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Design of an Aquatic Resource Status and Trends Monitoring Program for California

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Design of an Aquatic Resource Status and Trends Monitoring Program for California

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Doctor of Environmental Science and Engineering

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Leila Grace Lackey

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ABSTRACT OF THE DISSERATION

Design of an Aquatic Resource

Status and Trends Monitoring Program

for California

by

Leila Grace Lackey

Doctor of Environmental Science and Engineering

University of California, Los Angeles, 2012

Professor Richard F. Ambrose, Chair

Accurate estimates of wetlands and stream extent provide context for scientific investigations, enable informed management, and measure progress towards no-net-loss policy goals. However, the default approach for monitoring extent, comprehensive mapping, is prohibitively resource intensive over large areas, making it both impractical and statistically unreliable. Therefore, a number of national and state-level programs have begun to employ probabilistic approaches to monitor wetland and stream extent. These programs have proven practical for their intended applications but little information exists about the ability of the designs to meet diverse, state-level information needs such as accurately capturing rare wetland types or detecting small-scale changes in extent or spatial distribution. Here, I utilized a simulated sampling approach to empirically model and evaluate probabilistic methods for monitoring wetland and stream extent in California. The optimized design was then directly validated against comprehensive mapping through a pilot-scale implementation. The flexibility of the simulated sampling method enabled assessment of performance for a variety of objectives, beyond statistical precision. By employing this unique approach to sampling design, the results could be customized for the specific information needs of California. Results indicate that generalized random tessellation stratified (GRTS) sampling, a spatially balanced selection methodology, provides significant advantages over simple random sampling or stratification. In addition, fixed sampling locations over time provide the best power for both estimating the current extent and detecting changes in extent over time. The pilot-scale implementation was conducted in two separate regions of California. By using regions with existing, comprehensive aquatic resource maps, a direct comparison was made between the probabilistic and comprehensive mapping efforts. The pilot produced precise estimates of aquatic resource extent but suggested that mapping and classification methodologies may require more standardization in order to compare estimates between probabilistic approaches and comprehensive maps. In addition to systematic methodological differences, additional simulated sampling suggests that the expected accuracy of the 95% confidence interval is, in fact, dependent on the sample size. Final design conclusions from this dissertation will be recommended to the State of California for implementation in a California S&T program for aquatic resource extent.

The dissertation of Leila Grace Lackey is approved.

Peggy M. Fong

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2012

DEDICATION

For Danny, Liska, and Steve

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CHAPTERS TWO AND THREE

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CHAPTER FOUR

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CHAPTER ONE: INTRODUCTION

Context for and Current Status of Aquatic Resource Extent Monitoring in California

The State of California is commonly seen as a leader in environmental and aquatic resource protection. Several policy objectives, including a no-net-loss policy goal for extent and several State and Federal water quality protection legislative acts, are currently supported by a number of aquatic resource condition assessment programs (CWQMC 2008). However, few of these programs are statewide, include all aquatic resources, or monitor aquatic resource extent. As the State continues to invest in aquatic resource monitoring, protection and restoration, it has recognized the critical need for accurate and contemporary aquatic resource extent information (CWMW 2010). Previous approaches to aquatic resource extent monitoring have failed to meet this information need. Alternate approaches, such as mapping and estimation using randomly selected location, may be the best method for meeting the State information needs.

Need for Aquatic Resource Extent Information in California

Aquatic resources, such as wetlands, streams, and deepwater, provide numerous functions and services vital to the economy and the environment, such as flood control and prevention; water treatment, storage and distribution; biological habitat and food sources; irrigation and agriculture; aesthetic and recreational enjoyment; etc. (Millennium Ecosystem Assessment 2005). While not always the case historically, these economic, social, and environmental functions and services are now commonly recognized by scientists, managers, planners, and the public and are protected by federal, state, and local legislation (Costanza et al. 1997; de Groot et al. 2002).

Aquatic resources are one type of natural resource, a broad term that refers to physical, chemical, and biological environmental elements, typically with a perceived economic, social, or environmental benefit. Just as with other natural resources, aquatic resource extent inventories form the foundation and context for scientific investigation, policy and management decision making, regulatory action, development and resource extraction, conservation and restoration planning, and public inquiry (Cotterill and Foissner 2009; Nusser et al. 1998; O'Connell et al. 2004; Olsen and Schreuder 1997). Continual monitoring provides updates to said inventories, makes the information as relevant and useful as possible, and allows detection of trends in resource extent (Anderson 2002; Ståhl et al. 2010).

The US Environmental Protection Agency (USEPA) considers wetland resource mapping the primary component of their *Level 1-2-3* approach to wetland monitoring (USEPA 2006). Level 1 (L1) refers to landscape-level analysis and underpins rapid (L2) and intensive (L3) field-based assessments of condition. When combined, the three levels provide a complete picture of wetland extent, condition, functions, and stressors. This monitoring and assessment approach could easily be extended to all aquatic resources and to most natural resources. While there has been a tendency to focus on L2 and L3 tools, there is a growing recognition that the landscape level toolkit, such as probabilistic monitoring programs, must be further developed and refined in order to provide a stronger foundation for field-based assessment methods and results (Carletti et al. 2004; Mack 2006; Mita et al. 2007; Reiss and Brown 2007; Stein et al. 2009).

California aquatic resource assessment programs currently focus on condition. For example, the Surface Water Ambient Monitoring Program (SWAMP), operated by the State and

Regional Water Boards, monitors the physical, chemical and biological condition of California surface waters (SWAMP 2010). However, SWAMP does not include extent monitoring and has typically focused on stream assessments. The Resource Assessment Program (RAP), operated by the California Department of Fish and Game (CDFG), conducts wildlife, vegetation, and habitat condition assessments and coordinates assessments performed by other state agencies (CDFG). Habitat assessments for the RAP program include aquatic habitats but the program cannot provide a statewide estimate of aquatic habitat extent. Finally, the California Department of Parks and Recreation (CDPR) conducts an Inventory, Monitoring and Assessment Program (IMAP) for vegetation, habitat, and physical natural resources (CDPR 2001). IMAP includes aquatic resource habitat extent as one of its key endpoints but applies only to lands within the Parks system.

The State of California has attempted to quantify wetland extent and extent changes in two *State of the State's Wetlands* reports; one in 1998 and a second in 2010 (CNRA 2010; Wilson et al. 1998). The first report measured progress towards the no-net-loss policy goal by performing a regulatory accounting of changes in wetland extent (Wilson et al. 1998) . Regulatory accountings balance permitted destruction of wetland acreage, primarily authorized by Clean Water Act Section 404 dredge and fill permits, against wetland acreage gains via mitigation, restoration, and conservation. The 1998 report concluded wetland area had increased in California by 15,129 acres between 1996 and 1997. However, this extremely precise figure has several suspected inaccuracies including: failure to capture illegal or exempt wetland acreage losses, omission of non-regulatory or otherwise unreported restoration and conservation actions, inability to include natural changes in wetland extent, lack of physical verification to ensure reported wetland creation was successful, absence of wetland type information, etc.

The second *State of the State's Wetlands* report used the National Wetland Inventory (NWI) to estimate total wetland area in California at 2.9 million acres (CNRA 2010). The NWI is a nationwide map of wetland and deepwater extent, maintained by the US Fish and Wildlife Service (USFWS). However, the California portion of the NWI does not cover the entire state and variable map vintages and production methods decrease expected accuracy (Figure 1.1). The second report also evaluated no-net-loss progress, using regulatory accounting, but refrained from making a quantitative estimate of wetland gains or losses.

Between the first and second *State of the State's Wetlands* reports, several significant policy changes occurred at the State level. In the early 2000s, the California Environmental Protection Agency (Cal-EPA) began developing a coordinated Wetland and Riparian Area Protection Policy (WRAPP) to strengthen agency wetland regulation, assessment, and restoration activities. In 2007, the California legislature directed Cal-EPA to begin coordinating its wetland policies with the California Natural Resources Agency (CNRA) and other agencies around the state. The legislature charged CNRA and Cal-EPA with co-establishing a council to provide recommendations to improve the coordination and cost-effectiveness of water quality monitoring, enhance integration and data sharing across agencies, and increase public access to water quality information.

The two agencies created the California Water Quality Monitoring Council (CWQMC) and a number of working groups and technical committees, including the California Wetland Monitoring Workgroup (CWMW) for wetland-related activities. Inception of the CWQMC and the CWMW brought together all of the major state-level parties to ensure the policies enacted would be consistent with other efforts, and to expand the impact beyond CA/EPA and the WRAPP and its jurisdiction. In parallel with the 2010 *State of the State's Wetlands* report, the

CWMW released their current strategy document, *Tenets of a State Wetland and Riparian Monitoring Program (WRAMP)*, which the CWQMC directed all state agencies to follow (CWMW 2010; CWQMC 2010). As a result, the WRAMP document is now the primary statewide blueprint for California wetland and riparian monitoring.

The overarching objective in the WRAMP document is to: "produce regular reports on trends in wetland extent and condition and to relate these trends to management actions, climate change, and other natural and anthropogenic factors in a way that informs future decisions" (CWMW 2010). Tenets include a focus on basic questions, leveraging of existing programs, use of peer review, and implementation at the regional level with augmentation for regional needs. In addition, the document includes specific L1 recommendations such as adoption of a standardized statewide wetland definition, classification system, and mapping protocol and investigation of a probabilistic approach for aquatic resource extent. So far, the only completed component is a proposed wetland definition (Technical Advisory Team 2009).

The impetus for a probability-based extent monitoring program in California is a need for aquatic resource extent information that is (as compiled from the WRAMP and communication with the Technical Advisory Teams to the WRAPP Policy Development Team):

(1) statewide in scope but allows flexibility for regional or issue-specific investigations;
(2) includes all aquatic resources and provides extent information for specific aquatic resource types;
(3) and allows estimation of trends in extent and associates changes in extent with likely causes.

In addition, estimates of extent and changes in extent should be both accurate and precise, where accuracy refers to deviation of observations from the true value while precision refers to repeatability of observations. The next two sections will outline existing sources of extent information, and their shortcomings, and review the different approaches available for meeting California aquatic resource extent information needs. Then, the following chapters will discuss how design options for a probability-based program can be selected and optimized to meet California information needs. The dissertation work will perform the selection and optimization while providing a blueprint for creating similar programs to meet information needs for other natural resources and other locations.

Current Aquatic Resource Extent Information

Aquatic resource extent information for California is available from academic, state, and federal sources. While all provide useful information and satisfy the objectives they were established to fulfill, none currently meet California aquatic resource extent information needs. Academic studies can provide accurate, precise, contemporary, and California-specific extent information, estimate trends, and correlate trends with possible causative mechanisms; however, academic studies typically do not cover the entire state and rarely include all aquatic resource types (Glenn et al. 2006; Greer and Stow 2003; Phinn et al. 1996). State programs, such as those mentioned previously, either perform regulatory accounting or focus specifically on condition assessment (CDFG; CDPR 2001; CNRA 2010; SWAMP 2010; Wilson et al. 1998). Regulatory accounting may give contemporary, California-specific, statewide estimates of extent and trends for all, or most, aquatic resources but the estimates are unlikely to be accurate or precise, to provide type-specific information, or to correlate trends with possible causes. In addition, regulatory accounting cannot meaningfully support condition assessment as part of a level 1-2-3 monitoring strategy. Likewise, monitoring programs specific for condition assessment may provide contemporary, California-specific, statewide information on several aquatic resource types but, because they focus on condition, cannot provide accurate and precise extent and trend information.

Federal programs generally provide extent information, for a variety of aquatic resource types, and support condition assessment by providing basemap information, but only two of the four programs reviewed provide contemporary extent and trend information for the entire State of California. In addition, while all four of the reviewed programs include the entire United States, their accuracy, precision and specificity within California is limited. The four federal sources of aquatic resource extent information include two comprehensive mapping efforts, the National Hydrography Dataset (NHD) and the NWI, and two probability-based programs, the status and trends (S&T) component of the NWI (NWI-S&T) and the Natural Resource Inventory (NRI). Here, "status" refers to the current extent of aquatic resources and "trends" refers to changes in extent over time.

The two comprehensive programs (the NHD and NWI) estimate the total extent of aquatic resources by attempting to map all aquatic resources, nationwide. In contrast, the two probabilistic mapping programs (NWI-S&T and NRI) map a randomly selected portion of the locations according to the following approach (Figure 1.2). First, a regular grid is placed over the target area. Then, a randomly selected sample of the grid cells is selected and the area of aquatic resources within each cell is determined. Finally, the fraction of the entire area covered by aquatic resources is estimated by taking the average of the individual cells (Ståhl et al. 2010). All of the programs discussed here use aerial photography and remote sensing to produce the resource maps.

The first comprehensive program, the NHD, is a nationwide map of surface water features such as lakes, ponds, streams and rivers, produced by the US Geological Survey (USGS) (USGS 2000). The dataset was compiled from a mosaic of topographic maps and aerial photographs and includes definite channels at least 1 mile in length. This length minimum means the dataset may not include some ephemeral or seasonal streams, important components of California aquatic resources (Gasith and Resh 1999; Riley et al. 2005; Welsh et al. 2010). Variability in source map age, scale, and quality has resulted in a range of errors such as discontinuities in mapping or classification along topographic map boundaries; linework mismatches caused by mapping from planimetric, as opposed to topographic, maps; misclassifications or flow direction misidentifications; etc. The first iteration of the NHD was released in 2001 and a high-resolution version completed in 2007. Several of the known errors were partially improved by the NHD-plus, released in 2006 from a partnership between the USGS and the US Environmental Protection Agency (USEPA) (USEPA and USGS 2010). The NHD-plus also provides additional information for use in modeling and analysis applications. In conclusion, the NHD and NHD-plus have comprehensive, statewide coverage in California and provide extent information for a range of aquatic resources, although not for some wetlands or streams. However, the NHD and NHD-plus do not provide trend information and heterogeneity in map age and quality limit overall accuracy and precision.

The second comprehensive program, the NWI, began in 1974 and is operated by the US Fish and Wildlife Service (USFWS) (Tiner 2009). Wetlands are mapped from aerial photographs but, due to the significant costs associated with producing wetland maps by hand, the NWI is currently incomplete and includes maps from the 1970s to the 2000s. In California, approximately 80% of the state has been digitally mapped at this time (Figure 1.1). In addition, the wetland definition and classification system used may not adequately characterize wetlands in regions where evaporation exceeds precipitation — such as significant portions of California (Cowardin et al. 1979; NWI 1997). In conclusion, the NWI has statewide coverage in California, except for unmapped areas and provides wetland and deepwater extent information by type. However, trend information cannot be estimated and the variability in age and mapping methods (Figure 1.1), as well as the classification procedure, limit precision and accuracy in California.

The first probability-based S&T program, the NWI-S&T, was developed in response to some of the difficulties encountered when producing the comprehensive NWI (Tiner 2009). The probability based design adopted by the NWI-S&T maps approximately four thousand, randomly selected 4 mi^2 plots ever 5-10 years and has produced four nationwide reports since 1983 with another due in 2011 (Dahl 2006; Dahl 2000; Dahl and Johnson, C. E. 1991; Frayer et al. 1983). The reports include estimates of wetland extent and trends overall and for specific wetland types; see page 23 of Dahl (2006). The degree of regional coverage, and therefore the confidence in the change estimates, varies significantly from state to state but generally provides more contemporary information than the NWI. In summary, compared to the NWI, the NWI-S&T provides contemporary extent and trend information, for several aquatic resource types. However, California contains only 257 NWI-S&T plots, covering approximately 0.6% of the land area, and most are concentrated along the coast (Figure 1.3). The sparse and uneven coverage may reduce the accuracy and precision of NWI-S&T estimates for California-specific results.

The second probabilistic S&T programs, the NRI was created by the US Department of Agriculture (USDA) to produce regular reports on the condition and extent of a wide variety of land, soil, water and other resources, on non-federal lands (USDA 2007). Extent and trends estimates are produced every five years for most target variables, including wetlands and other aquatic resources. Estimates are also available for California. In summary, like the NWI-S&T, the NRI provides contemporary extent and trends information, for several aquatic resources. The NRI also provides specific, statewide information for California. However, some of the design and mapping approaches taken by the NRI, described in detail later, may affect the accuracy of the information obtainable from the NRI. As a result, the usefulness of NRI data in California is limited.

Approaches for Meeting California's Aquatic Resource Extent Information Needs

To reiterate, California's aquatic resource extent information needs requires a monitoring program that is

(1) statewide in scope but allows flexibility for regional or issue-specific investigations;
(2) includes all aquatic resources and provides extent information for specific aquatic resource types;
(3) and allows estimation of trends in extent and associates changes in extent with likely causes.

Aquatic resource extent information is needed in order to assess progress toward policy goals and to support the State's numerous condition assessment programs. It is clear from the previous review that there is no current source of this information. However, the review does point to certain approaches and design options most likely to provide the desired information.

The previous discussion includes three major approaches to aquatic resource S&T: regulatory accounting, comprehensive mapping, and probabilistic mapping. Incorporation of all three into a coordinated strategy could prove advantageous as each offers useful, but different, information and drawbacks. Regulatory accounting, such as was performed in the two *State of the State's Wetlands* reports, can provide contemporary, statewide information about wetland extent and trends (CNRA 2010; Wilson et al. 1998). However, methodological limitations negatively affect accuracy and precision.

In principle, comprehensive and probabilistic methods should both meet all of California's information needs. Comprehensive mapping is considered the gold standard because it can provide detailed, contiguous information, for every location in the target area, without assumptions or inferences (Thackway et al. 2007). Mapping based on a probabilistic sample of the target area cannot provide contiguous, statewide maps but can estimate extent and trend information for the State as a whole (McRoberts 2011; Olsen and Peck 2008; Olsen and Schreuder 1997; Thackway et al. 2007). In addition, probability-based approaches require significantly less resources, allowing for more frequent production of maps and extent and trend estimates; can still support condition assessment; and incorporation of model-based interpolation can provide resource extent estimates for each point in the target area without requiring comprehensive data (McRoberts 2011; Stehman 2000).

Most states incorporate regulatory accounting and comprehensive mapping but few also perform ongoing monitoring to update the comprehensive inventory. For example, the State of Oklahoma is digitizing all available NWI maps but has no ongoing mapping program (Oklahoma Water Resources Board). New York State mapped state-regulated freshwater wetlands between 1984 and 1987 but has only amended those maps, by public notice-and-comment, for changes of regulatory significance (New York State Department of Environmental Conservation). Finally, helped by its small size, the State of Delaware regularly updates their comprehensive wetland maps to reflect changes (Biddle 2010). The ease of producing and updating comprehensive maps makes a probabilistic S&T element unnecessary for this small state.

For larger states, one major downside of adopting a comprehensive S&T programs is: if insufficient resources prevent the program from completing the comprehensive map, and the areas mapped were not chosen randomly, the results cannot be generalized to the remainder of the target area without violating the assumptions of design-based inference (Hedt and Pagano 2011; Ludwig 2005). This statistical consideration, and the fact that a probabilistic approach has much lower costs, has led some states to start exploring probabilistic mapping approaches. For example, the Wisconsin Wetlands Inventory, began in 1978, attempted to update a comprehensive wetland map every 10 years; however, budget constraints have increased the interval to 20 years at best (Wisconsin Department of Natural Resources 1992). Therefore, the State of Wisconsin is now performing a pilot project in the Lake Michigan Basin to determine the feasibility of a probabilistic S&T program (personal communication with Thomas Bernthal, Wisconsin Department of Natural Resources).

The State of Minnesota is, so far, the only state implementing a statewide, probability-based approach to aquatic resource S&T, in addition to regulatory accounting and comprehensive mapping (Minnesota Pollution Control Agency 2006). The Minnesota Status and Trends (MN-S&T) program begin in 2006 and was designed and optimized using available NWI and NWI-S&T maps (Deegan and Aunan 2006). The program intends to produce statewide and regional extent and trend estimates every three years (Minnesota Pollution Control Agency 2006).

While not necessarily as coordinated as Minnesota, the federal government also operates regulatory accounting initiatives, including the recent *National Wetlands Mitigation Action Plan*;

comprehensive mapping programs, including the NWI, NHD and NHD-plus; and probabilistic S&T programs, including the NWI-S&T and NRI (USEPA 2002).

California currently operates regulatory accounting and comprehensive mapping programs but not a probability-based mapping program (CWMW 2010). The California Wetlands Portal and its Project Tracker provides public, online information on permitted actions and restoration efforts and the NWI is consistently expanded and updated by new maps produced by academic, state-funded, and federally-funded mapping programs (CWMW 2010; CWQMC). However, these two data sources do not meet all of California's aquatic resource extent information needs. Specifically, regulatory accounting cannot provide accurate or precise aquatic resource extent and trend information because the approach misses illegal, exempt, unreported, or natural changes and provides no mechanism for determining the total extent. In addition, comprehensive mapping has, in practice, failed to provide contemporary, statewide aquatic resource extent and trend information because of the significant time and resource costs associated with comprehensive mapping.

Structure of this Dissertation

A probability-based S&T program in California has the potential to provide statewide, contemporary, aquatic resource extent and trend information, for specific wetland types, capable of supporting condition assessment, and with additional information to associated trends with possible causes. Accuracy and precision of the extent and trend estimates will depend on appropriate selection of design and estimation options. The following chapters will discuss the selection process for three important components of all monitoring program designs: spatial sampling, temporal sampling, and estimation. These are also areas where different approaches can have significant impacts on cost-effectiveness, accuracy, and precision. Careful exploration will produce a set of design options to meet California aquatic resource extent information needs, expand the design literature, and provide useful information for others developing similar programs in other areas and for other natural resources.

Chapter 2 investigated the selection of a spatial sampling design for probabilistically measuring aquatic resource extent. Key parameters evaluated in this chapter were the sample selection method, use of stratification, and plot size. Results were optimized based on statistical accuracy and precision and on the aquatic resource information needs of California. Next, Chapter 3 evaluated temporal sampling designs for monitoring aquatic resource extent over time. This study compared the statistical performance of fixed, un-fixed, and a mixture of fixed and unfixed sample locations over time. Results were evaluated based on statistical performance and the practical considerations of monitoring aquatic resource extent. Finally, Chapter 4 presents the results of a pilot-scale implementation of the spatial sampling design optimized in Chapter 2. The pilot was conducted in areas of California with existing comprehensive maps for comparison. Results from these three chapters were then combined to produce a set of design recommendations for monitoring aquatic resource extent in California. Finally, Chapter 5 synthesizes the design recommendations and provides additional context for our results and conclusions.

Figures

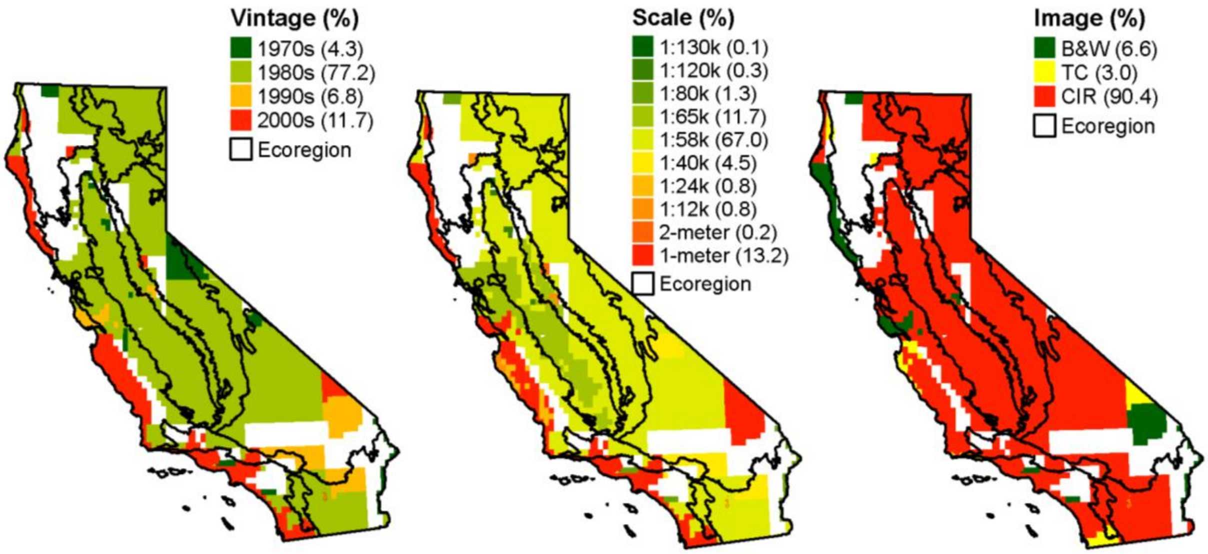


Figure 1.1. Areas within California with digital NWI coverage.
Geographic areas are color coded and plots show the fraction of the mapped area for each map year (A), mapping scale (B), and image type (C).

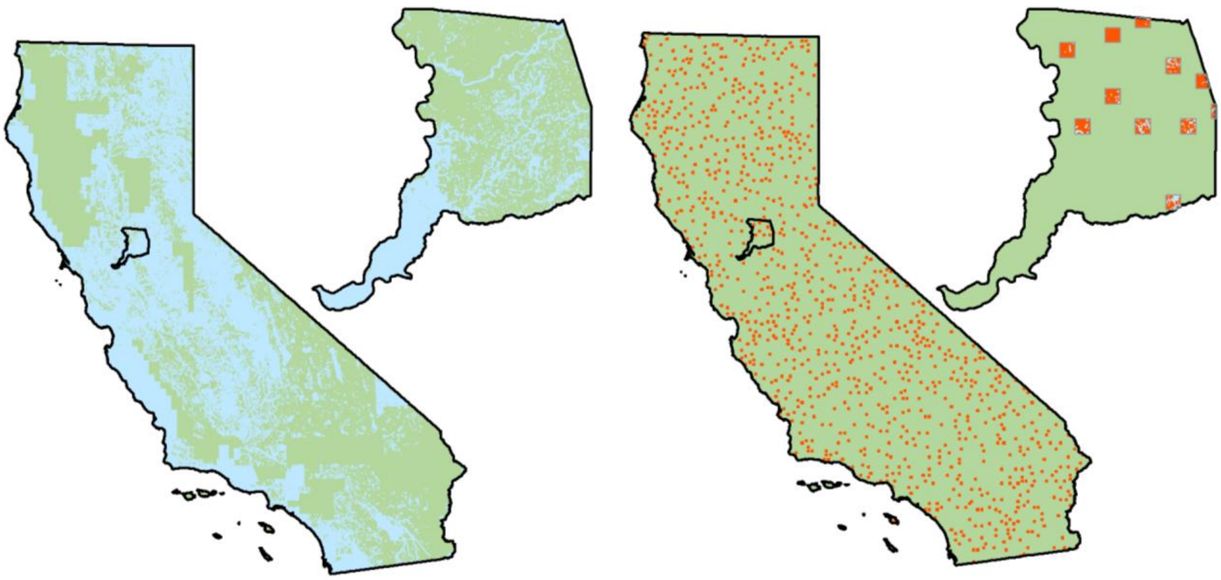


Figure 1.2. Comprehensive maps versus randomly selected points.
NWI wetland polygons in California and Marin County (insets) from the comprehensive map (A) and for randomly selected plots (B).

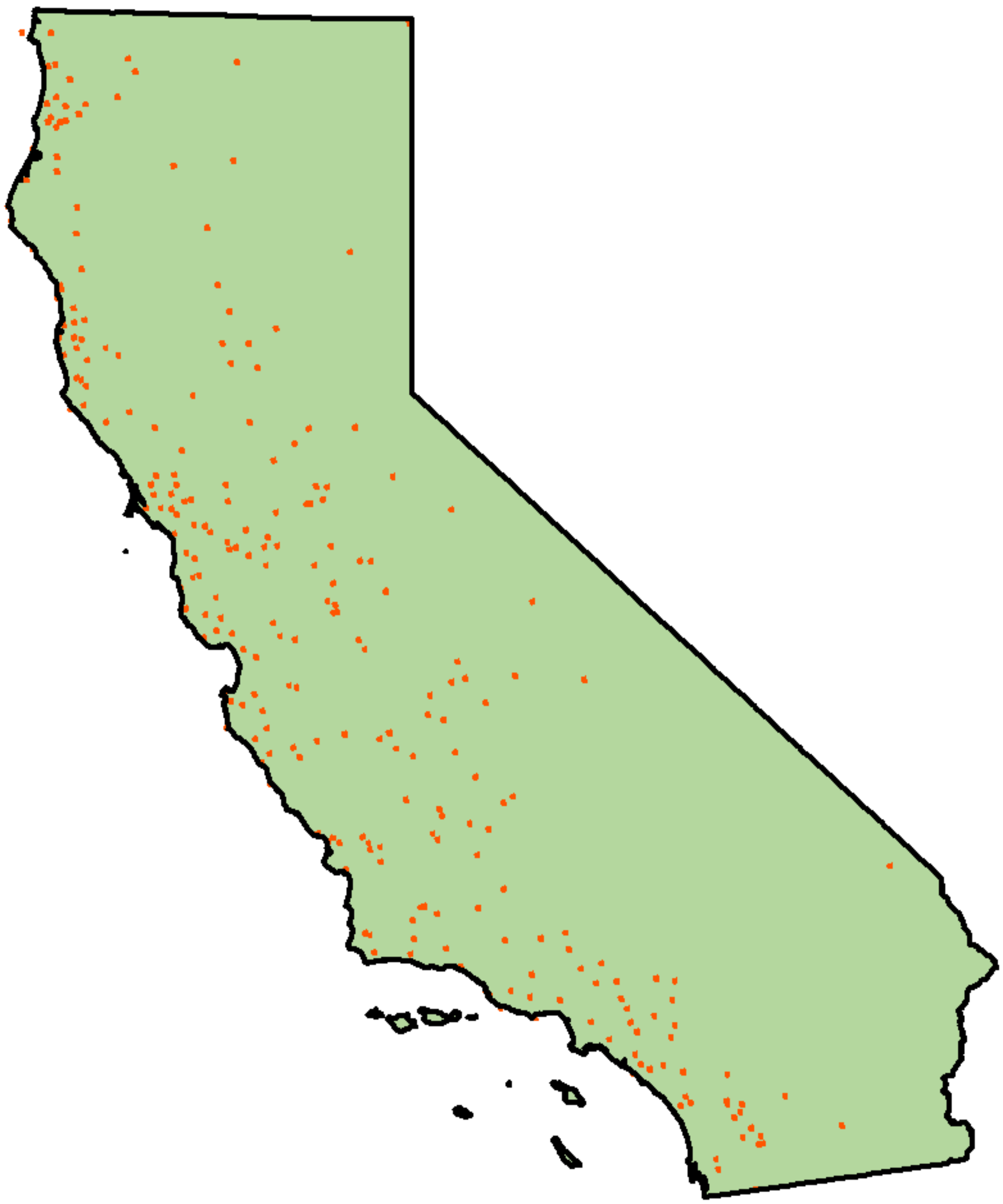


Figure 1.3. Location of NWI-S&T plots in California.

CHAPTER TWO: SPATIAL SAMPLING DESIGN

Design-based Sampling for Aquatic Resource Extent Monitoring in California

Abstract

Accurate estimates of wetlands and stream extent provide context for scientific investigations, enable informed management, and measure progress towards no-net-loss policy goals. However, the default approach for monitoring extent, comprehensive mapping, is prohibitively resource intensive over large areas, making it both impractical and statistically unreliable. Therefore, a number of national and state-level programs have begun to employ probabilistic approaches to monitor wetland and stream extent. These programs have proven practical for their intended applications but limited information exists about the ability of the designs to meet diverse, state-level information needs such as accurately capturing rare wetland types or monitoring small-scale changes in extent or spatial distribution. This study utilized simulated sampling to assess the performance of a suite of sample design options for monitoring the extent of wetlands and streams in California. The flexibility of the simulated sampling method enabled assessment of performance for a variety of objectives, beyond statistical precision, important for a proposed California monitoring program. By employing this unique approach for sample design, the results could be customized for the specific information needs of California. Results were able to support recommendations related to sample selection method, use of stratification, and appropriate plot sizes. This approach can also be extended to other state, tribal, and regional programs and to support development of local, probability-based approaches to monitoring the status and trends of wetland and stream extent.

