State of San Francisco Bay 2011
Appendix D

HABITAT – Baylands and Watershed Indicators
Technical Appendix

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Health Indicators of Intertidal and Watershed Wetlands

Background and Rationale

This report focuses on the health of Bay Area wetlands. The state is developing a wetland definition as part of its Wetland and Riparian Area Protection Policy. According to the proposed definition, wetlands are areas that, under normal circumstances, (1) are saturated by ground water or inundated by shallow surface water for a duration sufficient to cause anaerobic conditions within the upper substrate; (2) exhibit hydric substrate conditions indicative of such hydrology; and (3) either lack vegetation or the vegetation is dominated by hydrophytes (TAT 2009a). Shallow surface waters are less than 2 m deep. This is essentially the same definition used by the U.S. Army Corp of engineers (USACE) and U.S. environmental protection agency (USEPA) except that it explicitly includes areas that lack vegetation.

There are many kinds of wetlands, based on the recommended state definition. The main wetlands of the Estuary are the intertidal flats and marshes that adjoin the subtidal areas and open waters of the bays and straits. In the watersheds that drain to the Estuary, there are wetlands associated with lakes and ponds, rivers and streams, and springs and seeps. Some of these wetlands are perennial and others are seasonal. Seasonal wetlands can form along the beds of ephemeral or intermittent rivers and streams, and on hillsides and valley floors, where the root zone intercepts seasonally high groundwater. They also form in shallow depressions that fill with rain and slowly drain. Wetlands can be further classified according to differences in their sediment characteristics and plant community composition (Cowardin et al. 1979) and landscape position (Tiner 2003).

Wetlands provide many important services. They provide water and food, control pollution and flooding, and support wildlife and recreation. The types and level of services differ among the kinds of wetlands and their different settings. Some wetlands are managed to provide specific services, such as wastewater treatment or waterfowl hunting. Natural wetlands tend to provide broad suites of services. They are especially valued for their contribution to the native biological diversity of the region. Most of the region's rare and endangered plants and animals rely on wetlands for their survival. In the landscape context, wetlands are transitional between the wettest and the driest conditions. They tend to be very sensitive to changes in climate or land use that affect water supplies. In turn, wetlands influence water supplies by storing water and slowly releasing it downstream or into the ground. They increase the ability of landscapes to retain water and thus increase opportunities for land planners and managers to conserve water and use it wisely. The protection and restoration of wetlands can be significant aspects of our preparation and response to climate change, and of our overall approach to environmental health care. The public invests large sums of money each year to protect and restore wetlands through local, state, and federal policies and programs. The progress and success of these efforts can be assessed as the ability of wetlands to provide high levels of their desired or needed services.

There is an adequate foundation of scientific understanding to support separate assessments of many wetland services (e.g., Bedford and Preston 1988, Johnston et al. 1990, Brinson 1993, Carter 1997, Bayley 1999, MEA 2005). It is very unlikely, however, that funds and human resources will ever be adequate to assess every service for any kind of wetland. The assessments will need to focus on the services for which goals or benchmarks are set and for which adequate data can be assembled. The benchmarks are needed to define good health, or adequate service. The data are needed to track progress, improve our understanding, and to know when good health is achieved.

For this report, wetland assessment relies on a few metrics that integrate across many services. This approach is necessitated by a lack of benchmarks and data for all but a few wetland services, some of which are incorporated into separate assessments of water quality and wildlife support. The integrative metrics are useful for assessing the overall condition or health of wetlands.

Data Sources

Extent of Tidal Marshes and Tidal Flats

Definitions

For the purposes of this report, tidal marshes are intertidal areas at least 0.25 acres in size that support at least 5% cover of vascular vegetation, when viewed at a scale of 1:2,500. Tidal flats are similarly defined except that they lack at least 5% cover of vascular vegetation. For any given surface water salinity regime, tidal flats tend to occur lower in the intertidal zone.

The extent of marshes and flats is defined as their total acreage within the Estuary downstream of Broad Slough, which delimits the western boundary of the Delta. Measures of extent are based on maps. Expert mapping of marshes and flats requires defining their spatial limits using field surveys and remote data, such as aerial imagery. The upper (landward) margin of a marsh is recognized as a visually distinct shift from marsh vegetation to upland vegetation (e.g., Bodnar et al 1975, Culberson 2001). The lower limit is recognized as the shift to less than 5% cover of vegetation, which is generally abrupt. The upper limit of the flat is the same as the lower limit of the marsh. The lower limit of the flat is more difficult to discern. For the purposes of this report, the lower limit of tidal flat was taken directly from federal navigational charts, and represents an estimation of the mean lower-low tide datum, also referred to as the zero-tide height, based on depth soundings. The zero-tide contour is the conventional lower limit of the intertidal zone for the west coast of the U.S. (NOS 2000). The actual contour varies from year to year and the amount of this variability is generally unknown. The other boundaries of tidal flats and marshes are readily viewable in aerial imagery and have been verified in the field.

Tidal channels that are bounded by tidal marsh, almost entirely dewater at low tide, and are on average less than 200 ft wide are considered tidal marsh features, and are included in estimates of tidal marsh extent (see section below on marsh size). Marsh ponds, pannes, potholes, and other non-vegetated features of tidal marshes are also included in measures of marsh size. Portion of channels that are at least 200 ft wide and dewater at low tide are considered tidal flats features. Portions of channels that do not dewater at low tide are considered subtidal.

Historical Maps

There are multiple sources of maps of the tidal marshes and flats of the Estuary (U.S. Coast Survey 1857, U.S. Coast Survey 1897, Jones and Stokes Associates et al 1979, Dedrick 1989, Dedrick and Chu 1993, SFEI 2000, SFEI 2010). In order to describe historical and recent changes in the regional extent of marshes and flats, three strictly comparable maps are readily available. The earliest is the map of aquatic areas ca 1800 produced by SFEI in the 1990's as part of the Bay Area EcoAtlas (SFEI 2000, Goals project 1999). For the intertidal areas, the main source of data was the Topographic Sheets of the US Coast Survey that date from the mid nineteenth century (US Coast Survey 1857). For a thorough discussion of the early Coast Survey maps, go to the T-sheet User Guide (http://www.sfei.org/sites/default/files/T_sheet_user_guide_SFEI_highres_0.pdf).

