

Tidal Marsh Bird Population and Habitat Assessment for the San Francisco Estuary Under Future Climate Change Conditions

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SUMMARY

We modeled the spatial distribution of tidal marsh vegetation and bird species in the San Francisco Estuary (Suisun, San Pablo and San Francisco Bays) for the period 2010 to 2110 in relation to projected changes in sea-level rise, salinity, and sediment availability. The species distributions are projected with respect to four future scenarios with assumptions of low or high suspended sediment concentrations and low or high rates of sea-level rise (0.52 or 1.65 m/100 yr). We modeled the presence and absence of 15 tidal marsh plant species and five tidal marsh bird species across the entire Estuary. We used the models of bird species distributions combined with count data to generate population estimates under current and projected future scenarios for five tidal marsh bird species representing a range of federal and state special status listings: Clapper Rail, Black Rail, Common Yellowthroat, Marsh Wren, and Song Sparrow.

Under high sea-level rise and low sediment availability scenarios, high-marsh and mid-marsh habitat is largely replaced by low-marsh, mudflats and subtidal areas by 2110. Under all scenarios, spring and summer salinity were projected to increase throughout the Estuary. Under assumptions of high future sediment availability, tidal marsh habitat is expected to persist and be created in regions with high suspended sediment concentrations (300 mg/L), such as in Northern Marin county, along the Petaluma River and in southern San Francisco Bay, even under high sea-level rise scenarios. However, with less sediment availability, tidal marshes will only be sustainable over 100 years at locations with elevations currently at or above 0.3m above mean higher high water. Tidal marsh restoration efforts will be most successful if implemented during the first half of the 21st century, before rates of sea-level rise are projected to increase. Very little currently undeveloped, un-diked area exists that is predicted to support the migration of future tidal marsh habitat, though four times this area could be reclaimed by removing levees and other barriers to tidal action.

Models of plant species distributions were all heavily influenced by summer salinity and tidal range, but species exhibited individualistic responses to future scenarios depending on their tolerances to physical variables. The predicted probability of occurrence of species such as *Grindelia stricta*, *Schoenoplectus californicus*, *Schoenoplectus maritimus*, *Spartina alterniflora* and *Lepidium latifolium* are all predicted to decline from current levels by 2110 under most future scenarios. At the same time, the probability of occurrence of species such as *Salicornia pacifica* and *Spartina foliosa* are predicted to increase from current levels by 2110 under the high sea-level rise scenario.

The Estuary wide populations of Black Rails and Clapper Rails are predicted to increase from 19,400 and 1200 in 2010 to a mean across all future scenarios of 22,000 ($\pm 12,200$ s.d.) and 1500 (± 400 s.d.) respectively. The Estuary wide populations of Common Yellowthroats and Song Sparrows are predicted to decline under most future scenarios from 30,400 and 183,500 in 2010

to a mean across all future scenarios of 22000 (± 11800 s.d.) and 150,900 ($\pm 81,380$ s.d.) respectively. Under all future scenarios, the Estuary wide population of Marsh Wrens is predicted to decline from 118,800 to a mean across all future scenarios of 64,600 ($\pm 34,800$ s.d.). Uncertainty in future population numbers tends to increase across species in the second half of the 21st century, illustrating the effect of the range of projections from the two sea-level rise scenarios we tested, but also demonstrating that population estimates up to 2050 are relatively insensitive to uncertainty of future sea-level rise.

Projections for future tidal marsh bird populations were sensitive to both sediment and sea-level rise scenarios tested. All bird species except Clapper Rails are projected to experience population declines in high sea-level rise scenarios. Population declines are predicted to be reduced in the high sediment scenarios compared to low sediment; thus maintaining adequate sediment in tidal marsh systems can lead to positive population effects. Future bird population estimates were more sensitive to sea-level rise scenarios than to the sediment scenarios. The predicted responses of species vary among the different regions of the Bay. For example Black Rails are predicted to increase in abundance from current levels between 2010 and 2050 in San Pablo Bay, then decline in abundance from 2050 levels between 2050 and 2110. However, the abundance of Black rails is predicted to remain relatively stable in Suisun Bay from 2010 to 2110. The predicted regional differences in response to sea-level rise necessitate the development of region-specific adaptation strategies for sea-level rise in the Estuary that are flexible enough to deal with future uncertainty exhibited by the range of responses we predicted across future scenarios.

To identify areas most resilient to future sea-level rise, we mapped areas that are consistently projected to be of high value for each bird species individually and for suites of species. Maps ranking the suitability of sites across species and future scenarios can be used to prioritize conservation and restoration efforts. For example, Black Rails were found to have consistently high predicted probability of occurrence around the Petaluma River marsh complex and Coon Island in San Pablo Bay, and near Rush Ranch and Hill Slough in Suisun Bay. The Petaluma marsh complex and areas near the Napa Slough in San Pablo Bay, Point Edith Marsh, Rush Ranch and Montezuma/Grizzly Marsh in Suisun Bay and marshes along Coyote Creek in the San Francisco Bay are all predicted to consistently support high probabilities of occurrence for Black Rails, Common Yellowthroats, Marsh Wrens and Song sparrows across current and all future scenarios. These areas are consistently predicted to provide important bird habitat through time while also remaining resilient to climate change making them high priority sites for continuing conservation efforts.

We have made the species distributions and changes in tidal marsh elevations available in the form of interactive maps and for download as part of a decision support tool, the SF Bay SLR

Tool (www.prbo.org/sfbayslrc). The tool can be used, for example, by conservation planners at fine spatial scales to identify currently upland areas that are projected to be future marsh that could support tidal marsh habitat for plants and birds. The tool can also be used to identify future restoration sites which are projected to be resilient to climate change but are currently behind levees or other barriers to tidal influence. We are conducting outreach to decision makers to facilitate use of the tool for guiding the development of adaptation strategies for future sea-level rise.

INTRODUCTION

Tidal marsh habitat plays a critical ecological function in estuarine ecosystems on a global, national, and regional scale (Greenberg et al. 2006). On the one hand, this habitat supports an array of plant and animal species, many of them Threatened, Endangered or of Special Concern, including many endemic species or subspecies that have evolved adaptations to this saline environment (Greenberg et al. 2006). At the same time, the habitat has been severely altered and degraded, with more than 80% of the historic habitat in the San Francisco Estuary (Suisun, San Pablo and San Francisco Bays; Figure 1) lost since 1800 (Goals Project 1999, Takekawa et al. 2006). In addition to concern about historic changes and current dependency of biota, we must also understand and anticipate the impact of climate change, both changes that are already under-way and future change in order to best guide adaptive conservation and management of this habitat and its ecosystems. Tidal marshes occupy the zone between tidal mud flats and upland areas above the high tide line. As such these marshes are highly sensitive to increases in sea level (Kirwan et al. 2010). In addition to the impacts of sea-level rise on habitat availability, drowning of marshes in the future can have deleterious impacts on ecological function of the marsh ecosystem. Climate change is expected to bring about changes in salinity as well, which will affect plant species distribution and abundance in the marsh habitat. Bird and wildlife populations in turn are dependent on plant species for foraging, nesting, roosting, and as part of the food web supporting primary consumers (e.g., herbivorous invertebrates). Thus, birds and wildlife may be impacted by changes in tidal-marsh-associated plants, reflecting changes in salinity and inundation, as well as by the direct impact of changes in salinity and other physical variables on the animal fauna.

The project reported here is Phase II of a larger, three-phase effort to develop specific information to guide managers at the local and regional level in successfully managing wildlife populations, given current threats and future impacts of climate change. In Phase I, we have carried out a collaborative study to improve understanding of how marsh habitat will change in the future within the San Francisco Estuary. The first phase examined geomorphological change, including whether habitat is expected to be low marsh, mid-marsh, high marsh (Goals Project 1999), mudflats, sub-tidal, or upland, with respect to a series of scenarios designed to address key uncertainties. The uncertainties addressed the wide range in current estimates of sea-level rise (0.2 m to 2 m over 100 years; IPCC 2007, Vermeer and Rahmstorf 2009) and the poor understanding of whether marsh accretion can keep pace with rising sea-levels (Craft et al. 2009, Kirwin and Guntenspergen 2009) by examining different projections for future sediment supply, salinity, and sea-level rise using a dynamic sediment accretion model. Results indicate that tidal marshes can keep pace with high rates of sea-level rise if there is a sufficiently high supply of suspended sediments (Stralberg et al. in preparation). With intermediate to low sediment

availability, marsh sustainability is highly dependent on the rate of sea-level rise (Stralberg et al. in prep.).

The next two phases of the project consider the direct and indirect impacts of climate change on plants and birds in the tidal marsh habitat of the San Francisco Estuary. In Phase II, reported here, we have modeled the anticipated future distribution of key tidal marsh plant species and the distribution and abundance of focal tidal marsh bird populations. This builds directly on the geomorphological modeling framework and results described above (Stralberg et al. in prep.), and considers the same range of scenarios regarding sea-level rise, sediment supply, and salinity changes. Phase III, currently under way, is to consider demographic impacts of climate change, such as impacts of inundation on annual reproductive success or over-winter survival, and builds on the current modeling results regarding distribution and abundance.

In Phase II, we have focused on five tidal marsh bird species: Clapper Rail, Black Rail, Common Yellowthroat, Marsh Wren, and Song Sparrow. Each species is a year-round resident and is dependent on, or strongly associated with, tidal marsh habitat (Goals Project 2000). All but the Marsh Wren are species of high conservation concern. Two species are rails; the California Clapper Rail subspecies (*Rallus longirostris obsoletus*) is a Federally Endangered species and a focus of the USFWS Draft Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California (2009) while the California Black Rail subspecies (*Laterallus jamaicensis coturniculus*), is a California Threatened Species. The other three species are tidal marsh-associated songbirds. The Song Sparrow has three subspecies that are endemic to the San Francisco Estuary tidal marshes (*Melospiza melodia pusillula*, *M. m. samuelis*, and *M. m. maxillaris*; Marshall and Dedrick 1994), while the Common Yellowthroat has one subspecies endemic to the Estuary, the "Salt marsh Common Yellowthroat" (*Geothlypis trichas sinuosa*). Thus, these species are critical to consider in the context of several multi-million dollar restoration projects currently underway or are planned. We have also chosen these five focal species because of extensive Estuary-wide population studies, dating from 1996 (or earlier in the case of the two rails) to the present, conducted by PRBO and partners. Detailed information regarding distribution and abundance of these species is available for the last 5 to 15 years.

Specific Goals:

Our overall goal is to help ensure the conservation of San Francisco Estuary tidal marsh habitats and the birds and other wildlife dependent upon them in the context of sea-level rise, changing salinity, and other climate change related conditions. Phase II of our project (reported here) was designed to (1) assess the potential effects of climate change on tidal marsh habitats and bird populations, (2) identify priority sites for tidal marsh conservation and restoration, (3) develop a web-based mapping tool for managers to interactively display and query results, readily

updatable as more information becomes available, and (4) communicate conservation priorities to management agencies, conservation organizations and the public. This report deals explicitly with goals 1 and 2 above. Goal 3 has been completed and the results from Phase I and II are available online (www.prbo.org/sfbayslr). Goal 4 is ongoing and includes demonstrations and presentations given to government agencies, non-profit organizations and scientific meetings (see list of presentations given in Appendix 1).

To achieve goals 1 and 2 above we developed the following objectives:

- (1) Develop predictive models of focal tidal marsh plant species for current distribution Estuary-wide using available set of physical variables that includes elevation relative to tidal inundation and salinity. We then apply the predictive models, making projections into the future for plant distributions (i.e., presence or absence per 50 m x 50 m pixel) for 2030, 2050, 2070, 2090, and 2110 using four scenarios: low sea-level rise (52 cm over 100 years), moderate-high sea-level rise (165 cm over 100 years); low sediment supply, high sediment supply (Table 1). These incorporate variables that change in the future (elevation relative to tidal inundation; salinity) and those that are not anticipated to change (e.g., proximity to channels or the edge of bay),
- (2) Develop predictive models of focal tidal marsh bird species' current distribution Estuary-wide using available set of physical variables, comparable to that used for plant modeling (see [1]). We then apply these predictive models to the four future scenarios, from 2030 to 2110, as was done for plant species (see [1]). We also develop predictive models of focal tidal marsh bird species regarding abundance. We then sum over expected future habitat, to estimate current and future total population size for the entire Estuary and for the regions within the Estuary (San Francisco Bay, San Pablo Bay, and Suisun Bay).

Table 1. Combinations of future scenarios used for projecting the distributions of tidal marsh habitat, vegetation and tidal marsh birds. The values for the high and low sediment concentrations varied by study sub-regions. The low and high sea-level rise scenarios predict 0.52 m and 1.65m of sea-level rise by 2110, respectively.

| Name | Sediment Concentration (mg/L) | |
|-----------------------|--------------------------------------|-------------|
| | Low | High |
| South Bay | 150 | 300 |
| Redwood City | 50 | 150 |
| Hayward | 50 | 150 |
| San Francisco | 25 | 100 |
| Oakland | 50 | 100 |
| South Marin | 25 | 50 |
| East Bay | 50 | 100 |
| North Marin | 100 | 300 |
| Pinole | 50 | 150 |
| Petaluma River | 150 | 300 |
| San Pablo North Shore | 150 | 300 |
| Napa River | 100 | 150 |
| South Suisun | 150 | 300 |
| SE Suisun | 25 | 100 |
| Suisun Marsh | 25 | 100 |

METHODS

Study area

Our modeling efforts included the entire San Francisco Estuary of suitable current or future elevation for tidal marsh habitat, including Suisun, San Pablo, and San Francisco Bay's (Figure 1). Modeling was restricted exclusively to tidal marsh habitat as we did not have survey data incorporating vegetation or bird distributions in mudflats or upland areas. Vegetation and bird survey sites were located in tidal marshes throughout the region, extending as far upstream as Brown's Island at the western edge of the Sacramento/San Joaquin Delta (Figure 1). Sites, and survey locations within sites, were selected to sample the range of environmental conditions which occur throughout the Estuary in this habitat type.

Biological parameters

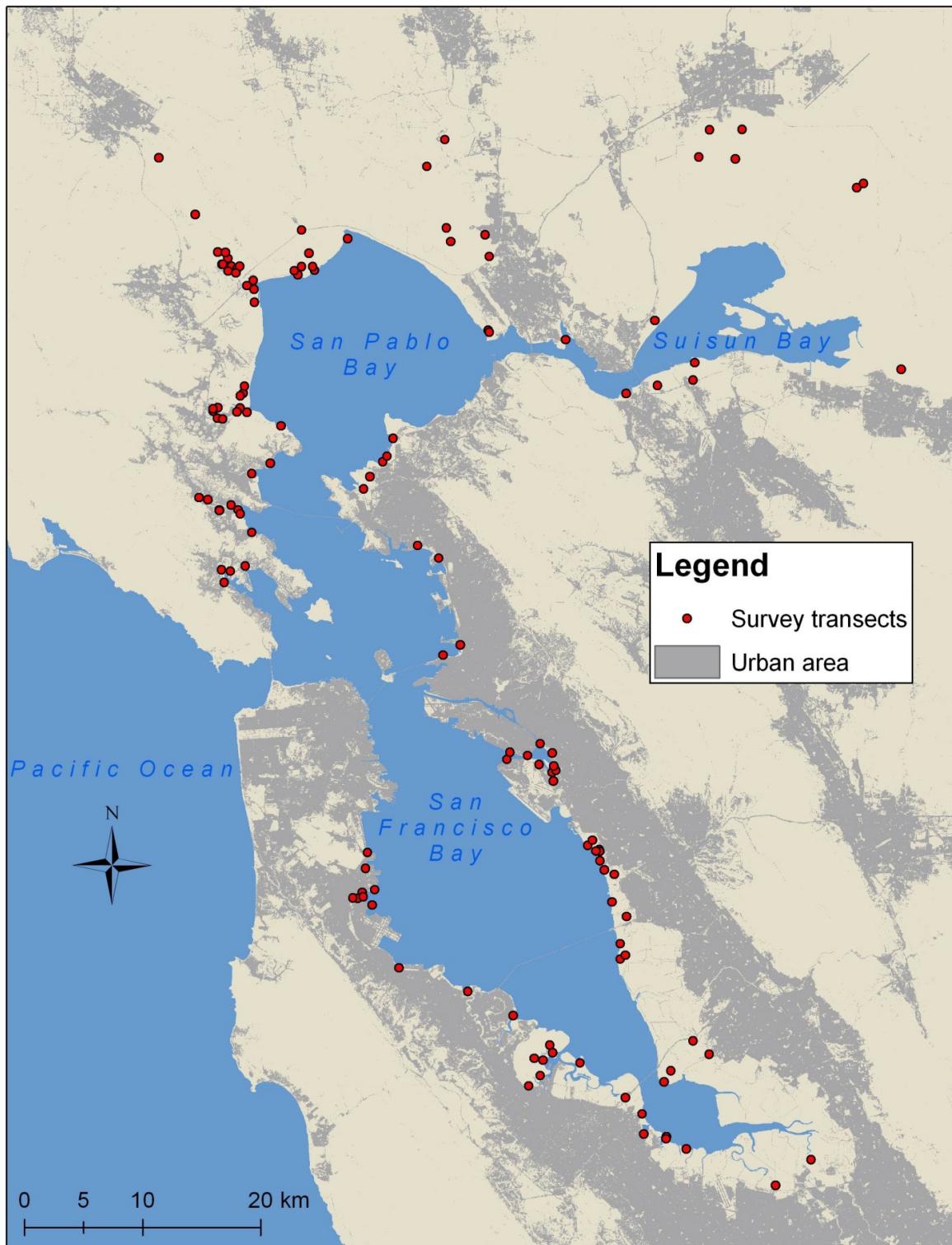
Survey methods

Vegetation was sampled at point count station locations (see below), with mature marshes sampled once every three to four years and younger marshes sampled more frequently (Spautz et al., 2006). Fifteen species were selected for initial modeling based on their prevalence in our surveys and their hypothesized importance in tidal marsh systems. Species modeled were: *Distichlis spicata* (salt grass), *Frankenia salina* (alkali heath), *Grindelia stricta* (gumplant), *Jaumea carnosa* (jaumea), *Juncus* spp. (reed), *Lepidium latifolium* (perennial pepperweed), *Phragmites australis* (common reed), *Salicornia pacifica* (pickleweed), *Schoenoplectus acutus* (tule), *Schoenoplectus americanus* (three-square bulrush), *Schoenoplectus californicus* (California bulrush/tule), *Bolboschoenus maritimus* (alkali bulrush), *Spartina foliosa* (Pacific cordgrass), "Spartina" which specifically refers to the hybrid between non-native *Spartina alterniflora* (smooth cordgrass) and other *Spartina* species, and *Typha* spp. (cattail). Presence/absence was scored as 1 if percent cover was greater than zero, and zero otherwise. Predictive models were developed for all fifteen species. We then chose eight species for mapping of results, based on a combination of predictive performance of the statistical model and our assessment of ecological importance.

Descriptions of the life history in the San Francisco Estuary of each of the five bird species except are described in Spautz et al. (2006), Black Rail, Common Yellowthroat Goals Project (2000), Spautz & Nur (2008a, 2008 b), Chan & Spautz (2008). Common Yellowthroat, Marshall and Dedrick (1994), Black Rail Evens and Nur (2002). and Liu et al. (2009).

Surveys for Clapper Rails required a specialized survey method (Evens 2002); these data were only available from 2005 to 2010. Details of the protocol and data handling are provided in Liu et al. (2009). All six years of survey data were used in the bird modeling to provide a long-term average for "current" distribution. Clapper Rail surveys were conducted at multiple survey locations in each marsh (1 to 21 survey points per site; median =5). Each survey point location ("point") was visited multiple times during the breeding season, from mid-January to mid-April, as per standard protocol.

Figure 1 San Francisco Estuary tidal marsh regions (Suisun, San Pablo, and San Francisco Bays) and survey transects used in the analysis. Each survey transect contained 1 to 21 individual point count locations.



Methods used for Common Yellowthroats, Marsh Wrens and Song Sparrows are described in Spautz et al. (2006) and Stralberg et al. (2010). More detailed information on surveys for these species and the survey methods for Black Rails are available in Liu et al. (2007). For these species, we used the most recent 10 years of survey data, i.e., 2000 to 2009, to provide a long-term average for “current” distribution. Data were from the breeding season (March to June), excluded juveniles, and were within 50m radius of the point count (Spautz et al. 2006). Most points were visited twice per year, rarely three or four times.

Analysis of Bird Survey observations

An important aspect of the Clapper Rail survey data is the large number of zeroes in the dataset, due to absence of the species at a site or low probability of detection during a survey (see “Estimating Absolute Abundance” section below). The Clapper Rail is absent from many sites around the Bay for reasons not captured by the set of covariates in our models. For model fitting, therefore, we only used sites with detections in any survey point and any survey year, thus ensuring that only sites where the rails are known to occur were included in the sample. Thus we did not make predictions for Suisun Bay (i.e., there were no surveys in this area that met the criteria). Filtered this way, the dataset still included points where the species was never detected, and numerous visits to points where no bird was recorded. The filtered dataset (Table 2) included 5,603 records of visits to points that were collapsed into 1,811 point-year records by taking the maximum number of detections at a point each year. That is, the number of point-year records is the sum across all years of all points included in the sample. If tape playback was used in any visit to a point within a year, the record for the point that year was attributed to indicate tape use. We considered the maximum number detected to be a better measure of true abundance than the mean number, due to the difficulty in detecting this secretive species.

For the other four tidal marsh bird species, survey data spanned years 2000 to 2009. The prevalence and probability of detection of these species was adequate for including all records from all sites surveyed, regardless of the history of detections of the species at the site. The summaries of numbers of records and detections, number of points and sites surveyed for each species are shown in Table 2. As with Clapper Rail, data were collapsed into maximum number of detections per point per year.