Introduction

Accurate estimates of wetland and stream extent and distribution provide a foundation for scientific, management, and policy actions. For example, probabilistic assessments of wetland and stream ambient condition rely on extent and distribution information to select sample locations and to determine generalizability of results (wetlands and streams are jointly referred to here as aquatic resources or features) (Albert et al. 2010). Extent also forms the basis for assessing progress towards state and federal no-net-loss policy goals (Mitsch and Gosselink 2000; Nusser and Goebel 1997). Finally, extent and distribution is critical for informed management decision making, such as for compensatory mitigation (Baron et al. 2002; Clare et al. 2011). Comprehensive inventory and mapping, such as the National Wetland Inventory (NWI), has been the preferred approach for monitoring aquatic resource extent and evaluating compliance with policy. Comprehensive maps have been preferred because they can provide detailed information for all locations without assumptions or inference; are easy to understand; and can readily convey information to policymakers and the public.

While comprehensive mapping is an attractive approach for monitoring extent and distribution, it is an inadequate approach for large areas. Under a comprehensive approach, the entire area must be mapped in order to provide unbiased estimates of area-wide parameters, such as total wetland area or total stream length (Gregoire 1999; Nusser et al. 1998). For large geographic areas, insufficient resources have frequently prevented timely completion and updating of comprehensive aquatic resource inventories (Ståhl et al. 2010; Tiner 2009). As a result, these comprehensive inventories have failed to provide estimates of total extent for a single point in time. In addition, temporal variability, evolution in mapping approaches and technology, and a “convenience” type approach to selecting mapped locations means production of estimates of total extent and trends is problematic or technically infeasible. In addition, if an estimate is produce, determination of the level of uncertainty in that estimate can be problematic. For example, the NWI, begun in the 1970s by the US Fish and Wildlife Service, has yet to produce a complete, national map of wetland extent (Tiner 2009). The current NWI covers less than two thirds of the country and is composed of maps produced between 1970 and the present. As a result, the NWI provides neither an estimate of current national wetland extent nor a clear mechanism for determining change with time.

In contrast to comprehensive inventories, statistical sampling and mapping employs a probabilistic approach to produce extent and trend estimates more frequently, and from significantly fewer resources (Olsen and Peck 2008). By mapping a portion of the target area, observations can be completed at a single point in time and repeated at regular intervals, enhancing ability to estimate extent and detect trends. While probabilistic sampling and mapping obviously does not produce a complete map of aquatic resource, the approach can provide unbiased estimates of area-wide extent and the uncertainties in that estimate (Albert et al. 2010). For example, while the NWI has yet to map the entire US, the NWI Status and Trends program (NWI-S&T) has produced five reports over the last thirty years (Dahl 2011). These reports catalog significant losses in wetland area between the 1950s and today. Similar probabilistic programs include the Minnesota Wetland Status and Trends Monitoring Program (MN-S&T), operated by the Minnesota Department of Natural Resources (MN-S&T); and the National Inventory of Landscapes in Sweden (NILS), operated by the Swedish Environmental Protection Agency (Kloiber 2010; Ståhl et al. 2010).

While the three programs mentioned above illustrate the promise of probabilistic sampling for monitoring aquatic resource extent, additional analysis and optimization is necessary to broaden the applicability of this type of monitoring for local applications. For example, it is unclear whether a national design, such as that used by the NWI-S&T, can meet state-level needs for extent and distribution information. Of the programs mentioned above, only the NILS was designed to also monitor streams, which have a significantly different landscape distribution from wetlands (Ståhl et al. 2010). In addition, new sampling and analysis tools have been developed since the NWI-S&T program was designed and none of the programs were designed based on a comprehensive evaluation of the variability in aquatic resource extent (Dahl 2011; Kloiber 2010; Ståhl et al. 2010). Finally, program design considerations typically extend beyond the statistical precision of a single estimate. For example, the State of California intends to utilize the S&T program maps as a sample frame for field-based studies of wetland and stream condition. This study used a model-based, simulated sampling approach to assess the statistical performance of different design options for monitoring wetland and stream extent in California. We then used study results to recommend a California S&T program design capable of satisfying the monitoring goals of the State of California.

Background: Program Objectives and Design Options

The primary objective for the California S&T program is to provide state-level estimates of the extent and distribution of aquatic resources and how this extent and distribution is changing with time. Estimates should also be provided for key resource subtypes and be customizable for various ecological, political, and administrative regions of interest. Existing probabilistic monitoring programs can provide a starting point for developing a sampling design capable of meeting the objectives of the California program. All existing programs treat the target area as a finite population of sample plots, laid out in a regular grid (Dahl 2011; Kloiber

2010; Ståhl et al. 2010). Therefore, the fraction of the target area covered by the aquatic resource of interest is easily estimated by design-based sampling and inference (Albert et al. 2010; Gregoire 1999).

This design approach is independent of the distribution of aquatic resources and does not require or utilize a pre-existing basemap of aquatic resources. Within this basic, probabilistic design, several key questions remain. Can the sample design balance measurement of wetlands, which have a patchy distribution, with measurement of streams, which are more evenly distributed? Can a probabilistic design adequately monitor rare wetland and stream types? Can the resulting sample be analyzed for all subpopulations and regions of interest? To explore these and other issues, we investigated three aspects of sample design: sample selection method, spatial stratification of the target area, and plot size.

Sample Selection Method

The sampling designs described here all start with a continuous grid placed over the target area. All grid cells within the target area (in this case California) are considered part of the population and the presence or extent of aquatic resources within a cell does not affect inclusion. Aquatic resource extent is then compared between cells based on the fraction of cell area covered by aquatic resources. Thus, comparisons are based on area, not number, and an individual aquatic feature may be split between several cells.

Statistically, each grid cell is considered an independent individual. However, geographically, adjacent grid cells are obviously closely related and the cell boundaries represent an artificial division of the landscape. Tobler's first law of geography, "everything is related to everything else, but near things are more related than distant things" (Tobler 1970). This concept, referred to as spatial autocorrelation, suggests that spatial relationships and proximity will affect our observations of aquatic resources and may influence the effectiveness of the spatial sampling design. For instance, spatially "clumped" locations may increase autocorrelation in the sample as the clumped observations effectively amount to repeated observations of a closely related subset of the landscape. In contrast, spatially "balanced" observations, spaced relatively evenly across the entire landscape, may reduce spatial autocorrelation and better represent the diversity across the population as a whole. Therefore, methods that produce a spatially balanced sample could theoretically improve the representativeness by reducing spatial autocorrelation (Chen and Wei 2009). This may also improve the accuracy and precision of the sample.

In previous simulation work, spatially balanced sampling methodologies have successfully reduced sample variance compared to non-balanced methods, such as simple random sampling (SRS), which can produce clustered samples (Theobald et al. 2007). Nevertheless, SRS is still commonly used, including by the NWI-S&T program, because of ease of implementation and communication of results (Dahl 2011). Systematic sampling is the simplest spatially balanced design to implement. In this approach, used by the NILS program, sampling locations are selected using a regularly spaced grid (Ståhl et al. 2010). However, systematic designs may align with spatial patterns in the population and unbiased variance estimation requires knowledge of the spatial variability of the population (Flores et al. 2003). Generalized random tessellation stratified (GRTS) sampling combines the advantages of SRS and systematic sampling and is used by the MN-S&T program (Kloiber 2010). GRTS provides better spatial balance than SRS by basing sample selection on a hierarchical, square grid placed over the sample area. GRTS also maintains a random distance between adjacent points by randomizing the selection order of grid cells (Deegan and Aunan 2006; Stevens and Olsen 2004). Supplemental File S2.1 provides an introduction to the technical details of the GRTS sampling methodology.

Stratification

Closely related to selection method is stratification, which can be utilized to improve sample accuracy and precision across heterogeneous areas. Conceptually, stratification benefits sampling accuracy and precision by dividing the population into homogeneous subsets. The expectation is that the homogeneous units will be better described if sampled and analyzed individually. Then, these more precise and accurate stratum-level estimates can be aggregated to produce a more precise and accurate estimate of the whole population. However, stratification can also reduce flexibility in sampling execution and analysis. For instance, complex reweighting procedures are required if sample estimates are required for subsets other than the sampling strata. In contrast, post-stratification is an extremely simple procedure if the entire study area is sampled equally. In addition, other methods, such as spatially balanced sampling, may more easily and reliably increase the accuracy and precision of the overall estimate. Finally, stratification to improve overall precision relies heavily on accurate knowledge of the population (information needs are described in detail below), which is not always available. Therefore, stratification may not be necessary or appropriate if results are not required for certain subpopulations or if there is insufficient pre-existing knowledge of the population to support the stratum allocations.

Stratification with proportional allocation is similar to spatially balanced sampling as the number of sample locations in each stratum is proportional to the size of the stratum. This approach, used by the NILS, can increase sample accuracy and guarantee adequate sample sizes for subpopulations of interest (Brus and Knotters 2008; Ståhl et al. 2010). In contrast, optimum allocation reduces sample variance by allocating sample locations to individual stratum according to both the size and the variance of the population within each stratum (Bosch and Wildner 2003). This approach, used by the NWI-S&T program, can produce a spatially representative sample and reduce sample variance, but requires accurate information about the spatial variability in the population (Dahl 2011).

Plot Size

Appropriate plot size is related to several factors, including measurement and analysis methods, population spatial characteristics, and study objectives. The aquatic resource S&T program under design here is based on measuring the fractional area of a square plot covered by aquatic resources. In this case, the average distance between aquatic resources, and the patchiness of the resource, can potentially affect program performance if plot size is not scaled appropriately (Rossi 2004).

For example, if the plot size is substantially smaller than the average distance between aquatic resources, or the resource is extremely patchy, sample plots will tend to either have very high or very low area density values. This could increase sample variance and increase the required minimum sample size. Examples of this resource distribution could include lacustrine-type wetlands and deepwater habitat. Under this situation, increasing the plot size could theoretically improve statistical performance. Increasing the plot size would also reduce the fraction of sample plots with zero values for aquatic resource density (referred to as null plots here) (Xiao et al. 2005). In addition, larger plots would provide more mapped information about aquatic resources and adjacent landscape elements, increasing inference and hypothesis formation capabilities (Bellehumeur et al. 1997).

In contrast to the first examples, if aquatic resources are closer together on average, or are more evenly distributed across the landscape, sample plots would theoretically show less variability and fewer extreme density values. This would decrease sample variance and decrease the required minimum sample size. Examples of this resource distribution include streams. Under this situation, larger plot size may not significantly benefit program performance and it may be possible to reduce the sample plot size. Reducing the plot size could reduce total program costs, as smaller plots are less expensive to image and map.

Therefore, a balance must be found between plot size, sample size, and information that considers the diverse goals of the sampling program. Unfortunately, while the existing S&T programs all use different plot sizes, only the MN-S&T program performed any statistical performance evaluations before selecting a plot size. The NWI-S&T program uses 10.4 km^2 (4 mi^2) plots while the MN-S&T program adopted 2.59 km^2 (1 mi^2) plots after testing the effect of various plot sizes on sample accuracy and precision (Dahl 2011; Deegan and Aunan 2006). The NILS program maps aquatic resources for 1 km^2 plots located at the center of 25 km^2 plots, for which additional information about land cover is simultaneously collected (Ståhl et al. 2010).

Methods

General approach

We developed an approach to evaluate sample design elements based on the full range of program objectives. We utilized simulated sampling because of its ability to provide empirical distributions of sample point estimates such as the mean wetland and stream density and the fraction of sampled plots lacking wetlands or streams (referred to here as the null fraction). Then, we utilized the empirical distributions to evaluate the statistical accuracy and precision of the sampling design.

We simulated the sampling scenarios using the best available geographic databases of stream and wetland extent in California, the National Hydrography Dataset (NHD) and the NWI. We evaluated twenty-eight different sampling conditions, by comparing the empirical sampling distributions of the sample mean and the fraction of sampled plots lacking the resource of interest, and by evaluating the relationship between estimated sampling errors and predicted sampling cost.

Geographic databases

We based simulations on digital NHD and NWI maps in California, available for 100% and 78% of the state, respectively (Figure 2.1). We assumed that each maps represented the true population of wetlands and streams in California. Importantly, we split NWI maps into two subsets for analysis because of a change in mapping methodology in the mid-1990s. A key step in NWI wetland mapping is production of a map of streamline position, similar to the NHD. Prior to the 1990s, these one-dimensional maps of streamline position were kept separate from two-dimensional maps of wetland extent. However, beginning in the 1990s, one-dimensional streamlines were buffered and combined with two-dimensional wetlands into a single map of wetland and stream extent. This change in procedure significantly increased wetland area, in terms of the total area of mapped polygons, as well as the spatial distribution of the mapped polygons. Therefore, we considered NWI maps with buffered streamlines (NWIb), covering 10% of California, separately from maps without buffered streamlines (NWI).

Results are provided separately for NHD, NWI, and NWIb datasets for three reasons. First, different units for stream density necessitated comparing the NHD, analyzed here as meters of streamline per square kilometer of landscape, separately from the NWI and NWIb, analyzed here as square kilometers of wetland per square kilometer of landscape. Second, different mapping methodologies for the NWI and NWIb could produce an artifact in the spatial variability structure if we combined the datasets for analysis. Third, the three datasets have separate spatial extents and the NWI and NWIb are non-overlapping (Figure 2.1).

Sampling approaches

We considered sampling conditions for three design categories: sample selection method, stratification, and plot size. We evaluated each option independently for twenty-eight combinations of design options. First, two sample selection methods were tested: SRS and GRTS. SRS, used by the NWI-S&T program, is the default probabilistic sampling method. GRTS, used by the MN-S&T program, is a spatially balanced sampling method, which offers several theoretical advantages over SRS, as discussed earlier. Numerous other sampling methods exist but we did not evaluate them for a few reasons. First, we considered spatially balanced sampling a potentially powerful mechanism for improving sample performance, as discussed earlier. However, we did not evaluate systematic sampling, another spatially balanced method and used by the NILS program, because a systematic sample of a study area cannot be easily modified for future needs, such as regional intensification. Any such modifications would require a completely new sample frame and sample draw and results could not be easily combined with the existing draw. Other commonly employed methods, such as probability proportional to size or cluster sampling, require significant prior knowledge about the population which we could not supply (Kozak and Zielinski 2007; Smith et al. 2003). Additional, more technical methods, such as poisson sampling, were computationally intensive without significant probability of improving sample performance (Williams et al. 2009).

Second, we compared stratified and unstratified sampling using SRS and GRTS sample selection (four combinations). We stratified along Level-III ecoregion boundaries (Omernik 2010). We chose ecoregions for stratification for two primary reasons. First, ecoregions represent relatively homogenous ecological units, consistent with the assumptions and motivations for statistical stratification. Second, aquatic resource density varied substantially between ecoregions (Table 2.1), consistent with assumptions about homogeneity. Additionally, ecoregions are a convenient combination of numerous physical, climatological, and biological variables. These variables could be used individually for stratification, but would not be as theoretically powerful as use of ecoregions. In addition, any attempt to combine variables would quickly complicate sampling and analysis and would eventually approximate ecoregion boundaries. Finally, several anthropogenic influence variables such as percent impervious surfaces, land use, protection from development, or political management unit are significant drivers of aquatic resource extent in California. However, these variables change quickly over time, making them unsuitable for a long-term monitoring program. In addition, accurate and reliable information about these anthropogenic variables is not always available.

When stratifying, we performed optimum allocation for variance minimization to allocate the total sample between individual strata (n_i):

$$n_i = n\left(\frac{N_i\sigma_i}{\sum_{i=1}^{k} N_i\sigma_i}\right) \qquad \text{for } i \text{ in } 1, 2, \ldots, \mathrm{k} \qquad (2.1)$$

Under optimum allocation, total sample size (n) is allocated based on the population size (N_i) and population standard deviation (σ_i) for each stratum i (for the NHD, we used the standard deviation of log-transformed streamline density; for the NWI and NWIb, we used the standard deviation of arcsine-transformed wetland density).

Third, we tested seven different plot sizes (1, 2.25, 4, 6.25, 9, 12.25, and 16 km^2) using the four combinations of sample selection method and stratification, creating twenty-eight total combinations. Because the number of plot sizes tested was such a significant driver of the simulation time required, testing was confined to the range of plot sizes used by the NWI-S&T, MN-S&T, and NILS programs — 10.36 km^2 (4 mi^2), 2.59 km^2 (1 mi^2), and 1 km^2, respectively (Dahl 2011; Kloiber 2010; Ståhl et al. 2010).

Dataset Preparation

To prepare the geographic datasets for the simulation, we first used the fishnet tool in ArcInfo to create seven square grids covering the State of California, one for each of the tested plot sizes (ESRI 2010). We applied a random offset to the bottom-left corner of each grid in both the x and the y direction. The offset was between zero and the nominal dimension of the grid (e.g., 1 or 3.5 km). We utilized the offset to reduce the probability that the fishnet tool would align grid cells with the California boundaries.

Next, we clipped grids to the boundaries of the three geographic datasets: the state boundary for the NHD; mapped areas without buffered streamlines for the NWI; and mapped areas with buffered streamlines for the NWIb (Figure 2.1) shows boundaries of the three datasets). The result was three separate grids for each of the seven plot sizes. In addition, the area of each grid cell now represented the portion of that cell which overlapped with the mapped area for that dataset. Next, we assigned an ecoregion to each grid cell based on the location of the cell centroid.

Then, we intersected grids with NHD streamlines and NWI and NWIb polygons. Intersection split streamlines and polygons according to plot boundaries and assigned the grid cell number to each streamline and wetland segment. The numbers were then used as an index for determining the total stream length and wetland area and each grid cell for the stream and wetland subtypes listed in Table 2.1. Finally, we computed streamline and wetland density for each grid cell by dividing the summed lengths and areas by the cell area.

By including stream and wetland subtypes, we could explore sample design performance for a range of resource densities, geographic distributions, and spatial heterogeneities. In addition, these subtypes are aquatic resource groups of interest for management and research purposes in California and accurate estimate of their extent is one of the objectives of the California S&T program. The palustrine, unconsolidated shore, seasonally flooded (PUSC) wetland subtype was used as a surrogate for rare wetland types in order to further test sampling performance (Cowardin et al. 1979). PUSC has also been used by the San Francisco Bay Area, Wetlands Regional Monitoring Program (wrmp.org) as a “classification cross-walk” to vernal pools, a unique and ecologically important wetland type in California (Duffy and Kahara 2011; Holland and Jain 1981).

Simulations

We conducted all sampling simulations in R version 2.13.1 (R Development Core Team 2011). Each of the 28 sampling designs was simulated 5,000 times for each dataset, a replication count used by Miller and Ambrose to give an adequate estimate of variability in the dataset (Miller and Ambrose 2000). Each repetition, we recorded sample estimates of mean density and null fraction for each feature type. GRTS samples were drawn using the *grts* function in the *spsurvey* package (version 2.2), developed for R and available the Comprehensive R Archive Network (CRAN) (Kincaid and Olsen 2011). SRS samples using the *sample* function in the *base* R package.

We utilized random number seeds for reproducibility of GRTS and SRS sample draws. Like most computer languages, R uses a pseudo random number generator (pRNG) to produce a sequence of numbers that lack any discernible pattern. While not fully random, pRNG's such as the *Mersenne-Twister* (the default in R) pass statistical tests for randomness (Matsumoto and Nishimura 1998). In addition, because pRNG's use an arbitrary starting value to produce a string of apparently random results, if the same starting value is used then the same string of numbers will be generated. Therefore, we ensured that simulation results could be reproduced exactly at a later date by setting the seed with a known value before simulating SRS or GRTS sampling.

Bias and Precision of the Sample Mean

The result of the simulations was empirical distributions of the two point estimates, mean density and null fraction, for each feature type and combination of sampling parameters. We utilized these empirical distributions to compare the performance of the different sampling designs. This section will describe the methods used to evaluate the empirical distribution of the mean, first to detect potential bias and second to determine the relative precision of each sampling design. Bias in the sample mean could indicate a systematic error in the sampling methodology, which over-samples a subset of the population and then fails to correct for this oversample during analysis. Improved precision (a smaller value as defined here) could indicate that the particular sample design is more reliable and a smaller sample size may be possible. Subsequent sections will describe analysis of the null fraction and estimation of sample errors compared to predicted costs.

We measured bias in the sample mean by subtracting the true population value (μ) from the mean of the empirical distribution of the simulated sample means ($\bar{x}_x$). We calculated true population values by taking the mean of all grid cells and dividing by the standard deviation of the empirical distribution (s_x):

$$d_{Cx} = \frac{\bar{x}_x - \mu}{s_x} \tag{2.2}$$

This relationship (d_{Cx}) is known as Cohen's d and is an alternative to a t-test for the difference of means (Cohen 1988). Because our replication rate was so large (5,000), a t-test would conclude that very small differences between $\bar{x}_x$ and μ were significant. However, Cohen's d does not consider the number of replications. Instead, the difference between the empirical distribution and the true value is only compared to the variability in the empirical distribution. Cohen's d cannot produce p-values for difference between means. However, traditional cutoffs for Cohen's d to define small, 0.2-0.5, medium, 0.5-0.8, and large, >0.8, effect sizes (Cohen 1988). These cutoffs indicate that a large difference between two values is one that is close to or exceeds the variability, while a small difference is less than half of the magnitude of the variability.

We computed the precision (p_x) of each sampling design as the ratio of the standard deviation and the mean (s_x and $\bar{x}_x$) of the empirical distribution of the sample mean, multiplied by the square root of the simulated sample size (n_s):

$$p_x = \frac{s_x}{\bar{x}_x}\sqrt{n_s} \tag{2.3}$$

Importantly, n_s is the size of the simulated sample draw, not the number of simulated repetitions. This sample size was different for each plot size, set in order to simulate an approximately equal simulated sample cost.

We multiplied the ratio of s_x to $\bar{x}_x$ by the square root of n_s because of the impact n_s theoretically has on s_x, the standard deviation of the empirical distribution of the mean. s_x is conceptually equivalent to the standard error of the sample mean (SEM), for a given sample size (e.g., n_s). SEM is commonly estimated as the sample standard deviation over the square root of the sample size. While this approximation may underestimate the true value, the effect is increasingly small for sample sizes above twenty (Gurland and Tripathi 1971). Therefore, multiplying s_x by the square root of n_s produces an indicator of the variability in the empirical distribution that, theoretically, is not influenced by n_s. As a result, p_x theoretically reflects the precision of the sampling method itself, instead of the impact a larger n_s would have on precision.

We compared p_x values between sampling conditions using an f-test for the radio of variances. This test typically has a null hypothesis that the ratio of sample variances is equal to one (i.e., s_{x1}^2 / s_{x2}^2 = 1). However, one can be replaced by any value and we chose the ratio of the mean of the sampling distributions ($\bar{x}_x$) over the simulated sample sizes, (n_s):

$$\frac{s_{x1}^2}{s_{x2}^2} = \frac{\bar{x}_{x1}^2 / n_{s1}}{\bar{x}_{x2}^2 / n_{s2}} \tag{2.4}$$

The equality in equation 2.4 can be re-arranged and, using equation 2.3, reduces to the equality $p_{x1} / p_{x2} = 1$. We defined statistical significance using bonferroni-corrected p-values. The bonferroni correction is used in cases of multiple comparisons, to account for the probability of a type-1 error (concluding a significant difference where none exists). The specific bonferroni correction is provided in the results when it is applied.

Sample Null Fraction

We compared the empirical distribution of null fraction values between sampling conditions to indicate potential differences in average mapped information. The null fraction (f_{null}) indicates the fraction of sample plots that are null for a particular aquatic resource type. Therefore, substantial differences between f_{null} values under different sampling conditions could indicate differences in the usefulness of sample plots, from the standpoint of mapping aquatic resources.

We used a different form of Cohen's d from the one described earlier to compare the means of the empirical distributions of the f_{null} statistic ($\overline{x}_f$) (Cohen 1988). For the same reason as above, we selected Cohen's d because the high replication number (5,000) used in this study meant a t-test for the difference between two f_{null} distributions would conclude that even very small differences in $\overline{x}_f$ were significant. In contrast, Cohen's d is not sensitive to the number of replicates performed. We calculated Cohen's d for f_{null} (d_{Cf}) as:

$$d_{Cf} = \frac{\overline{x}_{f1} - \overline{x}_{f2}}{s_{f1,2}} \qquad (2.5)$$

This form of Cohen's d compares the empirical distributions of two sampling conditions, instead of comparing the empirical distribution of one condition to the population value. As a result, the numerator is the difference between $\bar{x}_f$ for the two sampling conditions, instead of the difference between $\bar{x}_f$ and the true, population f_{null} value. In addition, the denominator is the pooled standard deviation for the two sampling distributions, instead of the standard deviation of a single empirical distribution. The same cutoffs mentioned above were used here to define small, 0.2-0.5, medium, 0.5-0.8, and large, >0.8, effect sizes (Cohen 1988).

Estimated Percent Error

The empirical distribution of sample means was also used to estimate the sampling error if the particular sampling conditions were applied to the state of California as a whole. We began with the formula for the confidence interval of the mean, defined by the sample mean, plus or minus an error term (*E*), where:

$$E = Z_{1-\frac{\alpha}{2}} \frac{s}{\sqrt{n}} \tag{2.6}$$

In the above, $Z_{1-\alpha/2}$ is the Z-value associated with a p-value greater than or equal to $1 - \alpha/2$, s is the sample standard deviation, and n is the sample size. Error can easily become percent error (E_P) by dividing by the sample mean ($\bar{x}_x$) and multiplying by 100%. However, this formula utilizes the sample standard deviation, whereas the standard deviation of our empirical distribution of the mean (s_x) is essentially equivalent to the standard error of the mean (Gurland and Tripathi 1971). Therefore, we multiplied s_x by the square root of n_s in order to approximate the sample standard deviation, s. Finally, to obtain a predicted percent error for a given sample size, we replaced n with a variable sample size (n_p):

$$E_P = \frac{Z_{1-\frac{\alpha}{2}}\left(\frac{s_x\sqrt{n_s}}{\sqrt{n_p}}\right)}{\bar{x}_x}100\% \qquad (2.7)$$

The above equation predicts the percent error of the sample mean as a function of predicted sample size.

Predicted Sample Cost

We based predicted sample costs using three different combinations of image acquisition and map production costs. Then, we used predicted costs to compare estimated percent errors between tested plot sizes. We developed predicted costs from the best professional judgment of experts in the fields of image acquisition and aquatic resource map production. We considered two general scenarios for image acquisition costs: use of no-cost, existing imagery from the National Agriculture Imagery Program and use of contract imagery from third party vendors. To predict contract imagery costs, we relied on the best professional judgment of an aerial photography company based out of Murietta, California. Based on experience in aerial photography and processing for a variety of applications, including scientific and technical work, this company predicted that the required image quality could be met at all plot sizes through a single-pass photograph. They also recommended a contract structure based on a per-plot fee, as opposed to payment for flight time. In their judgment, per-plot image costs would be between 150 and 450 USD per plot. Therefore, we considered three values for the imagery portion of predicted costs: (i) no-cost, existing imagery; (ii) 150 USD per plot, contract imagery; and (iii) 450 per plot, contract imagery.

We based predicted map production costs on the best professional judgment of two wetland-mapping groups, based out of Northridge and Richmond, California. Both groups have significant experience in aerial photo interpretation for stream and wetland mapping. Both groups reviewed their hour and contract records to produce estimates of the time and salary costs associated with all phases of stream and wetland mapping, including production and editing of the streamline network, delineation and classification of wetland polygons, and review and internal quality control on final maps. Both groups arrived, independently, at a rate of approximately 25 USD per square kilometer. We then combined this rate with the three imagery estimates to produce per-plot costs for each plot size scenario.

Map Production Cost Efficiency

Finally, we developed a measure of map production efficiency from the relationship between plot size, predicted costs, and the mean null fraction ($\bar{x}_f$). Our cost efficiency measure considers mapped plots with aquatic resources to be "useful" and plots that are mapped but do not contain aquatic resources to be "not-useful." This designation applies solely to secondary uses of the sample plots because all sample plots, including "null" or "not-useful" plots, were used to estimate the area-wide estimate of aquatic resource extent. Secondary uses of sample plots where the useful/not-useful designation is meaningful include utilization of sample plots for a sample frame for field-based assessments of stream and wetland condition.

We defined map production efficiency (e_m) using the per-plot cost ($cost_{plot}$) and the mean null fraction ($\bar{x}_f$):

$$e_m = \frac{n_s\left(cost_{plot}\right)}{n_s\left(1-\bar{x}_f\right)area_{plot}} \tag{2.8}$$

The cost per plot is calculated by the assumed imagery costs (no cost, 150 USD, or 450 USD), the product of the mapping costs (25 USD per mapped kilometer), and the plot size. After multiplying by the sample size (n_s), the numerator of e_m represents the total sample cost. The denominator is then the total area of the "useful" sample plots in a typical sample, based on the average "useful" rate and the area of each plot. Therefore, the resulting ratio is a cost per "useful" square kilometer of aquatic resource mapping.

Results

We provide results in three sections and use each section to make cumulative decisions about program design. First, sample selection method is considered, for all combinations of stratification and plot size, leading to selection of GRTS as the preferred sample selection method for this dataset. Second, we compare stratified and unstratified GRTS designs, for all plot sizes, leading to selection of an unstratified GRTS design, supported by all plot sizes and illustrated here by results for the 16 km^2 plot size. Third, we compared different plot sizes under an unstratified GRTS design.

Sample Selection Method

Mean wetland and stream densities were uniformly less variable for GRTS-selected samples compared to SRS-selected samples (Figure 2.2). We detected no substantial bias, as assessed by d_{Cx} between sample and population means and sample selection method (for all conditions, d_{Cx} between -0.01 and 0.004). Considering only total wetland and stream density, p_x values of GRTS-selected sample means were 5-33% lower than those for SRS-selected samples.

The observed decrease in p_x was not significantly associated with the expected spatial distribution of the resource. While the patchy NWI wetland resource had the largest percent decrease in p_x, 19-33%, the evenly distributed NHD streamline resource had the second largest, 8-20%, and the NWIb, which contains both streams and wetlands, had the smallest, 5-15%. However, tor the NHD, the difference between GRTS and SRS decreased as plot size decreased while the NWI and NWIb exhibited the opposite trend. For the NHD, the benefit of GRTS sampling decreased from a 16-19% reduction in p_x for 16 km^2 plots to an 8-11% reduction for 1 km^2 plots. For the NWI, GRTS-selection reduced p_x relative to SRS by 19-21% for 16 km^2 plots, and by 33% for 1 km^2 plots; for the NWIb, reductions were 5-8% and 10-15%, respectively. All differences between SRS and GRTS were statistically significant (f-test for equality of variance, all p-values less than the corrected p-value: 0.05 divided by 42; three populations times seven plot sizes times two stratification options).

Stratification

The effect of stratification on the precision of GRTS-selected samples was mixed for all plot sizes and data types. Neither stratified nor unstratified sampling produced biased estimates of the sample mean (as assessed by d_{Cx}) or showed any substantial relationship with bias (all d_{Cx} values between -0.03 and 0.05). Figure 2.3 illustrates the impact of stratification on p_x for the 16 km^2 plot size. We evaluated the other tested plot sizes in a similar manner and their results are consistent with the general conclusions of the 16 km^2 plot size, used below to illustrate the range of impacts of stratification on precision. We did not explore the impact of stratification on SRS, based on the conclusion from the previous section that GRTS sampling is preferred over SRS sampling. The values in each cell in Figure 2.3 are the result of comparing the p_x value for that specific region and resource type, under stratified GRTS sampling, to the corresponding p_x value under unstratified GRTS sampling. When reviewing this figure, the statewide results and the results for all wetlands or all streamlines should not be viewed as simply a combination of the results for individual regions or resource subtypes. While these results are related, the value for each cell was based on independent calculations and the statewide or all-type results were not weighted combinations of results for individual regions or subtypes.