The map of historical extent represents the expected average arrangement of wetlands and related habitats over the 400 years preceding Euro-American contact (i.e., 1400 to 1800 AD). For this region, the timeframe is characterized by moderate climatic variability with multiple droughts and wet periods lasting less than a decade each century. The period prior to 1400 AD is characterized by multi-decadal periods of persistent low and high rainfall and stream flow. The historical map also reflects indigenous land use practices, to the extent that they affected the distribution and shape of intertidal habitats. For example, there is anecdotal evidence that local tribes may have modified some tidal marsh pannes for salt production and waterfowl hunting.

Many additional sources of information were used to augment the Coast Survey maps. This included other eighteenth- and nineteenth-century maps, sketches, paintings, engineering reports, oral histories, explorers' journals, missionary texts, hunting magazines, oblique (land-based) photography, and interviews with living elders. Aerial photography did not exist at the time of the earliest maps. As part of the process of integrating these various data to create the best possible historical map, the relative certainty of each feature (e.g. a channel, marsh, or panne) was assessed, using a weight-of-evidence approach. In all regards, the historical map of tidal marshes and flats has a high level of certainty.

Modern Maps

Maps that are strictly comparable to the earliest historical compilation were made by SFEI during the late 1990s (SFEI 2000) and in 2010 (SFEI 2011). Other maps that pertain to the 1970s-80s (Jones and Stokes Associates et al 1979, Dedrick 1989) are not strictly comparable to each other or to the oldest and newest maps. They provide, however, a general picture of the declining extent of tidal marshes and little change in the extent of tidal flats during the latter half of the last century.

The standards used by SFEI to produce maps of marshes and flats published in 2000 and 2010 are essentially the same. The ca 2000 map integrates data collected from 1996-1999, and was revised in 2000 based on aerial imagery and an exhaustive account of the status of marsh restoration and mitigation projects to support the Wetland Tracker information system (CWQMC 2011). The ca 2010 map is an update of the ca 2000 map using the same kinds of data sources. The standards (http://www.wrmp.org/docs/SFEI%20MAPPING%20STANDARDS_01062011_v3.pdf) are being reviewed by state and federal interests as the basis for a statewide inventory of aquatic resources that would intensify the National Hydrographic Dataset (NHD) of USGS and the National Wetland Inventory (NWI) of USFWS. The regional pilot is called the Bay Area Aquatic Resource Inventory (BAARI; SFEI 2010) and serves as the base map for the Bay Area Wetland Tracker (http://www.californiawetlands.net/tracker/). The reader is referred to the BAARI standards as the source of detailed information about the methods and data sources used to produce the ca 2000 and ca 2010 regional maps of tidal habitats.

Methods and Calculations

The total acreage of tidal marshes and flats was measured separately for each of the three regional maps (ca 1800, 2000, and 2010) by compiling the acreage measures for each separate marsh and flat using the Geographic Information Systems (GIS) at SFEI. The separate marshes and flats were identified based on the definitions provided above (see also the section below on the sizes of marshes and flats).

Forecasts of the net change in extent for 2100 were derived by adding expected acres of projects for creating or restoring tidal marsh or flats to the ca 2010 map. Projects were included in the forecasts if they are represented by Notices of Intent, Environmental Impact Reports or Statements, environmental permit applications, existing permits, or strategic planning documents that are available to the public and well supported in concept by the community of agencies responsible for intertidal habitat protection. Much of information about the status of projects is available online as project-specific web sites and information linked to the Wetland Tracker, plus a backlog of project information that has been submitted to the Wetland Tracker but is not yet available online. Planned enhancements of the quality of existing marshes or flats were disregarded because they won't affect a net change in extent. Projects that are required as mitigation for unavoidable losses of marshes or flats due to land use were considered only in terms of their net effect. This necessitated accounting for both the acres of habitat mitigation and the associated acres of habitat loss. Such data are not always available. It is expected that the missing data are insignificant relative to the overall uncertainty of the forecasts. Some projects are represented by multiple plans that have accumulated over many years due to waxes and wanes in funding. In these cases only the most recent plans were considered. Some projects replicate information from over-arching, long-range strategic plans that they will help implement. Care was taken in these cases not to double-count expected changes in extent. Some agencies and interests have different names for the same project, necessitating a careful cross-reference between plans and places.

Despite these efforts to assure an acceptable level of quality for the data sources, the forecasts are very uncertain. The large-scale strategic plans, such as the Goals Project (Goals Project 1999), South Bay Salt Pond Restoration Project (Coastal Conservancy 2006) and the Suisun Restoration Plan (USBR et al. 2010) involve adaptively adjusting their habitat goals as experience is gained. They explicitly or implicitly provide a range of possible future changes in habitat extent. In these cases, the forecasts relied upon the more conservative expectations for change. Some projects that are just entering environmental review may not be implemented as currently designed, or at all. It was assumed, however, that they would be implemented as currently planned. Only the intended end points of projects were considered in the tallies of possible future net change, although it was recognized that some projects will initially provide shallow subtidal habitats or tidal flats before they evolve into tidal marsh.

Climate change and economic change will probably affect the outcome of every project. Perhaps most of the uncertainty relates to climate change. While the restoration of intertidal habitats is not inexpensive (Goals Project 1999, USFWS 2009), the public has supported initial implementation of large-scale projects in recent years. To some degree public support can be improved with education and outreach that highlights the success to-date. Climate change, however, can have a direct and lasting effect on project outcomes and it cannot be managed over the time span of these forecasts. The central question is whether or not newly created or restored marshes and flats, as well as those that have persisted until now, will survive increased rates of sea level rise, as affected by global climate warming. The question can be re-stated as whether or not supplies of new intertidal sediment will be adequate for flats and marshes to build upwards apace with sea level rise. Many factors and processes complicate the possible answers (e.g., French 1993, Orr et al. 2003, Callaway et al. 2007, Glick et al. 2007, Craft et al. 2009, Stevenson and Kearney 2009, US Climate Change Science Program 2009). The further into the future the forecasts are extended, the less certain and meaningful they are. The year 2100 is probably near the limit of reasonable forecasting (IPCC 2001, 2007). Despite these uncertainties, the forecasts in this report are based on the assumption that the planned projects will achieve their currently envisioned endpoints by about 2050, and that they will survive as envisioned to 2100.