Black Rails have not been found during our surveys within the San Francisco Bay (but they were present in San Pablo and Suisun Bay’s); they have widely been reported absent as breeders in San Francisco Bay (Evens and Nur 2002). Therefore we developed models without including the surveys for this region and our models are summarized without including these projections. However, we include extrapolations in our maps to illustrate potentially suitable habitat for the species. We note though, that projections in the San Francisco Bay should only be interpreted as suitable habitat and not as the probability of occurrence.

Table 2. Sampling effort, detection probability, and maximum count divided by mean count for bird species included in models..

| Species | #visits to | | | | Detection probability | Max count/Mean count |
|---------------------|------------------|----------------------|---|------|-----------------------|----------------------|
| | #sites (#points) | points (#detections) | #point-year records (#detections/#zeroes) | | | |
| Clapper Rail | 85 (536) | 5,603 (1,694) | 1,811 (1,260/1,183) | 0.38 | NA | |
| Marsh Wren | 51 (560) | 4,070 (5,203) | 2,238 (3,397/1,140) | 0.30 | 0.66 | |
| Common Yellowthroat | 41 (447) | 3,213 (1,478) | 2,238(1,131/1,579) | 0.31 | 0.76 | |
| Song Sparrow | 66 (627) | 4,486 (15,391) | 2,238(9,705/95) | 0.29 | 0.79 | |
| Black Rail | 36 (455) | 3,467 (372) | 1,424(366/1,150) | 0.06 | NA | |

Physical Parameters

Physical variables were initially selected for inclusion in the models based on previous work modeling tidal marsh habitat characteristics (Stralberg et al, 2010) and other published studies (Watson and Byrne, 2009, Table3). We were limited to variables for which data were available throughout the Estuary and which would also be available for modeling future scenarios.

Distance values were calculated using Euclidean distance in ArcGIS 9.3.1 and based on layers found in EcoAtlas (San Francisco Estuary Institute 1998). Mean percent rise (slope) was calculated using the slope function in spatial analyst tool box in ArcGIS 9.3.1 based on the elevation layer for a given scenario. “Marsh elevation relative to MHHW” for these analyses was characterized as the majority or mean of values within the 50 m radius of the center point of each 50 m x 50 m grid cell.

Initial elevation and layers derived from elevation were primarily based on Light Detection and Ranging (LiDAR) remote sensing data contributed by USGS (Knowles, 2010). The LiDAR data had a 5m spatial resolution with a $\pm 2 - 3$ cm vertical accuracy. The NAVD88 vertical datum of the LiDAR data was converted to a MHHW reference level using NOAA tidal gauge and benchmark data. Derived elevation layers such as slope and standard deviation of elevation, were all calculated in ArcGIS 9.3.1 (ESRI, 2009). Future marsh elevation layers were based on models of tidal marsh accretion (see above, Phase I; Stralberg et al. in prep.) and for each future scenario new derived elevation layers (mean slope in a 50 m radius circle, proportion of high, mid and low marsh habitat and marsh standard deviation) were calculated.

We chose two nonlinear sea-level rise scenarios based on the guidance provided by the US Army Corps of Engineers (ACOE, 2009), which recommends scenarios modifying curves proposed by the National Research Council to extrapolate intermediate and high sea-level rise scenarios (“NRC-I” and “NRC-III”, respectively). These scenarios project 0.52 m and 1.65 m of sea-level rise over the next century, with most of this change occurring within the second half of the century.

These rates are similar to recent high-end estimates (Rahmstorf 2007, Vermeer and Rahmstorf 2009), and to the draft State of California planning guidelines, which recommends planning for 0.41 m of rise in the next 50 years and 1.4 m in the next 100 years (Cayan et al. 2009).

Table 3. Abbreviations used and units of measurement for physical parameters included in models. Also indicated (static) is whether the parameter was allowed to vary by time interval in future projections.

| Variable | Abbreviation | Units | Static |
|---------------------------------------|----------------|--------------------------------|--------|
| Spring salinity | sprsalin | Practical Salinity Units (PSU) | No |
| Summer salinity | sumsalin | Practical Salinity Units (PSU) | No |
| Mean marsh elevation | mhhw10mean | Meters (MHHW) | No |
| Majority marsh elevation | mhhw10maj | Meters (MHHW) | No |
| Standard deviation of marsh elevation | mhhwsd | Meters (MHHW) | No |
| Tidal Range | meanhhw | Meters (MHHW) | Yes |
| High-marsh (0.2m to 0.3 m) proportion | mhhwHigh | Proportion | No |
| Mid-marsh (-0.2m to 0.1m) proportion | mhhwmid | Proportion | No |
| Low-marsh (-0.5m to -0.3m) proportion | mhhwlow | Proportion | No |
| Mean slope | slope | Percent Rise | No |
| Percent of area that is channels | channelpercent | Percent | Yes |
| Distance to edge of bay | distbay | Meters | Yes |
| Distance to nearest channel | distchan | Meters | Yes |
| Distance to nearest levee | distlevee | Meters | Yes |
| Distance to nearest urban area | urbdist | Meters | Yes |

Maps of current mean summer salinity were calculated based on a spatial interpolation of salinity observations throughout the bay. Observation at point locations were spatially interpolated to a 50 m x 50 m grid using local polynomial interpolation in ArcGIS 9.3.1. 100-year (2000-2099) salinity projections for 50 bay segments and delta outflow values were obtained from the USGS (Cloern JE et al, 2011). These salinity simulations were run as part of the CASCADE project. We used daily salinity projections based on the GFDL general circulation model for two emissions scenarios: B1 (low) and A2 (high). Because the SLR assumptions used in the salinity simulations were based on IPCC projections, they did not match our more recent estimates from the National Research Council (NRC). Thus we had to adjust the salinity projections to match the NRC projections (NRC-I and NRC-III). To accomplish this, separate regression models were developed for each bay segment and for each emission scenario (B1 and A2), in order to separate the effects of SLR and delta outflows (a function of precipitation) on daily salinity values. Linear regression models were specified as Salinity = SLR x Season + NetDeltaOutflow. Seasons were defined as consecutive three month periods, with spring defined as March, April, and May and summer defined as June, July and August. Regression models were then used to

predict daily salinity for the NRC-I and NRC-III SLR trajectories. The lower-end (NRC-I) SLR projections were based on models developed from lower emission (B1) salinity projections; high-end (NRC-III) SLR projections were based on models developed from higher emission (A2) salinity projections. Daily values were averaged over seasons and 20-year time periods to match our marsh elevation projections. Differences between current and future scenarios were calculated by taking the difference between the current layers and future projections. The differences were then added to the current layer to obtain future salinity projections which retain the spatial patterns of current salinity in the bay.

The effect of salinity on vegetation and bird distributions was hypothesized to be most extreme during the spring and summer seasons. Therefore, only these two seasons were included in the models. For vegetation models (see below), only summer salinity was included; summer salinity was hypothesized to have a more direct influence on vegetation distribution, and spring and summer salinity were highly correlated (Pearson's $r = 0.91$). For bird distribution models, we considered both salinity variables; whichever salinity variable had more relative influence in initial exploratory models was retained for final models. Additionally, bird models included bay region and observation year modeled as "factors." Clapper rail models included a factor for whether a tape playback was used during the survey.

Some variables were assumed to remain unchanged across future scenarios. We assume that the location of levees and urban areas will remain largely unchanged so the distance to levee or urban areas variables remains constant. Because we have no projections for how channels will change in the future or how tidal range will change we make the simplifying assumption that that distance to nearest channel, channel percent and tidal range will not change in the future.

Distribution and Abundance Models

Vegetation

To arrive at a candidate list of models, we initially created generalized additive models (GAMs) (Hastie et al. 2001) with cubic spline smoothers and penalty parameters with 4 degrees of freedom, following the default parameterization the gam package in R statistical software (R Development Core Team, 2010). We evaluated all possible combinations of predictor variables and included interactions between summer salinity and the majority elevation within 50m radius, thus allowing the effect of salinity to vary with location within a marsh. All models were ranked according to the Akaike Information Criterion (AIC) and all of the highest ranked models within 2 AIC units of the top model (model with lowest AIC value) were retained for further analysis (Burnham and Anderson 1998).

We made final species distribution predictions based on the model averaging of a bootstrapped ensemble of all retained candidate models to avoid over-fitting to a single model (Hastie et al.

2001). Ensembles were created by running 50 boot-strap iterations of each selected model per species to further avoid over-fitting (Efron, 1983). All iterations used a hold-out (20%) of the survey data to calibrate the model, with the remaining 80% split at random into 75% out-of-bag and 5% in-bag samples in each iteration. Predictions from each iteration and selected model were made for the entire San Francisco Estuary. A final ensemble prediction was made by taking the weighted average of all selected model iterations, where weights were based on deviance from cross-validation of each model in the ensemble against the hold-out set. Model predictions represented the weighted average probability of occurrence across all selected models and bootstraps.

Birds

Tidal marsh birds were modeled using boosted regression trees (BRT) (Elith et al. 2008) BRTs have been shown to be less prone to over-fitting than standard classification and regression trees and have better predictive performance than other statistical algorithms (Elith and Graham 2009). They also have the benefit of implicitly incorporating interactions into predictions and fit non-linear responses. We used BRTs rather than GAMs because we hypothesized that the relationships between bird occurrence and physical variables would be more complex than is the case for vegetation and, if so, BRTs are more efficient at selecting more complex models from a high dimensional space than GAMs.

BRTs require the specification of three important parameters which determine the complexity of fitted models: tree complexity, learning rate and number of trees. Fitting the BRT model requires the appropriate choice of all three parameters to maximize its predictive ability while reducing the likelihood of over-fitting on the training dataset (Elith et al. 2008). For each species, we tested all possible combinations of models with tree complexities of 1 to 5 and learning rates of 0.01, 0.005, and 0.001. The optimal combination of parameters was selected based on the predictive deviance using a 10 fold cross-validation of the models and the final number of trees fit. More trees with smaller learning rates tend to lead to more complex models. Therefore, optimal models were those which had the lowest predictive deviance with the smallest number of trees and with the highest learning rate.

We modeled predicted presence/absence in the surveyed area using BRTs with binomial link function fit to the entire dataset. We then predicted the probability of occurrence of the species in tidally influenced areas around the San Francisco Estuary using the predictions from these models. Areas which are not currently vegetated but are going to be affected by sea-level rise are included in all model years including the present so current predictions may be unrealistic in some areas such as young restoration sites that have the physical characteristics of suitable habitat but do not yet support mature vegetation. We also created binary presence/absence maps using a threshold value such that each cell was assigned either 0 (absent) or 1 (present),

depending on whether it exceeded the threshold value. For Clapper Rails we chose a low threshold value (0.1) due to the low predicted probably of occurrence throughout the Estuary. For all other species the threshold used was the species prevalence in the survey dataset (Liu et. al, 2005).

The accuracy of distribution models under current conditions was assessed by calculating the area under the receiver operating characteristic curve (AUC); AUC ranges from 0 to 1 and measures the ability of a model to discriminate presence from absence across all possible thresholds of the predicted probability of occurrence. An AUC value of 1 indicates that a model has perfect discrimination, while an AUC of 0.5 means a model discriminates presence from absence no better than random. $AUC < 0.5$ means that a model discriminates presence from absence worse than random. The sensitivity of each species to future scenarios was evaluated by calculating the mean probability of occurrence throughout the bay under each future scenario.

The goal of our study was to evaluate the sensitivity of species distribution to different future scenarios, not to test hypotheses about the correlations between biophysical variables and species occurrence or abundance. Even so, we were interested to know the relative influence of each variable in the final model. The algorithm we use estimates this relative influence by taking the number of times a variable was chosen for a split in a tree, weighted by the squared improvement to the model attributed to the split and then averaged across all trees (Elith et al. 2008) so that the influence of all variables in the models sums to 100%. Additionally, we generated partial response curves for each predictor variable in presence absence models by making predictions from the binomial boosted regression tree model to each environmental variable while holding all other variables at their mean value.

The number of detections per survey per ha was used as a measure of relative abundance (Nur et al. 1999). This variable was modeled using the log transformed maximum counts at a survey point from each survey year, with a Gaussian distribution of residuals. We then used the predicted presence/absence layer as a mask, such that abundance was predicted only for cells for which presence was predicted for that cell (i.e., above the threshold value). Thus, abundance was predicted conditional on the BRT predicting that the species was present in that cell. The total abundance of the Estuary was estimated by adding the predicted abundance across all tidal marsh cells in the San Francisco Estuary.

Analysis of observations of individual detected during avian surveys can be used to predict relative abundance and presence/absence for each species. However, in order to predict true, absolute abundance we need to estimate the probability that a species was present during a survey but was not detected (Buckland et al 1993; Thomas et al 2009, Royle et al. 2005). To do so we discriminated between true zeroes (points where the species was never detected because

it was absent) and apparent zeroes (points where the species was present but was not detected). To correct for apparent zeroes, that is, to account for probability of detection, we developed a detection correction factor. For Clapper Rails, we used the maximum counts per three visits to a point each year to fit a simple mixture model that includes a probability of detection parameter and a single count (Poisson) parameter (Royle et al. 2005). The estimated probability of detection is with respect to three visits to a point. Detection at two visits out of three implies failure to detect the individual at one of the visits, though it was presumed present. For Marsh Wren, Song Sparrow, Black Rail and Common Yellowthroat, there were enough detections in single visits, to fit a “mixture” model that estimated the probability of detection for a single visit for these species. In this analysis, zero detections is considered a mixture of true zeroes (species absent) and false zeroes (species present, but not detected). A single individual detected could mean that there was only one individual present or that more than one individual was present but only one was detected. We used the inverse of detection probably as our correction factor.

To estimate total absolute abundance for a species, we scaled the predicted relative abundance per surveyed area to fit the area of the prediction cell (50 m x 50 m). The estimate was then corrected for the estimated probability of detection for the species (see above). Finally, for the three songbird species, we adjusted maximum detections per visit to take into account over-counting because of double counting of individuals, and counting of transients. Detections of the two rail species were rare, and so no adjustment was necessary. For Song Sparrows, Common Yellowthroats, and Marsh Wrens, we determined the ratio of maximum counts per visit to mean counts per visit, and corrected the maximum count by the ratio of mean to maximum counts per visit (Table 2).

Priority areas

Areas that are consistently projected to be highly suitable for tidal marsh species represent a first order filter for prioritization of areas for conservation concern. For the tidal marsh birds we studied, our models identify species-specific locations that are insensitive to the variation in future environmental projections. Locations that retain high predicted probabilities of occurrence could prove vital for allowing tidal marsh bird species to seek refuge while other areas evolve in response to climate change. Areas that are predicted to support high abundance can also be designated as priority areas. We summarize projected results with respect to 21 scenario-time predictions: one prediction for the present (2010) and five time-specific predictions (2030, 2050, 2070, 2090, 2110) for each of the four scenarios modeled (low vs. high SLR; low vs. high sediment supply).

Maps of potential conservation priority for each species were generated by using a threshold (based on prevalence in the survey dataset) to convert the predicted values from binomial BRT

models to binary presence/absence maps (0 or 1 values) for each scenario-time prediction. The 21 resulting maps were then summed to create a new map with cell values ranging from 0 to 21 with 0 indicating pixels in which no maps predict the cell to be occupied by the birds in any scenario and 21 indicating the pixel is predicted to be occupied in all time-scenarios. We also generated a conservation priority map across all species, except Clapper Rail¹, in which the binary maps from all species were summed together with values ranging from 0 to 84. This final map highlights important areas for all tidal marsh bird species.

¹ Clapper rails were not included in the combined map because the maps for this species represent a different scale of occurrence, 100m vs. 50m radius circle.

RESULTS

Physical variables models

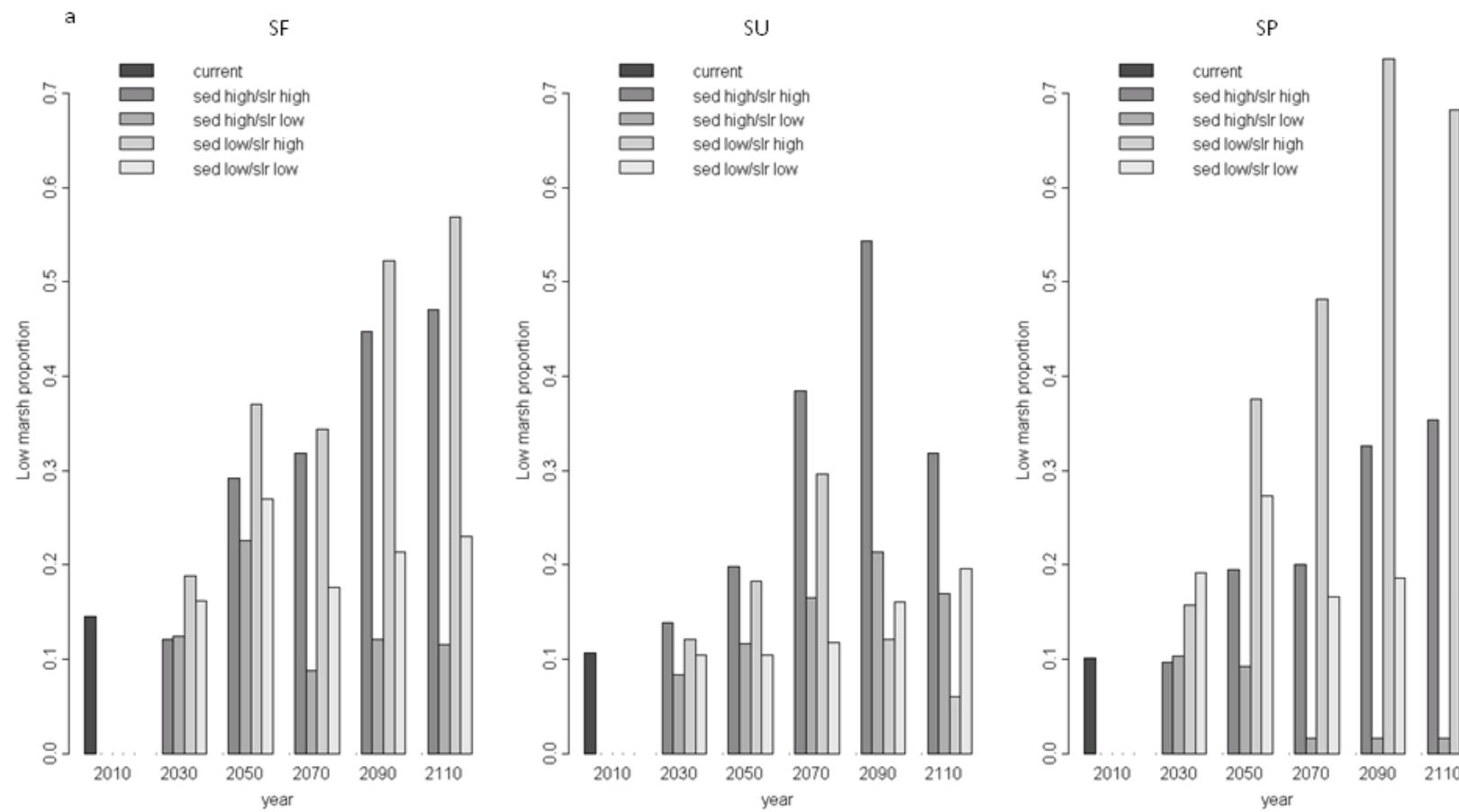
Detailed results from tidal marsh elevation modeling is provided in Stralberg et al., (in preparation). Here we provide brief summaries of the elevation and salinity projections to aid in the interpretation of vegetation and bird distribution modeling. We report on the changes of elevation at the scale of our vegetation and plant survey areas (50 m radius circle). In particular, we focus attention on changes in high marsh (0.2m to 0.3m with respect to MHHW), mid marsh (-0.2m to 0.1m) and low marsh (-0.5m to -0.3m).

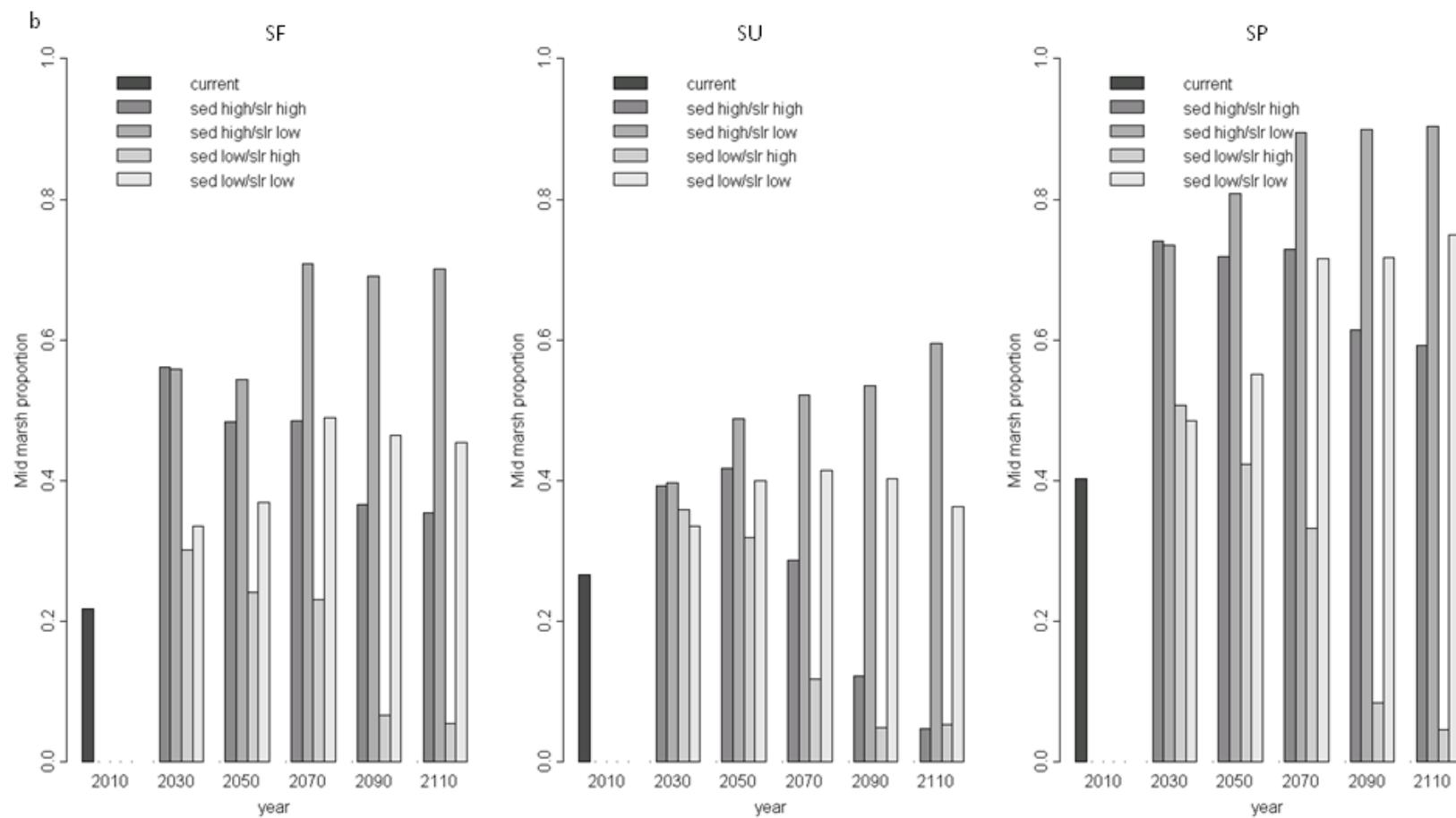
Our models project that tidal marsh habitat will have a dynamic response to sea-level rise. Under low rates of sea-level rise marsh accretion allows marshes to keep pace with increasing sea levels, particularly in areas with sufficient suspended sediment concentrations. However with high rates of sea-level rise combined with low suspended sediment concentrations, large proportions of tidal marsh are converted either to mudflats or subtidal areas.