In general, while stratification did significantly reduce sample variance for several ecoregions and aquatic resource subtypes, this benefit was largely limited to larger ecoregions and common aquatic resource subtypes. In contrast, stratification tended to increase sample variance, and therefore decrease sample power, for smaller ecoregions and less common aquatic resource types. At the statewide level (column 1 of Figure 2.3), stratification tended to increase p_x for the NHD and decrease p_x for the NWI and NWIb. We observed this trend for all plot sizes. For example, statewide effects on the p_x (column 1, rows 1, 7 and 14 of Figure 2.3) include a 1.5%, non-significant increase for the NHD to a 9.6%, significant decrease for the NWI and a 15.9% significant decrease for the NWIb (for this section, significance was for the f-test for equality of variance and the significance threshold was set at 0.05 divided by 280 — 20 aquatic resource types and subtypes times 14 geographic regions). When subtypes are considered for statewide results (column 1, rows 2-6, 8-13, and 15-20 of Figure 2.3), the impacts of stratification range from a 24% significant increase for 3rd and 4th order streamlines in the NHD to a 14% significant decrease for riverine wetlands in the NWI and a 34% significant decrease for lacustrine wetlands in the NWIb.

Ecoregion-level impacts of stratification on p_x included significantly positive and significantly negative impacts for all three datasets. We observed a similar mix of effects for all plot sizes. For example, the effects of stratification on the NHD (columns 2-14, row 1 of Figure 2.3) ranged from a 1.7% non-significant increase for the Klamath Mountains/CA High North Coast Range ecoregion to a 12.2% significant decrease for the Eastern Cascades, Slopes & Foothills ecoregion. Effects of stratification on the NWI (columns 2-14, row 7 of Figure 2.3) ranged from a 3.9% non-significant increase for the Sierra Nevada ecoregion to a 13.2% significant decrease for the Northern Basin & Range ecoregion. Finally, effects on the NWIb (columns 2-14, row 14 of Figure 2.3) ranged from a 1.6% non-significant increase for the Coast Range ecoregion to a 24.6% significant decrease for the Sonoran Basin & Range. Ranges widened when we considered wetland and stream subtypes (columns 2-14, rows 2-6, 8-13, and 15-20 of Figure 2.3).

Plot Size and Sample Cost

Smaller plot sizes produced more variable estimates of mean wetland and stream density (Figure 2.2) and were more likely to lack aquatic resources, i.e., to have higher null fractions (Figure 2.4). No plot size exhibited significant bias from the population mean nor was there any relationship between bias and plot size (all d_{Cx} values between -0.002 and 0.003). The impacts of plot size on precision and the null fraction were both significant for the NHD; CV for 1 km^2 plots was more than double the CV from 16 km^2 plots while the null fraction of NHD samples decreased tenfold, from 0.29-0.30 for 1 km^2 plots to 0.03-0.05 for 16 km^2 plots. The effect of plot size on variability was not significant for the NWI and NWIb but the effect on null fraction was. NWI sample null fractions decreased from 0.61-0.62 for a 1 km^2 plot to 0.21-0.22 for a 16 km^2 plot while NWIb sample null fractions decreased from 0.07-0.08 to 0.01-0.02. Marginal differences, e.g., between 1 and 2.25 km^2, in the null fraction became insignificant for the NWIb for plot sizes above 6.25 km^2 but were always significant for the NHD and NWI.

We observed two distinct relationships with predicted sample costs (Figure 2.5 and Table 2.2). Only the even plot sizes are shown in Figure 2.5 in order to reduce the number of lines on each plot; we also considered even plot sizes more realistic options for program implementation. We also do not show SRS and stratified sampling designs based on the conclusion, from the two previous sections, that GRTS unstratified sampling is preferred over SRS or stratification. First, we evaluated the relationship between plot size, total predicted sample cost, and estimated percent error. This relationship determines the ability of the sample design to meet its primary objective, reporting precise estimates of extent for aquatic resource types and subtypes. For the NWI and NWIb, smaller plots were always the least-expensive option for producing sample estimates at a given percent error (Figure 2.5 and Table 2.2). For the NHD, smaller plots were only less expensive when we assumed no-cost imagery. If contract imagery is assumed for the NHD, larger plots are more cost effective. This reversed relationship is likely caused by the lower overall variability for the NHD.

Second, we evaluated the relationship between plot size, total sample cost, and the area of aquatic resource maps produced by calculating e_m values. This relationship indicates the ability of the program to meet secondary objectives, such as serving as a sample frame for field-based studies of wetland or stream condition. Larger plots were always more cost effective for producing maps of aquatic features (Table 2.2). The difference between large and small plots was most significant for the NWI, because of higher null fractions, and when we assumed high-cost contract imagery, because of its significant impact on per-plot costs.

Discussion

The observed benefits from GRTS sampling are consistent with the theoretical basis of this sampling methodology. By increasing the diversity and balance of sampled landscapes, spatially balanced sampling is expected to minimize the potential impacts of small-scale autocorrelation on sample variance (Stevens and Olsen 2004; Stevens and Olsen 2003). Our results also suggest GRTS sampling may be a more effective approach, in some contexts, for reducing sample variance than use of stratification with optimum allocation. Stratum size and variability drive stratification with optimum allocation and optimization of sample variance is only possible for the objectives considered (Bosch and Wildner 2003). Therefore, stratification is most likely to benefit results for large strata or subpopulations, and is less likely to benefit, or may even harm, results for small strata or rare subpopulations. In contrast, the size or variability of individual strata is not a driver of spatially balanced sampling methods, such as GRTS, and therefore may provide advantages, as was observed here, for all strata and subpopulations.

As a set of recommendations, the results of this study have various levels of agreement with the designs of similar, existing programs and generally agree well with available sampling design literature. Our results have strongest agreement with the MN-S&T program, which also uses unstratified GRTS sampling (Kloiber 2010). The MN-S&T program also specifically evaluated the benefits of stratification according to ecoregion but determined it did not offer statistical advantages (Deegan and Aunan 2006). In contrast, the NWI-S&T and the NILS use stratification and systematic sampling, respectively (Dahl 2011; Ståhl et al. 2010). However, both of those programs chose their respective approaches in order to increase the spatial balance and improve the precision of the resulting sample. It is also important to note that the GRTS sampling methodology was developed after the NWI-S&T program was designed and none of these programs was specifically designed to monitor both wetland and stream extent. By considering both wetlands and streams, this work expands probabilistic sampling design beyond the monitoring of wetland extent. In addition, the results here are the first to demonstrate the advantage of using spatially balanced sampling for monitoring the extent of landscape elements.

Plot size is one area of significant differences between our results and existing S&T programs. However, these differences are consistent with the tradeoffs observed in this study between stream and wetland monitoring. The State of Minnesota employs a relatively small, 2.59 km^2 (1 mi^2) plot size in the MN-S&T program (Deegan and Aunan 2006; Kloiber 2010). Minnesota is also approximately 25% wetland, compared to approximately 5% in California, and is more ecologically homogeneous than California. High aquatic resource densities in Minnesota mean this state is unlikely to observe significant tradeoffs between plot size and mapped information (investigated here using the null fraction); therefore, a small plot size, which reduces sample error, is likely appropriate for Minnesota. This is similar to conclusions from this study that high stream densities and low overall variability in the NHD mean smaller plot sizes are adequate for monitoring streamline density in California. In contrast, the NWI-S&T program uses a relatively large, 10.4 km^2 (4 mi^2) plot size (Dahl 2011). The plot size for the NWI-S&T was not selected based on a systematic analysis of wetland density but our results suggest it may, in fact, be well selected for a national program. Nationally, wetlands cover approximately 5.5% of land area, very close to the density in California, and the NWI-S&T program monitors wetlands across diverse ecological settings, also similar to California (Dahl 2011). While the analysis from this study does not specifically lead to a specific plot size, the tradeoffs between plot size, sample error, sample information, and sample cost, perhaps the result of patchy wetland densities, mean that a larger plot size may be required to balance sample information needs against sample error. However, this tradeoff has significant cost implications for the program and the most pragmatic plot size may be the one that allows the design to meet targets for statistical accuracy.

While overall estimates of aquatic resource density are potentially achievable with acceptable levels of precision, estimates for rare or spatially limited aquatic resource types had significantly lower precision in this study. Accurate estimates for rare populations are significant issues for all probabilistic sampling designs. Options to address this issue typically lead to substantially different sampling designs, such as adaptive sampling, regional intensification, or modification of basemaps and target regions (Guisan et al. 2006; Smith et al. 2003). Each of these designs requires assumptions about the distribution of the rare population. However, these assumptions can potentially bias the resulting estimates if based on incomplete information or if applied imperfectly (Thompson and Seber 1994). Therefore, modification of the sampling design to address limitations in rare population measurement should only be pursued if monitoring objectives specifically emphasize accurate estimates for rare populations over other objectives.

Probabilistic monitoring clearly cannot replace comprehensive maps, which are essential for site-specific actions. However, this study supports the potential for probability-based monitoring of wetland and stream density as part of a coordinated strategy for monitoring wetland and stream extent and condition. In addition to providing estimates of wetland and stream density, with unbiased measures of uncertainty, probabilistic maps can serve as a sample frame for ambient, field-based assessment. At the time of this writing, less than ten percent of the State of California has wetland or stream maps produced within the previous ten years. Absence of an appropriate basemaps significantly handicaps field-based investigations of wetland or stream condition performed outside of recently mapped areas. Probabilistically mapped plots provide a cost-effective method for bridging this gap by providing spatially distributed, primary sampling units, suitable for a two-stage sampling approach. For example, a subset of mapped S&T plots could be randomly selected (the first stage of the sample), and individual stream reaches or wetlands could be selected from within each sample plot (the second stage of the sample).

Future work could include a prospective implementation of the unstratified GRTS design, with new map production from aerial imagery. If conducted in a study area with existing, comprehensive aquatic resource maps, the implementation could verify the performance of GRTS sampling and estimation, without relying on existing aquatic resource maps. The implementation could also include nested plot sizes to further explore and quantify the effects of plot size on sample error and sample information.

Implications for the California Program

The proposed California S&T program will monitor the extent and distribution of wetlands and streams across the state. Beyond selecting an efficient and accurate design, we employed simulated sampling in order to examine a number of design issues including the feasibility of reporting for subtypes and regions of interest, including rare subtypes, and how to balance monitoring wetland density with monitoring stream density. Our results clearly illustrate the advantages of a spatially balanced sampling method, GRTS. Results for stratification are mixed and highly dependent on the aquatic resource type and geographic area. As discussed previously, this lack of clear support for stratification, and the potential for stratification to reduce flexibility in sampling and analysis, led us to recommend against stratification for monitoring the spatial extent of wetlands and streams in California. Finally, results illustrated the relationship between plot size, sample error, sample information, and sample costs, enabling appropriate decision-making based on program objectives, priorities, and budget. Successful development of a simulated sampling approach in the R programming environment made these results possible. Potential limitations of these results include the incomplete extent and variable quality of source data, which could artificially increase population variance or reduce generalizability. However, while the results from the three datasets, including the statewide NHD dataset, may differ in magnitude, conclusions about program design elements are mutually consistent.

Results also show the feasibility and promise of a probabilistic S&T program in California. By providing a spatially balanced sample, GRTS significantly and consistently reduced sample variance, therefore increasing power to detect change, reducing the necessary sample size, and controlling sample costs. GRTS sampling provides additional statistical and practical advantages, not directly addressed here. These advantages, related to the local mean and variance estimator and the option to draw a master sample, are also a result of how the GRTS sample is drawn and analyzed. First, the local variance estimator reduces sample variance and increases statistical precision (Stevens and Olsen 2004; Stevens and Olsen 2003). The GRTS variance estimator is specific to the GRTS sampling methodology and could not be applied to a SRS or systematic sample. Second, the master sample list ensures that additional sampling locations can be added over time while maintaining the spatial balance of the entire sample. This is a direct result of how the GRTS sample is drawn. The master sample provides a practical, flexible, and statistically valid mechanism for substituting or adding sample plots, such as for a regional intensification (Theobald et al. 2007). These additional locations can also be added statewide or to regions of interest as the objectives of the monitoring program evolve. A master sample simplifies this process when multiple entities are involved with image acquisition, map production, and analysis. The master sample also removes the need to perform a supplementary sample draw, which requires GIS and statistical software expertise as well as access to the original sample frame.

Other simulated sampling studies have shown that stratification can be employed to reduce overall sample variance and to guarantee minimum sample sizes for subpopulations of interest (Jongman et al. 2006; Miller and Ambrose 2000). While stratification is commonly viewed as a significant improvement for many sampling approaches, we do not believe it is appropriate for the California S&T program. The mixed results for different regions and resource types do not, by themselves, provide consistent support for or against use of stratification. However, several key points should be made about the stratification results. First, the strata used in the simulations are not the only subregions the state will use for reporting results. Therefore, maintaining flexibility is a clear advantage for an unstratified design. Second, the allocations used in simulations were based on the stream and wetland distribution in the NWI and NHD. These allocations most likely do not represent the ideal allocation due to the incompleteness of the datasets and changes in the landscape since creation of the NHD and NWI. Therefore, these allocations are unlikely to be accurate for the implemented S&T program. The simulations represent a best case scenario where the information used for allocation is accurate and complete. If this best-case scenario cannot provide clear and consistent support for stratified over unstratified GRTS sampling, it seems less likely that an actual implementation, where the allocation information is incomplete and possibly inaccurate, will be successful.

Finally, by highlighting the implications of plot size on sample costs, the State will be able to balance available resources against program objectives. For example, if the State chooses to prioritize the statistical efficiency of the S&T program, the State could select a smaller plot size. However, if the State wants to produce more aquatic resource maps, for use in other programs as part of the coordinated aquatic resource monitoring effort mentioned above, the State could select a larger plot size. In addition, the relationship between plot size and sample costs suggests some potential design tradeoffs resulting from designing a sampling program to monitor different resource types. In California, stream density is significantly less variable than wetland density, reflecting the even spatial distribution of streams and the patchy distribution of wetlands. As a result, the most appropriate plot size for monitoring streams was not always the most appropriate plot size for monitoring wetlands, and depended significantly on assumptions made about imagery costs.

Tables

Table 2.1. Streamline and wetland density by Level-III ecoregion. Density is the sum of streamline length or wetland area, across the domain of interest, divided by the area of the domain.

Level-III Ecoregion	NHD Streamline Density (m km^{-2})					
	All	SO > 4	SO = 3-4	SO = 1-2	SO NA	SO = 1-2 & Int. Flow
Cascades	471	5	65	338	63	136
Central Basin & Range	764	39	89	403	234	272
Central CA Foothills & Coastal Mtns	1034	38	141	724	131	624
Central CA Valley	1100	86	92	246	677	211
Coast Range	868	36	134	665	33	193
E Cascades Slopes & Foothills	625	29	78	391	128	286
Klamath Mtns/CA High N Coast Range	787	36	102	630	19	226
Mojave Basin & Range	640	15	51	380	194	373
N Basin & Range	657	0	98	409	150	323
Sierra Nevada	863	27	117	667	53	316
Sonoran Basin & Range	808	16	61	455	277	423
S CA Mtns	1150	11	117	929	93	833
S CA/N Baja Coast	877	28	113	546	189	518
State	801	32	93	495	182	362

Level-III Ecoregion	NWI Wetland Density (km^2 km^{-2} × 100)[a]						
	All	Estuarine	Lacustrine	Marine	Palustrine	Riverine	PUSC
Cascades	3.06	--	0.89	--	2.13	0.04	0.003
Central Basin & Range	8.16	--	4.05	--	4.06	0.05	0.012
Central CA Foothills & Coastal Mtns	3.11	0.42	1.63	0.003	0.92	0.13	0.023
Central CA Valley	8.65	0.74	1.20	--	5.82	0.90	0.026
Coast Range	6.69	1.08	0.19	0.161	4.33	0.93	0.007
E Cascades Slopes & Foothills	8.77	--	3.42	--	5.27	0.09	0.013
Klamath Mtns/CA High N Coast Range	1.87	--	0.50	--	1.02	0.35	0.002
Mojave Basin & Range	2.89	--	2.59	--	0.24	0.06	0.002
N Basin & Range	20.84	--	8.16	--	12.67	0.01	0.006
Sierra Nevada	3.34	--	1.60	--	1.67	0.07	0.002
Sonoran Basin & Range	6.03	--	5.06	--	0.62	0.35	0.012
S CA Mtns	1.17	--	0.60	--	0.47	0.09	0.005
S CA/N Baja Coast	1.46	--	0.43	--	0.76	0.27	0.011
State	5.32	0.55	2.16	0.010	2.35	0.25	0.011

Level-III Ecoregion[b]	NWIb Wetland Density (km^2 km^{-2} × 100)[a]						
	All	Estuarine	Lacustrine	Marine	Palustrine	Riverine	PUSC
Central CA Foothills & Coastal Mtns	3.54	0.14	0.72	0.06	1.44	1.18	0.0046
Central CA Valley	4.97	--	0.29	--	2.93	1.75	0.0017
Coast Range	2.14	0.08	0.02	0.04	0.74	1.25	0.0025
Klamath Mtns/CA High N Coast Range	3.09	--	1.41	--	0.33	1.35	0.0003
Mojave Basin & Range	3.00	--	2.12	--	0.06	0.83	0.0010
Sierra Nevada	1.38	--	0.29	--	0.41	0.69	0.0019
Sonoran Basin & Range	59.25	--	56.80	--	0.75	1.70	--
S CA Mtns	3.08	--	0.18	--	1.45	1.46	0.0123
S CA/N Baja Coast	3.33	0.37	0.47	0.12	1.57	0.79	0.0459
State	3.24	0.17	0.67	0.09	1.22	1.10	0.0150

[a] Not all NWI and NWIb subtypes were present in all ecoregions
[b] NWIb extent did not include all ecoregions
[c] Palustrine, unconsolidated shore, seasonally flooded wetlands (Cowardin et al. 1979)

Table 2.2. Estimated costs for 10% predicted error and cost efficiency for producing maps (e_m) of wetland and stream resources by assumed imagery cost, data source, and plot size.

Plot Size	**Contract Imagery ($450)**			**Contract Imagery ($150)**			**Existing Imagery**		
(km^2)	**NHD**	**NWI**	**NWIb**	**NHD**	**NWI**	**NWIb**	**NHD**	**NWI**	**NWIb**
Estimated Cost for 10% Predicted Error (Thousand USD)									
1	163	991	1,026	60	365	378	9	52	54
2.25	108	992	952	44	404	388	12	110	106
4	98	1,118	980	44	508	445	18	203	178
6.25	80	1,306	1,042	40	660	526	21	337	269
9	69	1,395	1,177	38	775	654	23	465	392
12.25	70	1,587	1,246	42	958	752	28	643	505
16	71	1,890	1,418	46	1,223	917	33	889	667
Cost Efficiency, Mapping (USD km^{-2})									
1	672	1,258	513	248	463	189	35	66	27
2.25	267	451	234	109	184	95	30	50	26
4	152	234	141	69	107	64	28	43	26
6.25	104	148	99	53	75	50	27	38	25
9	79	10	76	44	59	42	26	35	25
12.25	64	83	62	39	50	38	26	33	25
16	55	68	54	36	44	35	26	32	25

Figures

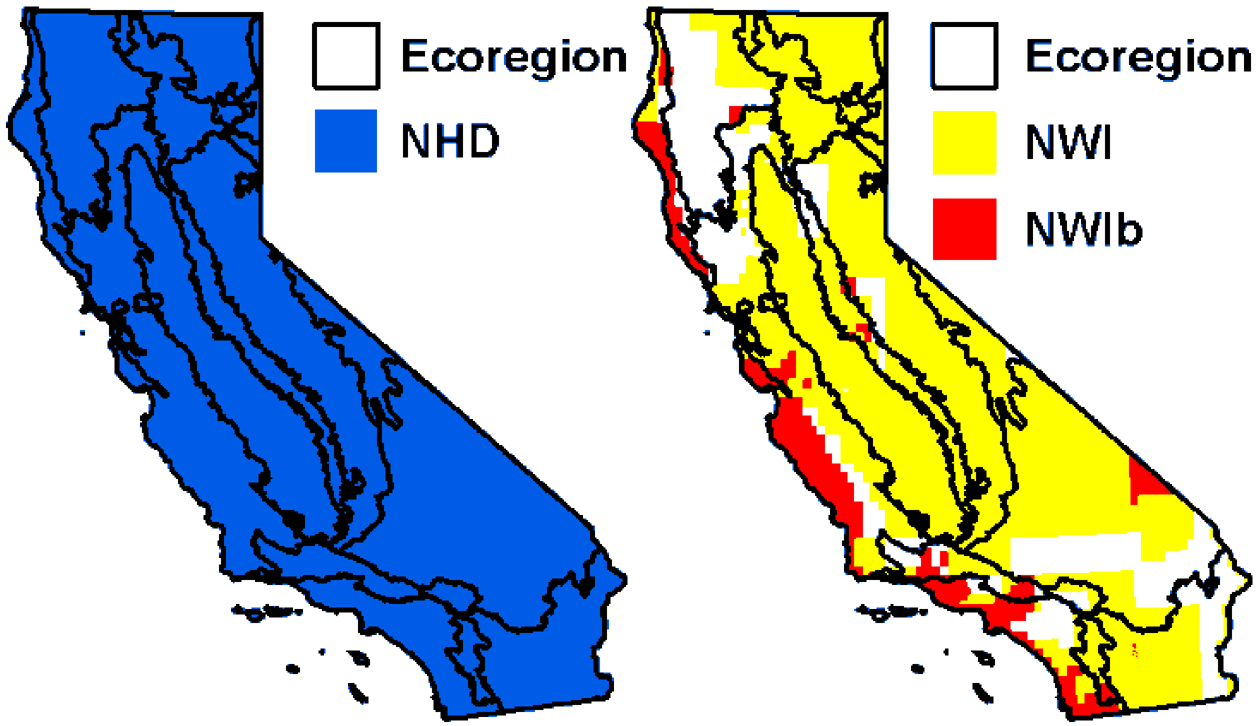


Figure 2.1. Level-III ecoregion boundaries and availability of NHD and NWI digital maps in California.
Mapping methodology divides the NWI into maps without (NWI; yellow) and maps with (NWIb; red) buffered streamlines.

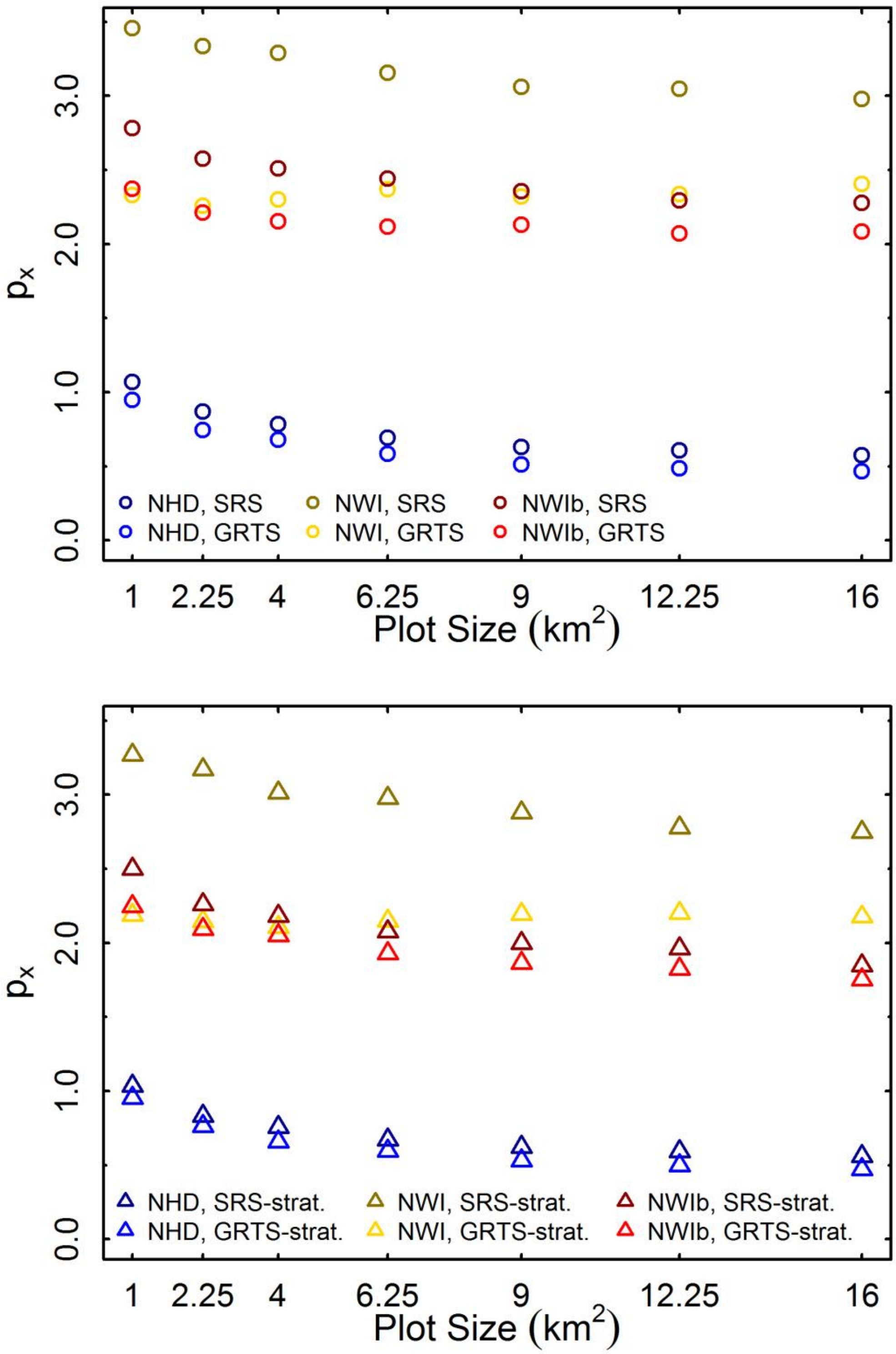


Figure 2.2. p_x by plot size and sampling method. p_x values are from the distribution of simulated sample means.

Results are shown for SRS (darker shades) and GRTS (lighter shades), with (triangle) and without (circle) stratification by ecoregion, for the NHD (blue), NWI (gold), and NWIb (red).

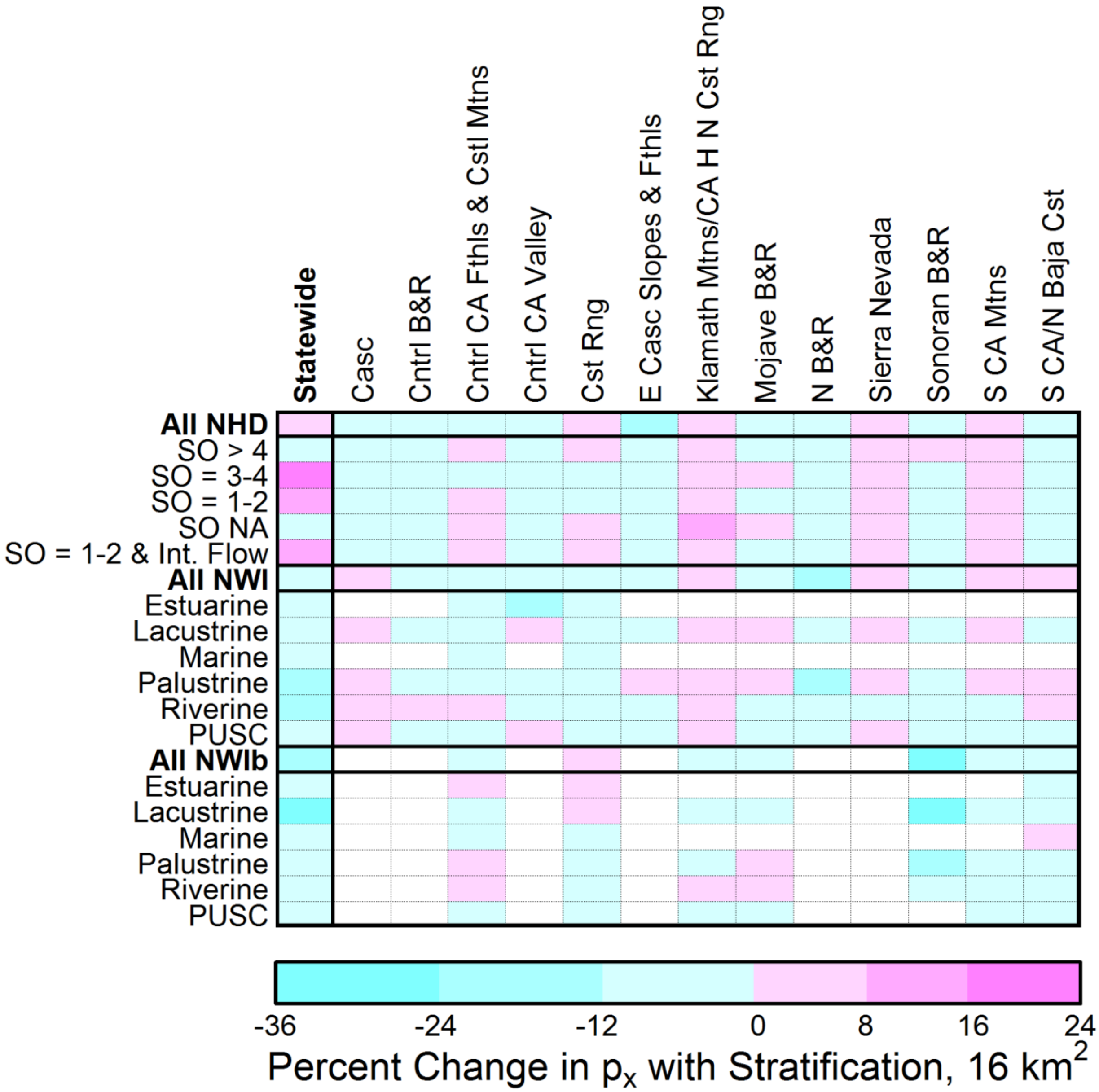


Figure 2.3. Percent change in p_x between unstratified and stratified sampling stratification.
Shown are results for 16 km^2 plots, sampled using GRTS. Wetland and stream types by row and ecoregions by column. Intensity of cell shading corresponds to degree of increase (magenta) and decrease (teal) in p_x.

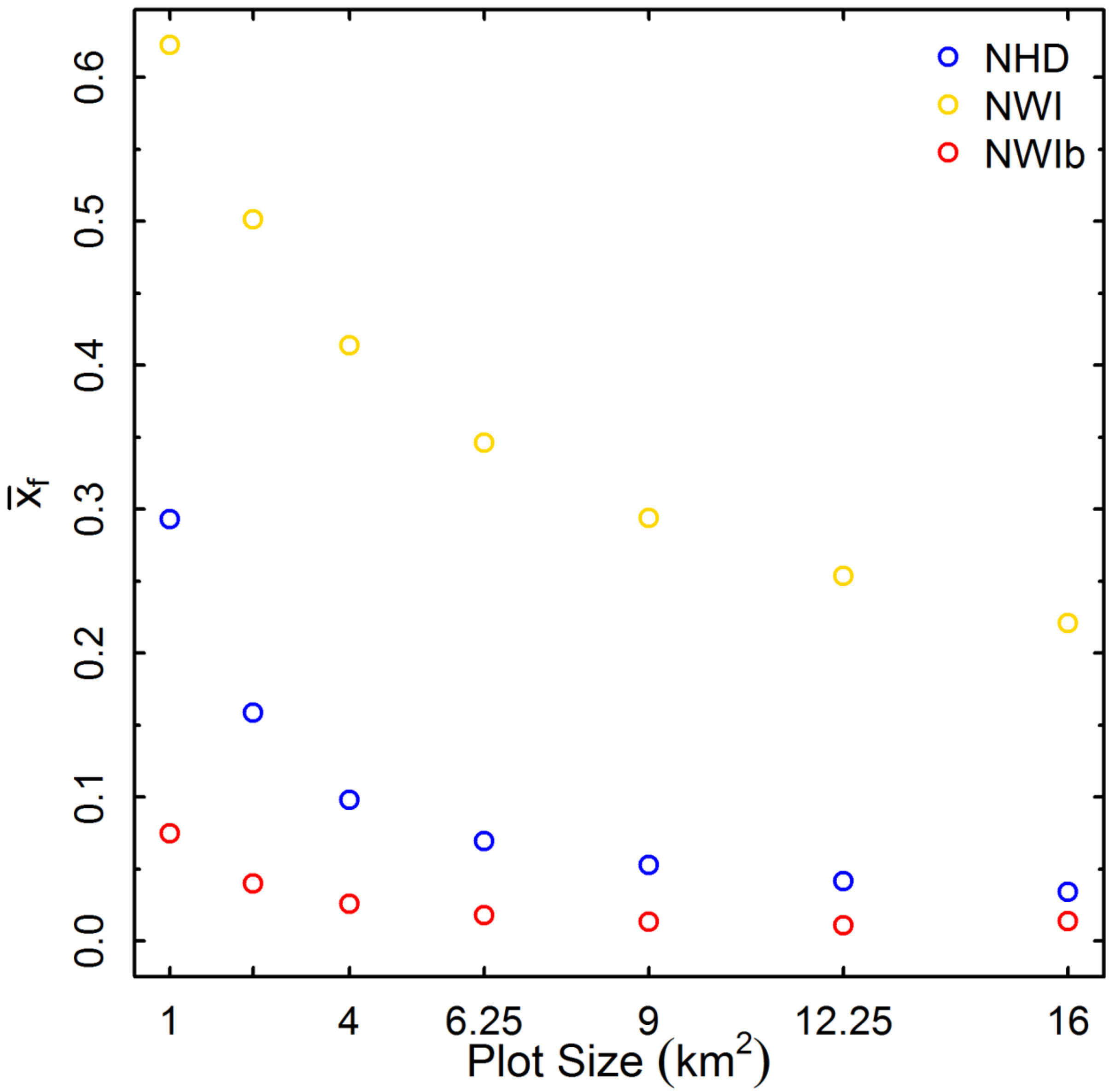


Figure 2.4. Mean fraction of sampled plots with null densities ($\bar{x}_f$).