Tidal Marsh and Tidal Flat Size

There are multiple approaches to assessing the size of tidal marshes and flats. Size can be measured based on extent, as explained above, and it can be measured based on function. For example, a marsh that is too small to support a viable population of one species of wildlife might be large enough for another species. An area of marsh that might be large enough to enhance flood control for one place in the Estuary might be too small for another place. In a general sense, whether a marsh or flat is large or small varies with the functions for which its size is being measured. Furthermore, the rules for deciding how to define the boundaries of a marsh or flat also vary with their functions. For example, whether or not a marsh is large enough to benefit a particular animal species depends on how much of the marsh it can safely access. In fact, most of the interest and concern about tidal marshes and flats relate to their function as habitat for native fishes, animals, and plants (USFWS 2010, BCDC 2008, SFBRWQCB 2010). Therefore, one useful way to look at marshes and flats is as habitat.

In 2002, SFEI began a study of intertidal habitat fragmentation in the Estuary as part of a west coast survey of estuary condition that was sponsored by the Environmental Monitoring and Assessment

Program (EMAP) of USEPA. A regional team of experts was assembled to recommend rules for using GIS to delineate patches of tidal marsh and tidal flat as habitat for different species of mammals and birds (Table 1). Geographic features, such as broad areas of pen water, major roads, and levees that tend to block or otherwise influence the dispersal or daily movements of these species were identified as patch boundaries. Different species required different rules, based on their different responses to the geographic features. The alternative sets of rules were applied to the historical map (ca 1800) and the most current map at that time (ca 2000). The results illustrate that the marshes and flats have become more fragmented for some species than for others (Collins et al. 2005). It is important to note that the default rules for defining separate tidal marshes for these two maps and for the more recent map (ca 2010) follow "Alternative 1" for tidal marshes as described in Table 1 below. In essence, the marshes and flats depicted in these three maps are bounded by geographic features that tend to inhibit the dispersal or daily movements of resident small mammals, such as the endangered Salt Marsh Harvest Mouse, and resident rails, including especially the endangered California Clapper Rail. These maps therefore generally represent the distribution and abundance of habitat patches for these species.

Methods and Calculations

The maps of the past and present distributions of patches of tidal marsh and tidal flats were used to assess changes in patch size. There are alternative approaches to such assessment (Forman and Godron, 1986). The simplest approach is to calculate the change in average patch size. However, the same differences in average size can result from a large variety of changes in the distribution and abundance of patches. For example, there might be a change in the number of large patches, or in the number of small patches, or there might be a change in the maximum or minimum patch sizes. These are ecologically important aspects of patch size that are not evident in simple measures of average size.

Another, more informative approach is to calculate the change in patch size-frequency. Size-frequency is the number of patches per category of size, when the categories together represent the complete size range. This approach reveals the change in abundance for each size category as well as the change in average size. It involves no assumptions about the importance of any particular patch size. This is an important consideration when the patches represent a variety of ecological functions for which optimal size might differ. Size-frequency analyses can help address a variety of concerns about habitat conservation including habitat fragmentation (e.g., Dorp and Opdam, 1987. Andrén1994, Dickson 2001), ecological connectectivity (e.g., Diamond1975, Brown and Dinsmore 1986, Lindenmayer and Nix 1993, Rosenburg et al. 1997,), and risks of local or regional extinction or recovery of wildlife species (e.g., Fahrig and Merriam 1985, Wilcox and Murphy 1985, Soulé 1987, Lindenmayer and Fischer 2006). The challenge is to identify meaningful patch size categories.

Table 1. Alternative rules for analyzing habitat fragmentation for tidal marshes and flats.

Patch Type	Patch Definition	Reference Species

Patch Type	Patch Definition		Reference Species
Tidal Flat	Alternative 1	Patch boundaries are any or all of the following: (A) the foreshore of adjacent marsh, (B) any non-tidal area at least 200 ft wide, (C) any area of open water at least 200 ft wide at low tide, (D) any man-made levee as shown on 1:24k scale USGS topographic quadrangles, (E) any “large channel” (i.e., tidal marsh channel or tidal reach of river or stream that is at least 200 ft wide from bank-to-bank for most of its length, or that receives perennial freshwater discharge, or that extends across the tidal flat to the subtidal zone), (F) any roads. Having considered all rules above, two patches that come together at a point are considered two separate patches because the point of intersection creates a place of such high risk of predation that two patches are ecologically separate.	Resident infauna, and vertebrate fauna resident in adjacent tidal marsh
	Alternative 2	Same as Alternative 1 above except disregard large channels (i.e., tidal flat Alternative 1 boundary type “E” above).	Shorebirds, large wading birds, intertidal fishes
Tidal Marsh	Alternative 1	Patch boundaries are any or all of the following: (A) the foreshore, (B) any non-tidal area at least 200 ft wide, (C) any area of open water at least 200 ft wide at low tide, (D) any man-made levee as shown on 1:24k scale USGS topographic quadrangles, (E) any roads (4 lane or larger), (F) any “large channel” (i.e., tidal marsh channel or tidal reach of river or stream that is at least 200 ft wide in cross-section from bank-top to bank-top at most points along the channel length or that receives perennial freshwater discharge). Having considered all rules above, two patches that come together at a point are considered two separate patches because the point of intersection creates a place of such high risk of predation that two patches are ecologically separate.	Resident intertidal rails (this rule set also defines marsh patches that are separate contributors to the tidal prism of a large channel or the Bay).
	Alternative 2	Same as Alternative 1 except disregard any man-made levees from rule D that partition or separate tidal marsh or muted tidal marsh.	Resident intertidal passerine birds (especially intertidal song sparrows)
	Alternative 3	Same as Alternative 2 except also disregard any man-made levees from rule D that partition or separate abandoned salt ponds (except where flooded) and diked managed marsh.	Resident intertidal small mammals, intertidal amphibians and reptiles
	Alternative 4	Same as Alternative 3 except include low-salinity and medium-salinity salt ponds, treatment ponds and mudflats, upland fill less than 60 meters wide, and disregard rules E and F and all channels regardless of their width as barriers.	Waterfowl and shorebirds
	Alternative 5	Same as Alternative 4 except include farmed baylands. (This patch represents partial habitat within the tidal area).	Raptors and medium to large mammalian predators

For this report, alternative size categories were tested relative to a set of three basic criteria: (1) does every category contain patches for each period (ca 1800, ca 2000, ca 2010); (2) does the number of patches in most categories change from one period to the next; and (3) are their separate categories for the large, medium-sized, and small restoration projects. The latter criterion was needed to make sure the analyses were sensitive to restoration efforts. The same categories were used for all three time periods, but different categories were selected for marshes and flats. Changes in size-frequency for tidal flats were insignificant because the patches of flats have remained very large. The analysis of patch size therefore focused on tidal marshes.