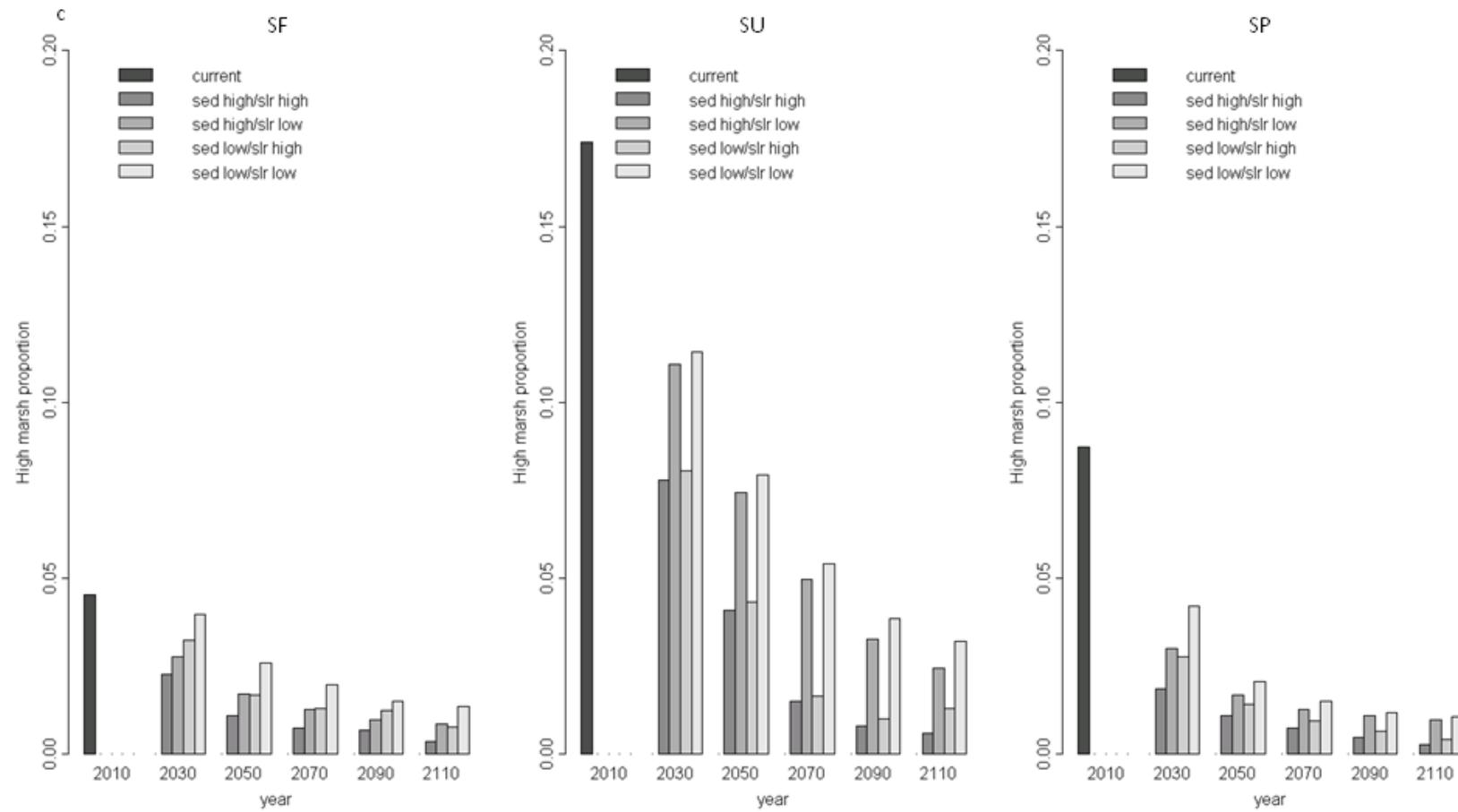
The mean amount of low marsh habitat occurring in the San Francisco Estuary is projected to increase from current levels for all scenarios except for the high sediment/low SLR rise scenario (Figure 2a). Currently low marsh habitat comprises, on average, about 10% of the area we sampled (Figure 2a). For the high sea-level rise scenarios, low-marsh habitat is projected to cover on average between 40-55% of the area of point counts in San Francisco Bay by 2090 and over 70% of the area in San Pablo Bay for the low sediment/high sea-level rise scenario for 2090 (Figure 2a). This increase in the proportion of survey areas simulated to be low marsh habitat in the future goes well beyond the range of what we observe in the San Francisco Estuary today, challenging our vegetation and bird distribution models to extrapolate projections to novel future conditions.

Future projections for mid-marsh habitat vary greatly with respect to each of the scenarios we tested. For all scenarios, mid-marsh habitat increases by 2030 across all bay regions (Figure 2b). The increase in mid-marsh habitat is attributed to marsh accretion which is able to outpace the increases of sea level from either sea-level rise scenario. However, as the rate of sea-level rise increases for both scenarios towards the middle of the 21st century, the amount of mid-marsh habitat available declines as accretion is projected to no longer be able to

Figure 2. Current and projected future mean low-marsh (a) mid-marsh (b) and high-marsh (c) for each 20 year period between 2010 and 2110 within San Francisco Bay (SF), San Pablo Bay (SP) and Suisun Bay (SU). Future projections are given for each combination of high/low sediment (sed) and high/low sea-level rise (SLR) scenario combinations





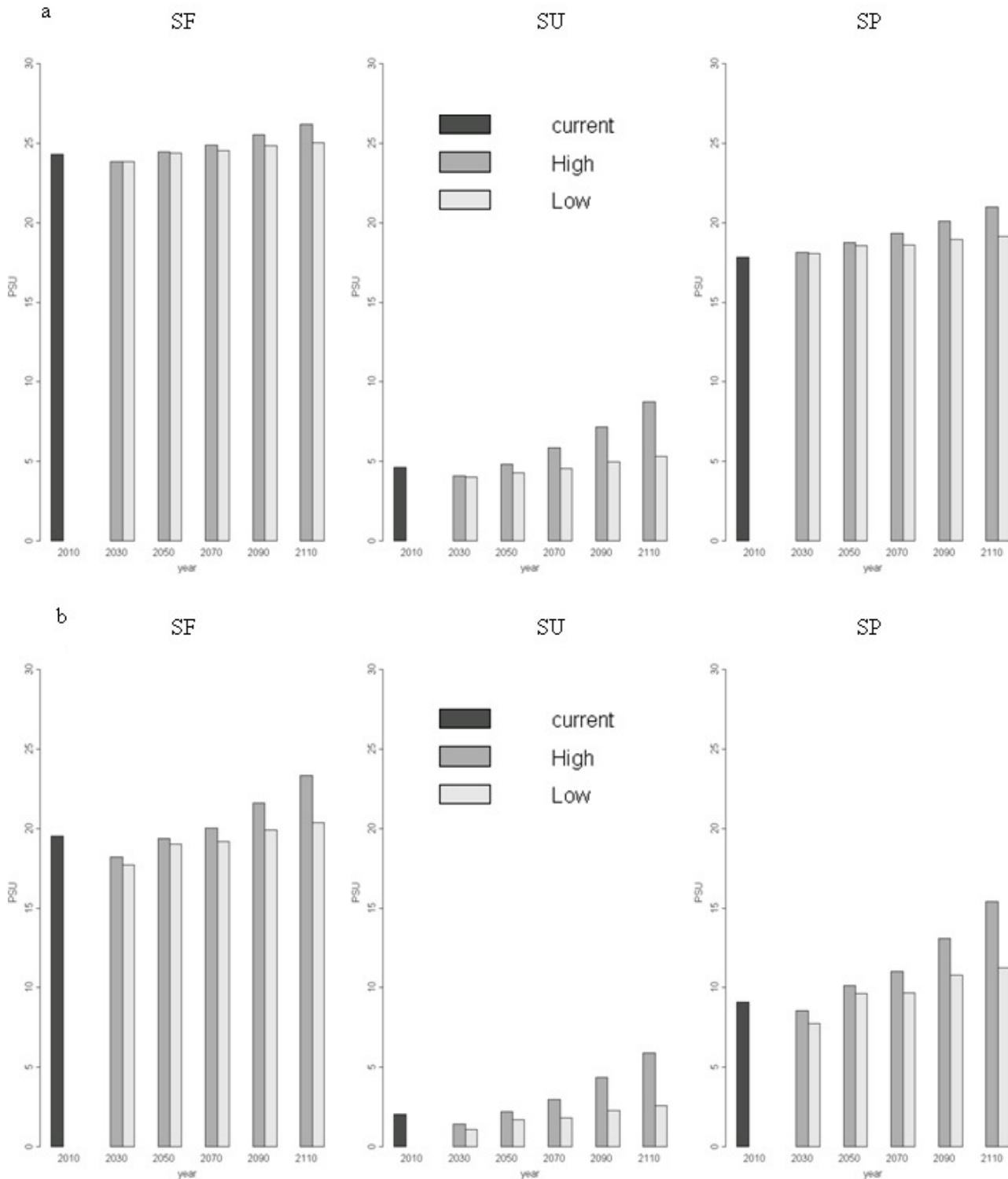


keep pace with increasing sea levels, especially for the low sediment scenarios. By 2090, the proportion of mid marsh habitat available is much lower under the high sea-level rise scenarios than for the low sea-level rise scenarios (Figure 2b).

Under all scenarios, high marsh habitat is projected to decline throughout the San Francisco Estuary by 2110, with larger declines projected for San Pablo and Suisun Bays compared to San Francisco Bay. Although the declines in high marsh habitat are greatest for the high sea-level rise scenarios, there is not as much variation across scenarios as there is for the mid and low marsh projections. However, high marsh may be under-represented under low sea-level rise and/or high sediment scenario projections. This is because the Marsh98 model does not take into account geomorphic processes within the marsh plain that produce heterogeneity within a marsh. Stochastic events such as extreme tides or storms may deposit material on the marsh plain creating high marsh but the Marsh98 model does not simulate these events.

In general, salinity is projected to increase throughout the San Francisco Bay Estuary. However, the future salinity projections for 2030 are lower than or equal to current levels in both summer and spring (Figure 3) except for summer in San Pablo Bay (Figure 3a). The lower than present salinities projected for 2030 are likely artifacts from the conversion of salinity projections from the IPCC derived sea-level rise scenarios to match the NRC sea-level rise scenarios. Salinity increases from current levels more for the high sea-level rise scenario than the low sea-level rise scenario.

Figure 3. Current (2010) and projected future mean summer (a) and spring (b) salinity for each 20 year period between 2010 and 2110 within San Francisco Bay (SF), San Pablo Bay (SP) and Suisun Bay (SU). Future projections are given for high and low sea-level rise (SLR) scenario combinations.



Distribution models

Projections for all species modeled and tidal marsh habitat under current and future scenarios are available for viewing online (www.prbo.org/sfbaysl).

Vegetation

The number of top models retained varied between 1 and 14 by species (Appendix 2). Almost all of the top models included summer salinity. Distribution models had good to excellent predictive accuracy across all species, with AUC ranging between 0.78 and 0.98 (Table 4.). Nine of the fifteen species modeled had AUC > 0.90 (excellent discrimination of presence from absence).

Our models project that plant species will have individual species-specific responses to sea-level rise (Figure 4). For example, *Grindelia stricta* and *Salicornia pacifica* increase in their mean probability of occurrence from 2070 forward under the high sea-level rise scenario, but decrease under the low sea-level rise scenarios. Under the high sea-level rise - low sediment scenario, the probability of occurrence of *S. pacifica* increases across much of the remaining tidal marsh habitat in San Pablo and Suisun Bays from 2050 -2110 (Figure 5 a-e). Under the low sea-level rise low sediment scenario, *S. pacifica* declines in the probability of occurrence, particularly in Suisun Bay where marsh accretion is relatively high under this scenario (Figure 5 f-j). On the other hand, *Schoenoplectus californicus* and *Spartina alterniflora* have the same directional responses to the different sea-level rise scenarios, with some variation in the magnitude of the response (Figure 4e and 4g).

Some species experience overall declines from current mean probability of occurrence in most future scenarios. *Grindelia stricta*, *Schoenoplectus californicus*, *Schoenoplectus maritimus*, *Spartina alterniflora* and *Lepidium latifolium* all decline in the probability of occurrence throughout the Estuary under most future scenarios. At the same time, species such as *Salicornia pacifica* and *Spartina foliosa* maintain or increase their current mean probabilities of occurrence under most future scenarios, particularly the high sea-level rise scenarios.

The probability of occurrence of the two exotic tidal marsh species we modeled, *Spartina alterniflora* and *Lepidium latifolium* both decline from current levels between 2030 and 2050 (Figures 4g and 4h). For *S. alterniflora*, the models project that the mean probability of occurrence throughout the Estuary will remain below current levels through 2110 but *L. latifolium* could see a return to the current mean probability of occurrence towards the end of the century, particularly under high sea-level rise scenarios (Figure 4h). In 2110, in Suisun Bay, under the low sea-level rise/low sediment scenarios, areas projected to have a high probability of occurrence for *L. latifolium* occur primarily in marsh habitat closer to the bay with interior marshes projected to have low to moderate probability of occurrence as opposed to the high probabilities of occurrence predicted, especially along marshes in the western part of the bay

under current conditions (Figure 6a and 6b). Projections from the model onto the high sea-level rise low sediment scenario show a dramatic decrease in the overall availability of tidal marsh habitat within Suisun Bay, but what remains is primarily projected to have high probability of *L.latifolium* occurrence (Figure 6c).

Table 4. Predictive model accuracy for tidal marsh vegetation species as measured by the area under the receiving operator characteristic curve (AUC). AUC ranges from 0 to 1. An AUC of 1 indicates a model which perfectly discriminate species presence from absence and 0.5 means a model discriminates presence from absence no better than random. The most influential variable in the model for each species is also shown.

| Species | AUC | Most influential variable |
|------------------------------------|------|---------------------------|
| <i>Distichlis spicata</i> | 0.79 | Tidal range |
| <i>Frankenia salina</i> | 0.87 | Summer salinity |
| <i>Grindelia stricta</i> | 0.78 | Summer salinity |
| <i>Jaumea carnosa</i> | 0.81 | Summer salinity |
| <i>Juncus</i> spp. | 0.98 | Tidal range |
| <i>Lepidium latifolium</i> | 0.84 | Summer salinity |
| <i>Phragmites australis</i> | 0.97 | Summer salinity |
| <i>Sarconia pacifica</i> | 0.97 | Summer salinity |
| <i>Schoenoplectus acutus</i> | 0.96 | Summer salinity |
| <i>Schoenoplectus americanus</i> | 0.96 | Tidal range |
| <i>Schoenoplectus californicus</i> | 0.95 | Summer salinity |
| <i>Bolboschoenus maritimus</i> | 0.94 | Summer salinity |
| <i>Spartina alterniflora</i> | 0.97 | Tidal range |
| <i>Spartina foliosa</i> | 0.85 | Summer salinity |
| <i>Typha</i> spp. | 0.97 | Summer salinity |

Figure 4. Mean predicted probability of occurrence of vegetation species across the San Francisco Estuary under current (2010) and future sea-level rise scenarios. The sea-level rise scenarios project a 0.52m and 1.65m increase in sea-level (“low” and “high” respectively) by 2110. The variation at each time period is from the different sediment scenarios and the error bars indicate ± 1 standard error. Selected native species shown are *Grindelia stricta* (a), *Salicornia pacifica* (b), *Schoenoplectus acutus* (c), *Schoenoplectus maritimus* (d), *Schoenoplectus californicus* (e), and *Spartina foliosa* (f). Exotic species shown are *Spartina alterniflora* (g) and *Lepidium latifolium* (h).

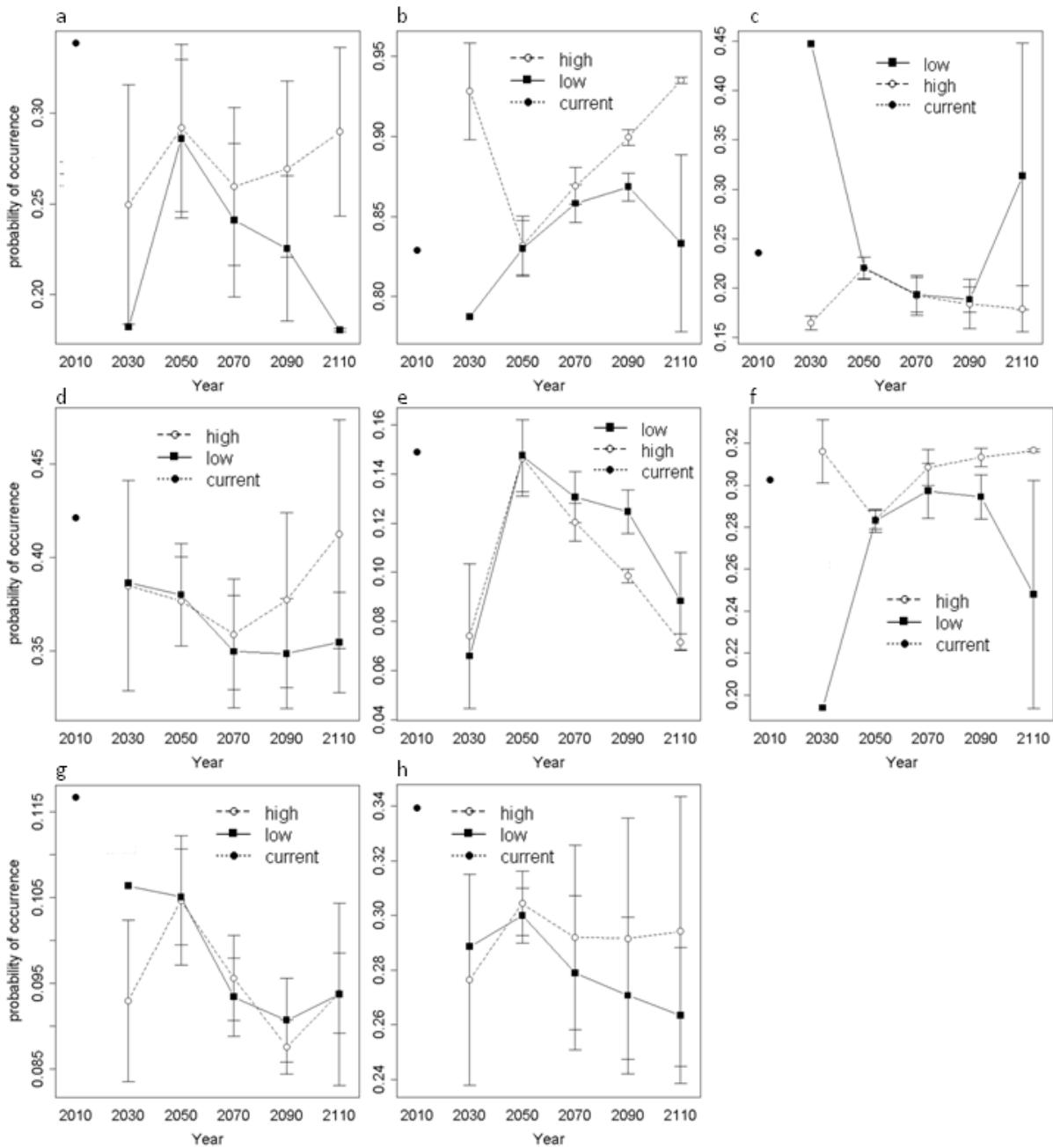


Figure 5. The predicted probability of occurrence for *Salicornia pacifica* under the low sediment/ high sea-level rise scenario (a-e) and low sediment/low sea-level rise scenario (f-j) for 2030 (a,f), 2050 (b, g) 2070 (c,h), 2090 (d,i) and 2110 (e, j).