Shown are results for GRTS, unstratified sampling. Points indicate the mean fraction of sampled plots lacking aquatic features for the NHD (blue), NWI (gold), and NWIb (red).

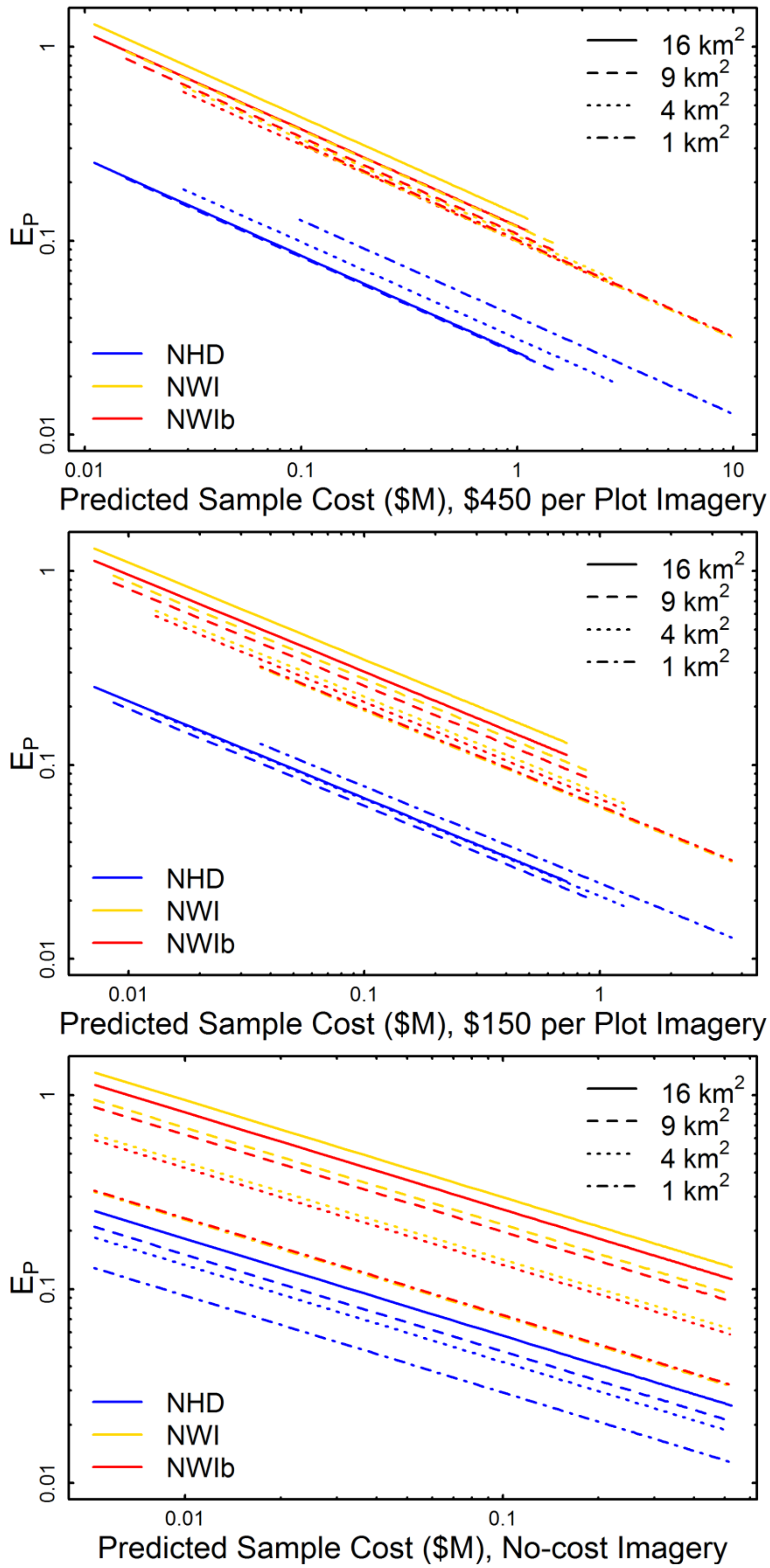


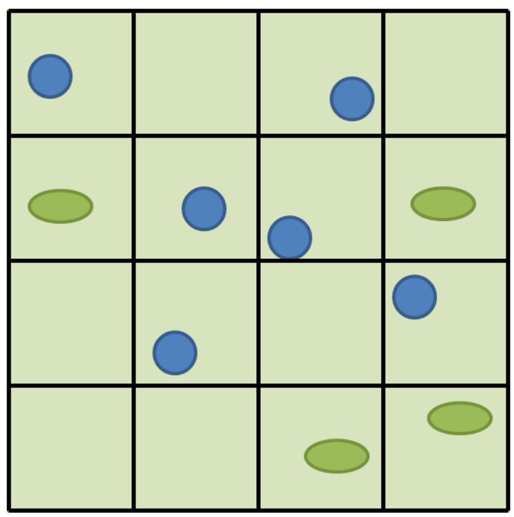
Figure 2.5. Estimated percent error by predicted sample cost.
Estimated percent error was calculated based on the 95% confidence interval for the mean. Lines show relationships by data type — NHD (blue), NWI (gold), and NWIb (red) — and plot size — 16 km^2 (solid), 9 km^2 (dashed), 4 km^2 (dotted), and 1 km^2 (dashed and dotted). Both axes have log scales.

Supplemental Files

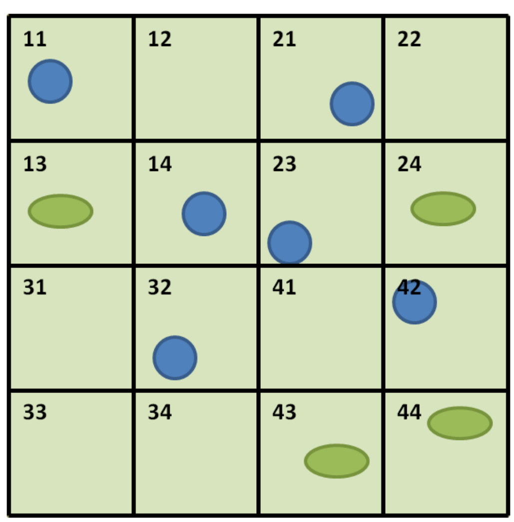
Supplemental File S2.1. Generalized Random Tessellation Stratified Sampling.

Generalized random tessellation stratified (GRTS) sampling can make use of the same design-based equations for determination of sample mean and variance as SRS. However, the method for selecting sample locations is distinctly different. Several open-source and proprietary computer programs have been developed to assist in GRTS sample selection. However, the method is the same for all.

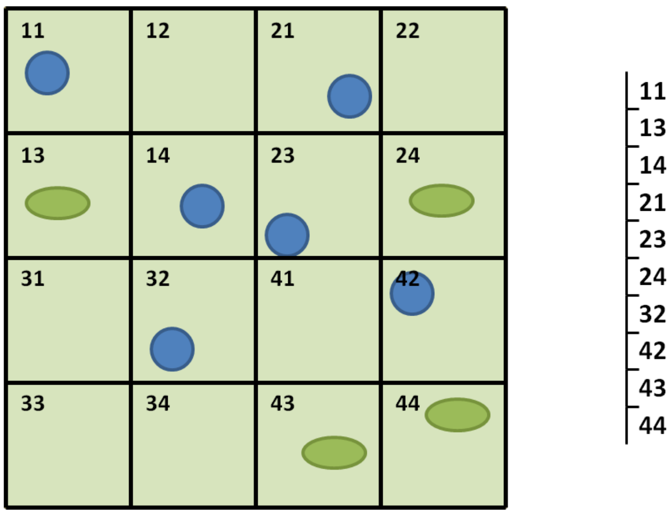
First, a regular grid is placed of the target area and subdivided until there is no more than one individual per grid cell. In our simulations, the grid is subdivided until the target cell size is reached.



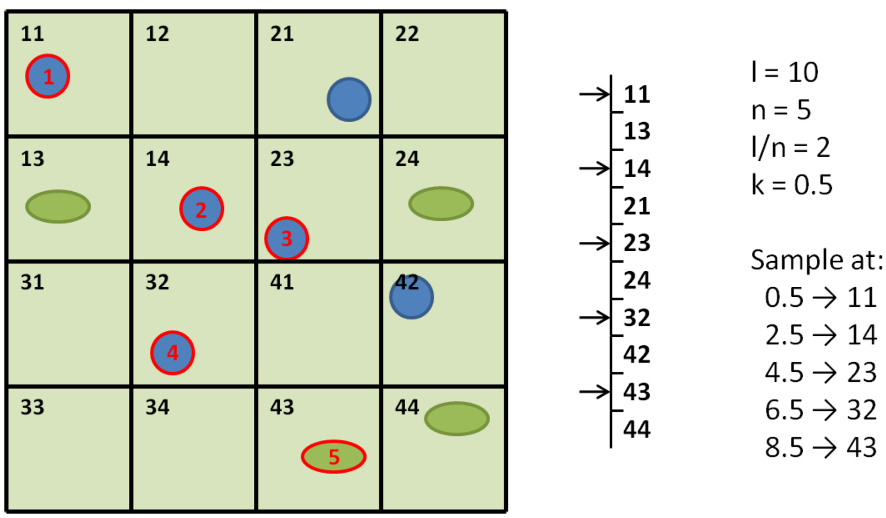
Second, a hierarchical address is assigned to each cell in the grid.



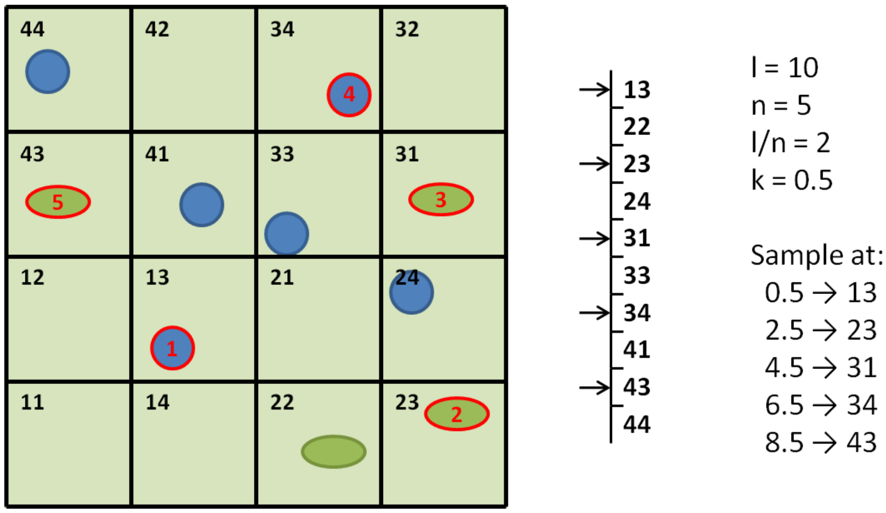
Third, the addresses are placed in order on a line if the cell contains the target. In our simulations, all grid cells overlapping California would be considered to contain the target.



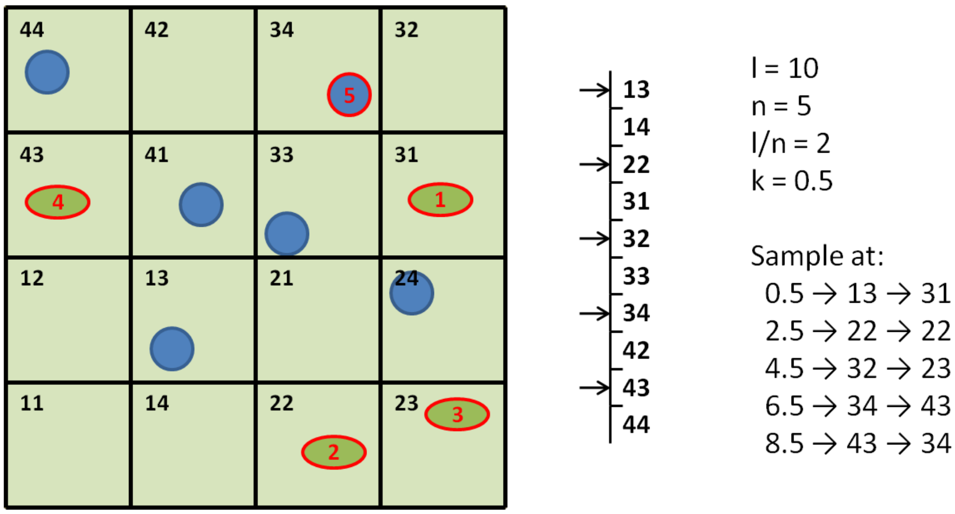
Fourth, a random starting point is placed on the line at a random location *k*, between *0* and *l/n* where *l* is the total length of the line and *n* is the desired sample size. Then, sampling occurs at *k* and all points $k + (i - 1) * (l/n)$ where $i = 2, 3, \ldots, n$. When sites are visited in order, the distance between subsequent points is generally minimized spatial balance is maintained. This also allows for substitution of sample locations from the same geographic area. This is advantageous when substitutions must be made in the field.



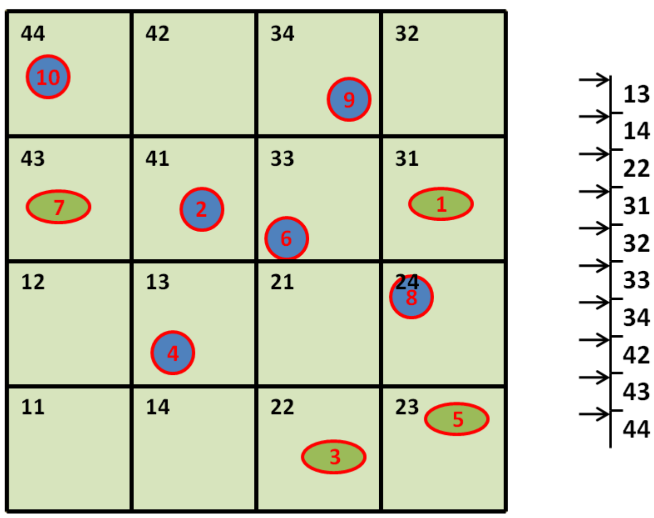
Alternatively, hierarchical addresses could be assigned randomly in the second step. This results in a greater median distance between sequential sample locations while still keeping locations within a somewhat limited area. Substitution of sample locations in the field is no longer efficient but the area observed by sequential locations is larger.



In addition, reversed, randomized hierarchical addressing can be used in the second step to map cells to the line based on the reversed address. This maximized the median distance between sequential sample locations. This is ideal for studies, such as ours, where there is no penalty for substituting a distant location. In addition, by maximizing the median distance, sequential samples are rarely concentrated in one subregion.



Finally, a GRTS master sample selects a larger than necessary sample from the population. This can be completed using any of the three addressing approaches. Once selected, the master sample provides a convenient mechanism for substituting sample locations while maintaining spatial balance. The master sample can even be sorted based on auxiliary variables and the sample will remain spatially balanced as long as points are selected in order.



I = 10
n = 10
I/n = 1
k = 0.1

Sample at:
0.1 → 13 → 31 (1st green)
1.1 → 14 → 41 (1st blue)
2.1 → 22 → 22 (2nd green)
3.1 → 31 → 13 (2nd blue)
4.1 → 32 → 23 (3rd green)
5.1 → 33 → 33 (3rd blue)
6.1 → 34 → 43 (4th green)
7.1 → 42 → 24 (4th blue)
8.1 → 43 → 34 (5th blue)
9.1 → 44 → 44 (6th blue)

CHAPTER THREE: TEMPORAL SAMPLING DESIGN

Monitoring Aquatic Resource Extent over Time: Fixed Versus Moving Sample Plots

Abstract

Accurate monitoring of wetland extent and changes over time provides context for scientific investigations, enables informed management, and measures progress towards no-net-loss policy goals. Previous simulate sampling work has supported probabilistic sampling and mapping using spatially balanced sampling methods for selecting observation and mapping locations. However, approaches for monitoring over time have not yet been evaluated. Existing monitoring programs employ some form of fixed sampling locations but sampling and monitoring theory suggests hybrid sampling approaches, where a mixture of fixed and moving locations are observed at each timepoint, may also be useful. This hybrid approach, termed sampling with partial replacement (SPR), theoretically provides improved balancing of status and trends monitoring. The study presented here utilized simulated sampling to assess the performance of fixed sampling locations, SPR, and moving locations. Modeled changes in wetland density over time were used as inputs for sampling simulations. Results support use of fixed sampling locations over time and, for the populations tested, show that SPR may under-estimate current values and changes over time.

Introduction

Regional and national wetland management strategies implicitly rely on knowledge of the extent and distribution of wetland resources, and how distribution changes with time. Knowledge of current status is required for appropriate prioritization of protection, restoration, management, and monitoring efforts (Euliss et al. 2008). Similarly, knowledge of trends is necessary for objectively determining program efficacy and efficiency (Fancy et al. 2009).

Unfortunately, many regional and national wetland mapping programs lack the capacity for routine assessment of wetland status and trends. Comprehensive mapping of all wetland resources is the gold standard for extent and distribution information, but is prohibitively time-consuming and expensive for large geographic areas. Design-based, probabilistic sampling and mapping can provide region-wide estimates quickly and accurately and has emerged as a possible alternative to, but not a complete substitute for, comprehensive approaches (Dahl 2011; Kloiber 2010; Ståhl et al. 2010). Due to their increased efficiency, design-based approaches are appropriately suited for providing status and trends estimates of wetland extent and distribution at a regional or national scale (Nusser et al. 1998).

Existing programs and simulation studies support the use of spatially balanced, probabilistic sample selection methods, with plot sizes reflective of the information needs of the program and the density of wetland resources (Rossi 2004; Ståhl et al. 2010; Theobald et al. 2007). Spatially balanced designs have been shown to reduce sample variance and increase precision, perhaps by reducing the impact of spatial autocorrelation (Stevens and Olsen 2004; Stevens and Olsen 2003). The effects of spatial balance may also exceed possible increases in precision through stratification (Chapter 2). Finally, appropriate plot sizes allow program managers to control program costs while still meeting landscape-level information needs

(Chapter 2). However, the previous work has focused primarily on design parameters for the initial selection of monitoring locations. Optimization has been confined to accurate and precise monitoring of the status of wetland extent and distribution, and has not explicitly considered monitoring of trends over time. These two information needs may lead to different sampling and monitoring strategies.

Most existing monitoring programs for aquatic resource extent employ fixed plot locations (Dahl 2011; Kloiber 2010; Ståhl et al. 2010). This design implicitly prioritizes detection of small trends over detection of differences in extent or trends across the study area. This design also assumes the initial draw of sample locations is sufficient to represent spatial variability and heterogeneity in the study area (Scott 1998). However, fixed sampling plots may produce inaccurate or imprecise estimates of status and trends if spatial variability is large or different temporal trends are found in different spatial areas (referred to as "spatio-temporal" trends in this paper). This could occur if the initial sample plot locations are, by chance, non-representative of the full population. In this situation, alternate temporal designs, such as using moving locations to observe a new sample at each timepoint, may provide increased power to accurately detect the temporal trends in the population. However, use of moving locations eliminates the possibility of paired change analysis, which is statistically powerful and may reduce mapping costs (Deegan and Aunan 2006; Thomas and Taylor 2006).

In contrast to the existing wetland programs, simulation studies in forestry and fisheries have suggested that sampling with partial replacement (SPR), can more efficiently monitor changes over time while also providing efficient estimates of status (Patterson 1950; Ranneby and Rovainen 1995; Warren 1994). SPR combines observations at fixed locations, which are observed at least twice, with observations at moving locations, which are observed only once.

SPR provides a balance between utilizing fixed sampling locations, which may under-represent spatial variability and heterogeneity, and moving sampling locations, which may reduce power to detect small changes (Scott 1998). In addition, SPR theory can combine observations from fixed and moving sampling locations if locations change due to factors such as loss of access, addition of new locations, or a concern that repeated observations could alter site conditions.

This study examined the efficiency and efficacy of SPR sampling and observation for monitoring the status and trends of wetland resources. In contrast to target variables in fisheries and forestry studies, wetland area density (i.e., area of wetland per square kilometer) is constrained between zero and one and typically has a substantial number of zero values. In addition, wetland density estimation is sensitive to the scale of observation due to the patchy and heterogeneous spatial distribution of resources. This study began by comparing SPR sampling to completely fixed and completely moving sample locations. Then, if SPR showed advantages over fixed and/or moving sampling locations, the ratio of fixed to moving sampling locations was optimized to balance status and trends (S&T) monitoring. The study was conducted for two regions in California and conclusions will be used to help develop a statewide status and trends monitoring program. The statewide program must be able to provide estimates for key resource subtypes and have flexibility to accommodate regional intensification and alteration of sampling protocols as motivating questions mature.

Temporal sampling designs were simulated on modeled changes in wetland extent. Simulated sampling can produce empirical distributions of sample point estimates, allowing us to balance status monitoring with trends monitoring, evaluate different ratios of fixed and moving plot locations, and consider the cost-effectiveness and reliability of the designs. The modeled changes in wetland extent were based on future impacts to wetlands as the footprint of developed areas expands. This driver of change was selected because the State of California is particularly interested in monitoring wetland losses due to development and land-use change. By considering two models of development impacts, one assuming impacts to wetlands are concentrated around metropolitan areas and one assuming impacts are evenly distributed across all developed areas, this study also provides preliminary indicators of possible impacts to wetlands under different development patterns. Results in these areas were based only on the modeled changes in developed area and thus should be used for hypothesis formation instead of planning or management actions.

Methods

General Approach

Future wetland losses were modeled in two separate regions using a simple, GIS-based approach. Existing land use was overlaid on current wetland maps and losses were simulated using two variations of development. The four modeled combinations of study area and impacts allowed us to evaluate the candidate sampling designs across a range of spatially and temporally heterogeneous populations. The primary purpose of the model was not to predict future changes in wetland area and density. Instead, the purpose was to produce a range of spatial and temporal trends for use in evaluating the temporal designs.

Temporal sampling and observation designs were simulated on the modeled changed in wetland density. Simulated sampling was selected because of its ability to provide empirical distributions of sample estimates such as the current mean wetland density, the change in wetland density since the previous observation timepoint, and trends in wetland density over time. Three temporal observation strategies were tested: fixed locations, moving locations, and SPR. Strategies were compared using the statistical accuracy and precision of the empirical distributions of the sample mean and changes and trends in the sample mean over time. Designs were selected and optimized to balance status and trend monitoring. Consideration was also given to the objectives of the California S&T monitoring program.

Study Areas

Two study areas were selected based on availability of high quality, contemporary wetland maps, the San Francisco Bay area (hereafter referred to as the Bay Area), and the central coast of California (hereafter referred to as the Central Coast) (Figure 3.1). Both areas were mapped primarily using 2005 National Agricultural Imagery Program (NAIP) imagery. Supplementary imagery, such as 2009 NAIP imagery, was used in limited areas but, overall, the maps were assumed to represent wetland extent and location for the year 2005. Bay Area wetland maps are viewable online through the California Wetlands Portal[1] and geodatabases were obtained directly from the map producer. Central Coast wetland maps are viewable and available for download as part of the National Wetland Inventory (NWI).[2] In the Bay Area, wetlands were classified according to a modified version of the California Rapid Assessment Method (Collins et al. 2008). The Central Coast area was classified consistent with NWI procedures (Cowardin et al. 1979).

[1] www.californiawetlands.net/tracker/ba/map
[2] http://www.fws.gov/wetlands/Data/Data-Download.html

Impact Scenarios

Two different impact scenarios were produced for each study area. The impact scenarios were based on simulated loss of wetlands in areas surrounding existing developed areas. Each scenario produced a map of wetland area density at eleven timesteps. Scenarios and density changes were developed and modeled in ArcGIS (ESRI 2010).

Expansion of impacted areas around currently developed areas was the driving force in the impact scenarios. US Census places, available as TIGER/Line shapefiles,[3] were used to spatially define existing development. Both incorporated places, which have legally defined boundaries, and Census-designated places, which are defined for unincorporated areas based on population were used (U.S. Census Bureau 2012). For simplicity, we refer to both incorporated and census-designated places as "places." We also considered which places had been designated as metro or micropolitan principal cities by the US Census. Metro and micropolitan areas are designated by the US Census by first identifying individual places with at least 50,000 or 10,000 individuals, respectively (U.S. Census Bureau 2012). These high-population places are considered the core of the metro or micropolitan area. Then, surrounding places with a high degree of economic and social integration with the core place are designated as part of the wider metro or micropolitan area. Finally, the largest place, by population, within each metro or micropolitan area is designated as the principal city.

Our first impact scenario, referred to here as "metro impacts," buffered each metro or micropolitan principal city by 1 km between each timestep, for a total of 10 buffers. In our second impact scenario, referred to here as "place impacts," all places were buffered by 0.5 km between each timestep. Existing places at time zero were erased from each buffer to remove

[3] http://www.census.gov/cgi-bin/geo/shapefiles2010/main

areas of “existing” impact. Protected areas, as defined by the California Protected Areas Database,[4] were also clipped from the buffers to account for current spatial protection policies. Resulting impact buffers under the two scenarios are given in Figure 3.2.

The resulting buffers were used to produce changes in wetland density as follows. First, a 500 m square grid (0.25 km^2) was generated for each study area and the area of existing wetlands was determined for each grid cell from the existing wetland map for each study area. This area was used as the baseline (time zero) value for both impact scenarios. Estimates were produced for total wetlands and for estuarine, lacustrine, palustrine, riverine, vernal pool and marine (for the Central Coast only) subtypes. Multiple subtypes were used in order to increase the range of spatio-temporal changes available for observation. We defined vernal pools using the palustrine, unconsolidated shore, seasonally flooded (PUSC) classification and chose vernal pools in order to evaluate the design for a rare wetland type in California (Cowardin et al. 1979; Holland and Jain 1981).

Second, for each timestep after baseline, grid cells were selected if their centroid fell within the corresponding impact buffer. If a cell was selected and contained wetlands, a 50% loss of wetland area was modeled. If a cell had no wetlands, or was outside of the development impact buffer for that timestep, no change in wetland area was modeled. A 50% decrease was considered appropriate in light of existing Federal and State programs whose goals are to avoid and minimize wetland losses associated with new development.

Third, modeled changes in wetland area were aggregated up to four larger grid sizes, 1, 4, 9, and 16 km^2. Aggregated wetland areas for each grid cell were then divided by either the cell area or, if the cell fell on the study area boundary, by the portion of the study area within the cell.

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[4] www.calands.org

The final product for each cell was wetland density, for all wetland types, at eleven timepoints under the two impact scenarios.

Finally, the resulting 1, 4, 9, and 16 km^2 grids were used to conduct the sampling simulations. The four grid sizes we used were based on previous work that described the close relationship between plot size, sample variance, mapping cost and sample error (Chapter 2). In brief, smaller plots produce higher sample variances but are less expensive to map. Since the ultimate selection of plot size will be made by CA program managers, who are concerned with both statistical value and financial costs, this study employed a range of plot sizes to determine if plot size influences conclusions about the use of a fixed, moving, or SPR design.

Sampling Simulations

Sampling and monitoring designs were simulated 5,000 times for each study area and grid size. Each repetition, one sample was drawn for each of the simulated temporal designs. The samples were then analyzed to produce point estimates of status and trends for each timepoint modeled under the two impact scenarios. We used 5,000 repetitions to assemble empirical distributions of the sample point estimates (Miller and Ambrose 2000). All simulations were performed in R using the spsurvey package (Kincaid and Olsen 2011; R Development Core Team 2011).

We used random number seeds for reproducibility. Like most computer languages, R utilizes a pseudo random number generator (pRNG) to produce a sequence of numbers that lack any discernible pattern. While not fully random, pRNG's such as the default in R, *Mersenne-Twister*, pass statistical tests for randomness (Matsumoto and Nishimura 1998). In addition, because pRNG's use one starting value to produce a string of apparently random results, if the same starting value is used, the same string of numbers will be generated. Therefore, we ensured that simulation results could be reproduced exactly at a later date by setting the seed with a known value before conducting the simulations.

To start each repetition, a master set of locations was selected by Generalized Random Tessellation Stratified (GRTS) sampling. Previous work has shown that unstratified GRTS sampling offers increased precision over non spatially-balanced methods such as simple random sampling (Chapter 2). Under a master sample approach, a single list of sample locations is selected at time zero and sample locations are selected as needed from the list. Beyond convenience, the GRTS master sample will maintain spatial balance in the observed sample as long as sample locations are selected in order from the list, and, if appropriate for the design, can provide new, previously unobserved locations at each timepoint.

Five separate master samples were selected at each repetition, one for each temporal design: fixed locations; moving locations; and SPR with three different ratios of fixed and moving locations (3 fixed to 1 moving, 1 fixed to 1 moving, and 1 fixed to 3 moving). The size of each master sample (*n*) was defined by:

$$n = N \cdot \rho_{plots}\left(\frac{n_f}{n_f + n_m} + t_{total} \cdot \frac{n_m}{n_f + n_m}\right) \tag{3.1}$$

where N is the total number of grid cells (i.e., the population size) and n_f and n_m are the number of fixed and moving locations, respectively, observed at each timepoint. We modeled a sample plot density (ρ_{plots}) of 5% of the population and 11 total timepoints (t_{time}). The plot density was selected somewhat arbitrarily, but was large enough to provide a reasonable sample size, 40, for the 16 km^2 grid. 5% is also low enough to preclude application of the finite population correction factor, which is recommended when the sample size is greater than 5% of the population size (Isserlis 1918). The finite population correction factor, which decreases the sample variance, is recommended because the calculation of sample variance assumes the population is much larger than the sample size. At a 5% sample rate, the finite population correction factor decreases the sample variance by only 2% and is therefore considered unnecessary.

After the five master samples were drawn, modeled wetland densities were used to calculate point estimates of status and trends at each timepoint. The type of sample, e.g., fixed or SPR, determined which analysis method was used. In the case of SPR, several analysis methods are available in the literature for estimating extent and trends. We selected two options and provide equations in Supplemental Table S3.1 along with references providing derivations. The following will provide a brief overview of the types of approaches used in each instance, and a brief explanation for why we selected these particular methodologies.

For fixed and moving samples, three types of point estimates were calculated (Table 3.1):

- The *mean* of all observed locations at that timepoint.
- The *difference* between the current mean and the mean from the previous timepoint.
- The *trend* component of an ordinary least squares regression of the sample mean over time. For example, for year 2015, the third timepoint, regression was performed between the sample means from timepoints 1-3, and the corresponding timepoint. The trend was taken as the slope of the regression.

As the difference and trend require at least two observation timepoints, only the mean was recorded for baseline and the difference and trend were recorded at all successive timepoints.

After recording point estimates for fixed and moving samples, estimates were recorded for SPR samples. As mentioned previously, multiple SPR approaches have been developed and the approach utilized has the potential to affect conclusions about the accuracy and precision of the SPR approach. A variance-minimizing approach was used here because it is considered statistically efficient and is simple to apply in simulation. Efficiency and ease of use are also important considerations for the State of California monitoring program.