Tidal Marsh and Wadeable Stream Condition

Definition

Tidal marshes are defined above (see the sections on extent and size for tidal marshes). A wadeable stream is a natural or artificial channel that can be safely crossed on the ground during low flow.

The condition of a marsh or stream is its existing potential or capacity to provide high levels of one or more of its needed ecosystem services. Ecosystem services are consequences of natural processes, functions, and management actions that benefit society (MEA 2005). In California, there are multiple, overlapping, and incompletely coordinated processes for to identify the kinds and levels of service that a marsh or stream should provide (see section below on benchmarks). For the purposes of this report, the conditions of marshes and streams are assessed relative to the conditions that generally correspond to high levels of a broad suite of services.

Estuarine Wetland CRAM

The data source for assessing the overall condition of tidal marshes is the statewide ambient survey of marshes conducted in 2007 (Sutula et al. 2008). The survey results are available online (http://www.cramwetlands.org/cramdisplay/), and are summarized in the recent State of the State's Wetland Report (Natural Resources Agency 2010). The details of the ambient survey including the sampling plan, sample size, and sample precision are provided in the survey report (Sutula et al. 2008).

The method used in the ambient survey of marsh condition is the Estuarine Wetland Module of the California Rapid Assessment Method for wetlands and wadeable streams (CRAM; Collins et al. 2008). A detailed explanation of CRAM plus the Estuarine Wetland Module used to assess tidal marshes are available online (http://www.cramwetlands.org/). CRAM is a standardized method used in the field by teams of 2-3 practitioners to assess the overall conditions of wetlands and wadeable streams relative to statewide networks of reference sites that represent excellent condition. The method assumes that, for any given kind of wetland or stream, the more structurally complex sites that are surrounded by more natural buffers and landscapes are likely to provide higher levels of their expected ecosystem services (Collins et al. 2008).

Riverine CRAM

Ambient surveys of the overall condition of wadeable streams have recently been conducted for the Napa River Watershed in Napa County and for the Coyote Creek Watershed in Santa Clara County. The survey results are available online (http://www.cramwetlands.org/cramdisplay/). The details of these ambient surveys of wadeable streams are provided in their separate survey reports (Sutula et al. 2008, SCVWD 2011).

Methods and Calculations

CRAM provides numerical scores for metrics that represent four basic attributes of condition: biological structure, physical structure, hydrology, and buffer-landscape context. Each metric can have one of four alternative scores that together represent the full range of possible conditions (Sutula et al. 2006, Stein et al. 2009). The metric scores for each attribute are summed to produce an attribute score for each site, and the attribute scores are summed to produce a site score. The attribute scores and the site scores are percentages of their maximum possible scores. It is assumed that every site has some ecological value, and therefore no site can have a zero condition score.

The ambient CRAM assessments of tidal marsh and stream condition were summarized as relative Cumulative Frequency Distributions (CFDs; NIST/SEMATECH 2001). CFDs were developed for the scores of each metric, each attribute, and all sites in each survey. This supported easy determinations of median and quartile scores. Since the surveys were based on probabilistic sample designs (Stevens and Olsen 2004), the CDFs could be used to estimate the percentage of marsh acreage or stream miles having scores above or below a particular score, or between any two scores, given the confidence limits of the CFDs.

Riparian Width

Definitions

The National Research Council of the National Academies has defined riparian areas as integral components of landscapes through which surface and subsurface movements of water interconnect aquatic areas and connect them to their adjacent uplands (Brinson et al. 2002). Riparian Areas are distinguished by gradients in biophysical conditions, ecological processes, and biota. They can include wetlands and portions of uplands that significantly influence the conditions or processes of aquatic areas. Based on this definition, every aquatic area including wetlands can be bounded by riparian areas. There is no minimum amount of plant cover, no requirement for particular kinds of cover, and the areas do not have to be natural.

Riparian areas have their own intrinsic ecosystem services (Gregory et al. 1991, Naiman et al. 2005). There are, for example, species of plants and animals that are largely restricted to riparian areas (e.g., Conard et al. 1977, Reed 1988, Fischer 2000, Bryce et al. 2002, RHJV 2004, white 2011), and riparian areas can serve as corridors for the dispersal, migration, and daily movements of terrestrial animals (Naiman et al. 1993, Fischer et al. 2000).

However, with regard to wetland and streams, riparian areas are generally regarded as buffers against external stressors, or as sources of materials that enhance wetland and stream services (e.g., Wenger 1999, Johnson and Buffler 2008, Ellis 2008.). In this regard, the kinds and levels of riparian services vary with riparian width (Wenger 1999, Polyakov et al 2005, Collins et al. 2006 and references therein). For example, the riparian area that stabilizes the banks of a stream tends to be narrower that the area that shades the same stream or supplies it with woody debris; the riparian area defined by hillslope processes that supply the stream with sediment tends to be wider than the area that provides woody debris. The broadest riparian areas tend to be defined by the spatial limits of effective habitats for riparian wildlife, especially riparian birds. This is the riparian concept that the SWRCB is considering while developing its Wetland and Riparian Area Protection Policy (http://www.swrcb.ca.gov/water_issues/programs/cwa401/docs/wrapp/tatmemo3_061610.pdf).

Riparian Buffer Decision Tool

Based on the riparian definition provided above, the USEPA, SWRCB, and the California Riparian Habitat Joint Venture have been sponsoring the development of a GIS-based tool for estimating functional riparian widths. The tool maps the riparian areas that correspond to bank or shoreline stability, shading, allochthonous input, sediment input, and runoff filtration, based on reported relationships among these services and topography, land use, and vegetation height (Collins et al. 2006). Pilot applications of the tool are occurring in Southern California (http://www.csun.edu/~centergs/data/SGR_FINAL_REPORT.pdf), the San Francisco Bay Area, (http://www.wrmp.org/docs/No569_WRMP_BasemapFactsheet_finalMay09.pdf) and the Tahoe Basin (http://tahoemonitoring.org/trt-charter.html).