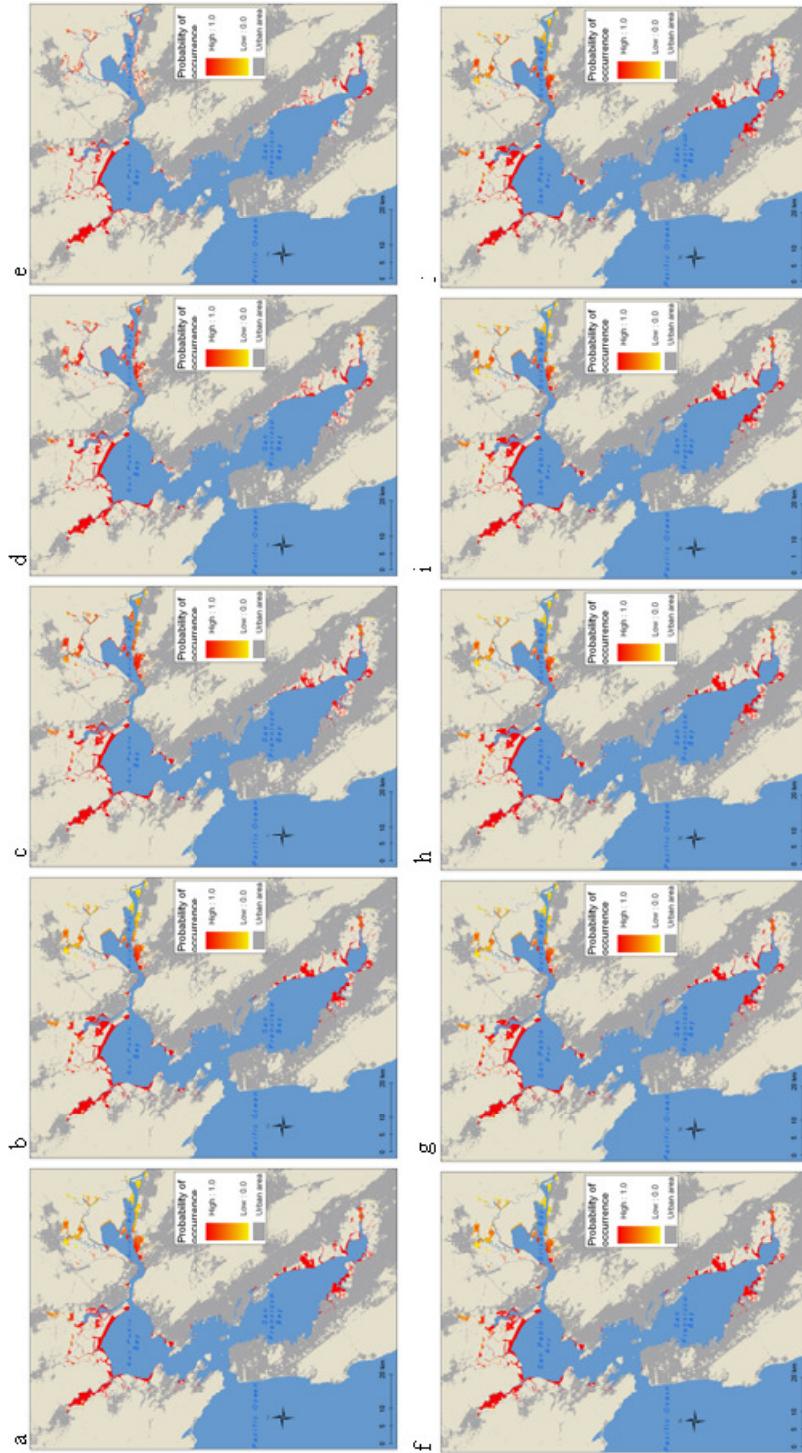
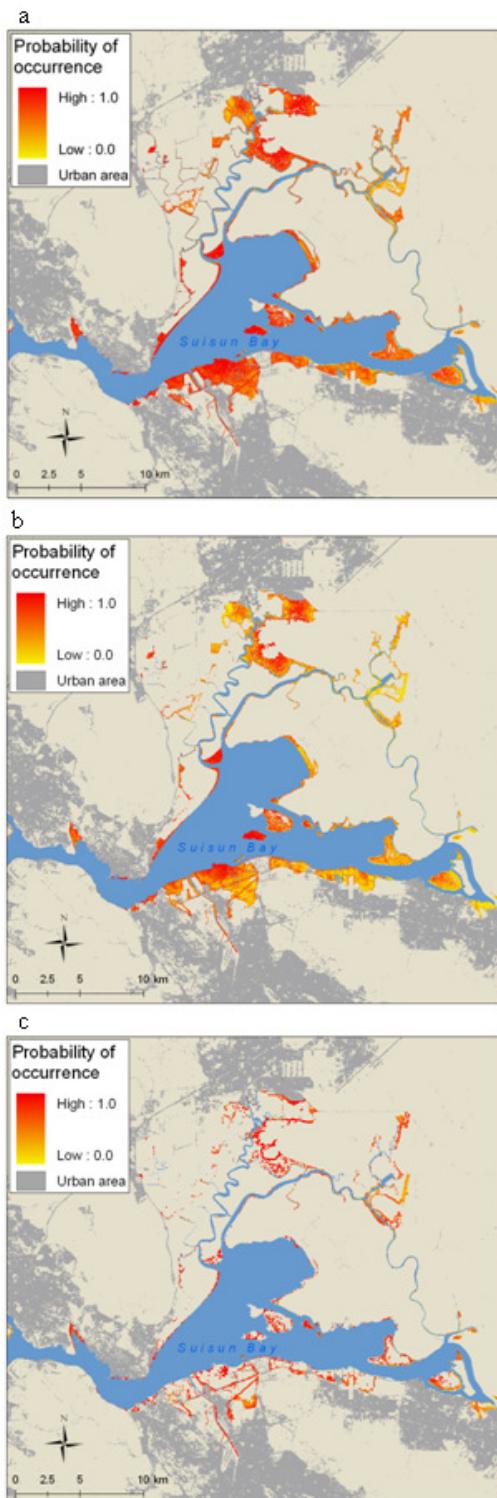


Figure 6. The predicted probability of occurrence for *Lepidium latifolium* under current conditions (a), low sea-level rise/ low sediments scenario 2110 (b) and high sea-level rise low sediment scenario 2110 (c).



Birds – current distributions and abundance

Clapper Rail

Occurrence models for Clapper Rail had a cross validated AUC of 0.73 (S.E. \pm 0.01). Tidal range was the most important predictor variable (19.2% relative influence, Table 3a.) with higher probability of occurrences found in areas with low tidal range (Figure 7a). Year was the second most influential predictor (15.1 % relative influence, Table 7a) with higher probabilities of occurrence in 2005-2007. Clapper Rails are predicted to be more likely to occur in areas with higher spring salinities (10.05 % relative influence, Figure 7b). The models also found a negative correlation between the mean elevation within a 50m radius and the probability of Clapper Rail occurrence (Figure 7c). Areas with the highest probability of occurrence of Clapper Rails were found in the North Bay; especially around China Camp State Park and near Steamboat Slough (Figure 8).

The cross-validated correlation between the observed and predicted density of Clapper Rails was 0.50 (S.E. \pm 0.05). The relative importance of predictor variables in the abundance models was similar to the binomial models. The main difference is that mean elevation had an increased relative importance (14.1%) in the abundance model and is more important than spring salinity (Table 5b). We estimate there are 1200 Clapper Rails currently in the Estuary and that San Francisco Bay has slightly more Clapper Rails than San Pablo Bay (700 vs. 500; Table 6).

Table 5. The percent relative influence of each predictor variable for each binomial model (a) and each abundance model (b) for Black Rails (BLRA), Clapper Rails (CLRA), Common Yellowthroats (COYE), Marsh Wren (MAWR), and Song Sparrows (SOSP). The influence is scaled to add to 100% across all variables. The most important variable for each species is indicated in bold.

| A | Variable | BLRA | CLRA | COYE | MAWR | SOSP |
|---|-----------------------------|-------------|-------------|-------------|-------------|-------------|
| | Bay region | 0.0 | 0.0 | 0.1 | 0.4 | 0.0 |
| | Percent Channel | 2.7 | 1.5 | 1.2 | 1.7 | 1.7 |
| | Distance to bay | 16.9 | 8.7 | 10.6 | 6.4 | 11.7 |
| | Distance to nearest channel | 9.1 | 4.6 | 0.8 | 2.5 | 10.0 |
| | Distance to nearest levee | 2.4 | 5.3 | 1.3 | 4.1 | 2.4 |
| | Tidal range | 9.9 | 19.2 | 10.6 | 21.5 | 2.6 |
| | Mean elevation | 3.3 | 7.5 | 2.4 | 3.2 | 27.2 |
| | High marsh proportion | 1.7 | 4.6 | 2.6 | 2.7 | 0.5 |
| | Low marsh proportion | 6.6 | 5.1 | 2.2 | 2.3 | 2.0 |
| | Mid marsh proportion | 1.7 | 7.0 | 2.6 | 4.6 | 4.1 |
| | Elevation variability | 5.3 | 3.2 | 1.6 | 2.4 | 6.3 |
| | Salinity | 4.0 | 10.1 | 52.4 | 27.0 | 26.8 |
| | Mean slope | 6.5 | 4.8 | 3.7 | 6.0 | 1.0 |
| | Tape | NA | 0.1 | NA | NA | NA |
| | Distance to urban areas | 7.8 | 3.3 | 2.4 | 3.7 | 0.4 |
| | Year (factor) | 22.0 | 15.1 | 5.5 | 11.5 | 3.5 |
| B | Variable | BLRA | CLRA | COYE | MAWR | SOSP |
| | Bay region | 0.0 | 0.0 | 0.2 | 2.5 | 1.9 |
| | Percent Channel | 25.3 | 0.7 | 1.2 | 1.9 | 1.7 |
| | Distance to bay | 12.0 | 5.1 | 10.4 | 7.3 | 4.7 |
| | Distance to nearest channel | 9.9 | 3.3 | 1.3 | 1.9 | 6.7 |
| | Distance to nearest levee | 8.1 | 3.1 | 4.3 | 3.5 | 7.9 |
| | Tidal range | 7.4 | 20.6 | 20.7 | 18.8 | 8.8 |
| | Mean elevation | 7.2 | 14.1 | 3.3 | 4.0 | 7.8 |
| | High marsh proportion | 5.9 | 4.3 | 2.5 | 2.5 | 2.6 |
| | Low marsh proportion | 5.5 | 5.0 | 2.0 | 2.4 | 4.6 |
| | Mid marsh proportion | 4.2 | 3.5 | 1.9 | 3.4 | 8.6 |
| | Elevation variability | 3.6 | 4.0 | 1.6 | 2.0 | 5.7 |
| | Salinity | 3.4 | 11.7 | 38.8 | 32.5 | 19.5 |
| | Mean slope | 3.1 | 2.9 | 3.2 | 4.3 | 3.7 |
| | Tape | NA | 0.1 | NA | NA | NA |
| | Distance to urban areas | 2.2 | 1.6 | 2.8 | 3.1 | 3.1 |
| | Year (factor) | 2.1 | 20.0 | 5.8 | 9.9 | 3.1 |

Figure 7. Modeled relationship between Clapper Rail presence/absence and summer salinity (a), tidal range (b) and mean elevation (c) while holding all other variables at their mean values; results from binomial boosted regression tree. The predicted response is scaled to have a mean of 0.

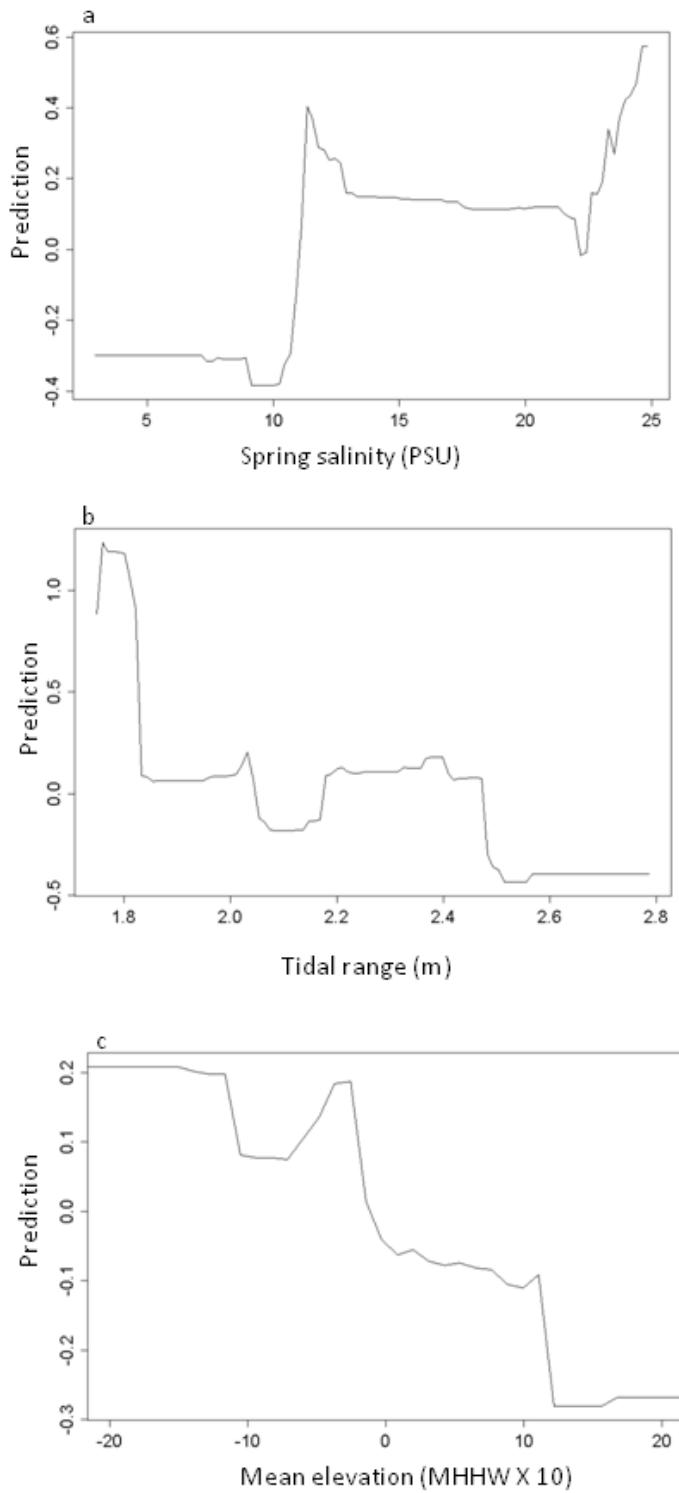
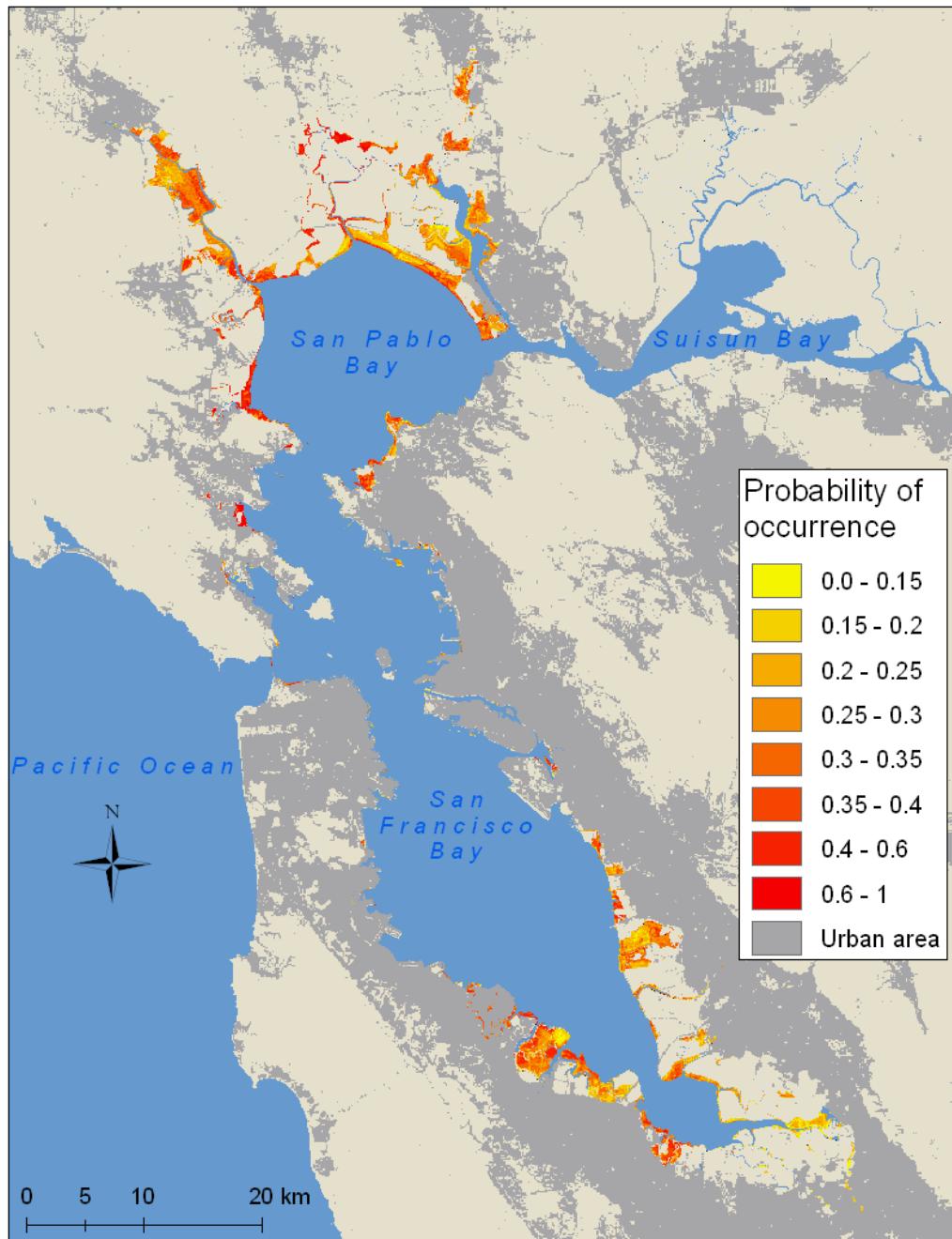


Figure 8. Predicted probability of occurrence of Clapper Rails under current (2010) conditions throughout San Francisco Bay and San Pablo Bay. Predictions were not made in Suisun Bay.



Black Rail

The occurrence models for Black Rails had the lowest predictive accuracy of any of the bird species modeled with a cross validated AUC of 0.64 (S.E. \pm 0.10). Year had the greatest influence on the model with considerable year to year variation in predicted probability of occurrence (22.0% relative influence, Table 5a). Black Rails showed a positive response to increasing distance from the bay, with peak estimated probability of occurrence (controlling for all other variables) occurring between six and ten kilometers from the bay (Figure 9). The distance to urban areas had greater influence on Black Rail predicted response as compared to other focal species (7.8% relative influence, Table 5a). Tidal range (9.9% relative influence) and distance to channel were also influential with Black Rail occurrence predicted to be more likely in areas with a lower tidal range and an increasing distance to channels. Marshes along the Petaluma River, including the ancient Petaluma marsh, marshes in the Sonoma/Napa creek area, especially Coon Island, and Suisun marsh area, including Rush Ranch are all predicted to have high probability of occurrence (Figure 10).

The predicted response of Black Rail occurrence to the proportion of low marsh was somewhat unexpected. The BRT predicted that Black Rails would have maximum probability of occurrence in areas with between 10% to 15% of low marsh habitat (Figure 9d). As expected, in areas with between 20% to 60% low marsh habitat, the model predicts a very low probability of occurrence. Above the 60% low marsh proportion, the model predicts an increasing probability of occurrence, which is unexpected based on expert knowledge. There are very few areas within the San Francisco Estuary currently with low marsh proportion greater than 70% and the increasing positive response predicted by the model is based on a few, highly influential records. Black Rails are thought to be more associated with mid to high marsh habitat and we suspect that the models are over predicting the probability of occurrence in areas with a high proportion of low marsh. However, due to the low prevalence of areas with high low marsh proportion within the San Francisco Estuary today, it is difficult to validate the model within this region of environmental space.

Black Rail abundance models had the same top five most influential variables as the binomial models, although the rankings within the top five differed slightly (Table 3b). The cross validated correlation between observed and predicted density was 0.18 (S.E. \pm 0.02). The models predict that 19400 Black Rails occur in the Estuary (Table 6). The population is split with 10,600 predicted to occur in the San Pablo Bay and 8,800 predicted to occur in Suisun (Table 6).

Figure 9. Modeled relationship between Black Rail presence/absence and distance to bay (a), tidal range (b) distance to nearest channel (c) and low marsh proportion (d) while holding all other variables at their mean values; results from binomial boosted regression tree. The predicted response is scaled to have a mean of 0.