Two types of variance-minimizing SPR were used: (i) SPR_2, which combines observations from the current and previous timepoint; and (ii) SPR_3, which combines observations from the current and two previous timepoints. In both cases, observations at the fixed and moving locations are combined to produce estimates of the current mean and the change in the current mean since the previous timepoint. In principal, SPR theory can be expanded indefinitely to accommodate all observation timepoints. However, in practice, such expansions quickly increase computational intensity and complexity. In addition, the basis for SPR theory is that past conditions at a given location are correlated with current conditions at the same location. As the time between the past and current timepoints increases, this assumption becomes less reasonable. Therefore, application of SPR beyond the two previous timepoints is typically only recommended after examining the appropriateness of this assumption.

Four different SPR point estimates were recorded (Table 3.1):

- The *SPR mean* for the current timepoint. *SPR mean* combines the mean of observations from the current timepoint with an extrapolated mean based on observations at previous timepoints.
- The *SPR change* since the previous timepoint. *SPR change* combines paired and unpaired differences between the current and the previous timepoint using a variance minimizing approach.
- The *difference* between the *SPR mean* for the current timepoint and the *SPR mean* from the previous timepoint. The *difference* does not consider paired and unpaired differences separately.

- The *trend* component of the ordinary least squares regression of the *SPR mean* over time.

All four estimates were calculated using both the SPR_2 and the SPR_3 approaches. Because SPR_2 and SPR_3 require two and three observation timepoints, respectively, the simple mean was calculated at baseline, SPR_2 only at the second timepoint, and both SPR_2 and SPR_3 after the second timepoint.

Analysis of Empirical Sampling Distributions

Sampling simulations produced empirical distributions of sample point estimates of status and trends (Table 3.1). To compare the different sampling and monitoring options, four summary measures were calculated from the empirical distributions.

First, Cohen's d (d_C) was used to measure bias between the mean of the distribution of sample point estimates ($\bar{x}$) and the "true" population value (μ), relative to the distribution standard deviation (s):

$$d_C = \frac{\bar{x} - \mu}{s} \tag{3.2}$$

The true population mean was defined as the mean wetland density of all grid cells in that study area and at that timepoint. Cohen's d cannot provide a p-value for bias but can provide an indication of effect size and is not influenced by sample size. Because sample size in this analysis is only limited by the number of repetitions performed, use of statistical tests sensitive to the number of observations could lead to conclusion that small, but not particularly meaningful differences were statistically significant. Traditional guidelines were used for Cohen's d to define small (0.2-0.5), medium (0.5-0.8), and large (>0.8), biases (Cohen 1988). These values are somewhat arbitrary, as with most statistical cutoffs and critical points, but define a large bias as approaching or exceeding the sample standard deviation. Similarly, a small bias is one that is less than half the sample standard deviation.

Second, the precision of sample estimates was compared using the standard deviation of the empirical distributions. Coefficient of variation (the sample standard deviation over the sample mean) was not used as small mean values for difference, change, and trends tended to produced extremely volatile ratios.

Third, the precision and reliability of sample estimates was compared by determining the fraction (f_E) of sample estimates (x) that fell outside of a range defined by the true population value (μ), the population standard deviation (σ), the simulated sample size (n), and the probability density function for the normal distribution (Z):

$$f_E = P\left(x < \mu + \frac{\sigma}{\sqrt{n}} \cdot Z_{0.025}\right) + P\left(x > \mu + \frac{\sigma}{\sqrt{n}} \cdot Z_{0.975}\right) \tag{3.3}$$

This interval is analogous to the sample confidence interval for a point estimate, but utilizes the population mean and variance. The f_E was developed to indicate the frequency with which a sample estimate was "extreme" relative to the corresponding population parameter.

Finally, the distribution of simulated means was used to estimate the percent sampling error (E_P) for an estimated sample size (n_e), if the temporal sampling strategy were applied to the entire State of California:

$$E_P = \left(Z_{1-\frac{\alpha}{2}}\left(\frac{s}{\bar{x}}\right)\sqrt{\frac{n}{n_e}}\right)100\% \tag{3.4}$$

In the above, $Z_{1-\alpha/2}$ is the normal distribution value with a cumulative probability of $1 - \alpha/2$, where α is the probability of a type I error. Estimated errors were compared against estimated sample sizes and predicted image acquisition and map production sampling costs, produced based on experience in these areas (Chapter 2). For image acquisition, two main options were considered, no-cost, existing imagery from the National Agriculture Imagery Program and contract imagery from third party vendors. For the contract imagery, costs per plot were assumed independent of plot size and lower and upper limits of 150 and 450 USD were used. Map production costs were assumed 25 USD km^{-2} for all plot sizes.

Results

Model Output

The GIS model produced total wetland density losses between 7.7 and 20% over the modeled time span (Figure 3.3). The place impacts scenario tended to produce greater losses of wetland density than the metro impacts scenario; differences between scenarios at the final timepoint were between 0.9 and 18 percentage points, depending on wetland type. The single exception was for palustrine wetlands in the Bay Area. In this case, under place impacts, 15.7% of original wetlands were lost, compared to 18.1% under metro impacts. Differences between the metro impacts and place impacts scenario were larger in the Central Coast than in the Bay Area (3-18 percentage points at year 50 for the Central Coast compared to 0.9-6.7 for the Bay Area), reflecting the larger differences between the place and metro impact buffers for the Central Coast compared to the Bay Area (Figure 3.2). Consideration of the empirical variograms (Supplemental Figure S3.1) demonstrated that the modeled losses did not significantly alter the spatial variability structure in the two populations beyond an expected decrease in semivariance as loss occurred.

Importantly, the model output also provided a heterogeneous set of inputs for the sampling and monitoring simulations. Losses for individual wetland types, by impact scenarios and study areas, varied from 0.9% for lacustrine wetlands in the Central Coast under the metro impacts scenario to 29.3% for PUSC wetlands in the Bay Area under the place impacts scenario. In addition, losses were variable over time, ranging from no change to 8.4% between sequential timesteps.

Status

SPR sampling and monitoring resulted in a small to medium negative bias for reporting mean wetland density. Moving sampling locations exhibited more bias than fixed locations, but the effect tended to lack directionality and was insubstantial compared to the bias observed for SPR. For example, Figure 3.4 illustrates the range of Cohen's d, d_C, values over time for all wetlands in the Bay Area and the Central Coast, using a 16 km^2 grid size. In this case, SPR sampling and monitoring produced Cohen's d values between -0.69 and -0.22, compared to -0.03 to 0.04 for moving locations and 0.00 to 0.02 for fixed locations. Study area, impact scenario, grid size and wetland type did not affect conclusions about bias.

Sampling and monitoring strategies had a more complicated relationship with precision. However, results as a whole still suggest fixed sampling may be more reliable than moving sampling locations or SPR sampling. Fixed sampling locations were more precise than moving locations for all grid sizes, wetland types, impact scenarios, and study areas. Compared to SPR, fixed sampling tended to be more precise in the Bay Area, but less precise in the Central Coast. For example, Figure 3.5 shows the extreme fraction, f_E, for the current mean wetland density in the Bay Area and Central Coast, using a 16 km^2 grid size. In this instance, fixed sampling locations were associated with extreme fraction values between 0.006 and 0.049 compared to 0.010 to 0.055 for moving locations. Importantly, the values of f_E for fixed locations was within the 0.05 limit expected from the theoretical basis for the measure (Equation 3.3). SPR values for f_E were more variable and differed significantly between the study areas. In the Bay area, extreme fraction values fell between 0.001 and 0.169 while in the central coast, the extreme fraction fell between 0.0004 and 0.0216. Grid size and wetland type did not affect conclusions.

Trends

Three different measures of change over time were examined: (i) the difference between the current mean and the previous mean; (ii) the trend in current means over time since baseline; and (iii) for SPR sampling and monitoring, the SPR change since the previous timepoint. Examination of bias and precision for these three measures provided further support for use of fixed sampling and monitoring locations. The result did not depend on the specific metric but, for SPR sampling, the change metric may be more precise than the difference. However, SPR change still exhibited a consistent negative bias.

Considering the difference between the current and the previous mean, none of the sampling and monitoring strategies exhibited substantial bias but fixed sampling locations were more precise than moving locations or SPR. For example, Figure 3.6 shows standard deviation, *s*, for the difference between means for all wetlands in the Bay Area and Central Coast, using a 16 km^2 grid size. Fixed sampling locations had standard deviations of 0.001-0.003 for the Bay Area and 0.0001-0.0008 for the Central Coast. In contrast, moving sample locations had standard deviations of 0.050-0.065 and 0.011-0.013 for the Bay Area and Central Coast, respectively; SPR sampling produced standard deviations of 0.024-0.062 and 0.0048-0.0090, respectively. The order of magnitude difference between the Bay Area and the Central Coast is reflective of an order of magnitude difference in total wetland density between the two regions.

Examination of the trend in the current mean (results not shown) was consistent with results for the difference. Fixed sampling locations had less bias and were more precise than moving locations or SPR sampling. In addition, SPR sampling showed a negative bias in the trend — a direct consequence of the negative bias in the SPR mean.

For SPR sampling and monitoring, the difference between the current and the previous mean was compared to the SPR change since the previous timepoint for both bias and precision (results not shown). This comparison showed some possible bias for SPR change, depending on the impact scenario and study area. The comparison also showed that SPR change might have higher precision than the difference between SPR means.

Estimated Percent Error

After reviewing the results supporting use of fixed sampling locations over moving locations or SPR, a cost-analysis was performed to compare predicted sampling costs to estimated percent error for the point estimates of mean, difference, and trend produced by fixed sampling and monitoring. Plot size and sample size were also considered.

The smallest plot size (1 km^2) was most cost effective for minimizing the percent error in estimates of the current mean wetland density and trends in mean density over time. Figure 3.7 shows estimated percent error versus predicted sample costs, when using existing imagery, for the Bay Area and the Central Coast. At a predicted cost of approximately \$100,000, 1 km^2 plots had estimated percent errors of 2.0% in the Bay Area and 3.7% in the Central Coast. In comparison, 16 km^2 plots had estimated percent errors of 11% and 12%, respectively. The impact scenario did not affect estimated percent errors and use of contract imagery did not affect conclusions about the most cost effective option for producing estimates of the current mean wetland density.

In contrast, the largest plot size (16 km^2) was most cost effective for estimating the difference in mean wetland density when contract imagery was used, but not when existing imagery was used. Figure 3.8 shows estimated percent error versus predicted sample costs for the Central Coast, based on use of either contract imagery (at $450 per plot) or existing imagery. Considering contract imagery first, at a predicted cost of approximately $500,000, 1 km^2 plots had estimated percent errors of 50% and 30%, under metro and place impacts, respectively. In contrast, 16 km^2 plots had estimated percent errors of only 31 and 17%. For existing imagery, the relationship was reversed. At a predicted cost of approximately $100,000, 1 km^2 plots had estimated percent errors of 27 and 17% while 16 km^2 plots had errors of 46 and 26%. The Bay area had a similar relationship between plot size, estimated percent error, and predicted costs.

Discussion

Temporal Sampling Design

Results support the use of fixed sampling locations for monitoring the status and trends of wetland density. Moving sampling locations did not show significant bias but did show slightly reduced precision, relative to fixed sampling, in the current mean, the difference of means, and the trend in means. SPR sampling, regardless of the ratio of fixed to moving sampling locations, exhibited a small to medium negative bias for estimating the current mean density of wetlands. This negative bias in the current SPR mean translated to a negative bias in estimation of trends in the mean over time. In addition to the negative bias, SPR sampling and monitoring often resulted in reduced precision in status and trends measures. Exceptions did occur where SPR was more precise than fixed sampling locations; however, occasionally increased precision cannot counteract a persistent underestimation of status and trends.

As this study was conducted in only two, relatively small, regions of California, the ability to generalize to the State as a whole, or to other types of spatial populations in other areas, remains an issue. In addition, sampling and monitoring was performed on simulated changes in wetland density, which may not be reflective of actual spatial or temporal patterns in wetland density change. However, the focus of this study was on determining the appropriate mechanism for monitoring changes over time, not on predicting changes over time. In addition, use of two study areas, two impact scenarios, and multiple wetland types greatly increased the diversity of spatial populations and temporal changes available as inputs for the sampling simulations. Finally, the conclusion that fixed sampling locations are the most appropriate strategies for monitoring the status and trends of wetlands in California was not dependent on the study area, the wetland type, or the impact scenario.

This study provides support for existing monitoring programs such as the Status and Trends component of the NWI, the Natural Resources Inventory, the Minnesota Wetland Status and Trends Program, and the National Inventory of Landscapes in Sweden (Dahl 2011; Kloiber 2010; Nusser and Goebel 1997; Ståhl et al. 2010). Each of these programs uses fixed sampling locations to monitor changes in extent of landscape elements such as wetlands. The results from this study strongly support fixed sampling locations as the optimum approach. However, previous work in forestry and fisheries has indicated that SPR sampling and monitoring is equivalent to, if not superior to, fixed sampling locations (Scott 1998; Warren 1994). Reasons for this difference could be related to the different populations and population distributions in question, and how those populations change over time. For example, the area wetland densities used to monitor wetland extent are constrained between zero and one and the population may contain a high proportion of zero values. Because variance-minimizing combination under SPR could place more weight on zero-value observations, with lower associated variance, SPR could be expected to under-estimate status and trends.

Fixed sampling locations are widely taken as the default for monitoring status and trends over time. Absent issues such as loss of access or a need to reduce the impact of observations on a site, fixed sampling locations are attractive for several reasons. For instance, fixed locations and paired differences are a conceptually simple mechanism for controlling unknown spatial and temporal confounders and for isolating trends. In addition, depending on the methodology involved, fixed locations may have reduced costs compared to moving or SPR sampling. Despite this conceptual preference for fixed locations, a perception remains that fixed sampling locations could somehow be biased to the influence of “extreme” and “non-representative” events occurring at a handful of sample locations. Our work, particularly through the f_E measure, illustrates that this perception of fixed locations is not reflective of their statistical robustness.

Modeled Impacts to Wetland Extent

Modeled impacts under the metropolitan and place expansion scenarios have potential implications for wetland loss prevention. No model output can be expected to completely represent reality. In addition, while the model used here was useful for selecting a temporal design for the S&T program, the model was purposefully simplistic and based on reasonable, but still arbitrary, rates of development and assumptions about the impacts of development on wetlands. Finally, these models only represent two relatively small areas of California.

Therefore, the model results should not be seen as anything beyond preliminary and should only be used to develop hypotheses that can be more extensively investigated through more rigorous modeling or through the implemented S&T program.

First, model results showed greater wetland losses under the place impacts scenario than under the metro scenario, even though the metro scenario used a higher rate of expansion. This suggests that there may be more vulnerable wetland area around less population-dense areas than around more population-dense areas. Therefore, wetland loss may be more likely to occur around less population-dense areas. This result is consistent with existing literature comparing dense and diffuse growth and development (Camagni et al. 2002; Echenique et al. 2012). In addition, some of the largest differences between the two impact scenarios occurred for PUSC wetlands, our surrogate for vernal pools. These rare and ecologically important wetlands have special management restrictions in California and the two impact scenarios suggest that protection of vernal pools may require additional focus on non-urban development. These conclusions are obviously strongly influenced by our assumptions for the rate of development expansion and the impact of development on wetlands. Differences between our modeled rates and assumptions and the true rates could significantly affect the validity of this result.

Second, model results also suggest the importance of land protection for reducing potential impacts on wetlands. For example, the Bay Area had lower rates of wetland loss than the Central Coast and the difference between place and metro impacts was larger in the Central Coast than in the Bay Area. These two results are likely driven by both the high number of metropolitan areas in the Bay Area and the increased amount of protected areas. The higher number of metropolitan areas reduced the potential difference between the two impact scenarios and the increased number of protected areas limited impacts to certain geographic areas, which contained a smaller fraction of the baseline wetlands in the Bay Area. Therefore, wetland loss rates in the Bay Area were slightly lower and less dependent on the impact scenario. However, these conclusions are directly influenced by our assumed development rates and assumptions about the absolute nature of land protection. In addition, these results may be highly specific to the geographic areas in question. While it is difficult to believe that increasing the amount of protected land will not help protect wetlands, it is also possible that this approach may not be the most effective approach overall given the logistical difficulties associated with developing protected areas and the potential importance of other factors for wetland loss (Gutzwiller and Flather 2011).

Conclusion

Monitoring of status and trends in a population often requires balancing an appropriate statistical design for status with an appropriate statistical design for trends. This study was undertaken to determine how fixed, moving, and SPR approaches compared for monitoring wetland extent over time in California. Results indicate fixed sample locations are preferable for both status and trends monitoring — perhaps a rare case where the two objectives have a supportable convergence of monitoring designs. This conclusion also supports the design of existing programs, which without exception employ fixed observation locations over time.

Tables

Table 3.1. First timepoint at which sample point estimates were calculated for a given sampling location design.

Sample Point Estimate[a]	Sampling Locations			
	Fixed	**Moving**	**SPR_2**	**SPR_3**
Mean	Baseline	Baseline	2	3
Difference	2	2	3	4
Change	--[b]	--[b]	2	3
Trend	2	2	3	4

[a] Calculation methods and explanations are available in the text and in Supplemental Table S3.1.
[b] Change is not calculated for fixed and moving sampling locations.

Figures

Figure 3.1. Location of study areas within California.
Areas were selected based on availability of high-quality, contemporary wetland maps.

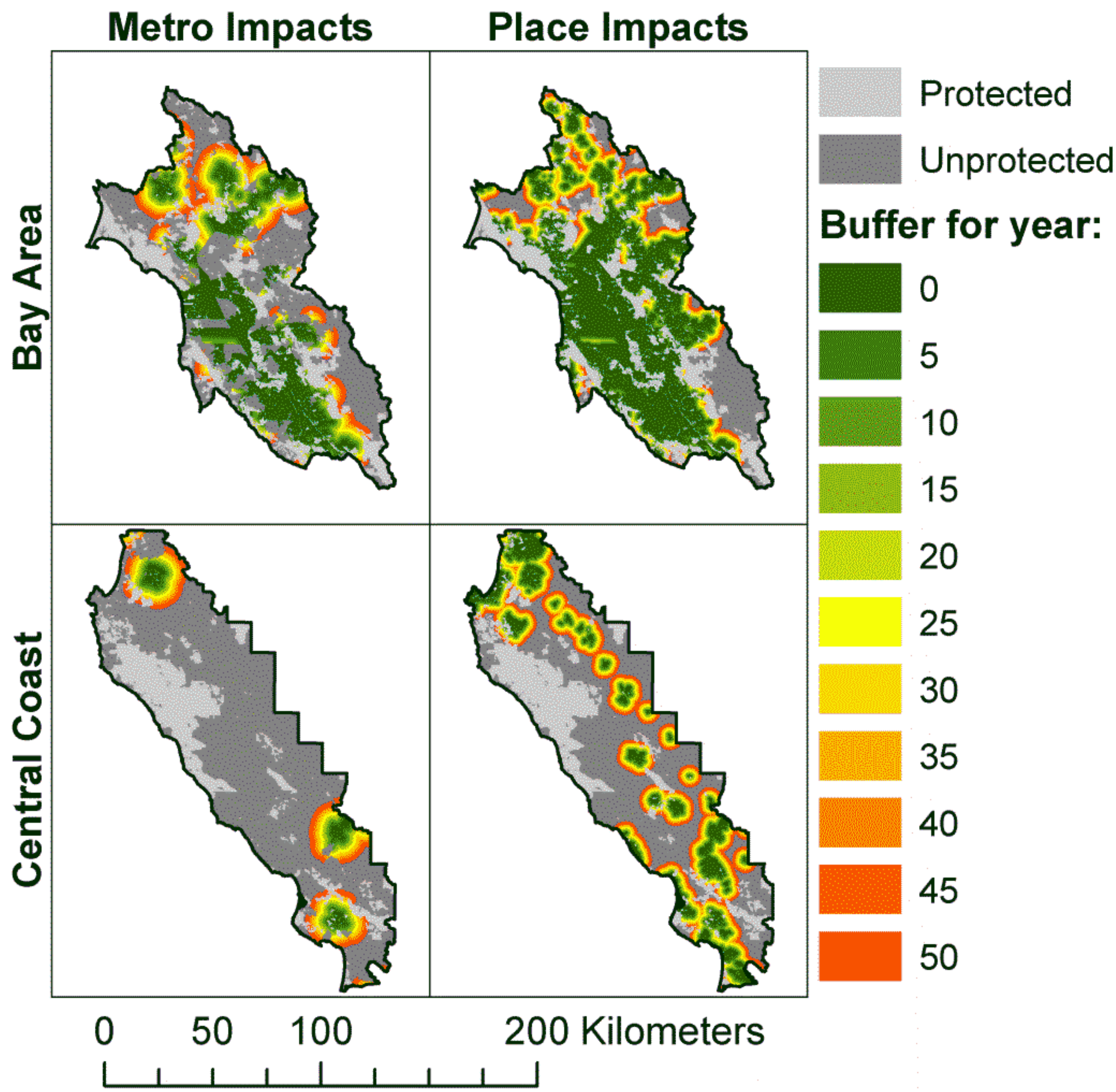


Figure 3.2. Buffers used for modeled impact to wetland area.
Shown are metropolitan (left) and place (right) impacts for the Bay Area (top) and the Central Coast (bottom) for the duration of the study period.

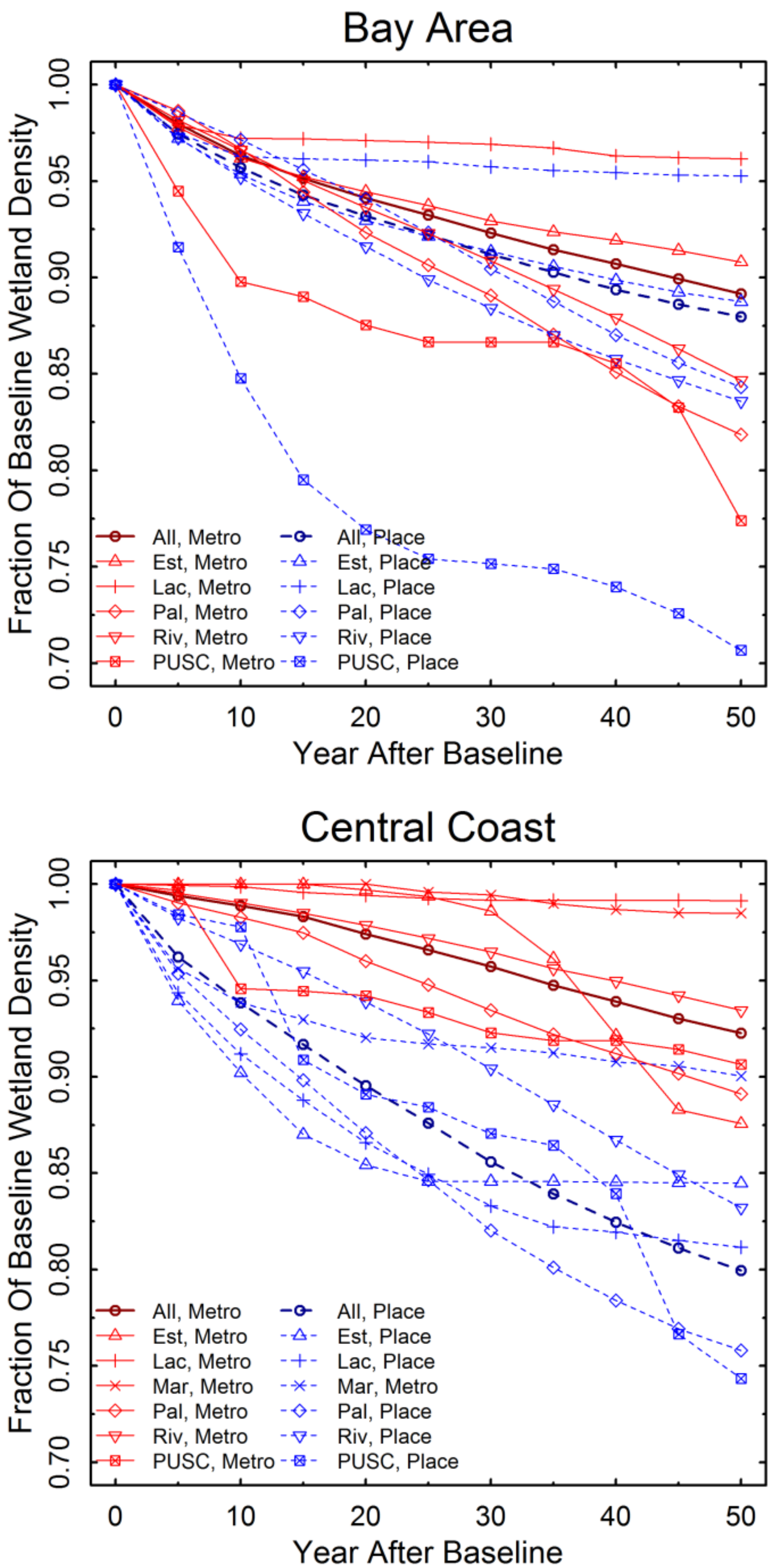
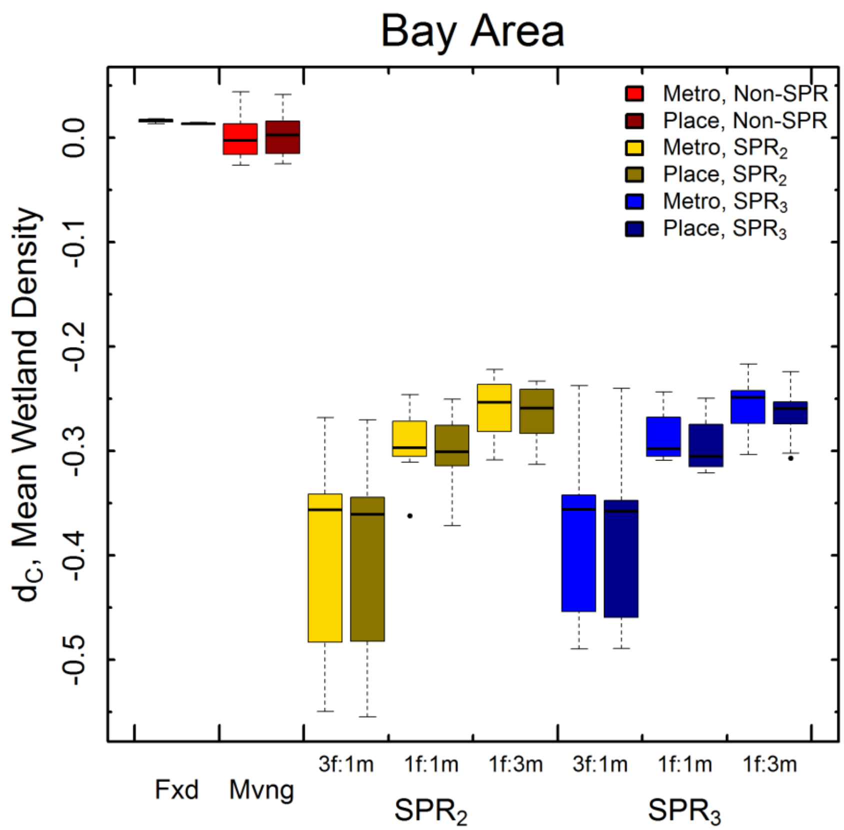


Figure 3.3. Results from modeled impact scenarios. *Red solid lines reflect the metro impacts scenario while blue dotted lines reflect the place impacts scenario. Symbols indicate the resource type.*



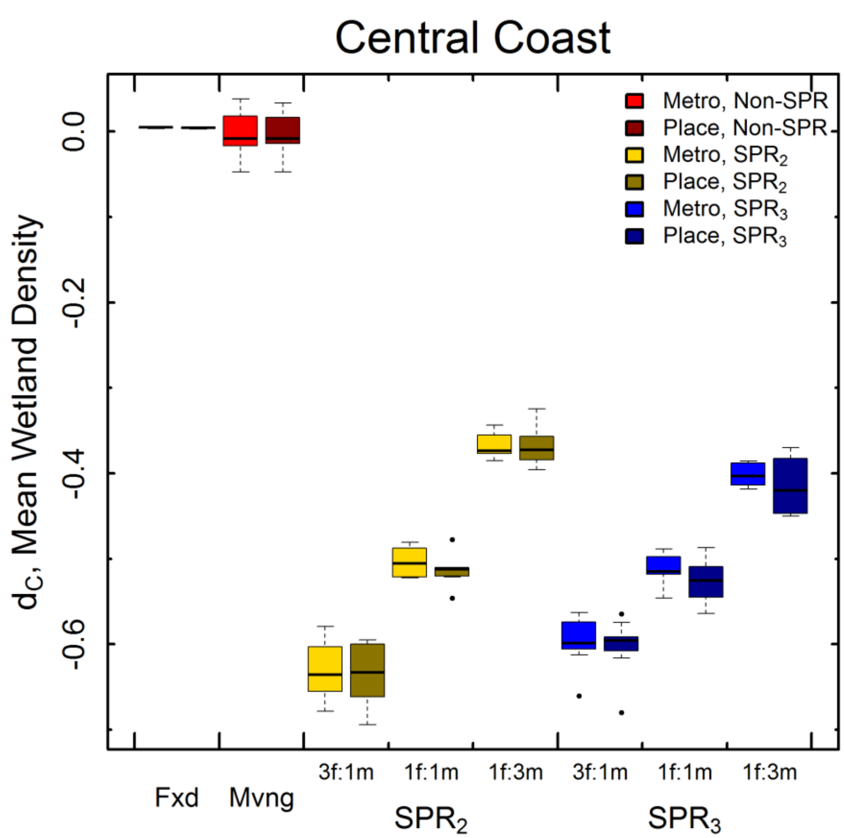
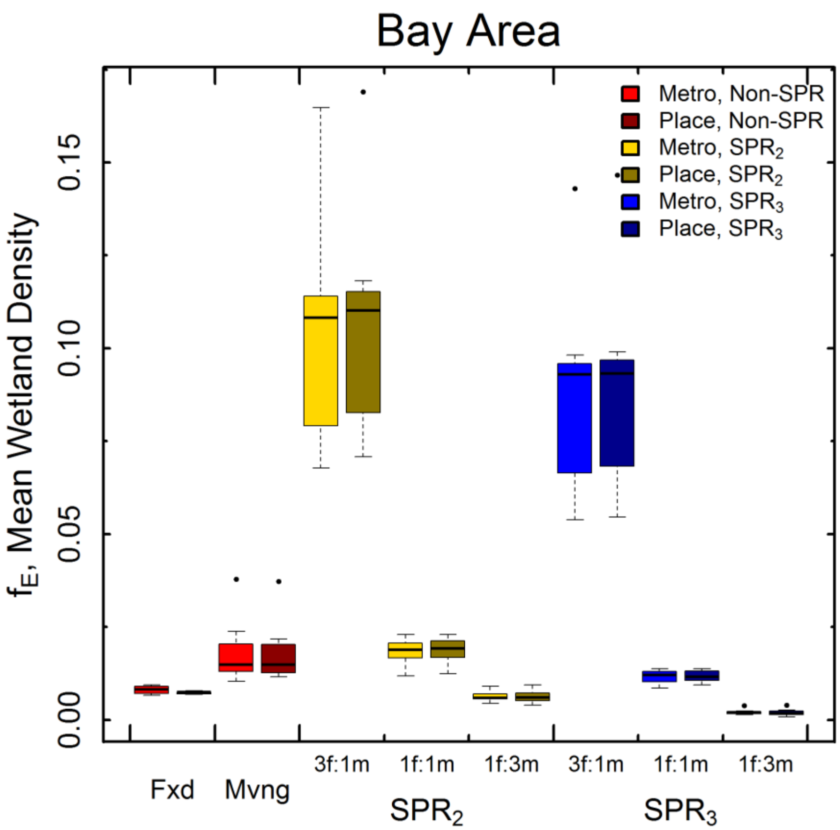


Figure 3.4. d_C for mean wetland density, 16 km^2 plot size.
Box and whisker plots show the distribution of values for all timepoints. Lighted shades of red, gold, and blue indicate the metro impacts scenario while darker shades indicate the place impacts scenario. Red indicates fixed or moving plot locations; gold indicates SPR based on two timepoints; and blue indicates SPR based on three timepoints.