Methods and Calculations

For this report, the Riparian Buffer Decision Tool was used to map the maximum extent of riparian areas for all the channels evident in the Bay Area Aquatic Resource Inventory (BAARI) for the two pilot watersheds. Riparian width was determined for both sides of each channel, beginning at the channel bank. Riparian width was mapped for all non-tidal channels longer that 30m. The minimum riparian width calculated by the tool is 1m. The maps are not constrained by any maximum riparian width. The riparian areas were classified as natural or unnatural, based on the degree to which the plan-form and/or structure of the associated channels had been modified. Each width class corresponds to a unique set of the riparian services listed above. It was assumed that, for any given location, the number of services that a riparian area is likely to provide increases with its width.

Stream Biological Integrity

Definitions

Biological integrity is a term that first appeared in the federal Clean Water Act in 1972. A variety of definitions have been developed since then (Cairns 1975, Karr and Dudley 1981, Hughes et al. 1982, Karr et al. 1986). It is commonly defined by USEPA as the capability of an aquatic area to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of a region (USEPA, http://www.epa.gov/bioiweb1/html/biointeg.html.) This is a practical working definition that is still broadly used. In 1981 USEPA produced a framework for developing indices of biological integrity (IBIs) that reflects this working definition. The framework continues to be revised as experienced with IBIs is gained (e.g., Barbour et al. 1996).

Benthic Macroinvertebrate Index and Benthic Index of Biotic Integrity

The water quality of streams is typically assessed based on chemical data, whereas their habitat quality is commonly assessed based on their form and structure as physical systems. One common approach to assessing the biological integrity of streams is to characterize their fish communities or benthic macroinvertebrate communities. The latter is more commonly used because many more streams support macroinvertebrates than fish. Aquatic insects are the most common organisms used in such assessments. It is generally understood that changes in structure (species composition and the relative abundance of species] of benthic macroinvertebrate communities reflect changes in environmental conditions (e.g., Vannote et a 1980, Vinson and Hawkins 1998, Brown and May 2000, LaBonte et al. 2001, Griffith et al. 2003), and that benthic macroinvertebrates are essential components of stream food webs (Vannote et al. 1980, Wallace and Webster 1996, Harding et al. 1998). Benthic bioassessments can integrate over time to provide robust measures of ecological impairment and rehabilitation (Hellawell 1986, Rosenberg and Resh 1993). Many states have incorporated benthic bioassessment into their stream monitoring programs to improve pollution control, guide abatement, and track regulatory compliance (Davis et al. 1996). Volunteer citizen science groups are increasingly using benthic bioassessment in watershed-based stream health care (Barbour et al. 1999, Clean Water Team 2011).

The Surface Water Ambient Monitoring Program (SWAMP) of the SWRCB has been developing regional IBIs for benthic macroinvertebrates (e.g., Rhyen and Ode 2006, Ode 2007, Rheyn et al. 2008). The development process has relied on many collaborators in different eco-regions to collect benthic data using the Benthic Macroinvertebrate Index (BMI) (Harrington 1999) along stressor gradients. The BMI focuses on the relative abundances of different groups of benthic macroinvertebrates that differ in their sensitivities to common stream stressors. Within a given eco-region, the BMI data can be calibrated to sites that are independently assessed as minimally impacted to developed a Benthos IBI (B-IBI). Local agencies within the Bay Area have been collaborating to develop a B-IBI for this region (Buchan et al. 2009). Starting in 2001, SWAMP has been surveying the integrity of Bay Area streams using, among other tools, the BMI (Breaux et al. 2005, Taberski et al. 2010). These SWAMP data were used to assess stream integrity for this report.

Methods and Calculations

The details of the BMI are available online (http://www.dfg.ca.gov/abl/Field/professionals.PDF). The method has varied somewhat since the Bay Area surveys began. Beginning in 2007, the focus of the method was switched from stream riffles to entire stream reaches (Ode 2007, Rehn and Ode 2007). Data for this region have been compiled for both sampling approaches, notwithstanding there differences. While the B-IBI is likely to be a more resolute than the BMI for assessing the integrity of Bay Area streams, it has not yet been fully implemented. In the meantime, the BMI has been determined to be useful for detecting differences in stream integrity that correlate to major stressors (Breaux et al. 2005, Taberski et al. 2010). In practice, the BMI is used to classify the health status of each assessed stream reach as excellent, good, fair, or poor.

Benchmarks

Each metric that is used to assess the condition of the region's aquatic habitats should ideally be calibrated to a broadly accepted, regional, numeric benchmark that represents the desired or needed condition. The benchmarks could be water quality objectives, habitat goals, population sizes to recover endangered species, or any level of ecosystem service that indicates excellent condition. This is not generally the situation, however. Some metrics that are directly linked to benchmarks are not adequately supported with data. Other metrics are well supported with data but lack a clear relationship to any benchmark. In the future, there should be a concerted effort to (1) identify the most important metrics for assessing the health status of the region's aquatic habitats; (2) establish benchmarks for the selected metrics that represent good health; and (3) fund efforts to collect the data that are needed to calculate the metrics.

<u>Tidal Marsh and Tidal Flat Extent</u>

The basis for the recommended benchmarks for the extent of tidal marches and flats is the 1993 California Wetlands Conservation Policy (http://ceres.ca.gov/wetlands/policies/governor.html). It's first two objectives are to ensure no overall net loss and to achieve a long-term net gain in the quantity, quality, and permanence of wetlands acreage and values in California. The state's anti-degradation policy is also designed to prevent declines in the quality of state's aquatic areas, although it focuses on the areas that have better conditions than necessary to provide adequate levels of service (SWRCB 1968). Neither policy stipulates a baseline acreage that cannot be further reduced, nor do they set a numerical target for how much more wetland acreage is needed. Instead, the Wetland Conservation Policy calls for regional and statewide wetland goals. Not long after this policy was published, a broad coalition of wetland interests including federal and state agencies began developing numerical (acreage) goals for intertidal habitats in the Estuary (Collins 1993, Goals Project 1999). It is reasonable to conclude that the tidal marsh acreage goals that were set in 1999 are consistent with, and have helped to implement the 1993 policy.