Black Rail

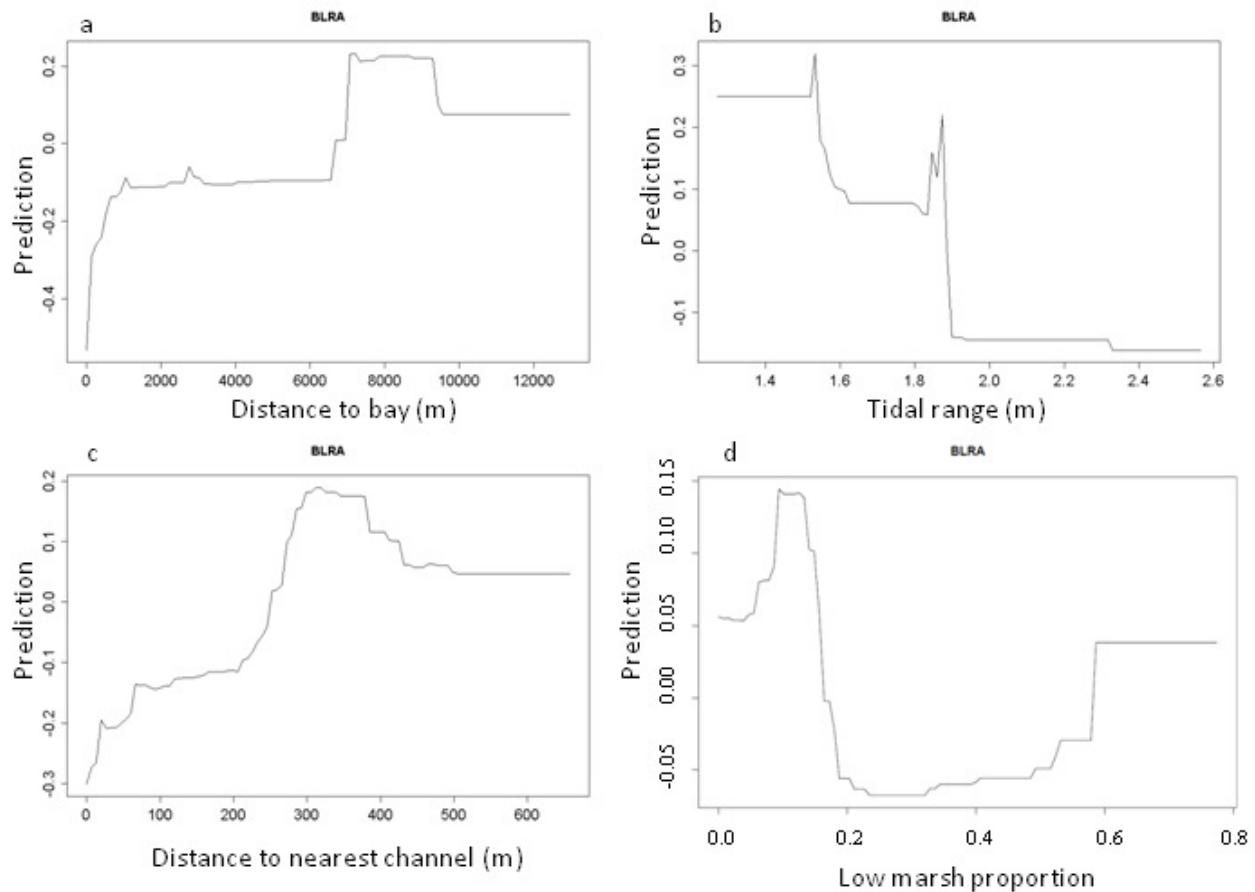


Figure 10. Predicted probability of occurrence of Black Rails under current (2010) conditions throughout Suisun Bay and San Pablo Bay. Map values were extrapolated to the San Francisco Bay and should be interpreted as the potential environmental suitability, not as a probability of occurrence.

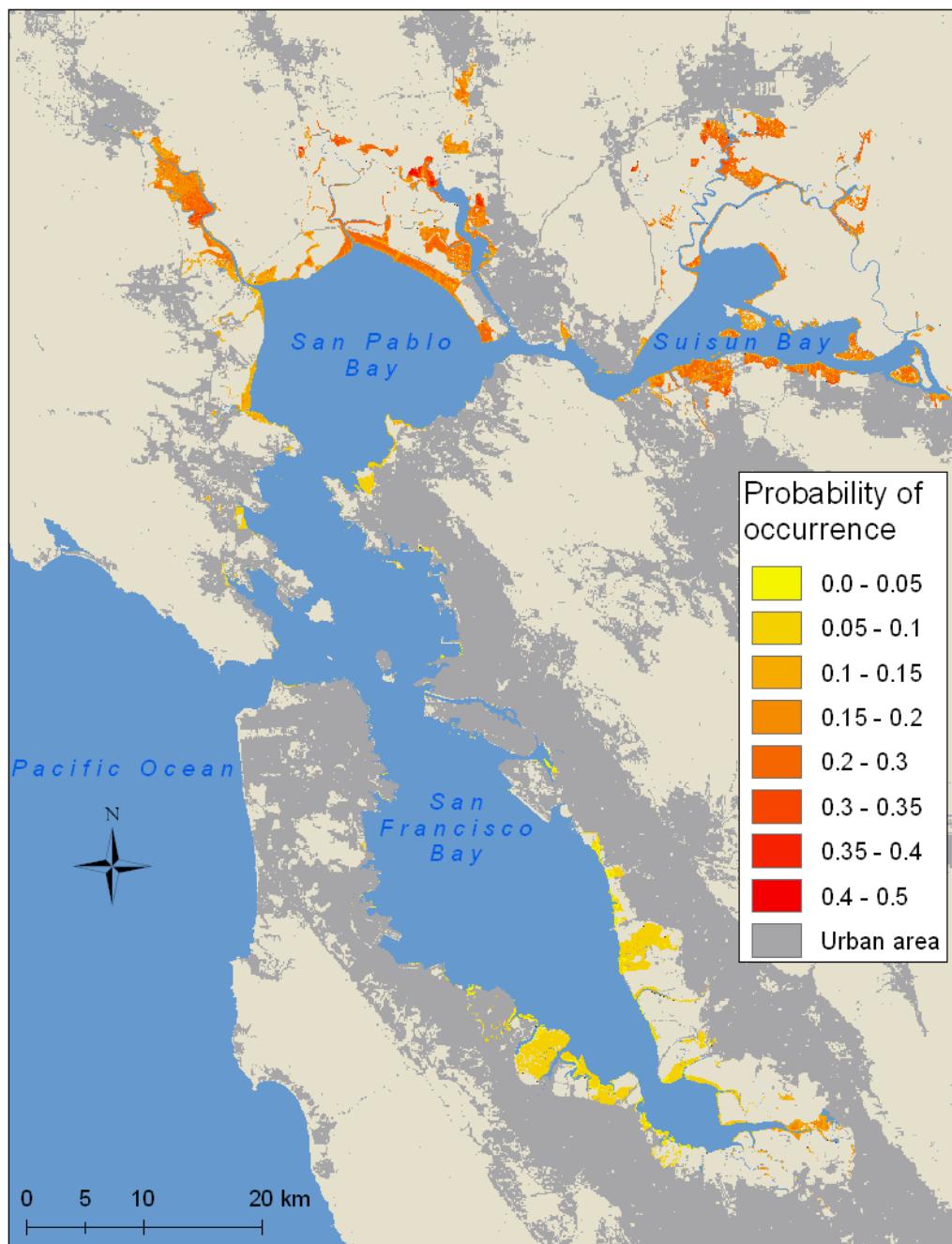


Table 6. Estimated current (2010) abundance for the five tidal marsh focal birds studied. Abundance is estimated for the entire San Francisco Estuary (Total), and bay sub-regions; San Francisco Bay (SF), Suisun Bay (SU) and San Pablo Bay (SP). The lowest (2110 Min) and highest (2110 Max) predicted abundance for the total Estuary at 2110 are also given.

| Species | SF | SU | SP | Total | 2110(Min) | 2110(Max) |
|---------------------|--------|--------|---------|---------|-----------|-----------|
| Black Rail | NA | 8,800 | 10,600 | 19,400 | 7000 | 3600 |
| Common Yellowthroat | 1,200 | 23,400 | 5,800 | 30,400 | 9000 | 35900 |
| Marsh Wren | 9,100 | 69,200 | 40,500 | 118,800 | 24,500 | 104,300 |
| Song Sparrow | 45,000 | 38,100 | 100,400 | 183,500 | 43,900 | 237,300 |
| Clapper rail | 700 | NA | 600 | 1,200 | 1200 | 2100 |

Common Yellowthroat

The predicted occurrence of Common Yellowthroats was strongly influenced by summer salinity (52.4% relative influence, Table 5a) with higher probabilities of occurrence predicted as salinity decreases. The strong influence of salinity resulted in a pronounced regional gradient in the predicted distribution of Common Yellowthroats. High probabilities of occurrence occur throughout Suisun Bay and Suisun Marsh which has lower salinity than other bay regions (Figures 11-12). Common Yellowthroats were also predicted to have higher probabilities of occurrence beyond 2 km from the bay. As a result, low probabilities of Common Yellowthroat occurrence were predicted in marshes along the bay edge throughout San Pablo and San Francisco Bays, but with moderately high probability of occurrence in the Napa River, including Pond 2A (Figure 12). Overall the models had high predictive ability with cross-validated AUC = 0.93 (S.E. \pm 0.01).

As with the binomial models, summer salinity was the most influential variable in the abundance models, but summer salinity was less influential overall (Table 5b). The rankings of the top four most influential variables did not change but higher predicted densities were more clearly associated with areas with low tidal range. The correlation between observed and predicted densities of 0.78 indicates that the abundance models also had relatively high accuracy. The models predicted a total Estuary population of 30,400 Common Yellowthroats, with Suisun having more than four times as many predicted Common Yellowthroats as San Pablo Bay, and over 20 times as many as San Francisco Bay (Table 6) .

Figure 11. Modeled relationship between Common Yellowthroat presence/absence and summer salinity (a) and distance to bay (b) while holding all other variables at their mean values; results from binomial boosted regression tree. The predicted response is scaled to have a mean of 0. Common Yellowthroat

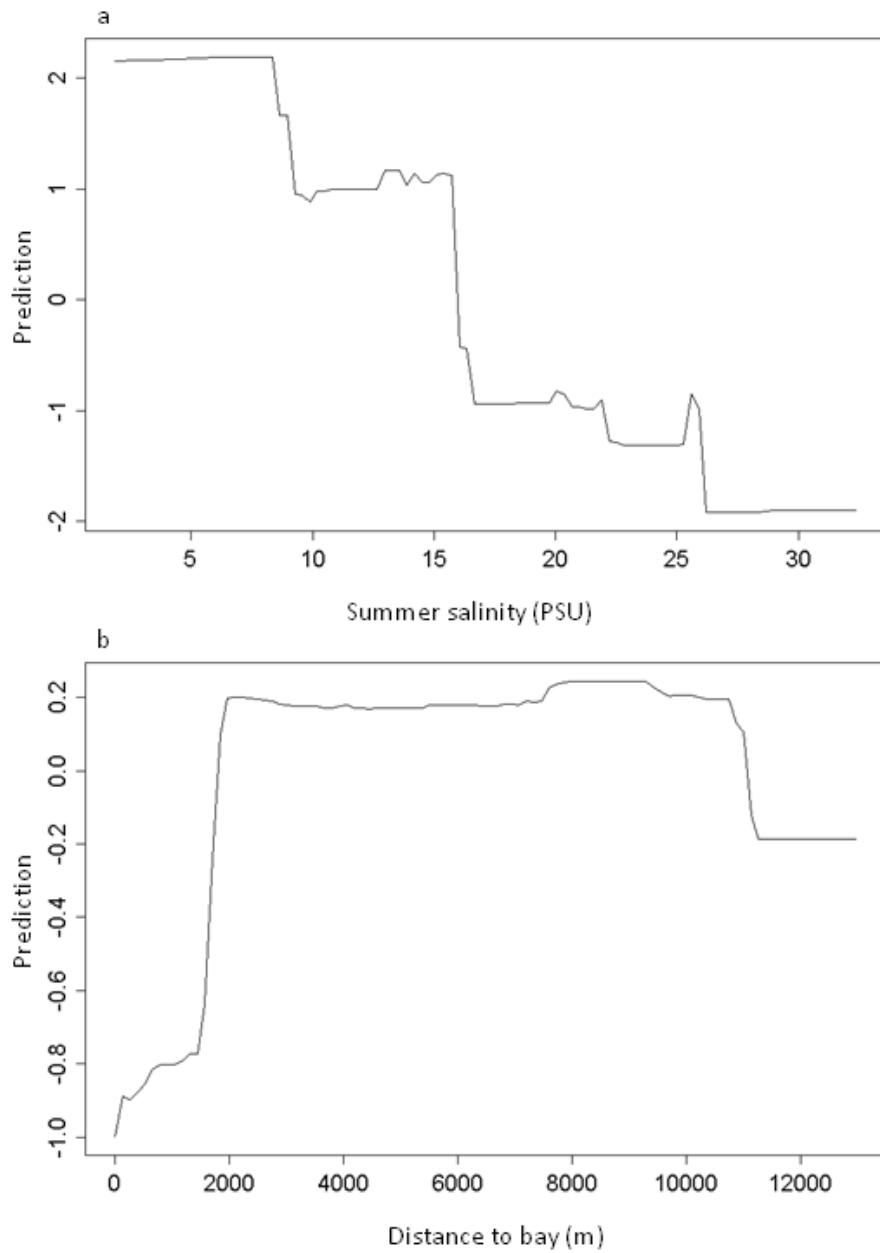
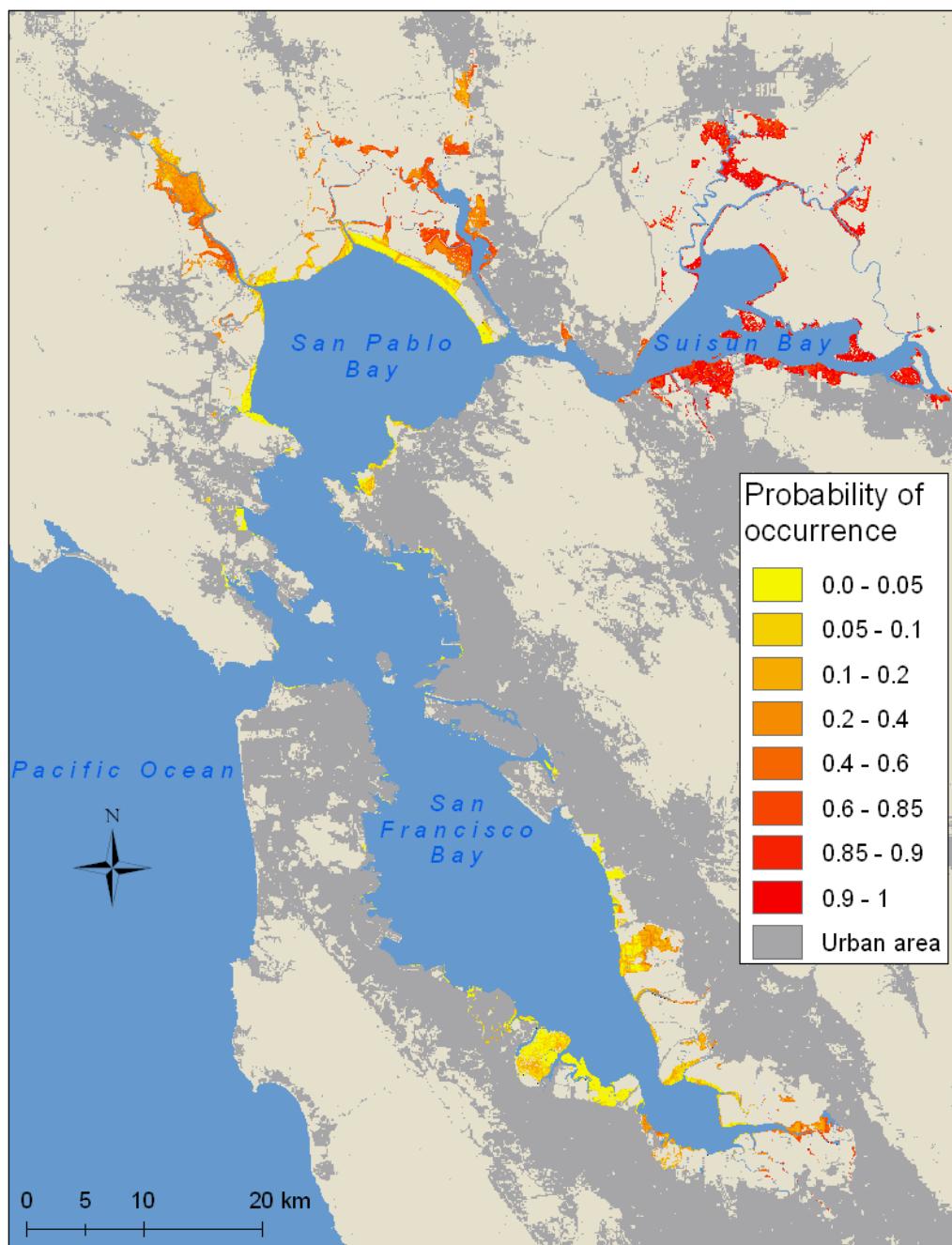


Figure 12. Predicted probability of occurrence of Common Yellowthroats under current (2010) conditions throughout San Francisco Estuary.



Marsh Wren

The predicted distribution of Marsh Wrens was also strongly influenced by summer salinity (Table 5a) with occurrence predicted to be less likely as summer salinity increased. Models also predict that Marsh Wrens are more likely to occur in areas with moderate tidal range (Figure 13). Marsh wren models had excellent predictive accuracy (AUC 0.94, SE \pm 0.01). Marsh wrens are predicted to occur in the high probability throughout Suisun and in the upper reaches of Napa River and Petaluma River in San Pablo; probabilities in San Francisco Bay are low except for Alviso marshes (Figure 14).

The rankings of the top five most influential variables did not change between the binomial and abundance models for Marsh Wrens (Table 5). Abundance models for Marsh Wrens had higher predictive accuracy than any other bird species ($R = 0.87$ between observed and predicted densities, S.E. \pm 0.004). Our models predict 118,800 marsh wrens to occur throughout the Estuary. The population in Suisun Bay is predicted to be 60% greater than in San Pablo Bay and seven-fold greater than in San Francisco Bay (Table 6).

Figure 13. Modeled relationship between Marsh Wren presence/absence and summer salinity (a) and tidal range (b) while holding all other variables at their mean values; results from binomial boosted regression tree. The predicted response is scaled to have a mean of 0.

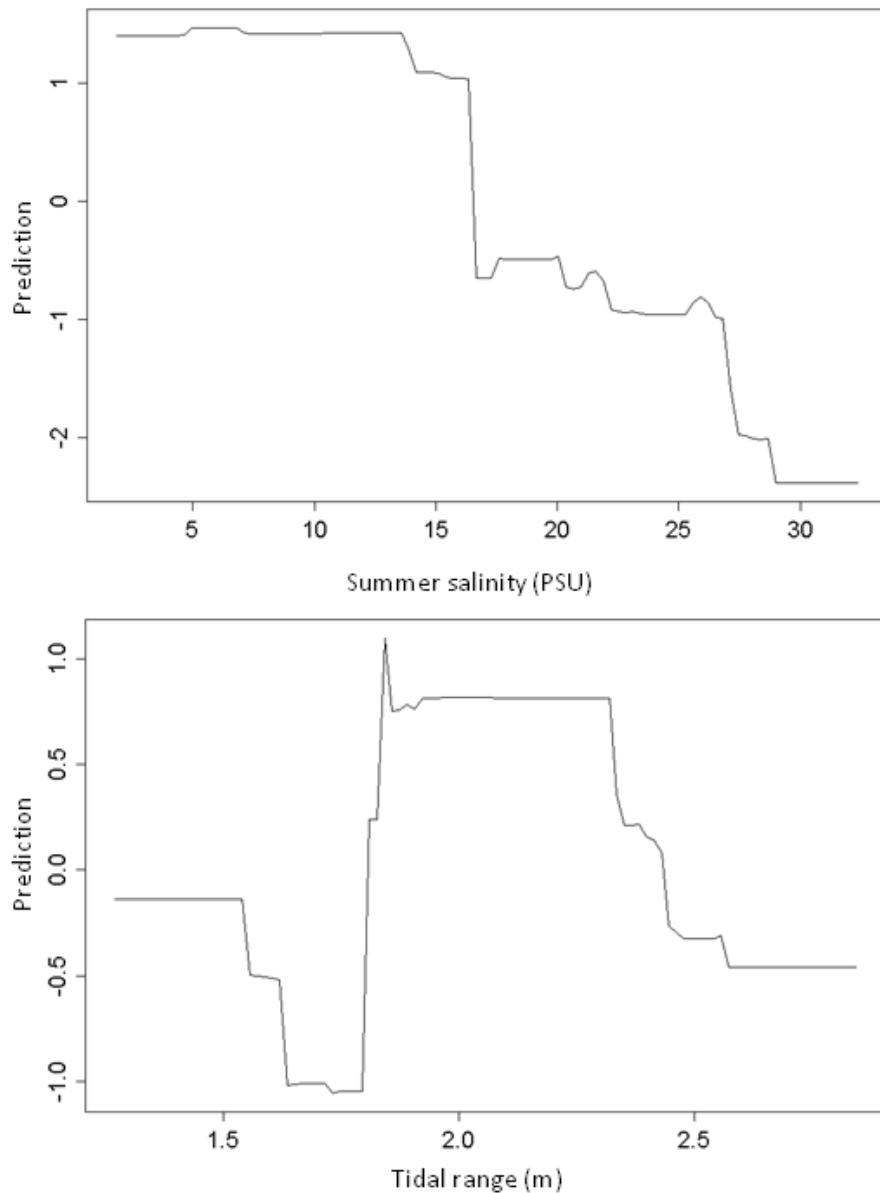
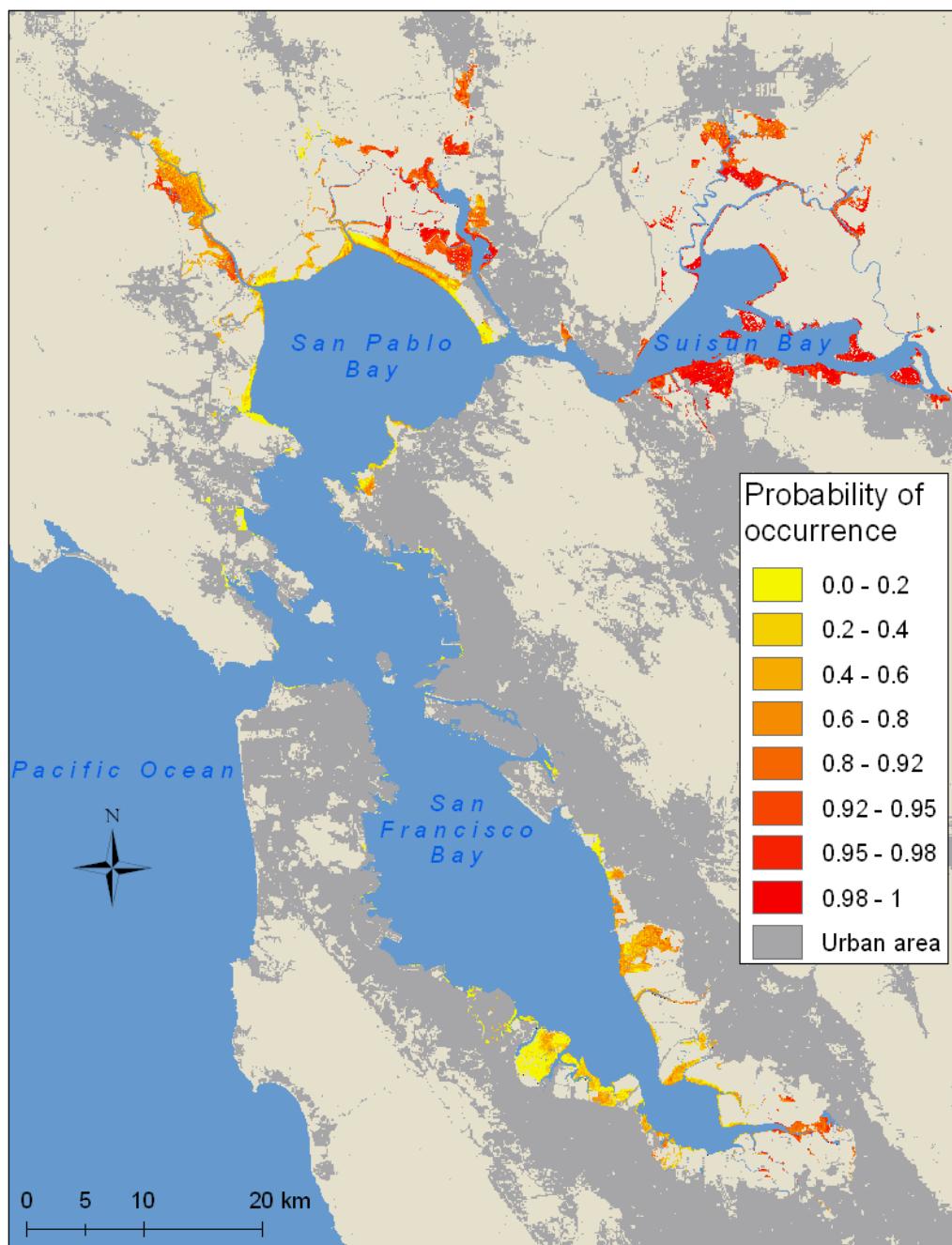


Figure 14. Predicted probability of occurrence of Marsh Wrens under current (2010) conditions throughout San Francisco Estuary.