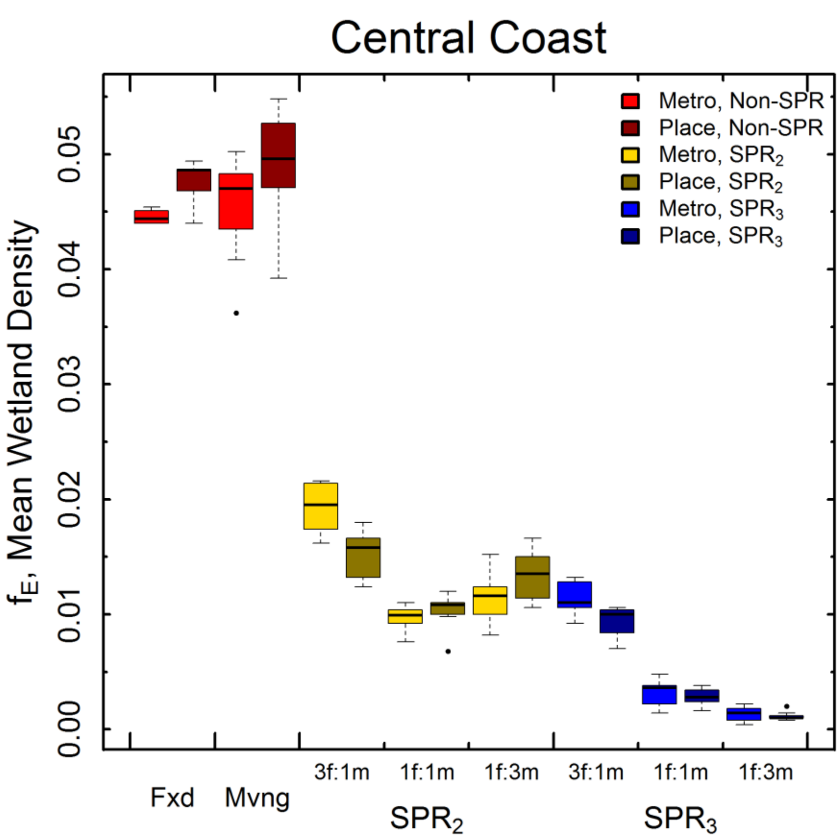


Figure 3.5. f_E for mean wetland density, 16 km^2 plot size.
Box and whisker plots show the distribution of values for all timepoints. Lighted shades of red, gold, and blue indicate the metro impacts scenario while darker shades indicate the place impacts scenario. Red indicates fixed or moving plot locations; gold indicates SPR based on two timepoints; and blue indicates SPR based on three timepoints.

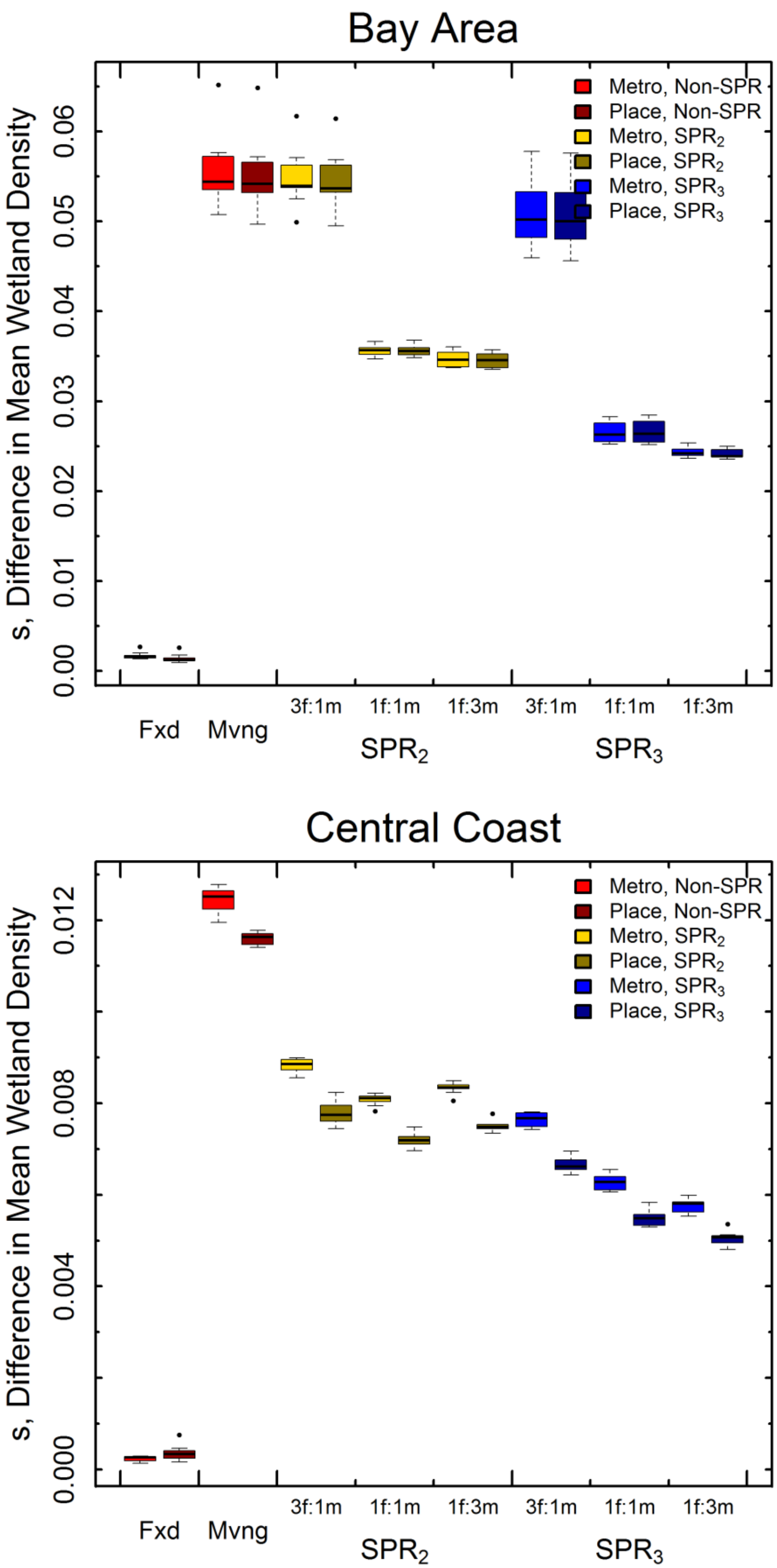
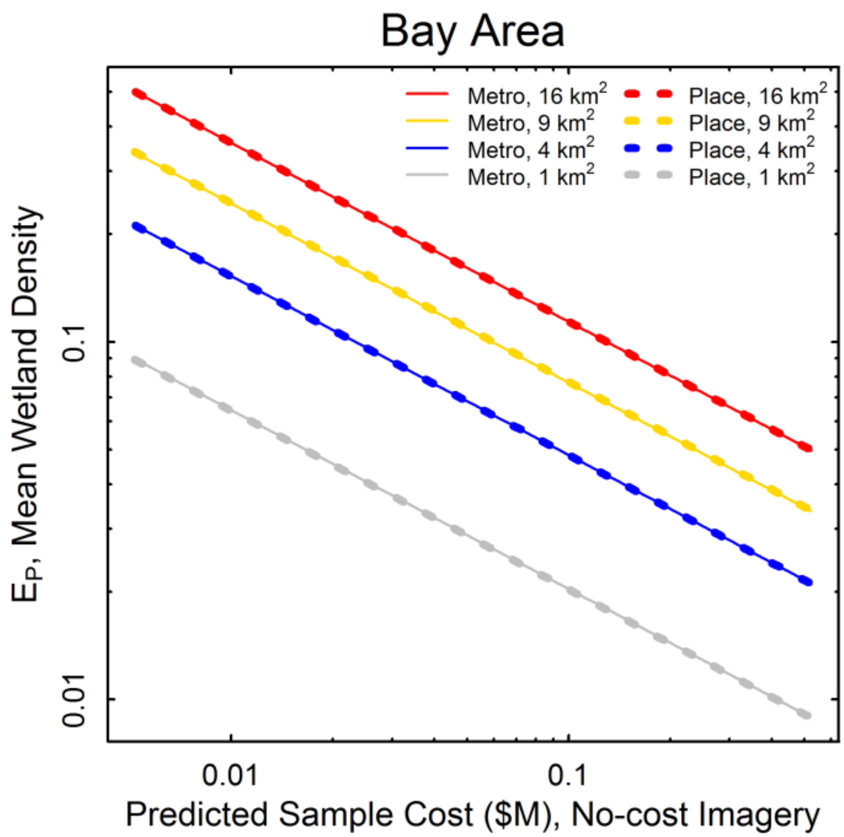


Figure 3.6. *s* for difference in mean wetland density, 16 km^2 plot size.
Box and whisker plots show the distribution of values for all timepoints. Lighted shades of red, gold, and blue indicate the metro impacts scenario while darker shades indicate the place impacts scenario. Red indicates fixed or moving plot locations; gold indicates SPR based on two timepoints; and blue indicates SPR based on three timepoints.



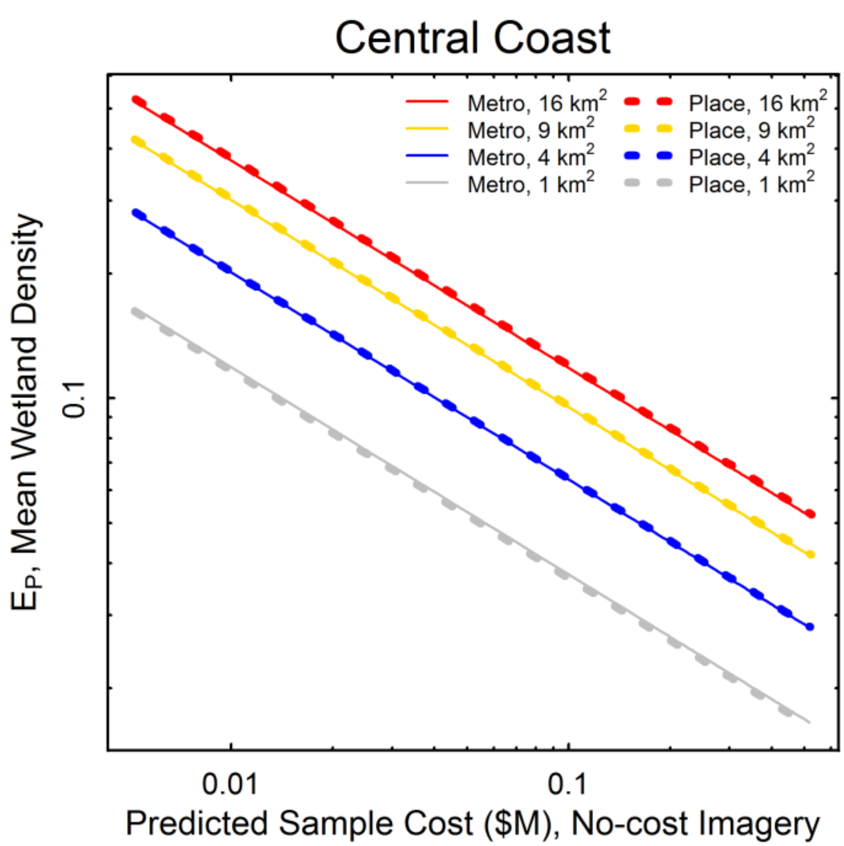


Figure 3.7. Estimated percent error, E_P, for mean wetland density as a function of predicted sample costs.

Both the Bay Area and Central Coast are shown. Results are shown for both metro (solid lines) and place (dotted lines) impact scenarios and costs assume use of existing (i.e., no-cost) imagery. Lines indicate the average from all timepoints and colors indicate the plot size.

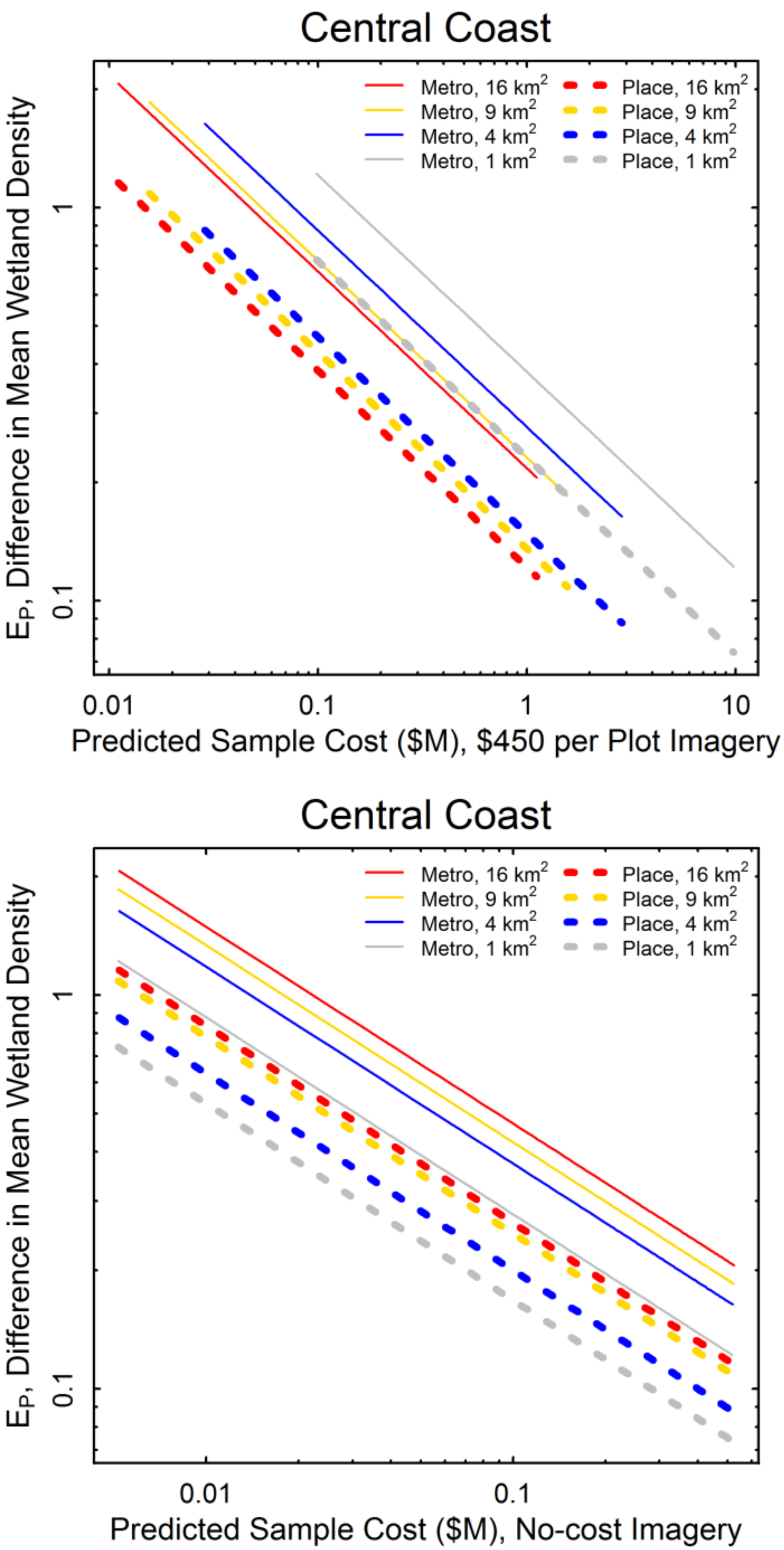


Figure 3.8. Estimated percent error, E_P, for the difference in mean wetland density as a function of predicted sample costs. *Only the Central Coast is shown. Results are shown for both metro (solid lines) and place (dotted lines) impact scenarios. Cost estimates for use of contract imagery assumed the upper limit for imagery costs ($450 per plot) and use of existing imagery assumed zero imagery costs. Lines indicate the average from all timepoints and colors indicate the plot size.*

Supplemental Tables

Supplemental Table S3.1. Point estimates by type of sample and analysis method.

Fixed and Moving Observation Locations	
Mean	$\bar{x}_t = \frac{\sum x_{t_i}}{n}$
Difference	$d_t = \bar{x}_t - \bar{x}_{t-1}$
Trend	OLS regression of mean against years since baseline
Mixture of Fixed and Moving Observation Locations Using Current and Previous Timepoint (SPR_2) (Cochran 1977; Cochran and Carroll 1953; Ware and Cunia 1962)	
Mean	$\bar{Y} = \frac{\sum \frac{\bar{Y}_j}{V(\bar{Y}_j)}}{\sum \frac{1}{V(\bar{Y}_j)}};\quad \mathrm{j} = 2,1$ $\bar{Y}_2 = \bar{Y}_{-2}$ $V(\bar{Y}_2) = \frac{s^2_{Y_{-2}}}{n_{-2}}$ $\bar{Y}_1 = \bar{Y}_{12} + \hat{\beta}_{Y_{12}X_{12}}\left(\bar{X}_{1\bullet} - \bar{X}_{12}\right)$ $V(\bar{Y}_1) = \left[\frac{\sum\left(Y_{12_i} - \bar{Y}_{12}\right)^2\left(1 - r^2_{Y_{12}X_{12}}\right)}{n_{12} - 2}\right]\left[\frac{1}{n_{12}} + \frac{\left(\bar{X}_{1\bullet} - \bar{X}_{12}\right)^2}{\sum\left(X_{12_i} - \bar{X}_{12}\right)^2}\right] + \frac{s^2_{Y_{12}} r^2_{Y_{12}X_{12}}}{n_{1\bullet}}$
Change	$\bar{C} = \frac{\sum \frac{\bar{C}_j}{V(\bar{C}_j)}}{\sum \frac{1}{V(\bar{C}_j)}};\quad \mathrm{j} = \mathrm{S, R}$ $\bar{C}_S = \bar{Y}_{12} - \bar{X}_{12}$ $V(\bar{C}_S) = \left(s^2_{Y_{12}} + s^2_{X_{12}} - 2COV(Y_{12}, X_{12})\right)/n_{12}$ $\bar{C}_R = \bar{Y}_{-2} - \bar{X}_{1-}$ $V(\bar{C}_R) = \left(s^2_{Y_{-2}} + s^2_{X_{1-}}\right)/n_{-2}$
Difference	Difference between current and previous mean
Trend	OLS regression of mean against years since baseline

Supplemental Table S3.1. *Continued.*

Mixture of Fixed and Moving Observation Locations
Using Current and Two Previous Timepoint (SPR$_3$)
(Cunia and Chevrou 1969)

Mean

$$\bar{Z} = \frac{\sum \frac{\bar{Z}_j}{V(\bar{Z}_j)}}{\sum \frac{1}{V(\bar{Z}_j)}}; \quad \mathrm{j} = 3, 2, 1$$

$$\bar{Z}_3 = \bar{Z}_{--3}$$

$$V(\bar{Z}_3) = \frac{s^2_{Z_{--3}}}{n_{--3}}$$

$$\bar{Z}_2 = \bar{Z}_{123} + \hat{\beta}_{Z_{123}Y_{123}}\left(\bar{Y}_{\bullet 2\bullet} - \bar{Y}_{123}\right)$$

$$V(\bar{Z}_2) = \left[\frac{\sum\left(Z_{123j} - \bar{Z}_{123}\right)^2\left(1 - r^2_{Z_{123}Y_{123}}\right)}{n_{123} - 2}\right]\left[\frac{1}{n_{123}} + \frac{\left(\bar{Y}_{\bullet 2\bullet} - \bar{Y}_{123}\right)^2}{\sum\left(Y_{123j} - \bar{Y}_{123}\right)^2}\right] + \frac{s^2_{Z_{123}} r^2_{Z_{123}Y_{123}}}{n_{\bullet 2\bullet}}$$

$$\bar{Z}_1 = \bar{Z}_{123} + \hat{\beta}_{Z_{123}X_{123}}\left(\bar{X}_{1\bullet\bullet} - \bar{X}_{123}\right)$$

$$V(\bar{Z}_1) = \left[\frac{\sum\left(Z_{123j} - \bar{Z}_{123}\right)^2\left(1 - r^2_{Z_{123}X_{123}}\right)}{n_{123} - 2}\right]\left[\frac{1}{n_{123}} + \frac{\left(\bar{X}_{1\bullet\bullet} - \bar{X}_{123}\right)^2}{\sum\left(X_{123j} - \bar{X}_{123}\right)^2}\right] + \frac{s^2_{Z_{123}} r^2_{Z_{123}X_{123}}}{n_{1\bullet\bullet}}$$

Change

$$\bar{C} = \frac{\sum \frac{\bar{C}_j}{V(\bar{C}_j)}}{\sum \frac{1}{V(\bar{C}_j)}}; \quad \mathrm{j} = \mathrm{S, R, U}$$

$$\bar{C}_S = \bar{Z}_{123} - \bar{Y}_{123}$$

$$V(\bar{C}_S) = \left(s^2_{Z_{123}} + s^2_{Y_{123}} - 2Cov(Z_{123}, Y_{123})\right) / n_{123}$$

$$\bar{C}_R = \bar{Z}_{--3} - \bar{Y}_{-2-}$$

$$V(\bar{C}_R) = \left(s^2_{Z_{--3}} + s^2_{Y_{-2-}}\right) / n_{--3}$$

$$\bar{C}_U = \bar{Z}_1 - \bar{Y}_1$$

$$V(\bar{C}_U) = V(\bar{Z}_1) + V(\bar{Y}_1) + 2Cov(\bar{Z}_1, \bar{Y}_1)$$

$$\bar{Z}_1 = \bar{Z}_{123} + \hat{\beta}_{Z_{123}X_{123}}\left(\bar{X}_{1\bullet\bullet} - \bar{X}_{123}\right)$$

$$V(\bar{Z}_1) = \left[\frac{\sum\left(Z_{123j} - \bar{Z}_{123}\right)^2\left(1 - r^2_{Z_{123}X_{123}}\right)}{n_{123} - 2}\right]\left[\frac{1}{n_{123}} + \frac{\left(\bar{X}_{1\bullet\bullet} - \bar{X}_{123}\right)^2}{\sum\left(X_{123j} - \bar{X}_{123}\right)^2}\right] + \frac{s^2_{Z_{123}} r^2_{Z_{123}X_{123}}}{n_{1\bullet\bullet}}$$

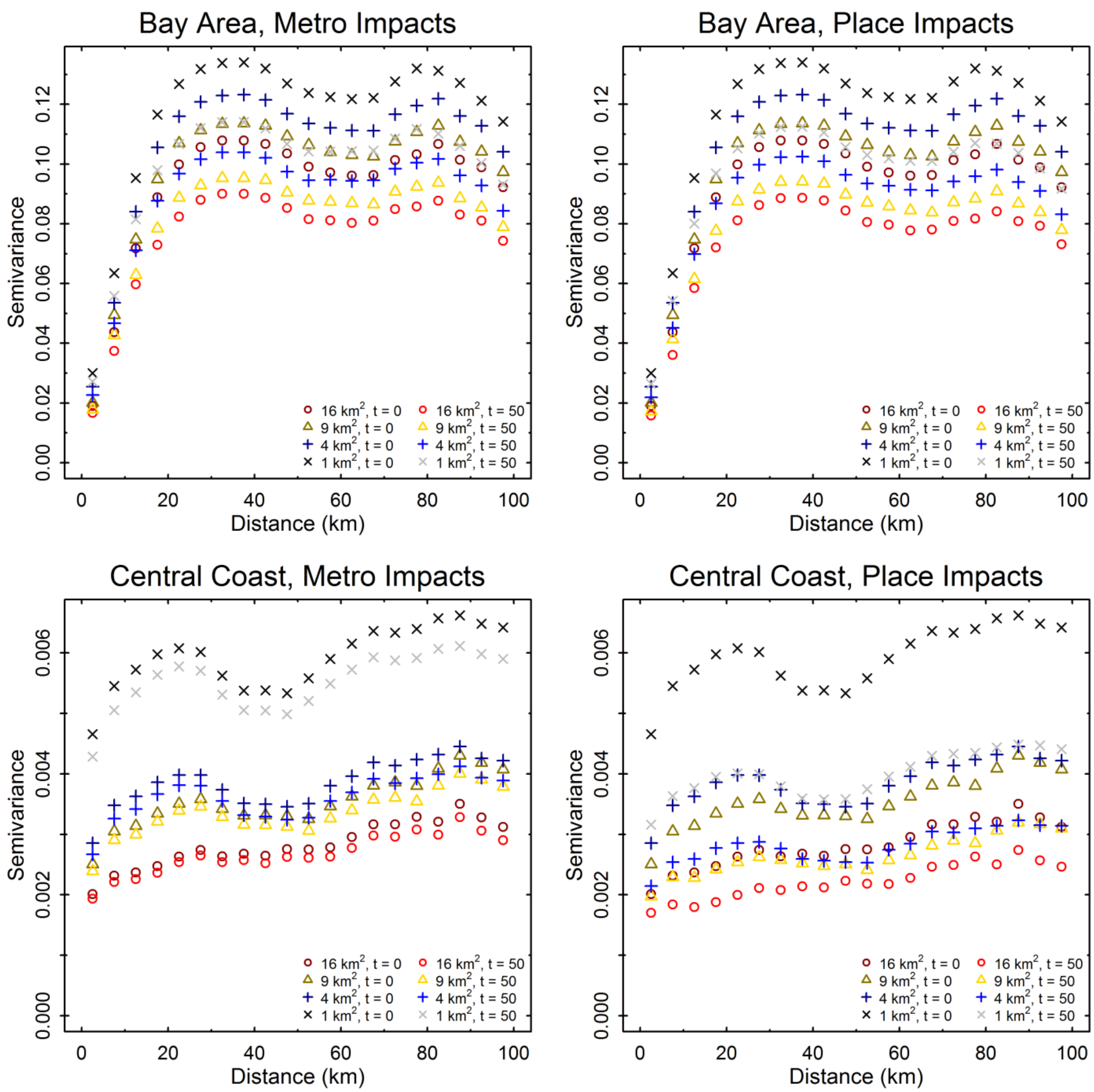
$$\bar{Y}_1 = \bar{Y}_{123} + \hat{\beta}_{Y_{123}X_{123}}\left(\bar{X}_{1\bullet\bullet} - \bar{X}_{123}\right)$$

$$V(\bar{Y}_1) = \left[\frac{\sum\left(Y_{123j} - \bar{Y}_{123}\right)^2\left(1 - r^2_{Y_{123}X_{123}}\right)}{n_{123} - 2}\right]\left[\frac{1}{n_{123}} + \frac{\left(\bar{X}_{1\bullet\bullet} - \bar{X}_{123}\right)^2}{\sum\left(X_{123j} - \bar{X}_{123}\right)^2}\right] + \frac{s^2_{Y_{123}} r^2_{Y_{123}X_{123}}}{n_{1\bullet\bullet}}$$

$$Cov(\bar{Z}_1, \bar{Y}_1) = \left(s^2_{Z_{123}Y_{123}} + s^2_{Z_{123}X_{123}} \hat{\beta}_{Y_{123}X_{123}}\right) / n_{123}$$

Difference Difference between current and previous mean
Trend OLS regression of mean against years since baseline

Supplemental Figures



Supplemental Figure S3.1. Empirical variograms for each study area and impact scenario. *Variograms are shown for each plot size at both baseline (t = 0; darker shades) and at the final timepoint (t = 50; lighter shades). Shapes indicate the plot size.*

CHAPTER FOUR: PILOT IMPLEMENTATION

Comparison of Probability-Based Sampling and Survey-Based Mapping to Estimate Wetland Extent in California

Abstract

Estimates of aquatic resource (wetland, deepwater, and stream) extent and distribution form the basis of state and federal monitoring, management and policy decisions. The default approach, comprehensive mapping, provides the most complete information on extent and distribution but is prohibitively expensive for large geographic areas. Probabilistic sampling and mapping provides a powerful alternative to comprehensive mapping by dramatically reducing time and resource costs. This study provides the first direct comparison between probabilistic and comprehensive mapping approaches for aquatic resources. We selected two regions of California that had recent, comprehensive wetland and stream maps. Thirty 4 km^2 sample plots were selected in each study area using generalized random tessellation stratified sampling and sample plot maps were produced from the same source imagery as the comprehensive maps. Sample estimates tended to underestimate mean aquatic resource density compared to comprehensive maps, in some cases by up to 60%. Differences were reduced, but not eliminated, by correcting systematic differences in mapping methodology between the new sample and the existing comprehensive maps — such as the assumed buffer width for mapped streamlines and the aquatic resource classification system. Inter-mapper variability and unidentified methodological differences may account for remaining differences between the sample and comprehensive map. We also considered the random sample draw had, by chance, selected a

non-representative sample of the study areas. Simulated sampling found that, even if systematic methodological differences are eliminated, a 30-plot sample could be expected to significantly under or over-estimate the true mean approximately 20% of the time. Increasing the sample size threefold was found to reduce this probability to 10%.

Introduction

Estimates of aquatic resource extent and distribution form the foundation of aquatic resource monitoring, policy, and management (Euliss et al. 2008). Comprehensive mapping of aquatic resources (defined here as streams, wetlands, and deepwater) has long been used as the default approach for determining regional or statewide aquatic resource extent (Rebelo et al. 2009). However, the time and money required to comprehensively map aquatic resources limits the frequency at which this can occur and/or limits this approach to small regions and subregions. In contrast, probability-based mapping can be completed quickly and cost-effectively for large geographic areas at regular intervals (Dahl 2011; Kloiber 2010; Ståhl et al. 2010).

Probabilistic mapping estimates aquatic resource extent by producing aquatic resource maps for randomly selected sample plots. Results from the sample plots can then be used to produce sample estimates for the general study area. While probabilistic mapping cannot replace comprehensive mapping, probability-based (also referred to here as design-based) methods can provide regional, statewide, and national estimates of aquatic resource extent (status) and changes in extent (trends) over time at a lower cost than comprehensive mapping (Dahl 2011; Kloiber 2010; Nusser and Goebel 1997; Ståhl et al. 2010). These estimates can also be supplemented by model-based interpolation methodologies, which can produce area-wide estimates as well as interpolated values for each location in the sampled area (Aubry and Debouzie 2001; Gregoire 1999).

Several existing state and national programs utilize probability-based sampling and mapping to estimate aquatic resource extent and distribution. Examples include the status and trends (S&T) component of the National Wetland Inventory (NWI-S&T), the Minnesota Wetland S&T program (MN-S&T), and the National Inventory of Landscapes in Sweden (NILS) (Dahl 2011; Kloiber 2010; Ståhl et al. 2010). The implicit assumption of these programs is that a design-based approach will approximate the results of comprehensive mapping within the confidence limits defined by the statistical design or model used. However, no direct comparison has yet been performed between probabilistic and comprehensive mapping to explore the accuracy of this assumption or the implications of the choice of model based estimation methods.

This study compared sample estimates of aquatic resource density with comprehensive values for two representative regions of California in order to assess the accuracy of the probabilistic approach. The study also functioned as pilot-scale testing of the proposed California S&T program design, developed in previous work (Chapter 2). We evaluated two potential drivers of differences between the sample estimates and the comprehensive values: (i) systematic differences in methodology and (ii) the likelihood of obtaining a non-representative sample.

Methods

<u>General Approach</u>

We conducted probabilistic mapping in two regions of California (Figure 4.1). Sample plots were selected using a protocol previously optimized for California aquatic resources

(Chapter 2). Three different mapping groups produced sample plot maps. Inter-group quality checks to increase consistency and accuracy. Once completed, we compared mean aquatic resource density between the sample and comprehensive maps. Next, we conducted plot-by-plot comparisons between the new sample maps and the existing comprehensive maps to examine systematic differences in mapping methodology and classification. Finally, we utilized simulated sampling to estimate the probability of selecting a non-representative sample.

Study Areas

We selected study areas in the central and south coast of California (Figure 4.1) based on availability of comprehensive, high-quality aquatic resource maps. Comprehensive maps were produced from recent (2005), freely available imagery. Therefore, we could perform a direct comparison between probabilistic and comprehensive approaches by using the same source imagery to produce new aquatic resource maps for the sampled plots. In addition, the two study areas were ecologically and geographically distinct and had different types and densities of anthropogenic land use. This allowed us to evaluate the probabilistic approach in two distinct settings. Comprehensive central coast maps were downloaded from the National Wetland Inventory (NWI). We obtained south coast maps directly from the producer, but these maps will be made publicly available through the NWI at a future date.