The Goals Project recommended that the Estuary downstream of the Delta should have no fewer than 100,000 acres of tidal marshes. This goal represents about 50 percent of the total acreage of tidal marshes that existed historically. The Goals Project also recommended how the acreage should be allocated among major subregions of the Estuary. This report, however, only addresses the extent of marshes relative to the overall regional goal.

No quantitative goal or benchmark has been set for increasing the extent of tidal flats. However, the 1993 California Wetlands Conservation Policy in conjunction with the state's anti-degradation policy suggests that the amount of tidal flat that existed in 1993 is the minimum acceptable extent for the future. In other words, the future extent of tidal flat should not be less than what existed in 1993. As stated in this report, the 1993 extent of tidal flats is about half the historical extent, and is therefore commensurate with the goal set for tidal marsh.

This report assumes that the extent of tidal marshes and flats that existed in 1993 is represented well enough by the ca 2000 map (SFEI 2000). Since the ca 2000 map is a compilation of information spanning the period 1993-2000, the change in extent that occurred in this period can be estimated. The change was positive and very small, relative to the total amount tidal flats. It mostly resulted from levee breaches designed for tidal marsh restoration, and therefore represented tidal flats that are expected to evolve into tidal marsh.

Tidal Marsh Size

The benchmark for tidal marsh size is the historical size-frequency of marshes ca 1800. There are three main technical questions about this benchmark: what is the correct set of rules for mapping individual patches; what is the correct set of patch size categories; and why is the historical size-frequency a reasonable template for the future.

Correct Patch Mapping Rules

Patches are defined by geographic features or changes in land cover that delimit selected functions or ecosystem services. For this report, patches were defined as habitat for resident wildlife, especially rails and small mammals, based on the best professional judgment about the kinds of features and land cover that inhibit their dispersal and daily movements (see Alternative 1 of Table 1 above). These rules can be refined as information about the behavior of these species increases. Studies to-date of the behaviors, habitat preferences, and movements of these species do not refute the mapping rules used in this report (Shellhammer et al. 1982, Geissel et al. 1988, Shellhammer 1989, Albertson 1995, Foin et al. 1997, Bias 1999, Albertson and Evens 2000, Hulst 2000, Shellhammer 2000, Schwarzbach et al. 2006, Overton 2007, Casazza et al. 2008). It is important to note, however, that these rules are general and not absolute. Individuals within these species may not follow the rules. For example, most resident populations of animals contain a small number of individuals that tend to disperse much further than the rest (Murray 1967, Koenig et al. 1996).

Correct Size Categories

The historical and modern maps of tidal marshes and flats are exhaustive. The sum totals of all the individual patches of marshes and flats are equal to their total regional acreages. No areas large enough to map are left out, based on the mapping standards. This is an essential requirement of any effort to assess extent.

There are many different sets of size categories that meet the same selection criteria. And, given that the maps are exhaustive and follow the same mapping standards, different sets of criteria can be applied to them. One of the basic advantages of standardized mapping procedures that generate exhaustive maps is that many different patch definitions and size categories can be applied to them to answer different questions. The criteria used in this report (see *Methods and Calculation* in section above addressing tidal marsh and flat size) support an analysis of the effects of restoration and mitigation projects on the overall extent and size-frequency of tidal marshes and flats. A variety of qualified size categories were tested and the one selected was most sensitive to the effect of past and proposed projects. Given a different set of criteria, a different set of size categories might be optimal.

Historical Size-frequency

Three basic assumptions underlay the decision to use the historical (ca 1800) patch size-frequency of tidal marshes as the model for their future patchiness. First, it is assumed that the current size-frequency distribution, which reflects almost two centuries of tidal marsh fragmentation, is not an appropriate benchmark or goal for the future. The patchiness that existed at the starting dates of the State Wetland Conservation Policy of 1993 and the anti-degradation policy of 1968 might indicate the maximum acceptable amounts of fragmentation, but they do not represent the needed deceases in fragmentation. Second, it is assumed the historical size-frequency sustained the native species that are currently threatened or endangered. Although the increased fragmentation of their habitats is only one factor in the declining abundance of these species, it has likely increased the negative effects of other factors. For example, as the marsh patches have gotten smaller, the ratio of their edge length to their surface area has increased, as has the distance between patches (Collins et al 2005), which in theory has increased the risk of predation, exposure to external stressors, and failure to disperse (Troll 1971, Forman 1995, Turner 1989, 2005, Fahrig 2002). It should be noted however, that declines in the total quantity of habitat and in its quality can out-weight the effect of fragmentation (Harrison and Bruna 1999). Third, larger habitat patches are usually better than smaller patches for sustaining local animal populations (e.g., Andrén 1994, Kolozsvary and Swihart 1999, Lindenmayer and Fischer 2006). The historical landscape included much larger tidal marsh patches than exist today.

The vertebrate communities of tidal marshes exhibit a high degree of endemism. Many species are entirely restricted to tidal marshes, and some are restricted to marshes of one or a few estuaries (Greenberg and Maldonado 2006, Greenberg et al. 2006, SBSPRP 2007). A reasonable assumption is that these species have adapted to the particular characteristics of the marshes they inhabit, including their hydrology, salinity regimes, vegetation, predators, as swell as the natural patchiness of their habitats.

This emphasis on categorical environmental patchiness as a determinant of community structure is common but not without controversy. The central concern is that the patch-based approach to the analyses of the distribution and abundance of plants and animals disregards the interactions between individuals or populations and gradients in their key resources and limiting factors (e.g., Cushman et al. 2010a,b). There are, however, gradients in habitat patch size within the geographic distribution of a species, and, for animals, these gradients usually include patches that are too small to support viable populations. In other words, patch size can be limiting for animals in highly fragmented habitats (Wilcox and Murphy 1985, Fahrig and Merriam 1985, Fahrig 2002).