Song Sparrow

Song sparrows are relatively common throughout the Estuary and occurred in approximately 96% of our surveys. Unsurprisingly, the binomial models predicted high probabilities of occurrence throughout the Estuary (mean probability of occurrence throughout the bay = 0.98, Figure 15). The binomial model for Song Sparrow has good accuracy at discriminating presence from absence (AUC = 0.84, S.E. 0.02). Mean tidal marsh elevation had the greatest influence on the predicted probability of occurrence of Song Sparrows with higher probabilities of occurrence predicted in areas with mid to high marsh elevations (Table 3a). The models predicted lower probabilities of occurrence in areas with high spring salinity, which was the second most influential variable in the model (Table 5a). Song sparrows are predicted to occur with high probability in most of the Estuary. However, in the most up river areas, e.g., in Napa River, Hayward marsh/Cogswell marsh, Alviso marsh, and Middle Bair Island, Song Sparrows are predicted to have a low probability of occurrence.

Mean elevation was less influential in the abundance models for Song Sparrow (Table 5a) while year was the second most influential, highlighting the amount of year to year variation in abundance for this species. Both mean tidal marsh elevation and proportion of mid marsh were moderately influential on the abundance model and predicted higher Song Sparrow densities in areas with moderate amounts of mid-marsh habitat. Song sparrow densities were predicted accurately ($R = 0.73$, observed vs. predicted densities). While the binomial models predicted little regional differences in the probability of occurrence, the abundance model found some heterogeneity in abundance in space (Table 6). The abundance of Song Sparrows across the Estuary in 2010 is predicted to be 183,500. The model predicts the highest Song Sparrow abundance in San Pablo Bay (over 100,000), with more than twice as many birds than are predicted to occur in either San Francisco Bay or Suisun Bay (Figure 16).

Figure 15. Modeled relationship between Song Sparrow presence/absence (a & b) or abundance (c) and mean elevation within 50m (a), spring salinity (b), and the proportion of mid-marsh habitat (c) while holding all other variables at their mean values; results from binomial boosted regression tree. The predicted response is scaled to have a mean of 0. The predicted response in all plots is scaled to have a mean of 0.

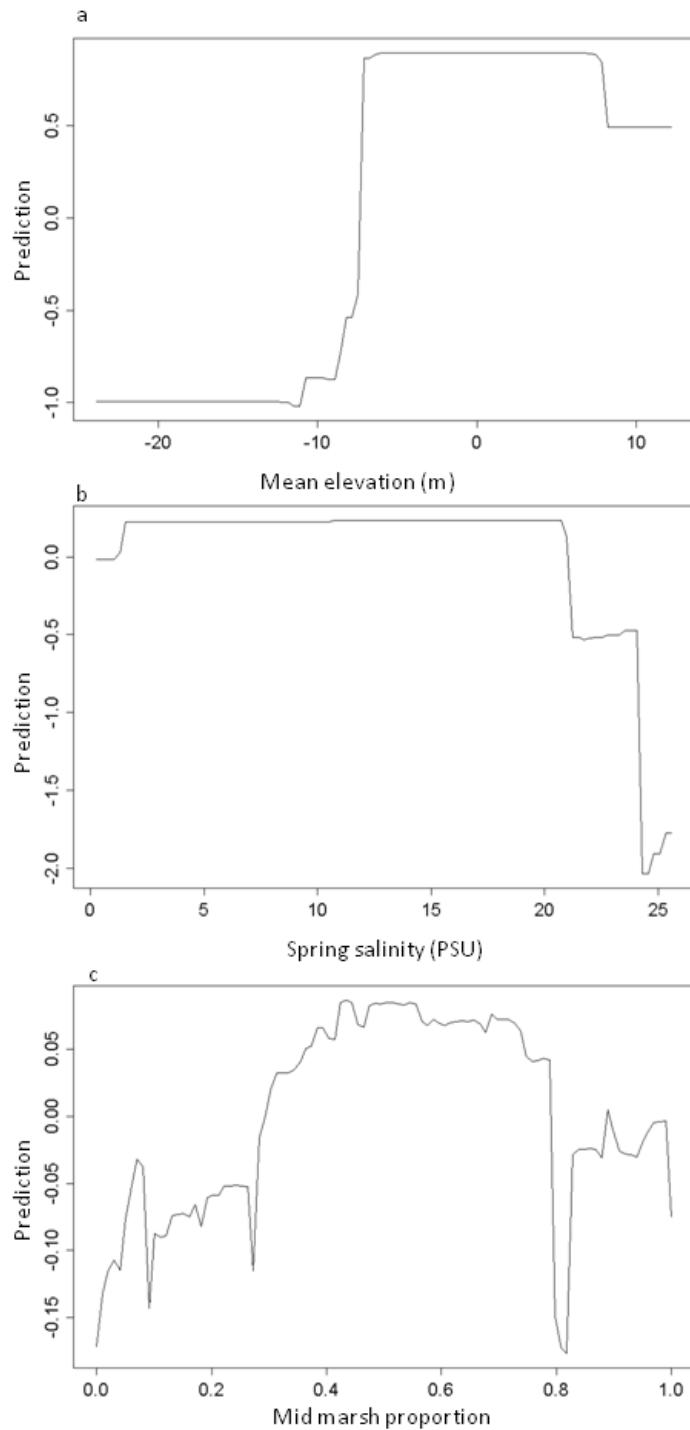
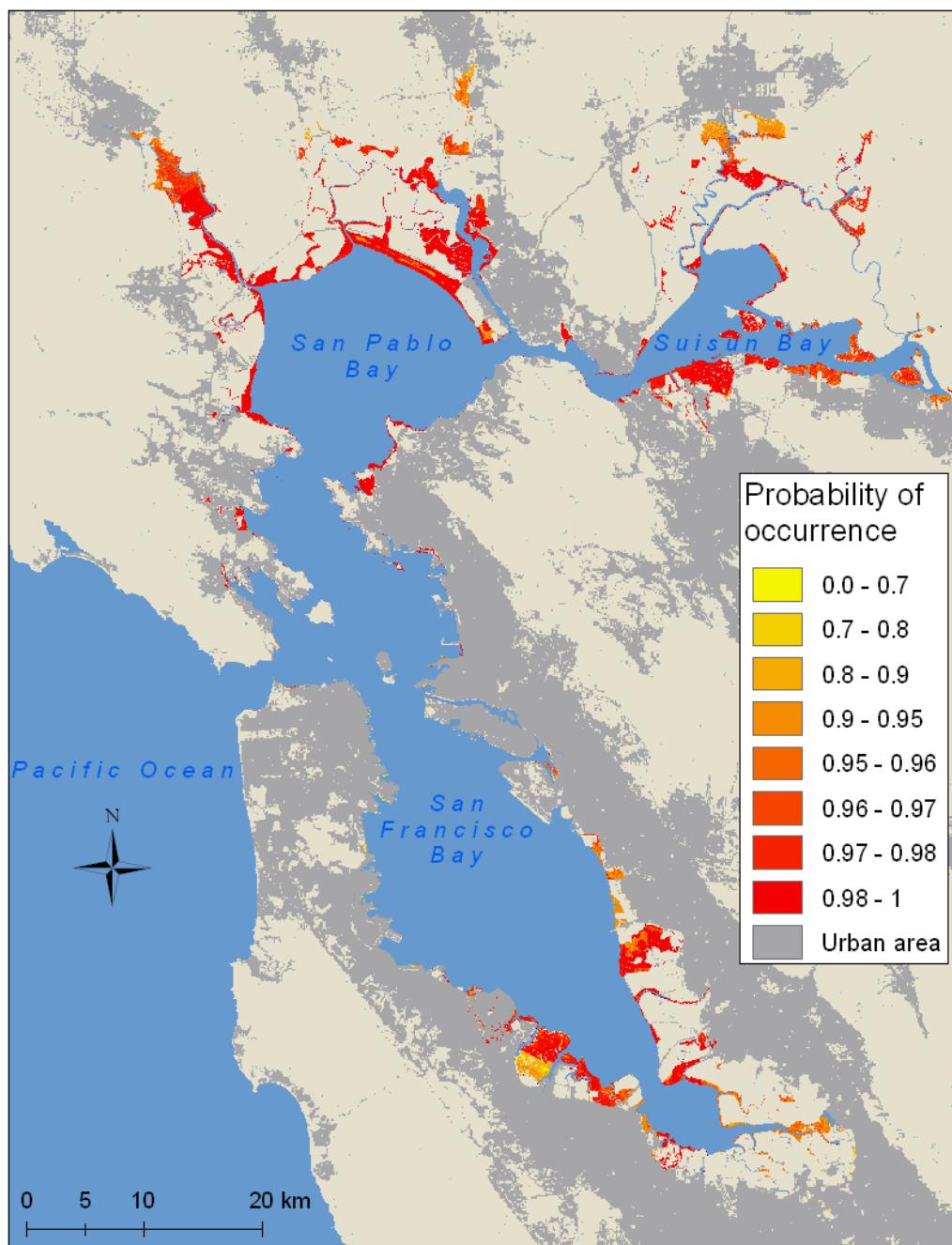


Figure 16. Predicted probability of occurrence of Song Sparrows under current (2010) conditions throughout San Francisco Estuary.



Birds – future distributions and abundance

We found that our projections for future tidal marsh bird populations were sensitive to both the sediment and sea-level rise scenarios we tested. The influence of the sediment and sea-level rise scenarios varied for some species through the time periods studied. For example, the projections for Black Rail and Song Sparrow abundance were higher at 2030 and 2050 under the high sediment scenarios than under the low sediment scenarios (Figure 18b and 18e). Between 2050 and 2090, the influence of the sea-level rise scenarios becomes more important as populations for Black Rail and Song Sparrow are projected to be higher under the low sea-level rise scenario than either sediment scenario with the high sea-level rise scenario (Figure 18b and 18e). The change in importance of the sea-level rise scenario is coincident with the increasing rates of sea-level rise projected by both scenarios after 2050. In almost all cases, high rates of sea-level rise are projected to have a deleterious effect on tidal marsh bird populations by 2110. For Common Yellowthroat and Marsh Wren, populations are always projected to be equal to or lower than current population estimates under the high sea-level rise scenario, irrespective of the sediment scenario used (Figure 18c and 18d). For all species, the projected abundance is almost always higher under the high sediment compared to the low sediment scenarios indicating that marsh accretion could buffer the effects of future sea-level rise on tidal marsh birds if sufficient sediment is available (Figure 18, Table 7).

Table 7. Projected abundance for Black Rails, Clapper Rails, Common Yellowthroats, Marsh Wrens and Song Sparrows at 2110 across the San Francisco Bay Estuary given different future sediment x sea-level rise scenarios.

| Sediment/Sea-Level Rise Scenario | Black Rail | Clapper Rail | Common Yellowthroat | Marsh Wren | Song Sparrow |
|----------------------------------|------------|--------------|---------------------|------------|--------------|
| Low/Low | 26600 | 1300 | 27000 | 83100 | 180300 |
| High/Low | 35800 | 1700 | 35900 | 104300 | 237300 |
| Low/High | 7000 | 1200 | 9000 | 24500 | 43900 |
| High/High | 18400 | 2100 | 16100 | 46400 | 142100 |
| Mean | 22000 | 1600 | 22000 | 64600 | 150900 |
| Standard deviation | 12200 | 400 | 11800 | 35800 | 81400 |

For each species except Clapper Rail, populations throughout the Estuary are projected to be higher under the low sea-level rise scenarios (Figure 18). For Clapper Rails, there is a greater difference in projected abundance due to the sediment scenarios as compared to the differences due to sea-level rise scenario, with highest abundances generally projected for the high sea-level rise scenarios (Figure 18). In contrast to other species, there was almost no regional differences in the projected change in Clapper Rails abundance across time. Clapper rail models project a decline in populations between 2010 and 2030 in both San Pablo and San

Francisco Bay (Figure 17). After 2030, Clapper Rail populations, across regions and scenarios are projected to increase from 2030 levels. For the low sediment scenarios, Clapper Rails remain below or equal to 2010 population levels across all years. For the high sediment scenarios, Clapper Rail population projections return to 2010 levels by 2050 and remain greater than 2010 levels through 2110 (Figure 18).

The changes in projected Clapper Rail populations are consistent with projected changes in spring salinity and tidal marsh elevations. The decline in projected Clapper Rail abundance is consistent with the decline in spring salinity between 2010 and 2030 (Figure 18 and 3). After 2030, spring salinity is projected to increase, coinciding with increasing Clapper Rail abundance through 2110. Additionally, under the high sediment scenarios, mid marsh habitat is projected to increase from current levels throughout the Estuary at 2030. The models for Clapper Rails predict a negative response to increasing mid-marsh elevation so the decline in populations at 2030 from current levels is due to a combination of decreasing salinity and increasing elevation.

Figure 17. The estimated abundance for Black Rail (BLRA), Common Yellowthroat (COYE), Marsh Wren (MAWR), Song Sparrow (SOSP) and Clapper Rail (CLRA) across the total San Francisco Estuary (Total), San Francisco Bay (SF), San Pablo Bay (SP) and Suisun Bay (SU). The error bars show the variation due to future scenarios and indicate ± 1 standard error.

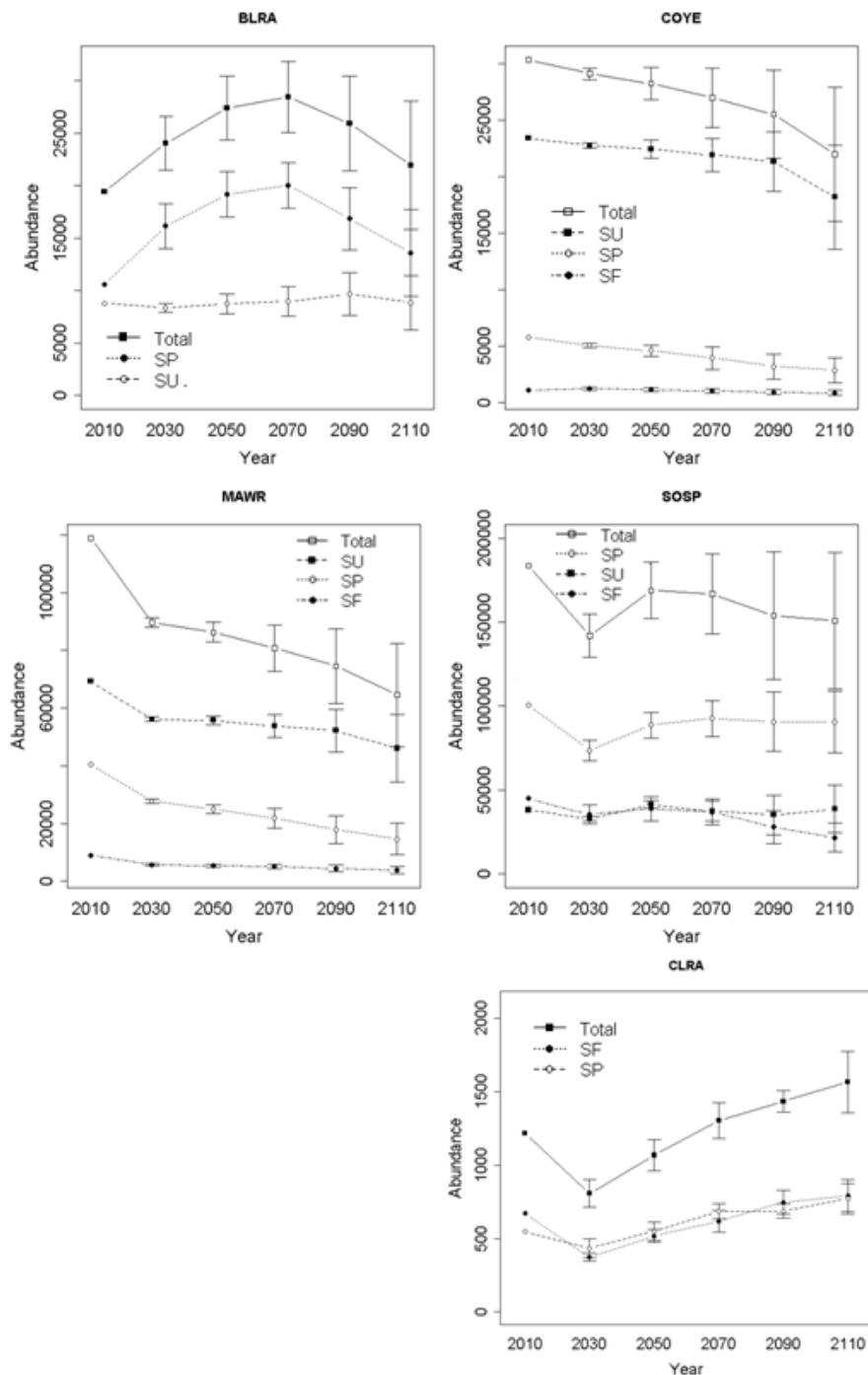
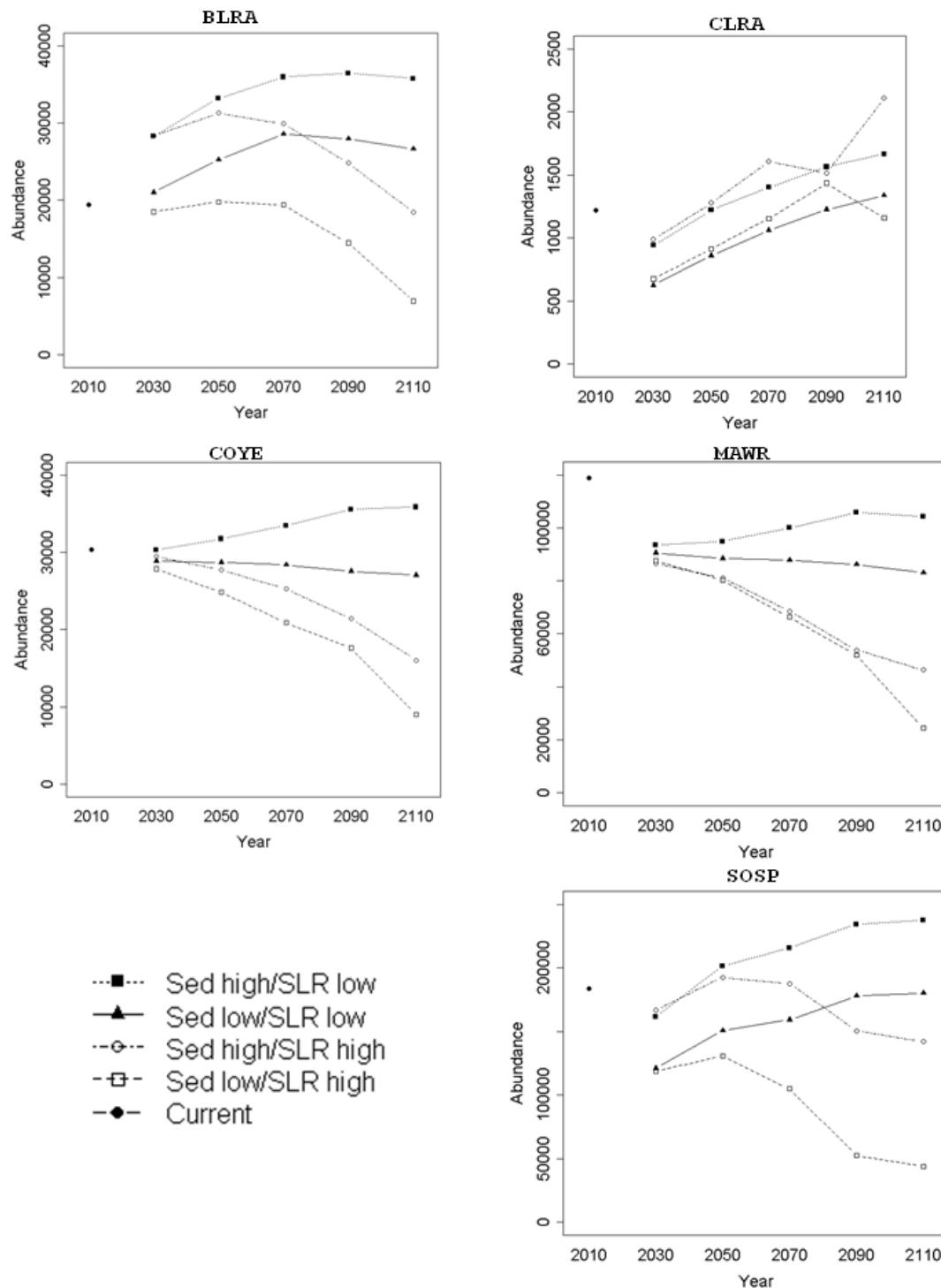


Figure 18. The projected abundance of Black Rails (a), Clapper Rails (b), Common Yellowthroats (c), Marsh Wrens (d) and Song Sparrows (e) for each combination of the sediment/ sea-level rise scenarios. For all species except Clapper Rails, current predictions are an average of predictions for years 2000-2009, for Clapper Rails the predictions are an average for years 2005-2010. Future projections are made for each 20 year period between 2030-2110.