It should be noted that the producer of the comprehensive map for the south coast was also involved in production of sample plot maps in this study — producing 25 of the 60 sample plot maps (Group 2 in Figure 4.1). This mapping group was also involved in defining the methodological approach used for the current study. This overlap doubtless reduced methodological differences between the sample and comprehensive maps in the south coast. In turn, this methodological overlap allowed us to contrast the likely methodological consistency in the south coast with the potential methodological differences in the central coast.

Sample Draw

We selected sample plots using Generalized Random Tessellation Stratified (GRTS) sampling (Stevens and Olsen 2004). First, a 4 km square (16 km^2) grid was produced in ArcGIS for each study area (ESRI 2010). Next, grid cells were converted to points, exported as a shapefile, and the GRTS sample draw performed in R using the *spsurvey* package (Kincaid and Olsen 2011; R Development Core Team 2011). We selected thirty plots, without stratification, in each region. Selected plots were divided between the three mapping groups (Figure 4.1; mapping group 3 did not produce maps for the south coast).

When we divided sample plots between map production groups, we took care to maintain the integrity of the GRTS sample. GRTS sample selection produces an ordered list of locations. Importantly, the sample will maintain spatial balance as long as sample locations are used in order from the list (Larsen et al. 2008; Stevens and Olsen 2004). Spatial balance in the final sample will theoretically reduce spatial autocorrelation (i.e., natural correlation between values at neighboring locations) and increase the diversity of locations observed. Therefore, we assigned sample plots to each map production group based on the order of the GRTS sample list, and not by geographic subdivision of each study area. In the central coast, three mapping groups produced sample plot maps. Therefore, we assigned locations 1-10 of the sample list to group one, locations 11-20 to group two, and locations 21-30 to group three. In the south coast, two mapping groups produced sample plot maps. Therefore, we assigned locations 1-15 to group one and locations 16-30 to group two.

Sample Map Production

Mapping groups produced maps consistent with the draft aquatic resource mapping standards for the State of California. All mapping groups produced maps directly in ArcGIS. Base imagery for all maps was 2005 National Agriculture Imagery Program (NAIP) imagery, available as natural-color with 1 m pixel resolution, and provided at no cost from the US Department of Agriculture. Secondary imagery included the 2009 NAIP imagery, available as color-IR with a 1 m resolution. Auxiliary data included 1:24,000 digital raster graphic (DRG) from the US Geological Survey; 10 m digital elevation model (DEM) from the National Elevation Dataset; the National Hydrography Dataset (NHD); and soil maps from the Natural Resources Conservation Service.

The first step in aquatic resource mapping was generation of a linear stream network from the 10 m DEM. Next, the mapper edited the network using base imagery and auxiliary data. The mapper used auxiliary data sources, particularly the DRG and NHD, when the exact streamline position was not apparent in the imagery — such as when obscured by vegetation. The final stream network was then converted to polygon features and aquatic resources associated with or adjacent to the network were delineated. Finally, the mapper delineated isolated aquatic resources. The result was a map of aquatic resources composed of polygons for streams, wetlands, and deepwater.

The final step in map production was 100% quality assurance and control (QA/QC). Fully delineated and classified maps were transferred to a different mapping professional who then completely reviewed the maps for delineation and classification. The reviewing mapper performed edits directly within a copy of the draft map geodatabase, resulting in a final, QA/QC'ed geodatabase for the study plot. Because group 2 was so familiar with the south coast area, they did not perform QA/QC on any south coast plots.

<u>Sample Mean Estimates and Comprehensive Mean Values</u>

We produced sample estimates of aquatic resource density by three methods: the simple mean and variance estimators, the GRTS estimator of mean and local variance, and block kriging. The GRTS estimator of the mean utilizes the Horvitz-Thompson estimator, making it equivalent to the simple mean (Diaz-Ramos et al. 1996; Kincaid and Olsen 2011). The GRTS local variance estimator uses a moving window of points around each sample point and is statistically more precise than the traditional sample variance estimator (Stevens and Olsen 2003). Kriging is a form of model-based estimation and spatial interpolation that uses the mean and variance of the sampled locations to estimate values for a prediction grid. We used ordinary kriging for interpolation and block kriging to compute an area-wide mean and variance. We assumed a spherical semivariogram model based on visual examination of empirical semivariograms.

We produced comprehensive mean density values by first intersecting the comprehensive aquatic resource map for each region with the 16 km^2 grid used for the sample draw. Then, we calculated an aquatic resource density for each grid cell by dividing the area of aquatic resources in the cell by the area of the cell. This produced a population of density values that could be used to easily calculate a mean aquatic resource density value for the comprehensive map.

Systematic Differences between Sample and Comprehensive Maps

We identified two major systematic differences between the sample and comprehensive maps: (i) the assumed buffer width for streams and (ii) the classification system. These differences are not and should not be considered exhaustive. However, these differences were easily evaluated in a GIS-based analysis and could be identified without a detailed examination of standard operating procedures. Additional possible differences could be related to internal standards for mapping streams and seasonal or ephemeral wetlands. Stream mapping can differ between map producers due to, among other things, assumptions about the minimum drainage area required to generate a stream headwater and whether to map a stream that is, for example, predicted by the DEM but not visible in the source imagery. Seasonal or ephemeral resources present a particular issue because they may not contain visible or surface water when the source imagery is obtained. When this is the case, map producers rely on secondary imagery to determine the existence and boundaries of the seasonal or ephemeral resource. Therefore, which secondary imagery sources to use and how much to rely on them compared to the primary imagery source may have a significant impact.

Buffer Widths for Stream Networks

Converting a linear stream network, representing the centerline of each stream segment, into a two-dimensional polygon, representing the bank-to-bank extent of each segment, requires an initial assumption about the typical bank-to-bank width for the network. The assumption is used to buffer the linear network and produce initial polygons. Map producers then adjust the initial polygons based on the bank locations visible in the aerial imagery. However, the assumed bank-to-bank width remains a critical component of the mapping methodology for two reasons. First, the remote imagery used for aquatic resource mapping typically has a pixel size of approximately 1 m. At this resolution, low-order and/or ephemeral streams are often too narrow for the imagery alone to clearly support a bank-to-bank width. Second, if vegetation obscures stream banks, delineation from the imagery alone is impossible. In both of these cases, the mapper relies solely on the assumed buffer. However, while the assumed buffer represents the best professional judgment of the mapper about the typical, two-dimensional extent of stream resources, the buffer is still a simplifying assumption.

This study considered two different approaches to stream network buffering: (i) the best-professional judgment of the map production groups; and (ii), the standard buffer width used for aquatic resource maps in the NWI — the comprehensive map source used for comparison. We chose to consider both buffer widths for three reasons. One, the map production groups involved in this study considered a differential buffer (defined below) to be the best representation of how streamlines are distributed in the landscape. All three mapping groups have substantial experience producing aquatic resource maps in California. We considered it unwarranted to abandon this expertise based on the assumptions of the NWI (a national program). Two, the assumed buffer has a significant impact on mapped aquatic resource area because is so often used without modification. As a result, the assumption can significantly affect the calculated area for an individual stream reach. Therefore, the NWI standard buffer width was required to provide a meaningful comparison between the sample and comprehensive maps. Three, maintenance of both buffer widths enabled us to estimate the magnitude of the effect this simple assumption has on mapped aquatic resource area.

The mapping professionals in this study utilized differential buffer widths based on stream order. First order streams, defined by Strahler stream order, were buffered to 0.5 m (1 m bank-to-bank width); second order streams were buffered to 1 m (2 m bank-to-bank width); and third order or higher streams were buffered to 2 m (4 m bank-to-bank width). Mappers then expanded assumed widths when supported by the imagery but in the majority of cases — particularly for lower order streams — left banks as buffered. The resulting polygons represented the best professional judgment of the mapping groups used to produce the sample plot maps.

After mapping and QA/QC was completed, we created a second layer for the stream network using the standard NWI buffer width of 2.5 m. Then, we overlaid the newly buffered stream network on the sample aquatic resource maps and retained the portion of the new buffer area that fell outside of the originally mapped area.

Classification

Map producers classified aquatic resource polygons using the draft California aquatic resource classification system (CARCS). CARCS is a functional classification derived from the hydrogeomorphology (HGM) classification system used by the US Army Corps of Engineers (Brinson 1993) but modified for California aquatic resources. The CARCS system combines a hierarchical classification, based on hydrogeomorphology and landscape connection, with optional modifiers for vegetation, anthropogenic influence, flow regime, and substrate. We provide the draft version of CARCS used in this study as Supplemental File S4.1.

Polygons in the newly produced sample maps were delineated and classified based on the hierarchical, hydrogeomorphic component of the CARCS classification. However, the pre-existing comprehensive maps were classified according to the Cowardin classification used by the NWI (Cowardin et al. 1979). Therefore, we developed a preliminary crosswalk (Table 4.1) to facilitate comparisons between sample and comprehensive maps. In the South Coast, polygons were attributed using the HGM classification, including a fluvial designation, in addition to the NWI classification. We used the fluvial designation to enhance the crosswalk for palustrine and riverine resources. Some palustrine resource can be defined as functionally riverine and would be classified as riverine under the CARCS classification. For example, a vegetated island, sandbar, or floodplain area would be classified as palustrine under NWI but is functionally riverine under CARCS. Another example would be a vegetated streambed that contains substantially different vegetation from the surrounding upland. The streambed is functionally riverine but would be considered palustrine under NWI. Therefore, the fluvial designation allowed us to identify the functionally riverine palustrine resources and to compare them to riverine resources in the sample maps.

Likelihood of a Non-Representative Sample

We considered whether a larger sample size than the one we used might improve the accuracy of sample estimates; i.e., we asked whether the 95% confidence interval of the sample mean actually contains the comprehensive mean value 95% or more of the time. We also asked whether there was a relationship between the sample size and the likelihood that the 95% interval contains the true value. To answer these two questions, we conducted simulated sampling using the comprehensive maps for each study area. Simulated sampling is a process whereby a pre-existing "population" of data values is repeatedly and artificially sampled to simulate the result of repeating the sampling procedure.

In this study, the “population” was the sample grid used for the original sample draw. Each repetition, a sample of cells was selected and the 95% confidence interval for the sample mean was calculated. We then determined whether the interval contained the comprehensive mean value defined previously. This process was repeated 5,000 times using 10 different sample sizes. We selected a repetition count of 5,000 based on prior experience with simulated sampling (Miller and Ambrose 2000). The 10 sample sizes ranged from 10 to 100 sample plots. Because new maps were not produced for the entire study area, we could only use the comprehensive map for the simulated sampling. Therefore, the simulation only determines the accuracy of the sampling procedure as a result of sample size, and controls for methodological differences.

Results

The results section contains three primary sections. First, we compared sample mean estimates to comprehensive mean values. Sample mean estimates were produced by using the individual sample plot maps to estimate the mean aquatic resource density for the study area. Second, we evaluated systematic differences between the sample and comprehensive maps. This evaluation included (i) sample estimates under different methodological assumptions and (ii) plot-by-plot comparisons between the new sample maps and the portion of the comprehensive map within each sample plot. The plot-by-plot comparisons allowed us to control for a non-representative sample. Third, we determined the likelihood of drawing a non-representative sample using the simulated sampling approach. Because the simulations considered only the pre-existing, comprehensive maps, these results controlled for systematic methodological differences.

Probabilistic Estimates vs. Comprehensive Values

Sample estimates of mean aquatic resource density were 39% lower than comprehensive mean values in the central coast and 60% lower in the south coast (Figure 4.2 and Supplemental Figure S4.1). The estimation method used did not influence accuracy but did affect estimated variance. The GRTS estimator produced a 12-15% narrower 95% confidence interval than the traditional estimator did while the block kriging interval was 67-77% narrower than the GRTS and traditional intervals.

Possible causes of the difference between the sample estimates and comprehensive values include inter-mapper variability, systematic methodological differences, and the possibility of a non-representative sample. As no two mapping groups mapped the same sample plot, this study could not address inter-mapper variability. Known methodological differences (the assumed stream width and the classification system) were explored through plot-by-plot comparisons, as discussed below. Finally, the possibility of a non-representative sample was evaluated via simulated sampling and results are provided below.

Systematic Methodological Differences

Assumed Stream Width

The assumed stream buffer width had a significant impact on sample estimates of mean aquatic resource density (Figure 4.3a). Use of a 2.5 m buffer increased the estimated mean density by 58% in the central coast, and 43% in the south coast, relative to sample mean density based on a differential buffer. In the central coast, this increase meant that the sample mean density was statistically equivalent to the comprehensive mean value. In contrast, in the south coast, the sample estimate was still 43% lower than the comprehensive map mean value.

Plot-by-plot comparisons between the sample and comprehensive maps also showed that the assumed buffer width had a significant systematic effect on the sample and comprehensive maps. In the south coast, the sample maps were expected to be more methodologically consistent with the comprehensive maps due to the overlap in map producers. Considering just the maps for the 30 plots, use of the 2.5 m buffer reduced the average plot-by-plot differences to just 5% from 37% when a differential buffer was used (Figure 4.3c). However, in the central coast, plot-by-plot differences changed from an average of 14% lower to 71% higher (Figure 4.3b). This suggests that additional methodological differences may exist between the sample and comprehensive maps in the central coast.

Aquatic Resource Classification

In the central coast, the sample estimate of the mean density of depressions and slopes was 88% lower than the comprehensive mean value for palustrine density (Figure 4.4a). The average plot-by-plot difference, calculated by considering just the maps within the plot boundaries, was 78% in the central coast (Figure 4.4b). However, the palustrine classification includes some resources that would be classified as riverine under the CARCS classification. Therefore, we cannot know if the plot-by-plot differences in the south coast reflect additional systematic methodological differences, as suggested by the previous section, or a failure of the classification crosswalk.

In contrast, a fluvial designator was available for the comprehensive south coast map. In this study area, the sample estimate for mean depression and slope density was 83% lower than the comprehensive mean for all palustrine resources and 53% lower than the comprehensive mean for non-fluvial palustrine resources (Figure 4.4a). However, plot-by-plot differences decreased from an average of 62% to 0.6% (Figures 4.4c, and 4.4d). This result suggests very low systematic differences between the south coast sample and comprehensive maps for the mapping of depressions, slopes, and non-fluvial palustrine resources. However, this methodological consistency is also expected in the south coast because of the overlap in map producers between the comprehensive and sample maps.

Moving on to riverine resources in the central coast, the sample estimate of the mean density of riverine resources was only 2% higher than the comprehensive mean density value for riverine resources (Figure 4.4a). However, this similarity may be spurious. First, this sample estimate of riverine resource density is based on a differential buffer, instead of a 2.5 m buffer. Second, the riverine classification does not include all functionally riverine resources as illustrated previously. As expected, the estimated mean density of riverine resources, based on a 2.5 m buffer, was 114% higher than the comprehensive mean value for riverine (Figure 4.4a). In addition, the average plot-by-plot differences increased from 120% higher to 270% higher (Figure 4.4b). These results suggest potential methodological differences in the central coast that resulted in more mapped streamlines for the sample plot maps compared to the comprehensive maps. However, we cannot eliminate the possibility that an enhanced classification crosswalk, through use of a fluvial designator for palustrine resources, could reduce the differences between the sample and comprehensive maps.

In contrast, in the south coast, the most logically consistent comparison available, between riverine based on a 2.5 m buffer and the sum of riverine and fluvial palustrine, was in the best agreement. The sample mean estimate under this comparison was only 22% lower than the comprehensive map mean density and plot-by-plot differences averaged only 1% (Figure 4.4a and 4.4d). Use of the narrower, differential buffer option decreased sample plot density for riverine resources and reduced the agreement between the sample and comprehensive maps (Figures 4.4c and 4.4d). Agreement was also reduced for comparisons to riverine resources alone in the comprehensive map reduced agreement (Figure 4.4c). However, as indicated previously, methodological consistency was also expected in the south coast.

The low number of sample plots containing estuarine, lacustrine, and marine resources prevented evaluation of differences for these resource types.

Likelihood of a Non-representative Sample

In addition to methodological differences between the comprehensive and sample maps, there was a possibility that the randomly selected sample, by chance, did not accurately represent the full range of aquatic resource density present in the comprehensive map. To explore this possibility, we first compared the distribution of comprehensive map density values against the actually sampled plots (Figure 4.5a and 4.5b). This figure contains only the densities from the comprehensive map and therefore ignores the methodological differences in mapped aquatic resource density identified in the previous sections. However, our intent in this exercise was to evaluate the possibility that, by chance, the randomly sampled locations did not represent the comprehensive map. Use of densities from the newly produced maps would confound this evaluation.

The comparisons in Figure 4.5a and 4.5b suggest that the randomly selected locations tended to represent the more common aquatic resource densities in the comprehensive map. However, because of the right-tailed distribution of the comprehensive map values, this random sample sometimes resulted in a statistically significant underestimation of the true mean for the comprehensive map. For example, in the south coast, the mean of comprehensive maps for the sampled locations had a 95% confidence interval from 0.028 to 0.048, while the area-wide comprehensive mean was 0.055 (Figure 4.5b).

Figure 4.5c provides the results of the simulated sampling. Considering just a sample size of 30, the sample 95% confidence interval contained the true value 78-80% of the time. For the largest simulated sample size (100), the interval contained the true value 89% of the time. This may explain the observation that, even though the south coast sample maps appear to have only systematic differences compared to the comprehensive maps, the south coast sample estimates remain significantly lower than comprehensive map mean values.

Discussion

Design-based estimates of mean aquatic resource density were lower than values from existing comprehensive maps. This appeared driven primarily by two factors. First, systematic methodological differences, including the assumed streamline buffer width, accounted for a significant portion of the paired differences between sample and comprehensive maps in the south coast. Additional systematic differences remained in the central coast; however, we were not able to identify or quantify these errors and plot-by-plot comparisons were handicapped by the absence of an unambiguous crosswalk between the classification systems used for the sample and comprehensive maps. In addition, methodological consistency in the south coast may have been driven by the overlap in map producers between the comprehensive and sample maps. Second, the relatively small sample sizes used in this study may have under-sampled the highest density areas by chance — introducing a random bias into our sample estimates of mean aquatic resource density. Unfortunately, this study was not able to quantify inter-mapper variability between the map producers because sample plots were never mapped by more than one producer. The California S&T program will need to address these three factors by adopting a standardized internal methodology to ensure internal consistency; developing rigorous classification and methodological crosswalks before attempting to compare S&T results with an outside data sources; and quantifying and monitoring inter-mapper variability under the standardized methodology.

Assumed Buffer Width and Other Systematic Differences

This study considered the effect of a systematic difference in assumed buffer width between aquatic resource maps. Assuming a consistent buffer width largely erased systematic differences the south coast, where we expected greater methodological consistency due to an overlap in map producers between the comprehensive and sample maps. However, in the central coast, adopting a consistent buffer width actually increased average differences between the sample and comprehensive maps. These results suggest additional systematic differences between mapping methodology used. Possible differences could include mapping rules for producing the stream network and for use and reliance on auxiliary data sources when mapping seasonal or ephemeral wetlands (Jana et al. 2007; Schmid et al. 2005; Smith et al. 1998). It is important to note that these conclusions regarding systematic differences are based on average plot-by-plot differences and improvements in methodological agreement did not necessarily increase agreement between sample estimates of mean density and comprehensive values.

While standardizing the assumed streamline buffer significantly reduced differences between sample estimates and comprehensive values in the south coast, we do not recommend that a 2.5 m buffer be the default assumption for stream and wetland mapping as part of an S&T program. In an implemented S&T program, the primary focus should be on accurately mapping wetland and stream extent and ensuring internal consistency between sample maps and across time. Therefore, mapping procedures and assumptions should be defined based on the best professional judgment and experience of map producers, preferably validated and supplemented by field mapping and groundtruthing. Standardization of mapping procedures and rigorous training and quality control measures can then be incorporated into the S&T program to ensure internal consistency. Comparisons between outside data sources and S&T maps and sample estimates should then only be conducted after reviewing the methodology used to create the outside data source.

Aquatic Resource Classification

Analysis of type differences in this study was confounded by the ambiguity in the classification cross-walk. Addition of a fluvial/non-fluvial descriptor in the south coast corrected a deficiency in the crosswalk for riverine and palustrine NWI subtypes. Under the CARCS classification system, all functionally riverine resources are classified as such (Supplemental File S4.1). However, riverine resources are only classified as riverine under the NWI classification system if they are (i) scoured or unvegetated or (ii) intermittent streams with no vegetation differences between the streambed and surrounding upland (Cowardin et al. 1979). Other vegetated, functionally riverine resources are classified as palustrine under Cowardin et al. The fluvial designation in the south coast allowed us to identify functionally riverine resources that had been classified as palustrine under the NWI classification. This allowed us to compare riverine resources in the sample maps to all functionally riverine resource in the comprehensive maps.

This result reflects the differences between functional and biologically based aquatic resource classification systems (Brinson 1993; Cowardin et al. 1979). The difficulty of cross-walking between a functional system, such as HGM, and the biologically based NWI has been previously identified by several individuals (Abdullah and Nakagoshi 2007). For example, Brooks et al. estimated reclassification accuracy between NWI and HGM at approximately 60% (Brooks et al. 2011). As with potential methodological issues mentioned above, internal consistency and reliability should be the primary motivation for an S&T program. Therefore, an adequate crosswalk between CARCS and Cowardin is not strictly necessary.

However, while the California S&T program will primarily report results for the CARCS classification, accurate crosswalking of results to the Cowardin system will ensure that California results can be compared to the national NWI-S&T program estimates. In addition, while CARCS is the preferred system for California aquatic resources, a number of different aquatic resource mapping efforts use the Cowardin classification system. Therefore, the ability to crosswalk the California S&T program to Cowardin will increase the potential applications of the maps and the S&T results. Based on this, the most obvious solution is to include delineation and classification of aquatic resource polygons by both the CARCS and the Cowardin system to ensure compatibility of type-specific estimates with the NWI-S&T and other aquatic resource mapping efforts. Further work for the California S&T program will be to determine the most efficient approach for mapping and classifying according to both CARCS and Cowardin.

Sample Size, Accuracy, and Precision

The relatively small sample size used may have under-sampled the highest density areas by chance, suggesting that a larger sample size should be used to increase the probability that the full range of population values is sampled (Banik and Kibria 2010). We based sample sizes for this study on an expected error of plus or minus 25%, which we achieved if a 2.5 m buffer was assumed. However, the results from the simulation study suggest that the sample 95% confidence interval of the mean is more appropriately described as (for the sample size used in this study) an empirical 80% confidence interval (i.e., 80% of the time, the interval contains the true population mean). Tripling the sample size improved this result to an empirical 90% confidence interval. In addition, these simulation results considered only the confidence interval based on the GRTS sample variance estimator and assume that systematic differences between the sample and the population are nonexistent. Due to these factors, we suggest either an increase in the project sample size or recognition that the sample variance estimator may be "over-optimistic" and underestimate the true variability in the sample mean. Options for addressing this issue include non-parametric approaches to estimating sample variability, such as bootstrapping and permutation (Bonate 1993; Henderson and Lewis 2008).

Inter-Mapper Variability

Finally, while the current study could not directly address inter-mapper variability, previous results in the literature have identified a wide range of accuracy rates for aquatic resource mapping, depending both on the vegetation types and resources being mapped (Corcoran et al. 2011; Hirano et al. 2003). Results from our study for plot-by-plot differences in the south coast suggest that inter-mapper variability map be as low as 5%. However, exact estimates are so far unavailable. Inter-mapper variability could potentially be managed, and possibly reduced, by standardizing mapping and classification procedures for sample map production. Use of intercalibration plots and quality control measure can also be used to reduce differences between individual mappers and to provide more consistent maps of aquatic resource extent.

Tables

Table 4.1. Crosswalk between CARCS and the Cowardin/NWI wetland classification system for the two study areas.

Sample Maps	Comprehensive Maps	
CARCS *South & Central Coast*	**NWI** *South & Central Coast*	**NWI + HGM** *South Coast*
Depressions + Slopes	Palustrine	Non-fluvial Palustrine
Riverine	Riverine	Riverine + Fluvial Palustrine
Estuarine	Estuarine	Estuarine
Lacustrine	Lacustrine	Lacustrine
Marine	Marine	Marine

Figures

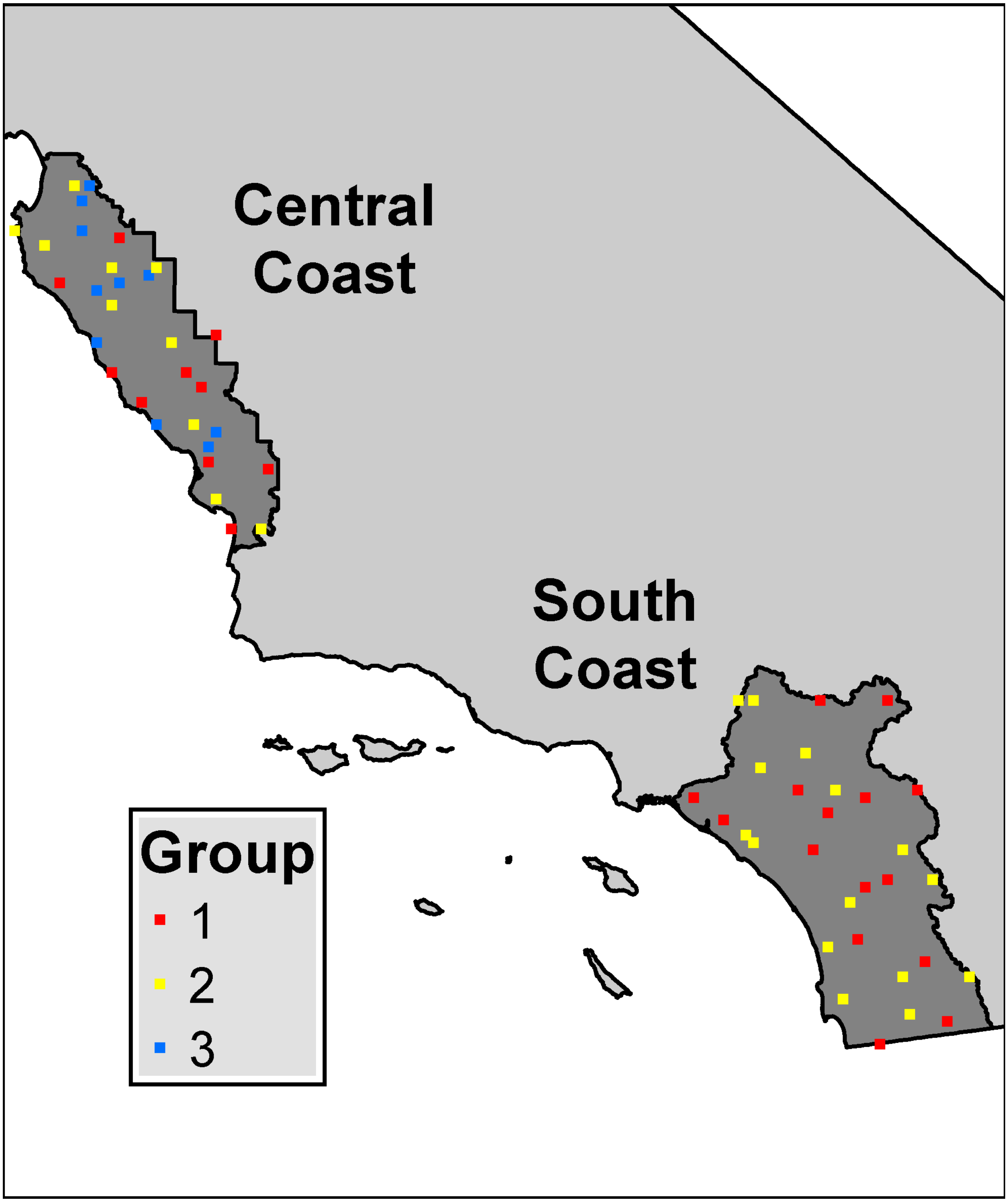


Figure 4.1. Study areas and sample plot locations by mapping group.
The third group did not produce sample plot maps in the south coast region.

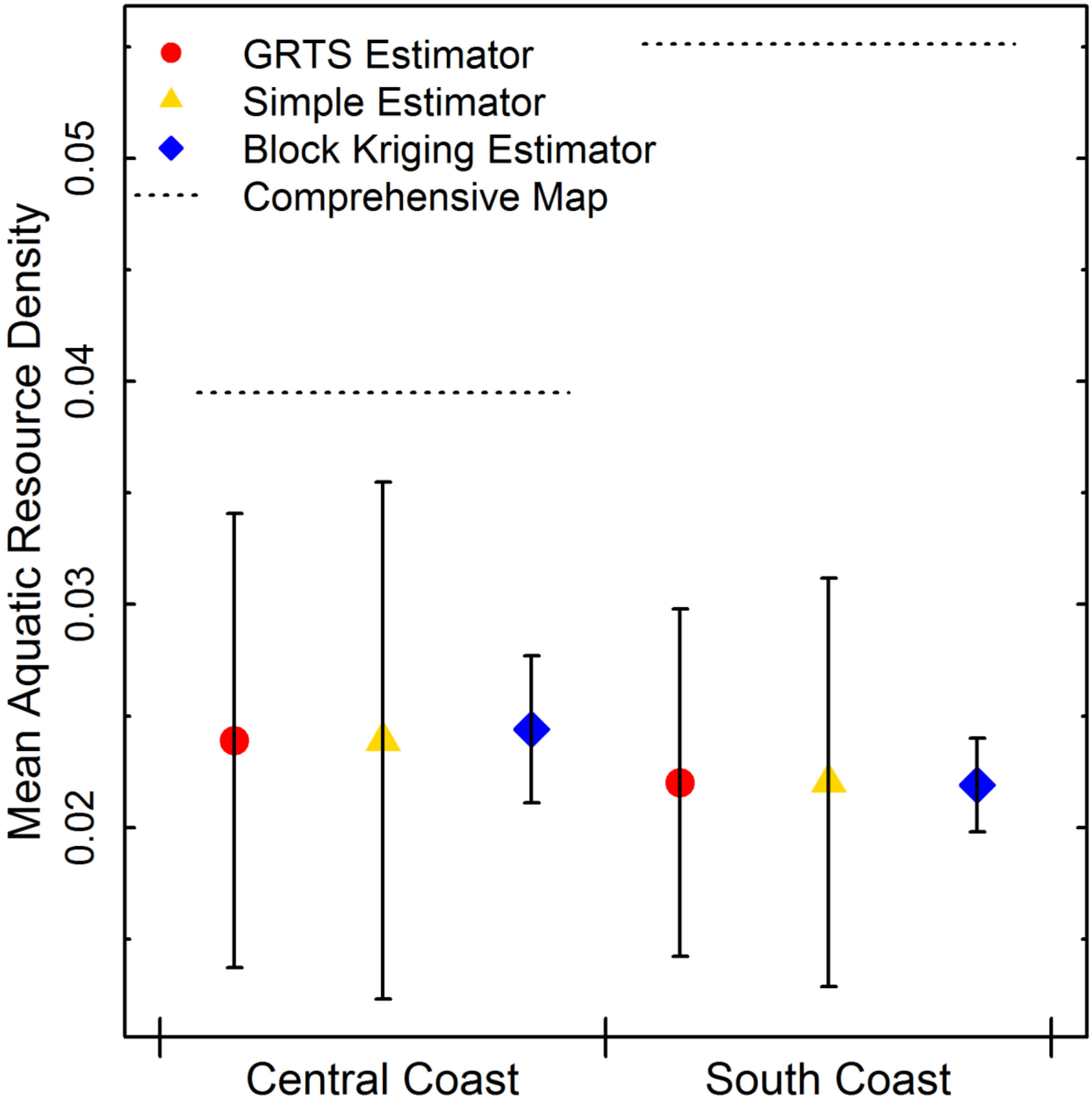


Figure 4.2. Mean aquatic resource density by study area.
Red circles indicate the estimated mean of sample maps, plus or minus the 95% confidence interval, based on the GRTS variance estimator. Yellow triangles are based on the simple mean and variance estimator and blue diamonds are based on block kriging. Dotted lines are the mean density from the comprehensive map.