There are numerous studies of tidal marsh animals in the Estuary that clearly indicate their distributions vary along environmental gradients independent of patch size (e.g., Atwater and Hedel 1976, Shellhammer 2000, Albertson and Evens 2000, Watson and Byrne 2009). This is not unusual for estuaries that are characterized by strong gradients in salinity and other physical factors. It does not necessarily mean, however, that patch size is not important. It means that patch size is one of many inter-relating factors that together affect the distribution and abundance of tidal marsh species over time. In the absence of any known optimal patch sizes for tidal marsh species in the Estuary, and given the negative effect of past habitat fragmentation on the prospects for their survival, setting an initial benchmark for future patch sizes that reflect the historical, natural patch size-frequency seems reasonable.

Tidal Marsh and Stream Condition

Careful analyses of the CRAM scores for both tidal marshes and wadeable streams revealed that the lower site scores were generally due to low scores for the physical structure attribute. Mean scores for this attribute were lower for Bay Area marshes than for marshes in other regions of the state. Mean scores for Bay Area streams were lower than for stream along the north coast.

For tidal marshes, the low scores for physical structure were mainly due to low scores for the metrics for topographic complexity and physical patch richness. This can be explained in part by the early stages of evolution of many of the marshes that were assessed. Tidal marshes gain physical complexity as they naturally evolve upwards through the intertidal zone (Redfield 1972, Orson et al. 1987, Kirwan and Murray 2007). Most of the Bay Area marshes have developed rapidly due to excessive sediment supplies resulting from an influx of hydraulic mining debris during the late 1800s (Atwater et al. 1979, Nichols et al. 1986, Dedrick and Chu 1993), and increased erosion in local watersheds due to nineteenth and twentieth century land use changes (Collins 2006, McKee and Lewicki 2009). These relatively young, rapidly accreted marshes lack the physical structural complexity of the remnant, higher, ancient marshes of the Estuary, which tended to get higher CRAM scores for physical structure. Higher scores for physical structure were also obtained for older marshes along the north coast. These findings support the recommendation in this report that the benchmark for future marsh condition should focus on CRAM scores for physical structure that are comparable to the natural, older marshes of this region and the north coast. Natural evolution of existing newly restored low-elevation marshes should eventually achieve this benchmark.

For wadeable streams, the low scores for physical structure were mainly due to the entrenched state of most of the assessed stream reaches. Entrenchment is caused by an increase in flows, relative to the size of the sediment loads that the stream must transport, or a decrease in sediment loads relative to the flow, or both (Lane 1955, 1957, Schumm 1969). It greatly increases the range of flows that are contained within the stream channel, which in turn increases the tendency of the stream to incise its bed (Schumm et al. 1984, Rosgen 1996). This in turn increases the degree of entrenchment until the bed encounters material that resists erosion, or the channel reaches a new equilibrium between the flow, the sediment load, and the channel form. The result of such chronic entrenchment is a loss of floodplains, riparian vegetation, large woody debris, persistent pools, and other features that together support many of the ecological services expected of rivers and streams. In many cases, managers must intervene to engineer a stable channel and/or to adjust upstream inputs of water and sediment. Both approaches are expensive, and they are not mutually exclusive. The former approach usually involves restoring floodplains, which in the Bay Area usually involves purchasing expensive lands. The latter approach usually involves changes in land use that can be politically challenging. It is unlikely, however, that the needed services of Bay Area streams can be attained and sustained unless their entrenched state is corrected. It seems appropriate therefore to set a benchmark for steam health that focuses on restoring the natural complexity of the streams, which will require long-term reductions in stream entrenchment.

Riparian Width

For the purposes of this report, which is focused on wetlands and streams, riparian areas are primarily regarded as buffers that protect wetlands and streams from external stressors. Riparian areas can provide one or more buffer services. In other words, they can buffer against multiple kinds of stress. Riparian areas can also provide their own ecological and social services, such as riparian wildlife support and recreation.

For any given topographic side slope and vegetation community, each riparian service tends to require a certain range in riparian width. The functional riparian widths can overlap, and greater widths tend to provide more kinds and higher levels of service. Setting benchmarks for riparian buffers therefore requires knowing what services are needed, and knowing how the existing riparian structure and setting must be modified, if at all, to provide the needed services. Benchmarks can therefore vary from place to place, depending on the stressors involved and what other services, besides buffering, are needed.

The approach to setting riparian benchmarks that was adopted for this report emphasizes the relationship between riparian width and riparian service, and recognizes that different stream reaches may have different benchmarks. The approach is similar to that used for tidal marsh size. According to this approach, future riparian areas should be distributed among categories of width according to their historical distributions. The rationale for this approach is the same for riparian areas and tidal marshes (see section above on Tidal Marsh Size). There are two main technical questions about this approach as it pertains to riparian areas: what is the correct set of riparian width categories, and why is the historical distribution of riparian areas among these categories a reasonable template for the future.

Correct Width Categories

The western literature concerning the relationships between riparian buffer width, structure, and buffer functions or services have been summarized multiple times since the 1980s (e.g., Clinnick 1985, Phillips 1989, Barling and Moore 1994, Desbonnet et al. 1994, Collier et al. 1995, Mander et al. 1997, Wenger 1999, Collins et al. 2006). These summaries provide a basis for recommending width categories. The correlations are not precise, however, and the categories tend to get wider (i.e., inclusive of larger areas of a landscape), with distance away from the waterbody. This is because the services that extend furthest from the waterbody are mostly about the support of riparian wildlife species that sometimes have large home ranges. Each of the more physical services, such as bank stability, shading, and allochthonous input do not extend as far as the wildlife support functions, are their extents are less variable. The categories of riparian width devised for this report are remarkably applicable to historical maps of riparian areas. This is further evidence of the general usefulness of the categories.

Historical width-frequency

The argument in favor of using the historical riparian width frequencies as a model for the future parallels that for tidal marsh patch size (se section above on Tidal Marsh Size). First, it is assumed that the current width-frequency distribution, which reflects almost two centuries of increasingly intensive landscape modification, is not an appropriate benchmark or goal for the future. The riparian widths that existed when the State Wetland Conservation Policy and the anti-degradation policy were enacted might indicate the minimum acceptable widths, but they do not represent the needed increases in width. Second, it is assumed the historical width-frequency sustained the native species that are currently threatened or endangered, including steelhead and salmon. Although the loss of riparian areas is only one factor in the declining abundance of these species (see section above on Stream Condition), it has likely increased the negative effects of other factors. For example, the loss of riparian forests along the Napa River and elsewhere in the region has decreased shading and the input of woody debris, which in turn have caused an increase in stream temperature and a decrease in stream channel complexity (e.g., Napolitano et al. 2003, Stillwater Sciences and Dietrich 2002). Third, as sated above, wider riparian areas tend to provide higher levels of more kinds of riparian services (Collins et al 2006). The historical landscape included much wider riparian areas than exist today.