Although most species are projected to experience population declines across the Estuary over the next 100 years, Black Rail populations are projected to increase initially and then return to near 2010 levels by 2110 with the direction of change at 2110 depending on the future scenario applied (Figure 18). While Black Rail populations in San Pablo Bay are predicted to remain above 2010 levels from 2030 to 2090, the population in San Francisco Bay is projected to remain relatively stable through time (Figure 17). The pattern could reflect a positive response to marsh accretion early in the century which is replaced by marsh drowning during the second half of the century as sea-level rise accelerates under both sea-level rise scenarios. The increase in the proportion of low marsh through time in the Estuary reflects the same pattern of increase in Black Rails. As mentioned above, the model may be projecting too high an abundance in areas with high proportions of low marsh due to the lack of areas with these conditions in the current environment.

Under all future scenarios, Song Sparrows are projected to decline between 2010 and 2030 (Figure 17). From 2050-2110, Song Sparrow projections under some scenarios return to 2010 levels (high sediment/low sea-level rise scenarios), while other models project population declines (low sediment/ high sea-level rise; Figure 17). The variation due to future scenario across Song Sparrow projections increases steadily through time, a pattern which is consistent across most species (Figure 17). Song sparrow populations are projected to remain highest in San Francisco Bay relative to other bay regions throughout the next century (Figure 17). Song sparrow populations are projected to decline under most scenarios in San Pablo Bay in every time period while the Suisun populations remain relatively unchanged (Figure 17).

CONSERVATION AND MANAGEMENT IMPLICATIONS

The modeling presented in this study provides the first comprehensive, Estuary-wide estimates of areas of concentration for all five tidal marsh bird species, and does not rely on previous surveys at specific marshes. The compilation of data from multiple years (6 years for Clapper Rails; 10 years for the other four tidal marsh bird species) allows more robust inferences about differences in occurrence or in relative density than would be possible from surveys conducted at specific marshes in disparate years. Previous field studies of Black Rails in the breeding season have generally not been conducted in San Francisco Bay; our modeling confirms the assumption of low likelihood of occurrence. Thus, except for Alviso marshes, the habitat potential of San Francisco Bay marshes with respect to Black Rails appears to be quite limited. Protection of current and future areas of high concentration, such as the Petaluma Marsh and Coon Isalnd in San Pablo Bay and Rush Ranch and Hill Slough in Suisun Bay, is thus a higher priority for this species (Figs. 10 & 20).

Our results highlight differences among regions of the Estuary, not only in terms of current patterns of distribution and abundance, but also with respect to anticipated changes in population trajectory. Regional differences reflect substantial differences in physical characteristics of the regions, such as reduced tidal range and salinity in Suisun compared with San Pablo and San Francisco Bays. Management of these species needs to be tailored to the regions, including prioritization of one region vs. another.

Trends projected for the period 2010 to 2030 generally accord well with recent trends observed. Clapper Rails are predicted to show strong declines from 2010 to 2030, declines exceeding 50%; from 2005 to 2010, the species' abundance has declined over 50% ($P < 0.001$; Liu et al. 2009, PRBO unpublished). Black Rails are predicted to increase in San Pablo Bay; since 1996, their densities have increased at an average annual rate of 4.08% (cumulative increase over 12 years is 61.6%; San Francisco Estuary Indicator Team, 2011; State of the Birds 2011, in prep.). The species is not expected to increase in Suisun, and no significant overall trend has been observed from 1996 to 2008.

There may be some discrepancies between recent observed trends and future predictions. Common Yellowthroats are predicted to decline across all three regions. In recent years (2000-2009), an annual 5.1% decline in this species has been observed in San Francisco Bay ($P = 0.19$; PRBO unpublished), but has shown a 5.3% increase in Suisun Bay ($P = 0.24$; PRBO unpublished). For Song Sparrows, estimates of recent (2000-2009) trends of changes in population sizes have shown annual 2% decreases in population in San Pablo and Suisun Bays and a 2% annual increase in San Francisco Bay but these trends were not found to be significant (PRBO unpublished). Observed trends are somewhat consistent with the projected behavior from 2010 to 2110. However, the model predicts a greater decline in population from 2010 to 2110

then would be expected from observations from San Francisco Bay Song Sparrow populations. Marsh Wrens are predicted to decrease in abundance from current levels across all scenarios, with similar patterns of decrease across all bay regions. Recent trends (2000-2009) indicate declines of Marsh Wrens in San Francisco Bay (3.1% annual decline, $P = 0.49$; PRBO unpublished) and San Pablo Bay (1.2% annual decline, $P = 0.67$; PRBO unpublished) which is consistent with the future model predictions. Yet we observed a positive trend in Suisun Bay (7.3% annual increase, $P = 0.12$; PRBO unpublished) for Marsh Wrens which is inconsistent with the population declines predicted by the model. The differences between observed and predicted trends may be more related to changes in habitat availability, which the empirical trends do not take into account whereas our modeling does.

Our models of species response to future changes in biophysical conditions indicate that there will be substantial changes in the distribution and abundance of plant and bird species throughout the San Francisco Bay Estuary. These changes will be manifested at regional to local scales. For example, as noted above, the direction of population change for tidal marsh birds relative to present day may vary by bay region. At the same time at the local scale, the spatial distribution of individual plant species within a marsh will likely change as species track suitable biophysical conditions. The management of these systems will require flexibility in order to adapt to the changes which actually occur.

Our models illustrate the value of considering a range of future scenarios. The variation in future predicted bird abundance across the four scenarios tested is high for all species. If we had only chosen to model using the low sea-level rise scenario, our results would predict population increases from current levels by 2110 for most species as opposed to the declines in abundance from current levels predicted for most species using the high sea-level rise scenario. As future projections of sea-level rise are refined, we will be able to narrow the range in estimated future abundance of the species studied. Similarly, we have more confidence in near term predictions given the smaller variation in future sea-level rise projections at 2030 as compared to 2110. Given the variability in future outcomes from our models we recommend the development of adaptation strategies which are flexible enough to change as future projections of sea-level rise and habitat response become more certain.

The future projections we provide in this report and which are also available online are best used to identify the range of possible responses under the different future scenarios we have tested. The actual value projected at a pixel is unlikely to precisely predict the probability of occurrence of a tidal marsh plant or bird. If the models indicate large changes at a site, i.e. the models show highly suitable habitat disappearing due to marsh drowning, then the maps should accurately indicate areas vulnerable to sea-level rise. Below we demonstrate how

consensus predictions across scenarios through time can use the range of projections from future scenarios to test the sensitivity of specific locations to the sea-level rise.

Priority Areas

There are very few areas in which Clapper Rails are predicted to be present in 18 or more of the scenarios modeled (Figure 19). Areas where Clapper Rails are predicted to occur consistently across scenarios include Steamboat Slough, Muzzi marsh, Petaluma marsh and in China camp (Figure 19). Given the rarity of sites which are projected to be resilient to future climate change scenarios, the results here suggest that these sites should be a high priority for conservation.

Comparing maps of the summed predictions from each species illustrates how certain areas may provide complementary protection, the same location is consistently predicted to be suitable across scenarios and species, but that the locations of important habitat can also vary for different bird species (Figure 20). All focal species, except Clapper Rail, are projected to occur in marshes around Coyote Creek in the south bay in 15-20 of the scenarios. Clapper Rails were predicted to be absent in all scenarios in the same area. The Petaluma marsh complex is predicted to be of consistently high values (areas within the marsh selected by 15-21 of the scenarios) for Black Rails, Common Yellowthroats and Marsh Wrens but is mostly predicted to be unsuitable for Clapper Rails and Common Yellowthroats across all scenarios (Figures 19-20).

Maps of the predictions summed over the four tidal marsh species that were modeled at the same scale (all species except Clapper Rail) reveal locations of high conservation priority relative to the four species. Rush Ranch and Montezuma/Grizzly marsh in Suisun Bay, and Black John Marsh along the Petaluma River and sites along the Napa Slough are projected to be suitable habitat for Black Rails, Common Yellowthroat, Marsh Wrens and Song Sparrows across most scenarios (75 - 84 scenarios predicting presence, Figure 20, 21). Figure 21 has the added benefit of highlighting marshes that remain as marsh habitat across all the scenarios (i.e. resilient to drowning), pixels which are converted to non-marsh habitat from any scenario are dropped from the analysis. So any pixel with values from 0-84 in figure 21 is projected to remain as marsh habitat through 2110 across all scenarios.

Most importantly, high priority areas that are currently upland but projected to be suitable future marsh habitat can be readily identified by conservation planners at fine (site-specific) scales using our online web tool. These areas are very limited in San Francisco Bay making maps at the Estuary-wide scale less useful than interactive maps that allow the user to focus on his/her area of interest (Figure 22-23). Having decision-makers with site-specific knowledge use the web tool to identify high priority upland areas and plan for the protection and eventual restoration will be instrumental in maximizing future tidal marsh.

Figure 19. Areas of highest conservation importance for Clapper Rails. Map values indicate the sum of predictions from boosted regression tree binomial models for Clapper Rails. Model predictions have been converted to presence/absence maps for each scenario and then summed across scenarios. Values of 21 mean all scenarios (1 current + 20 future scenarios) predict presence at a cell and illustrate areas insensitive to uncertainty in future climate change.

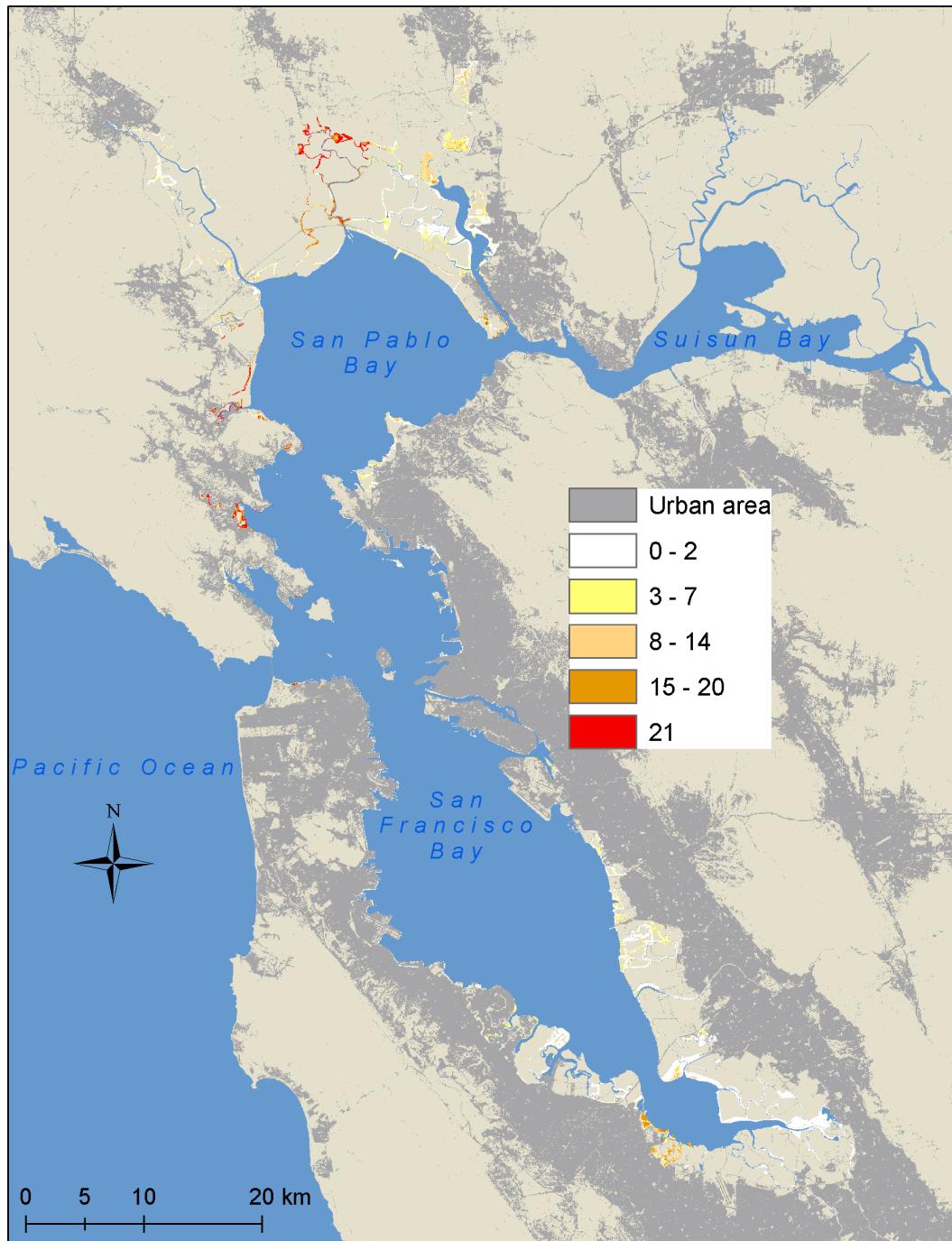


Figure 20. Areas of highest conservation importance for Black Rail (a), Common Yellowthroat (b) Marsh Wren (c) and Song Sparrow (d). Map values indicate the sum of predictions from boosted regression tree binomial models. Model predictions have been converted to presence/absence maps for each scenario and then summed across scenarios. Values of 21 means all scenarios (1 current + 20 future scenarios) predict presence at a cell and illustrate areas insensitive to uncertainty in future climate change.

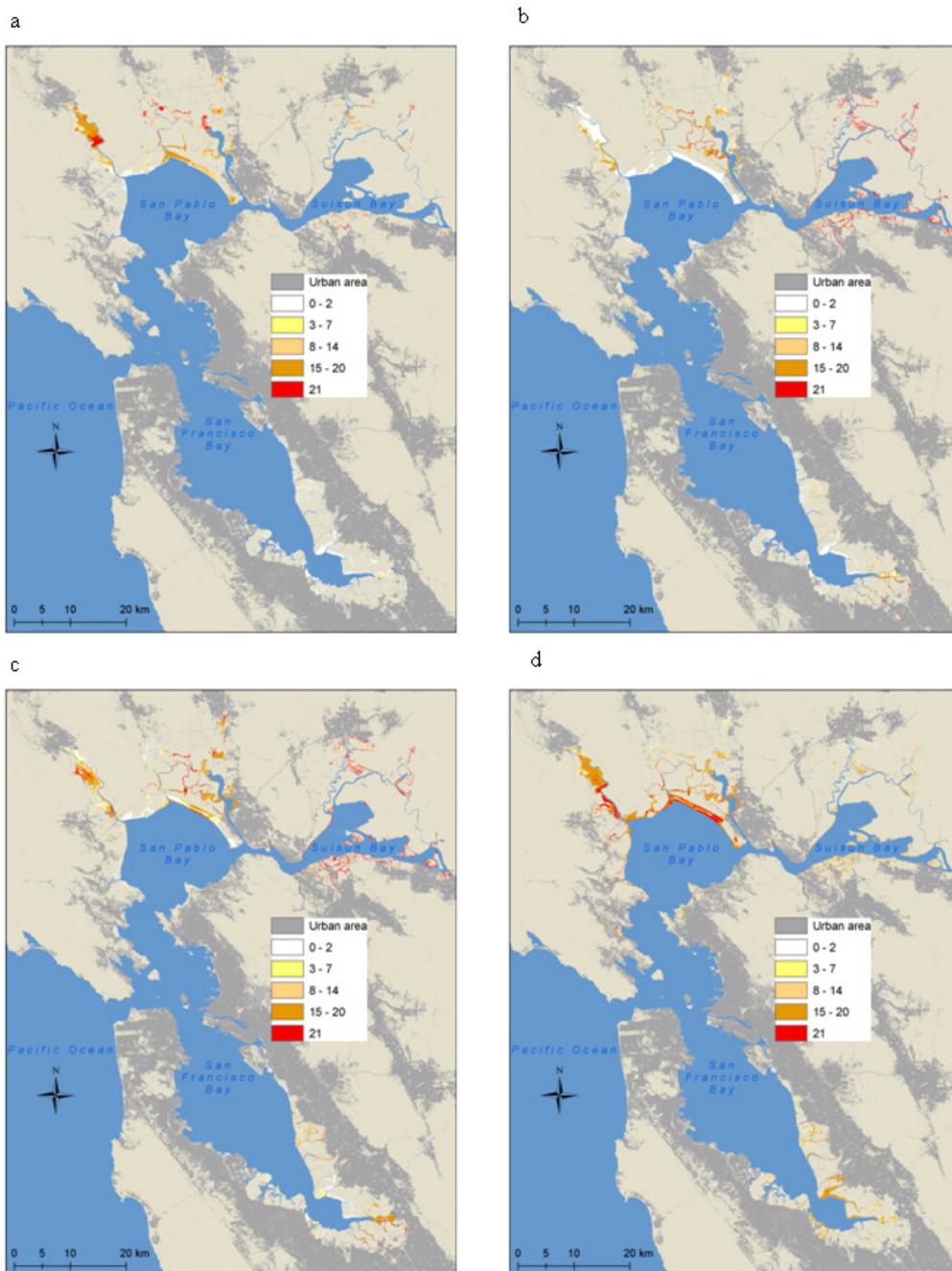


Figure 21. Areas of highest combined conservation importance for Black Rail, Common Yellowthroat, Marsh Wren and Song Sparrow. Model predictions have been converted to presence/absence maps for each scenario and then summed across scenarios and species. Values of 84 means all scenarios (1 current + 20 future scenarios x 4 species) predict presence at a cell and illustrate areas insensitive to uncertainty in future climate change.

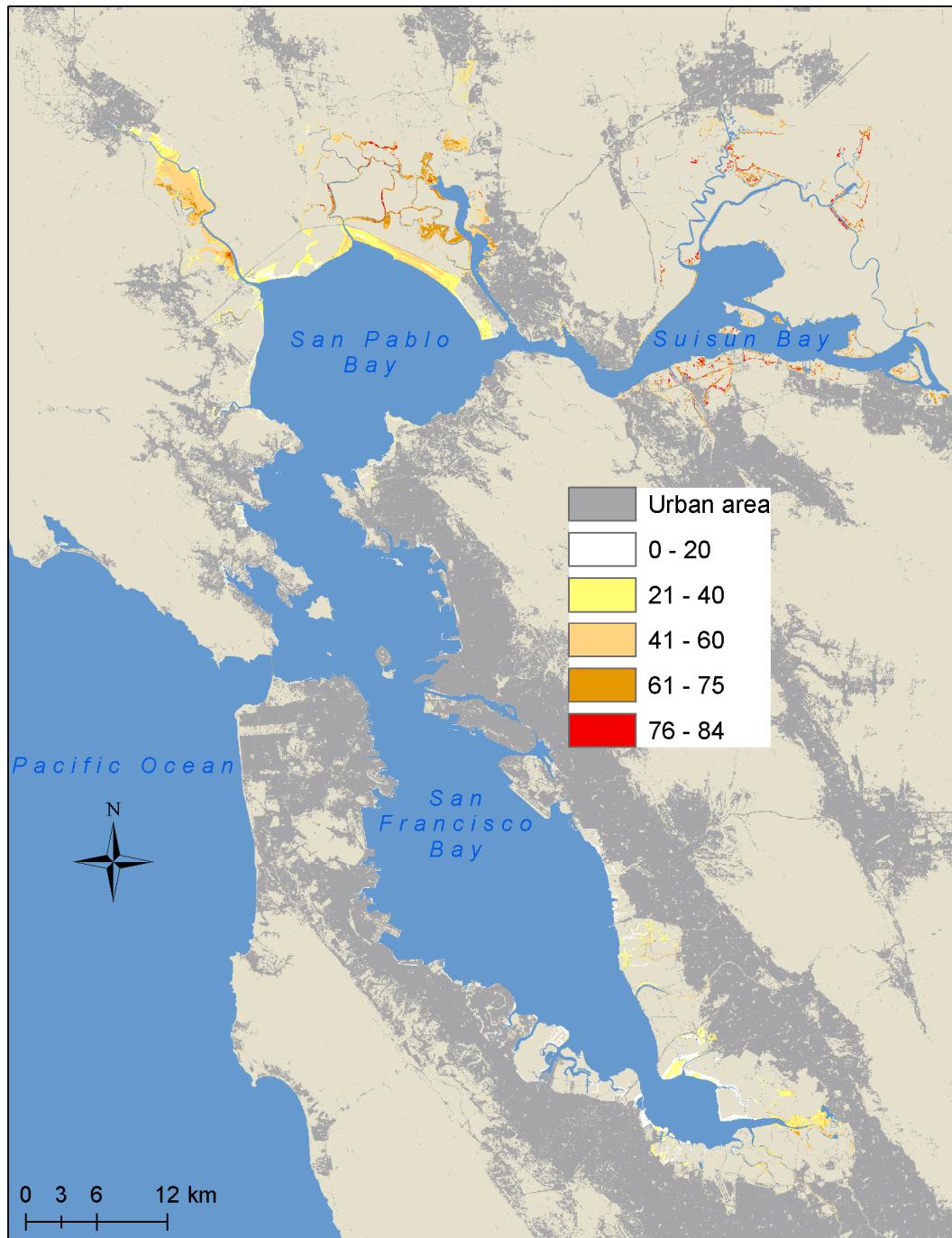


Figure 22. Screen shot of sea-level rise web tool (www.prbo.org/sfbayslrisr) showing current potential tidal marsh habitat in the northern portion of San Pablo Bay with all levees removed (left image) and projected tidal marsh habitat in year 2110 (right image) with box outline indicating the area of zoom extent in Figure 22. Projected tidal marsh is under high sea-level rise, low sediment scenario.

