YELL	B	Fall	158	93	65
YELL	B	Spring	21	10	12
YELL	C	Fall	158	75	82
YELL	C	Spring	21	9	12