Stream Biological integrity

Stream integrity can be defined in terms of abiotic as well as biotic factors and processes. It might be defined as the absence hydromodification (USEPA 2007, Mohamoud et al. 2009), the persistence of geomorphic stability (Leopold et al. 1964, Heede 1980, Rosgen 1994, Trush et al. 2000), or the support of reference communities of plants and animals (KARR 1999).

These various bases for defining stream integrity are broadly covered by the beneficial uses of waterbodies that are defined and designated by the SWRCB and its Regional Water Boards under the Porter-Cologne Water Quality Control Act Beneficial uses are the needed ecosystem services of a water body. (http://www.waterboards.ca.gov/laws_regulations/docs/portercologne.pdf). For example, water filtration is a marsh process that functions to improve water quality, which is a service incorporated into the beneficial use called "Estuarine Habitat;" the support of biological diversity is one of many services of marshes and streams that are incorporated in the beneficial use called "Wildlife" (http://www.swrcb.ca.gov/rwqcb2/water_issues/programs/planningtmdls/basinplan/web/bp_ch2.shtml).

Beneficial uses indicate the kinds, but not the levels, of services that a marsh or stream should provide. The needed levels of service are represented by water quality objectives. These are numerical or narrative descriptions of minimum conditions or levels of service that must be sustained to ensure that the waters of the state can support their designated beneficial uses. The SWRCB has initiated a process to set water quality objectives for streams (http://www.swrcb.ca.gov/plans_policies/biological_objective.shtml), and intends to set them for wetlands (http://www.swrcb.ca.gov/water_issues/programs/cwa401/wrapp.shtml).

As part of the state's effort to set water quality objectives for streams, the Surface Water Ambient Monitoring Program (SWAMP) of the SWRCB has been developing regional IBIs for benthic macroinvertebrates (B-IBIs) (e.g., Rhyen and Ode 2006, Ode 2007, Rheyn et al. 2008). The development process has relied on many collaborators in different eco-regions to collect benthic data using the Benthic Macroinvertebrate Index (BMI) (Harrington 1999) along stressor gradients. The BMI focuses on the relative abundances of different taxonomic groups of benthic macroinvertebrates that differ in their sensitivities to common stream stressors. Within a given eco-region, the BMI data can be calibrated to sites that are independently assessed as minimally impacted to developed a B-IBI. Local agencies within the Bay Area have been collaborating to develop a B-IBI for this region (Buchan et al. 2009).

The regional B-IBI will provide one method for scoring the biological integrity of streams relative to a regional standard. The SWRCB is encouraging practitioners to classify the level of integrity of each assessed stream reach as excellent, good, fair, or poor. There is no established benchmark, however, for the proportion of assessed reaches that should exist in any class. Based on the state's policies for wetland conservation and anti-degradation, it seems reasonable that the conditions existing at the time of the policies represent the minimum proportion of assessed reaches that have good or excellent levels of integrity. The benchmark for the future should indicate improved stream integrity region-wide. This means that the proportion of stream reaches classified as having good or excellent levels of integrity should increase. Given that only about 60% of the assessed reaches have these high levels of integrity at this time, an increase of 15% to reach a regional benchmark of 75% seems appropriate. As with any of the benchmarks, if monitoring shows that this one is not likely to be obtained, it can be revised downward. If it is likely to be surpassed, it can be revised upward. If additional indicators of integrity such as the algal IBI (Fetscher and McLaughlin 2008) are implemented in the region, they can be integrated together with the B-IBI to produce a more robust assessment of stream integrity.

Next Steps

The reported assessments of wetland and stream health are rudimentary. While the approach of assessing health status relative to established health goals or benchmarks is useful and workable, few benchmarks have been set and the data needed to set benchmarks or track progress towards them are scarce. One large step moving forward toward a more comprehensive assessment will be for the regional community of wetland and stream interests to prioritize the aspects of health (i.e., the wetland and stream services) that must be assessed, and then to develop benchmarks that define their ideal state. These decisions should be made with an aim to track the performance of wetland and stream protection policies and programs in the context of climate change.

The assessments of aquatic habitats in watersheds are especially weak because so few watersheds have been sampled. Although the assessments include some of the larger watersheds in the region, they do not represent the full regional range in watershed size, geology, dominant land use, or climate. The ambient assessments of watershed health should be extended throughout the region.

The capacity to track changes in habitat extent and overall condition using new maps and rapid assessment is increasing. The Wetland Tracker information system is being expanded to cover more wetland types with additional functionality including automated watershed delineation, on-screen mapping, and automated data summaries at user-defined scales. These new functions are being developed by SFEI with local partners. The intent is to develop local data stewards who can revise and update BAARI as needed and build CRAM into their local monitoring efforts. CRAM trainings continue to gain popularity as CRAM is incorporated into state and federal regulatory and management programs. CRAM and Wetland Tracker are being merged to enable the public to view and summarize CRAM results and mapping results for watersheds and for the region as a whole. These developments should improve the ease of assessment and the sharing of information about aquatic resources.

Maps and rapid assessment will not be able to track conditions for all the needed services of wetlands and streams. Field-based, quantitative measurements will be needed is some cases. For example, maps and rapid assessment by themselves cannot assess changes in the size of key wildlife populations or in the levels of contaminants. Standardized, quantitative methods of assessment already exist for some of these concerns, but others will be needed after the services and concerns are reviewed and prioritized.

In the context of setting benchmarks, tracking progress toward them, and reporting the results to the public, the importance of standardizing the data collection methods, providing adequate data quality assurance and control, and maintaining a common data library with broad accessibility cannot be overemphasized. These are essential elements of a comprehensive regional environmental monitoring program that can exist. The State of the Estuary reports can continued to foster public understanding and political will to improve the health of the Estuary and its watersheds. This should catalyze the support that is needed from the regional community of wetland and stream scientists.

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