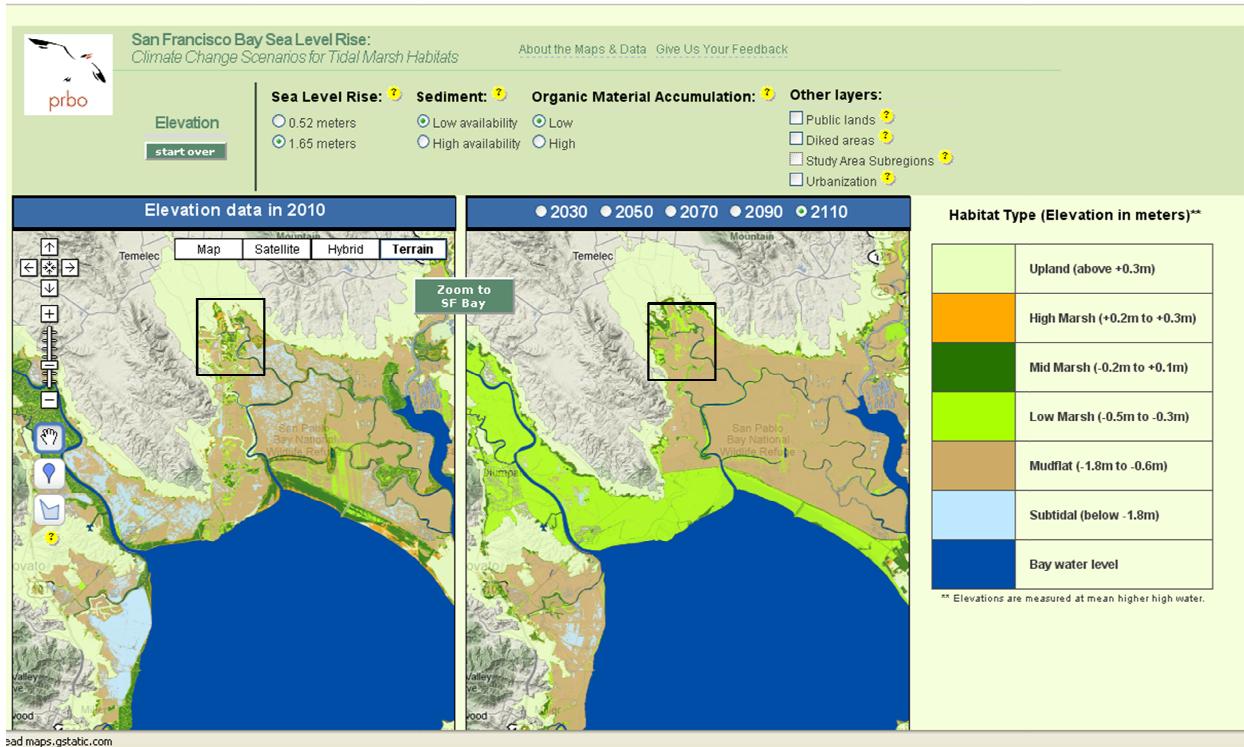
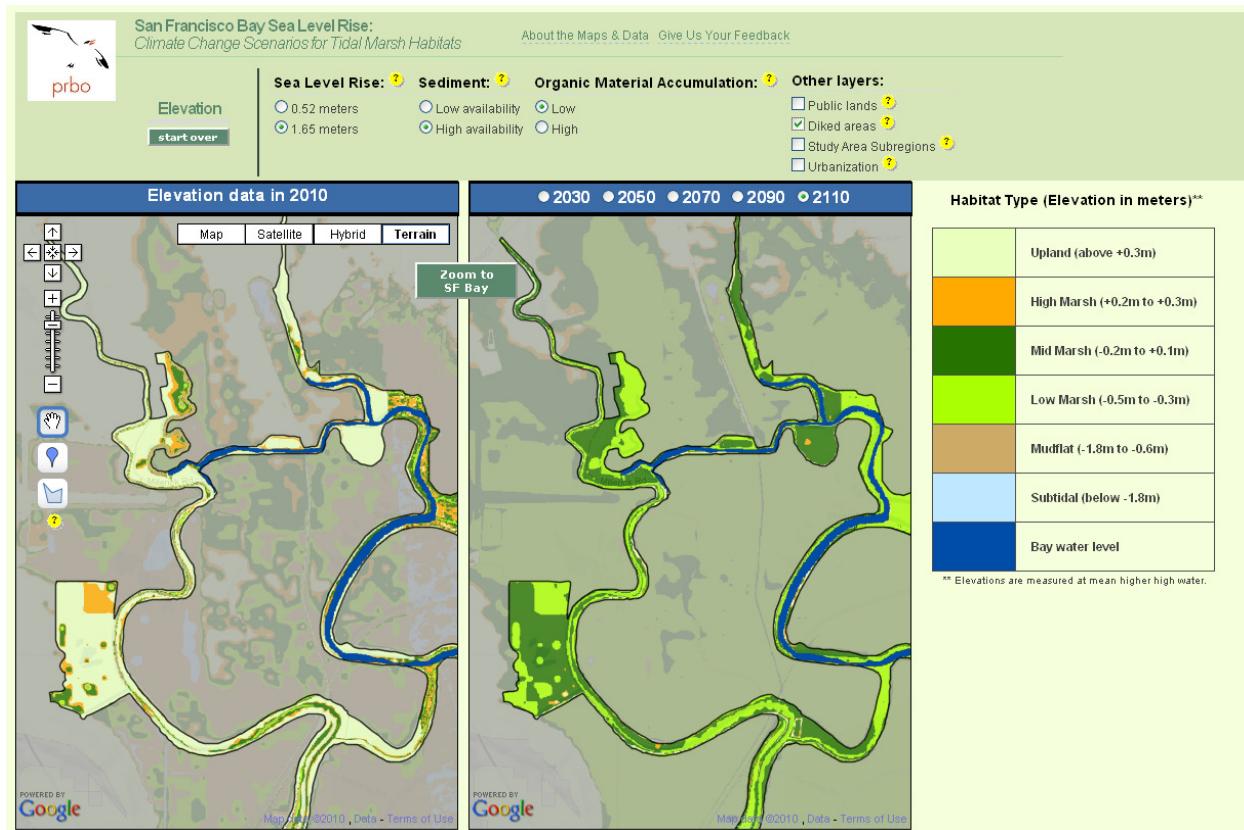


Figure 23. Screen shot of sea-level rise web tool (www.prbo.org/sfbayslr) showing current potential tidal marsh habitat within the existing levee system at Sonoma Creek (left image) and projected tidal marsh habitat in year 2100 (right image). Comparison of the two images reveals opportunities for protecting current uplands projected to become suitable tidal marsh habitat. Full restoration opportunities visible under opaque levee layer mask or by unclicking “diked areas” button online. Projected tidal marsh is under high sea-level rise, low sediment scenario.



Restoration results and recommendations

Stralberg et. al. (in preparation) provide a coarse estimate of the amount of sediment needed to achieve marsh stability given future sea-level rise. Under a low rate of SLR and high sediment concentrations (≥ 200 mg/L), models show that mid marsh restoration could be achieved over the next century with initial bed elevations 2 m or more below MHHW (i.e., subtidal). With very high sediment concentrations (250-300 mg/L), mid marsh habitat could be expected within 20 years at subtidal locations, while close to 60 years would be necessary with 150 mg/L. For intermediate (100 mg/L) and low (25-50 mg/L) sediment concentrations, successful mid marsh restoration would be expected only with mudflat and mid or high marsh starting elevations, respectively. Under a high rate of SLR, however, mid marsh restoration would only be achieved

over a 100-year time period given starting elevations well above MHHW (current upland areas), or very high sediment concentrations. With very high sediment concentrations (200-300 mg/L), mid marsh habitat could be constructed or restored even in areas that are currently subtidal. At lower sediment concentrations, mid marsh could initially be restored from low- and mid-marsh starting elevations below MHHW, but would not persist more than 80 years (40 years at very low sediment concentrations).

Our models indicate that marsh accretion will be hampered by the accelerating rate of sea-level rise during the second half of the 21st century. Successful tidal marsh restoration will be more successful if implemented before the second half of the 21st century when the rate of sea-level rise is not as steep and allowing more time for marsh accretion. In Phase I, we found that the availability of suspended sediment is key to marsh sustainability and restoration success (Stralber et al., in preparation). Given that sediment concentrations in the San Francisco Estuary may be declining (Wright and Schoellhamer, 2004) we recommend developing adaptation strategies that involve restoring natural sources of sediment (e.g., creek restoration and removal of barriers to downstream sediment transport) or the strategic re-use of clean dredge material to raise bed elevations pre-restoration or to apply to a site repeatedly to maintain mid marsh elevations. Other creative process-based or artificial means of maximizing sediment concentrations locally and Estuary-wide should be investigated. Effort should be made to identify and protect upland areas projected to become future marsh under high sea-level low sediment scenarios.

Furthermore, future restoration and land acquisition priorities should be informed by the availability of adjacent upland sites for lateral marsh expansion. Although our spatial analysis revealed relatively little area naturally available to accommodate future marshes we found that four times as much area could be reclaimed by removing levees and other barriers to tidal action as opposed to not removing the structures. Unfortunately, the large majority of areas with elevations suitable for marsh expansion are already urbanized and thus considered unavailable. The existing opportunities for marsh expansion into upland areas (diked or not) may be evaluated using our web-based tool (Stralberg et al., in preparation) Finally, if restoration choices are constrained, efforts should be concentrated in high sediment areas (e.g., Petaluma and Napa rivers and southern portion of San Francisco Bay)with better prospects for long-term sustainability.

LIMITATIONS OF THE STUDY AND OPPORTUNITIES FOR FUTURE RESEARCH

Values for several parameters used for modeling species distributions were assumed to remain static in future scenarios. At certain spatial scales, the distance to certain features such as the distance to the bay will likely remain unchanged. However other variables such as distance to channels will undoubtedly change. We have attempted to strike a balance between including variables which are hypothesized to be important in limiting the distribution of tidal marsh species and including variables which can plausibly be modeled under future conditions. We believe the static variables we used will remain relatively unchanged in the future making the simplifying assumption warranted. If the variables do change greatly then our results will be less reliable. However, unanticipated changes in features can be readily incorporated into future iterations of our models as new information becomes available.

Summer and spring salinity were found to be important in both vegetation and bird distributions. For present day conditions, we used spatial interpolation from point level observations to calibrate our models. However, salinity can vary considerably within tidal marshes and our spatial interpolations may give an accurate but imprecise estimate of salinity at fine spatial scales. In addition, the future salinity projections we use have considerable uncertainty so our future vegetation and bird projections are somewhat limited by the reliability of salinity models. Our models could be improved by incorporating site-specific measures of salinity into model calibration and improved projections of future salinity.

Our models are based on statistical correlations between the occurrence or abundance of birds and physical characteristics of the environment. The correlations between the occurrence of birds and physical may indicate a direct relationship in which the species is adapted to a specific range of the physical variable, or simply that a bird may be associated with a plant that covaries with the physical variable. Our models do not provide a mechanistic link between the species and the physical variables, but the models inform our follow up work where we will explicitly model demographic response to changes in physical characteristics such as increased risk of nest flooding based on the decline in high marsh habitats projected throughout the San Francisco Estuary.

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APPENDICES

Appendix 1. List of meetings, workshops and conferences where staff have presented the results of this study and the online decision support tool.

1. SF Bay Joint Venture Restoration Committee
2. Bay-Delta Conference, Sediment Management Workshop
3. California Vulnerability and Adaptation Study
4. SF Bay Joint Venture Management Board
5. North Bay Managers Meeting
6. East Bay Regional Parks District
7. PACLIM conference

Appendix 2 Top generalized additive models (GAM) predicting vegetation species' occurrence, selected based on AIC. All models within two AIC units of the model with the lowest AIC were retained for further modeling.

| Species | Rank | Model | AIC |
|----------------------------|------|---|--------|
| <i>Distichlis spicata</i> | 1 | salinsum + distbay + meanhhw + slope + mhhw10maj*salinsum | 18.75 |
| | 2 | salinsum + distchan + distbay + meanhhw + slope + mhhw10maj*salinsum | 19.24 |
| | 4 | salinsum + distbay + meanhhw + mhhw10maj + slope + mhhw10maj*salinsum | 19.52 |
| | 5 | salinsum + distchan + distbay + meanhhw + mhhw10maj + slope + mhhw10maj*salinsum | 20.16 |
| | 6 | salinsum + distchan + distbay + meanhhw + mhhw10maj + slope | 20.69 |
| Species | Rank | Model | AIC |
| <i>Frankenia salina</i> | | salinsum + channelpercent + distlevee + distbay + mhhw10maj + mhhwsd + | |
| | 1 | mhhw10maj*salinsum | 425.43 |
| | 2 | salinsum + channelpercent + distlevee + distbay + meanhhw + mhhw10maj + mhhwsd + mhhw10maj*salinsum | 425.43 |
| Species | Rank | Model | AIC |
| <i>Grindelia stricta</i> | 1 | salinsum + distlevee + distbay + meanhhw + slope + mhhw10maj*salinsum | 598.36 |
| | 2 | salinsum + distlevee + distbay + meanhhw + mhhwsd + slope + mhhw10maj*salinsum | 598.88 |
| Species | Rank | Model | AIC |
| <i>Jaumea carnosa</i> | 1 | salinsum + distchan + distlevee + distbay + meanhhw + mhhw10maj + mhhwsd | 88.03 |
| | 2 | salinsum + distchan + distlevee + distbay + meanhhw + mhhw10maj + slope | 89.33 |
| | 3 | salinsum + distchan + distlevee + distbay + meanhhw + mhhw10maj + mhhwsd + slope | 89.42 |
| | | salinsum + distchan + channelpercent + distlevee + distbay + meanhhw + mhhw10maj + | |
| | 4 | mhhwsd + slope | 89.95 |
| Species | Rank | Model | AIC |
| <i>Juncus spp.</i> | 1 | salinsum + distlevee + distbay + meanhhw + mhhw10maj*salinsum | 59.38 |
| | 2 | salinsum + distlevee + distbay + meanhhw | 60.43 |
| Species | Rank | Model | AIC |
| <i>Lepidium latifolium</i> | 1 | salinsum + channelpercent + distlevee + distbay + meanhhw + slope | 465.64 |

| | 2 | salinsum + distchan + channelpercent + distlevee + distbay + meanhhw + slope | 467.4 |
|------------------------------|------|--|--------|
| Species | Rank | Model | AIC |
| <i>Phragmites australis</i> | 1 | salinsum + distlevee + slope | 132.5 |
| | 2 | salinsum + distlevee + distbay + slope | 132.58 |
| | 3 | salinsum + channelpercent + distlevee + distbay + slope | 133.06 |
| | 4 | salinsum + distlevee + mhhwsd + slope | 133.18 |
| | 5 | salinsum + distlevee + mhhw10maj + slope | 133.48 |
| | 6 | salinsum + distlevee + distbay + mhhw10maj + slope | 133.65 |
| | 7 | salinsum + distlevee + mhhwsd | 133.85 |
| Species | Rank | Model | AIC |
| <i>Salicornia pacifica</i> | 1 | salinsum + mhhw10maj | 145.51 |
| | 2 | salinsum + distlevee + mhhw10maj | 145.9 |
| | 3 | salinsum + distbay + mhhw10maj | 145.99 |
| | 4 | salinsum + mhhw10maj + mhhw10maj*salinsum | 146.97 |
| Species | Rank | Model | AIC |
| <i>Schoenoplectus acutus</i> | 1 | salinsum + distbay + mhhwsd | 202.13 |
| | 2 | salinsum + distchan + distbay + mhhw10maj | 202.24 |
| | 3 | salinsum + distbay + mhhw10maj + mhhwsd | 202.31 |
| | 4 | salinsum + distbay + mhhw10maj | 202.43 |
| | 5 | salinsum + distbay + mhhwsd + slope | 202.75 |
| | 6 | salinsum + distchan + distbay + mhhwsd + slope | 203.3 |
| | 7 | salinsum + distchan + distbay + mhhw10maj + mhhwsd | 203.36 |
| | 8 | salinsum + distbay | 203.61 |
| | 9 | salinsum + distchan + distbay + mhhwsd | 203.62 |
| | 10 | salinsum + distchan + distbay | 203.73 |
| | 11 | salinsum + distchan + distbay + meanhhw + mhhw10maj + mhhwsd + slope | 203.93 |
| | 12 | salinsum + distbay + meanhhw + mhhwsd + slope | 204.05 |
| Species | Rank | Model | AIC |

| <i>Schoenoplectus americanus</i> | | | |
|------------------------------------|------|---|--------|
| Species | Rank | Model | AIC |
| <i>Schoenoplectus americanus</i> | 1 | salinsum + distlevee + distbay + meanhhw + mhhwsd + mhhw10maj*salinsum | 250.27 |
| | 2 | salinsum + distlevee + distbay + meanhhw + mhhw10maj*salinsum | 250.7 |
| | 3 | salinsum + distlevee + distbay + meanhhw + mhhw10maj | 250.9 |
| | 4 | salinsum + distlevee + distbay + meanhhw + slope + mhhw10maj*salinsum | 251.25 |
| | 5 | salinsum + distlevee + distbay + meanhhw + mhhw10maj + mhhwsd | 251.38 |
| Species | Rank | Model | AIC |
| <i>Schoenoplectus californicus</i> | | salinsum + distchan + channelpercent + distlevee + distbay + mhhwsd + | |
| | 1 | mhhw10maj*salinsum | 239.77 |
| | 2 | salinsum + distchan + channelpercent + distlevee + distbay + mhhw10maj + mhhwsd | 239.81 |
| | 3 | salinsum + distchan + channelpercent + distlevee + distbay + slope + | |
| | 4 | mhhw10maj*salinsum | 241.33 |
| <i>Schoenoplectus californicus</i> | 4 | salinsum + distchan + channelpercent + distbay + mhhw10maj + mhhwsd | 241.41 |
| | 5 | salinsum + distchan + distlevee + distbay + mhhwsd + mhhw10maj*salinsum | 241.49 |
| Species | Rank | Model | AIC |
| <i>Schoenoplectus maritimus</i> | | salinsum + distchan + distlevee + distbay + meanhhw + mhhw10maj + | |
| | 1 | mhhw10maj*salinsum | 372.42 |
| | 2 | salinsum + distchan + distbay + meanhhw + mhhw10maj + mhhw10maj*salinsum | 373.43 |
| | 3 | salinsum + distchan + distbay + meanhhw + mhhw10maj + slope + | |
| | 3 | mhhw10maj*salinsum | 374.36 |
| Species | Rank | Model | AIC |
| <i>Spartina foliosa</i> | 1 | nsum + distchan + channelpercent + distbay + meanhhw + slope + mhhw10maj*salinsum | 447.4 |
| Species | Rank | Model | AIC |
| <i>Spartina alterniflora</i> | 1 | salinsum + channelpercent + meanhhw + mhhw10maj | 176.59 |
| | 2 | salinsum + channelpercent + meanhhw + mhhw10maj*salinsum | 176.72 |
| | 3 | channelpercent + distbay + meanhhw + mhhw10maj | 176.91 |
| | 4 | salinsum + distchan + channelpercent + meanhhw | 177.02 |
| | 5 | channelpercent + distbay + meanhhw + mhhw10maj*salinsum | 177.06 |
| | 6 | salinsum + channelpercent + distbay + meanhhw + mhhw10maj | 177.13 |
| | 7 | salinsum + channelpercent + distbay + meanhhw + mhhw10maj*salinsum | 177.29 |

| | | |
|----|--|--------|
| 8 | salinsum + channelpercent + meanhhw + mhhw10maj + mhhw10maj*salinsum | 177.31 |
| 9 | salinsum + channelpercent + distbay + meanhhw | 177.63 |
| 10 | salinsum + distchan + channelpercent + distbay + meanhhw | 177.64 |
| 11 | salinsum + distchan + distbay + meanhhw | 177.99 |
| | salinsum + channelpercent + distbay + meanhhw + mhhw10maj + | |
| 12 | mhhw10maj*salinsum | 178.14 |
| 13 | salinsum + distbay + meanhhw | 178.44 |
| 14 | salinsum + distbay + meanhhw + mhhw10maj + mhhw10maj*salinsum | 178.47 |

| Species | Rank | Model | AIC |
|-------------------|------|------------------------------|-------|
| <i>Typha spp.</i> | 1 | salinsum + mhhw10maj + slope | 25.92 |