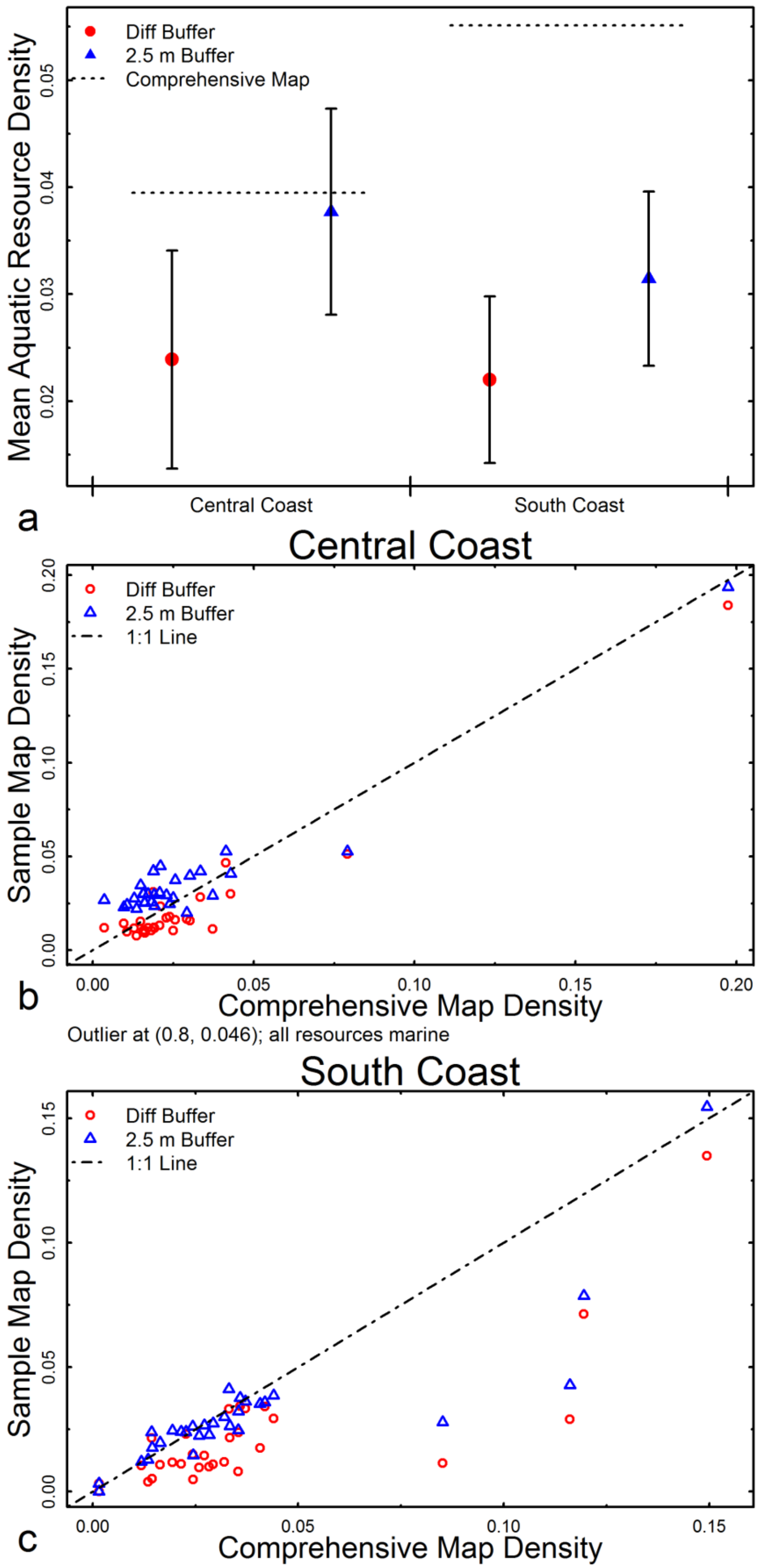


Figure 4.3. Comprehensive versus sample map density by study area.
(a) Red circles indicate the estimated mean based on a differential streamline buffer width, plus or minus the 95% confidence interval from the GRTS variance estimator. Blue triangles are based on a 2.5 m buffer. Dotted lines are the mean density from the comprehensive map. (b-c) Points compare aquatic resource density for the comprehensive map (x-axis) against density for the sample map (y-axis) for individual plots in the central coast (b) and south coast (c) based on a differential buffer (red circles) and a 2.5 m buffer (blue triangles) for the sample maps. The dashed-dotted black line provides a 1:1 reference.

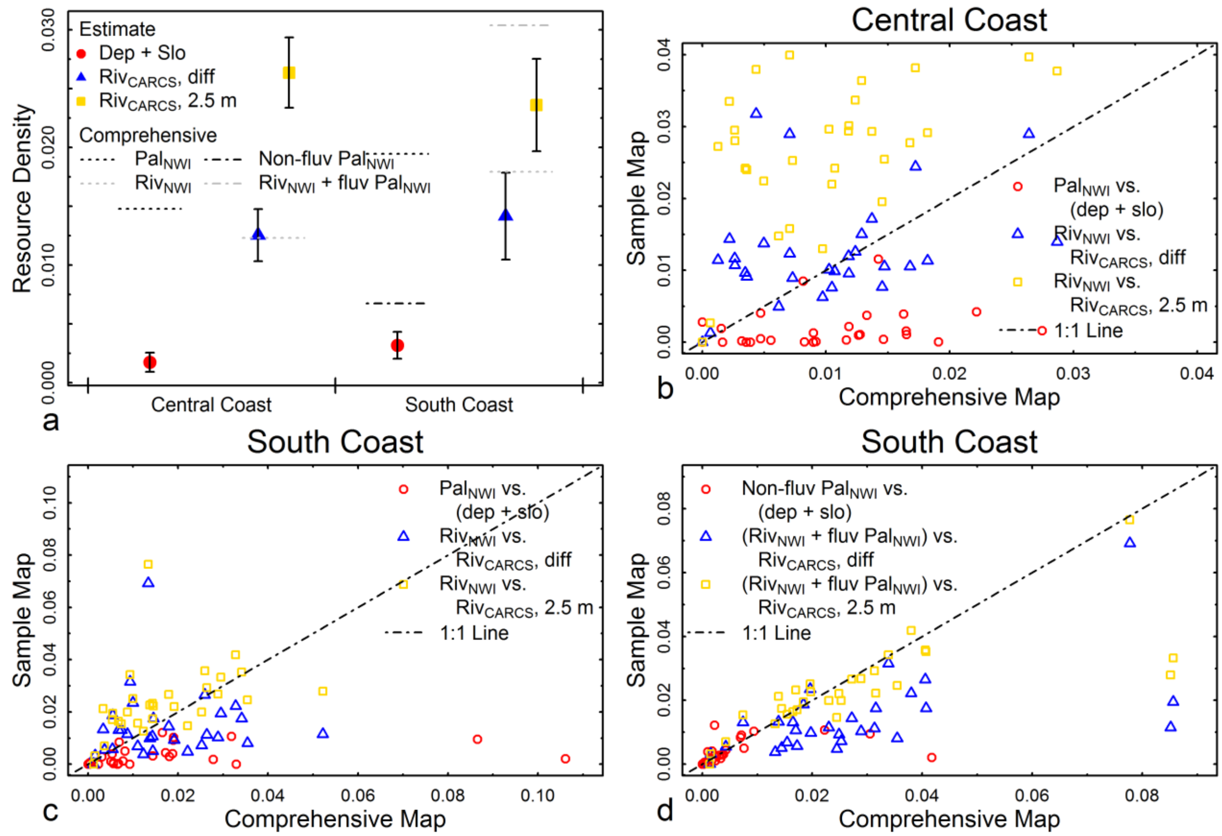


Figure 4.4. Comprehensive versus sample map density by study area and resource type.

(a) Red circles indicate the estimated mean density of depressions and slopes. Blue triangles indicate mean riverine density based on a differential streamline buffer width and yellow squares indicate a 2.5 m buffer. Intervals represent the 95% confidence interval from the GRTS variance estimator. The black dotted line provides the mean palustrine density from the comprehensive map while the black dashed-dotted is the mean non-fluvial palustrine density. The grey dotted line is the mean riverine density from the comprehensive map and the grey dashed-dotted is the mean riverine plus fluvial palustrine density. (b-d) Points compare aquatic resource density for the comprehensive map (x-axis) against density for the sample map (y-axis) for individual plots in the central coast (b) and south coast (c-d). The dashed-dotted black line provides a 1:1 reference. (b-c) Red circles compare palustrine against the sum of depressions and slopes; blue triangles compare riverine against riverine based on a differential buffer; and yellow squares compare riverine against riverine based on a 2.5 m buffer. (d) Red circles compare non-fluvial palustrine against the sum of depressions and slopes; blue triangles compare riverine and fluvial palustrine against riverine based on a differential buffer; and yellow squares compare riverine and fluvial palustrine against riverine based on a 2.5 m buffer.

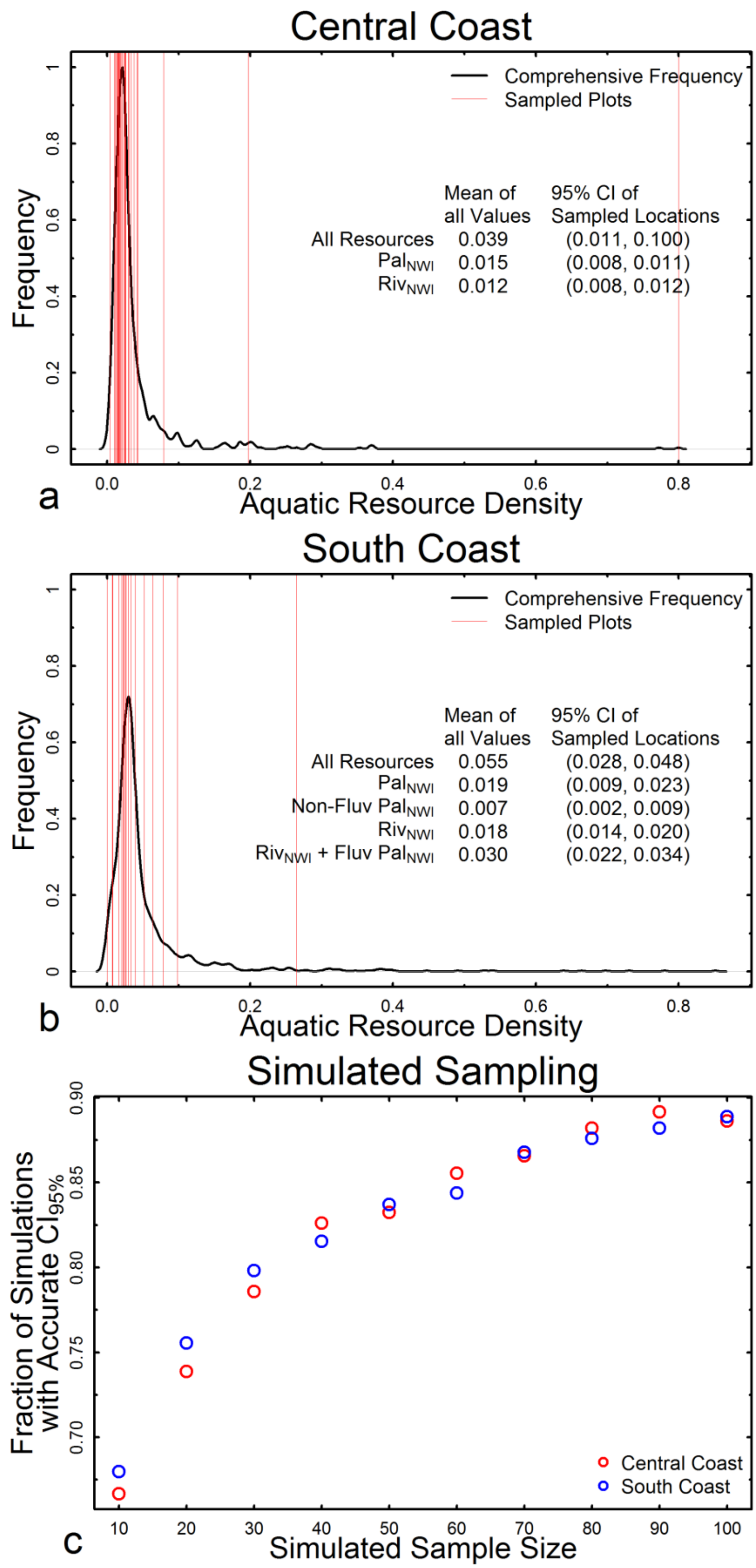
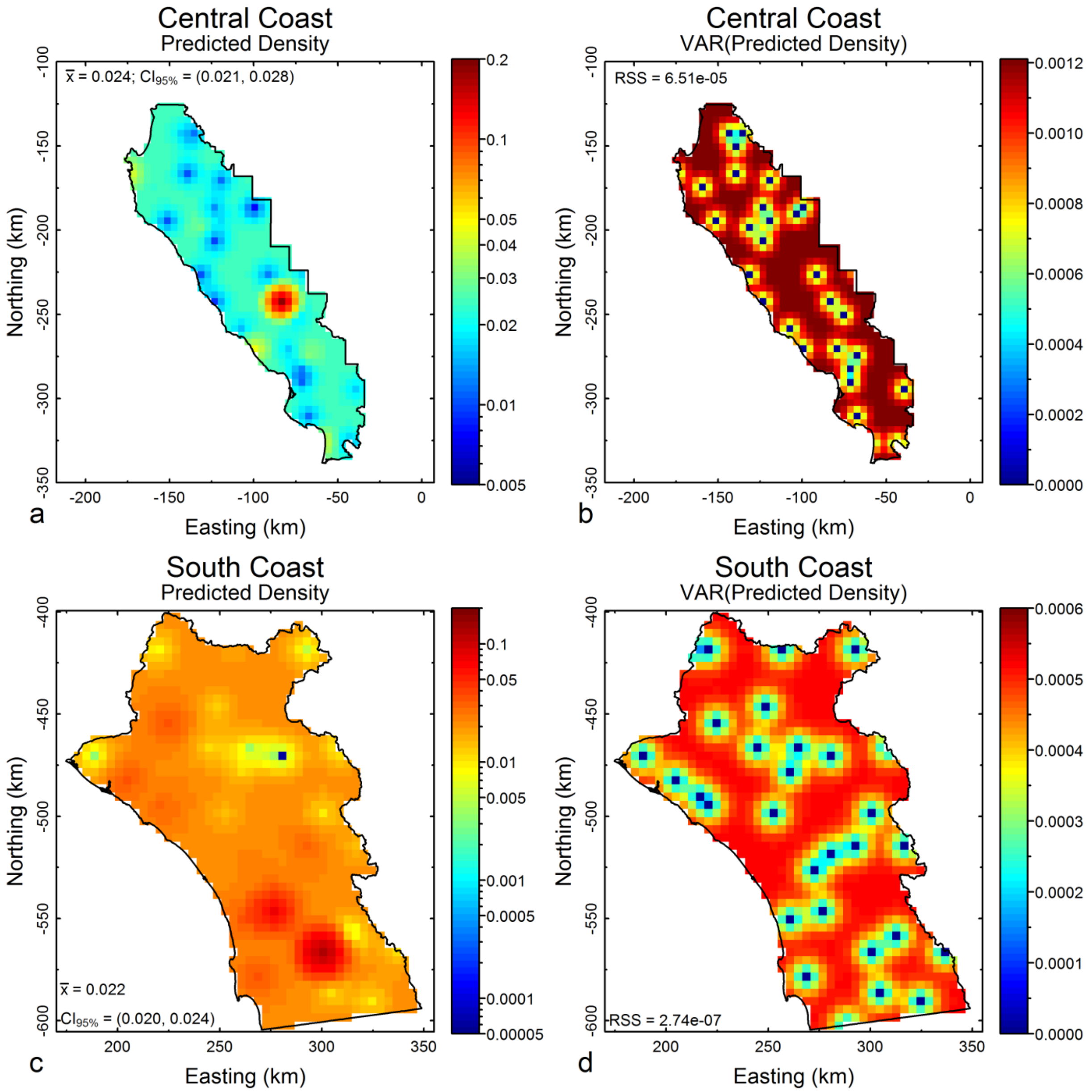


Figure 4.5. Distribution of comprehensive aquatic resource density values and accuracy.
(a-b) Solid black line indicates the relative frequency of aquatic resource densities in the comprehensive map for the central coast (a) and the south coast (b). Red lines indicate the plots included in the random samples in Figure 1. The table provides the mean of all values for selected resource types as well as the 95% confidence interval for the sampled locations, indicated by red lines. The interval is based on the GRTS variance estimator and uses the aquatic resource densities of the comprehensive map, not the newly produced sample maps. (c) Points indicate the results of 5,000 simulations of each sample size (x-axis) for the central coast (red) and the south coast (blue). The y-axis is the fraction of the simulations with a 95% confidence interval that contained the comprehensive mean.

Supplemental Figures



Supplemental Figure S4.1. Results from ordinary kriging.

Interpolated density (a&c) and variance (b&d) for the central coast (a-b) and south coast (c-d). Ordinary kriging was used with a spherical semivariogram model. Inset means and 95% confidence intervals (a&c) are the block kriging results for each area. Inset residual sums of squares (b&d) are from the ordinary kriging results.

Supplemental Files

Supplemental File S4.1. Draft version of California Aquatic Resource Classification.

Background and Need for Classification

Purpose

The purpose of this memorandum is to recommend an aquatic resource classification system to the State Water Board's Wetland and Riparian Area Protection Policy (WRAPP) Policy Development Team and the California Wetlands Monitoring Workgroup (CWMW) with enough additional information to establish and support a link to policy. Previous memorandums have established the Technical Advisory Team (TAT); proposed a wetland definition; recommended a landscape framework for deepwater, wetlands, aquatic support areas, and uplands; and recommended a methodology for identifying and delineating wetlands and aquatic support areas. Subsequent memorandums will propose riparian area identification and delineation approaches, a wetland and riparian mapping methodology, and a wetland and riparian mitigation and assessment methodology. As envisioned, the classification system would be used to support implementation of WRAPP and provide the common language necessary for data and information sharing between and within State agencies and partners. The classification language could be used for mapping and status and trends assessment as well as condition or functional assessments at the project, watershed, regional, and statewide scales.

Uses for the Classification System

This classification system is recommended to the Policy Development Team for use in the State Water Board's Wetland and Riparian Area Protection Policy. It will also submit the system to the CWMW and the California Water Quality Monitoring Council (CWQMC) for possible recommendation to and use by other state agencies. Coordinated use of a common classification has been recommended by the CWQMC to support implementation of the recommendations in the 2010 *State of the State's Wetlands Report* and *Tenets of a State Wetland and Riparian Area Monitoring Program (WRAMP)* (CNRA 2010; CWMW 2010).

The primary proposed use for this classification system is a "common language" of shared terminology and definitions, principally applied during remote or field-based mapping and investigation of aquatic resources. This common language will have several benefits for the State of California. First, it will provide coherence between programs operated by different agencies, at different levels of government. This will enable information exchange, data aggregation, and implementation of Level 1 assessments such as the California Aquatic Resource Inventory (CARI) or the Status and Trends (S&T) program, both currently under development. These types of assessments will enhance aquatic resources management and planning under WRAPP and WRAMP. Second, accurate and specific classification in Level 1 mapping will provide a foundation for selecting appropriate Level 2 and 3 assessment tools (Stein et al. 2009). Standardizing the link between individual aquatic resource types and specific assessment methods will increase the weight of evidence behind individual results and will support tool refinement in general. These benefits will help protect and improve resource condition as part of a project management.

It is important to note, although the classification system is intended to support regulatory applications, it is not intended be a jurisdictional determination. Delineation of jurisdictional boundaries is governed by agency-specific regulations and requires field-based assessment of hydrology, soil/substrate conditions, and/or vegetation. While this classification system was developed with agency uses in mind, it is founded in scientifically-based definitions of non-wetland open waters, wetlands, and aquatic support areas, recommended by the WRAPP TAT in earlier technical memos (Technical Advisory Team 2009a; Technical Advisory Team 2009b). Furthermore, the classification system does not require field surveys and can be applied using remotely sensed imagery and appropriate auxiliary information. Finally, because the classifications are meant to provide a common language between agencies, the system should be applied uniformly, independent of specific agency jurisdictions. Thus, while the classification can provide a foundation for overall agency wetland management, it cannot be used to determine agency jurisdiction. It is our contention that this basis means the classification system will support the broadest possible range and variety of regulatory and management programs, consistent with the goals of the CWQMC (CWMW 2010).

Goals for California's Classification System

Recognizing the intended uses and users for the proposed classification system, the classification system was designed to provide accurate and reliable, descriptive categorizations of California's aquatic resources; to support the regulatory, management, monitoring and assessment needs of the primary agency users; and to increase the efficiency of classification. The intent is the classification will form a coherent basis for all aquatic area mapping, assessment, planning, management, monitoring, and regulatory actions by California agencies, including the assessment of impacts, mitigation planning, and monitoring.

Development of the classification system was guided by a set of goals, consistent with the intended uses of the system. The goals are to:

1. Cover all aquatic resource types including wetlands, non-wetland open water, aquatic support areas, streams, and channels (as defined by the TAT in earlier memos) but excluding groundwater, subsurface flow not directly associated with a surface aquatic resource, and isolated aquatic support areas.
2. Help infer beneficial uses and functions, consistent with the State and Regional Water Boards.
3. Support and be consistent with WRAMP, including the California Aquatic Resources Inventory (CARI) and the California Rapid Assessment Method (CRAM).
4. Crosswalk with the National Wetlands Inventory (NWI) and the Coastal and Marine Ecological Classification Standard (CMECS).
5. Have a flexible hierarchical structure that supports classification at different levels of detail and can incorporate more detailed regional or site-specific information, if available.

Methodology

Development of the classification system began with a review of existing national and state classification systems (Appendix I) by a technical advisory committee (TAC) consisting of state and federal agency staff, national experts, and program managers from other states with experience in wetland mapping and classification. The TAC evaluated each system against a common set of criteria (Appendix II) to determine potential applicability and use in California. Based on this review, the TAC developed, discussed, reviewed, and revised several iterations of the proposed classification system. Draft products resulting from the TAC's initial efforts were tested and refined by trial application to wetlands throughout California. The draft classification system was then vetted through regional and statewide wetland agency workgroups including the CWMW. Input from these reviews was used to refine the classification to produce the proposed draft-final version for submission to the Policy Team.

Results

Existing classification systems were reviewed according to the criteria agreed to by the TAC, but no existing system satisfied all of California's needs and applications (Section 3.1). Instead, the TAC recommended creation of a new system (Section 3.2) based on a functional classification approach similar to HGM and CRAM. In addition, HGM and CRAM terminology and definitions would be used to maintain continuity with existing agency programs in California. Finally, elements of LLWW, NWI, and CMECS would be incorporated to support creation an aquatic resource classification system instead of a system applicable only to wetlands. By not basing the system solely on CRAM, the result is both consistent with existing assessment tools (e.g., CRAM), and indicates where additional tool development is needed. This approach also reduces the likelihood the classification system will require revision as assessment tools are developed, revised, and expanded.

Summary and Comparison of Existing Classification Systems

USFWS, Cowardin: defines wetlands using vegetation, soil/substrate, and hydrologic regime (Cowardin et al. 1979). This classification is used by the National Wetland Inventory for mapping and status and trends programs (Dahl and Bergeson 2009). Focus is on grouping similar ecological units and habitat functions. System is well developed for mapping and remote sensing and used nationally. However, crosswalk to CRAM, HGM, and other classifications is ambiguous, particularly for "palustrine" wetlands. In addition, Cowardin may not optimally classify wetlands in arid regions — where evaporation exceeds precipitation (NWI 1997). Finally, function can be only partially inferred and is not the foundation of the method.

CRAM: methodology was developed, and wetland types defined, specifically for California's assessment needs (Collins et al. 2008). Existing modules are based on HGM wetland types and were developed in response to California's policy needs. CRAM, by definition, provides excellent support for condition assessment; includes many rare wetland types for California, such as vernal pools; and is fully consistent with WRAMP. However, CRAM modules are not comprehensive, modification and refinement are ongoing, and new modules will be created with time.

HGM: classification based on geomorphic setting, water source, and hydrodynamics (Brinson 1993). Function and ecology are inferred based on these three properties. Classification and delineation manuals exist for California and significant overlap exists with CRAM. However, HGM is not fully consistent with WRAMP and some modification may be required in order to capture all rare wetland types in California.

LLWW: classifies wetlands using landscape position, landform, water flow path, and waterbody type (Tiner 2003). Can be applied independently or as a hydrogeomorphic supplement to Cowardin; therefore, providing information about abiotic functions. However, LLWW was developed based on East-coast wetland types and would require additional development in order to capture wetland types important to California.

Ramsar: non-hierarchical list of habitat-based definitions created to classify wetlands of international importance (Ramsar 1996). Used internationally, Ramsar recognizes the usefulness of form, hydrology, and function as the basis of classification. While Ramsar wetland types provide a common vocabulary, the system lacks a classification procedure. In addition, Ramsar does not include all functions or habitats of importance in California.

California Forest Practice Rules: set of regulatory definitions developed by the California Department of Forestry and Fire Protection to support California's forestry policy (Forest Practice Program 2011). System has a direct connection to and application in policy but was not developed as a comprehensive classification system and does not support ambient assessment.

Canadian Wetland Classification System: ecosystem based classification approach for Canadian wetlands (National Wetlands Working Group 1997). Broad classification groups are based on abiotic parameters and more narrow classification units on biotic and ecological characteristics. System provides excellent consideration of structure, hydrology, and biology but is specific to Canadian wetlands and includes many classes not relevant in California.

Discussion of Elements for the California Aquatic Resource Classification System

Following review of existing classification systems, the TAC developed six specific recommendations for the California Aquatic Resource Classification System (CARCS):

1. Include open water, wetlands, and aquatic support areas
2. Provide information about the most important elements for a functional classification of aquatic resources
3. Define and classify using CRAM terminology and elements of HGM and LLWW
4. Provide clear divisions between remote and field-based components
5. Allow flexibility through additional modifiers or finer classifications
6. Cross-walk with Cowardin/NWI and CMECS

First, the TAC defined the scope of the proposed classification system to include open water, wetlands, and aquatic support areas; the proposed classification system does not address riparian areas. Open water, wetlands, and aquatic support areas are mutually consistent and exclusive aquatic resource area categories, identified and defined in previous TAT memoranda, reflect the diversity of possible uses, and rooted in the landscape moisture gradient concept

(Technical Advisory Team 2009b). The TAT has not addressed a definition or mapping approach for riparian areas at the time this system was developed. Additionally, the riparian definition could potentially include uplands, aquatic support areas, wetlands, and open water. That is, riparian areas could potentially fail to represent a mutually consistent or exclusive category within the classification system. Once the definition and mapping procedures for riparian areas are developed, their classification can be easily integrated into this system as a system of modifiers or as a separate classification hierarchy.

Second, the TAC agreed functional aquatic resource classification involves six physical and biological characteristics: hydrogeomorphology, landscape setting or connection, anthropogenic influence, dominant vegetation type, water regime, and substrate class. The listing order does not necessarily define one element as more important than another, as different applications have different information needs. However, the TAC recognized hydrogeomorphology and landscape setting provide critical context for the other four elements, are strong predictors of function, and are relatively easily applied using remote sensing. Therefore, the TAC developed the classification system based primarily on the mandatory identification of attributes of hydrogeomorphology and landscape setting, combined with the optional identification of other elements to support additional details of classification. The proposed modifiers, if used, would further refine expectations for the function and performance of a particular aquatic resource. Depending on the specific application, one or more of the modifiers could be considered essential.

Third, the TAC agreed a classification system consistent with the wetland classification in CRAM would provide for a seamless integration between Level 1 mapping and Level 2 condition assessment, as called for by the CWMW and the SB 1070 Monitoring Council in the WRAMP and the State Water Board in the WRAPP. Other classification systems can be cross-walked with CRAM modules but any cross-walk inherently misses information. However, the CRAM wetland classification does not fully address all wetland types in California. Therefore, the TAC chose to combine elements from HGM and LLWW with the CRAM classification in order to assemble a complete classification system. CRAM, HGM, and LLWW define wetland types based on hydrogeomorphology and landscape setting, consistent with what the TAC viewed as the most important elements to include in the classification system.

Fourth, the TAC recommended clear distinctions between what should be mapped remotely and what should only be mapped based on field-based information. Some of the important elements of aquatic resource classification cannot be accurately or reliably mapped from remotely sensed information alone. For example, remotely sensed imagery often does not provide the resolution necessary to identify dominant plant species or to classify sediments. In addition, water regime classification systems can require information about how water levels change daily, monthly, and yearly and this information must be available for several successive years. However, these elements are still critical for understanding function and performance. In these cases, mandatory inclusion of dominant species, sediment type, and water regime would most likely reduce the accuracy and reliability of the aquatic resource classification in cases of remote classification (a likely majority of applications). Therefore, the TAC's members recommend genus- and species-level vegetation information, water regime, and substrate be identified only from field-level assessments.

Fifth, the TAC recognized the diversity of agencies and individuals that will use the proposed classification system. This includes both geographic, aquatic resource type, and programmatic diversity. To satisfy this wide range of needs, the proposed classification system focuses on robust terms applicable in multiple geographic regions and useful in multiple agency contexts. To facilitate specific regional and agency needs, the TAC supports addition of modifiers and further development of the hierarchical scheme at the regional, agency or institution level, as long as the added classification elements are consistent with the required components of the system proposed here.

Sixth, the TAC acknowledged existing national wetland and aquatic resource mapping and classification efforts such as the National Wetland Inventory (NWI, based on the Cowardin classification, and to some extent including LLWW classification) and the draft Coastal and Marine Ecological Classification Standard (CMECS) developed by the National Oceanic and Atmospheric Administration (Standards Working Group 2010). To provide consistency with these mapping and classification systems, the TAC supports creation of a crosswalk between the proposed classification system and the Cowardin/NWI classification and CMECS.

Recommended California Aquatic Resource Classification System

<u>Structure and Approach</u>

The proposed classification system consists of six elements important in aquatic resource classification. The first two elements (hydrogeomorphology and landscape position) are <u>hierarchical</u> (e.g., aquatic area classes occur within major classes, and aquatic area subtypes occur within types; see Table B.1), and classification elements represent mutually exclusive categories. These two elements are anticipated to be identified using remotely sensed information, are mandatory, and must be applied in all cases (these two elements are also most relevant for crosswalks to other classification systems). The remaining four elements (anthropogenic influence, hydrology, substrate, and vegetation) are <u>modifiers</u>. The hierarchy and the anthropogenic influence and vegetation modifiers can all be applied remotely while the water regime and substrate modifiers should only be applied based on field information. Modifiers may not always comprise mutually exclusive categories. The use of modifiers is recommended, but is not required in order to apply the recommended classification. The system can be cross-walked to CMECS and the Cowardin (1979) classification system at the higher levels. Although wetland definitions recommended by the TAT for the WRAPP are used to guide the overall structure of the classification system, the mapping does not constitute a jurisdictional determination.

The first component of the complete system is a hierarchical classification based on hydrogeomorphology and landscape setting. This hierarchy comprises the required components of the classification system and is easily translated to expected functions (Table B.2). Suggested additional modifiers cover anthropogenic influence, vegetation, water regime, and substrate. The hierarchy and the anthropogenic influence and vegetation modifiers can all be applied remotely while the water regime and substrate modifiers should only be applied based on field information. In addition, utilization of VegCamp is strongly encourage and is fully consistent with the proposed system. However, VegCamp requires specialized expertise and training as well as field information. Therefore, it is not included as part of the proposed classification system.

The hierarchical component categorizes aquatic resources based on hydrogeomorphology and landscape position. Hydrogeomorphology indicates the dominant characteristics of water source and dynamics for aquatic resources. Landscape setting can indicate potential influences of "place" on functions, beneficial uses, or resource condition.

The hydrogeomorphology classes are consistent with CRAM classes. However a comprehensive set of CRAM modules does not yet exist for all hydrogeomorphologic classes; the exact types of wetlands covered by each module is shifting based on changing understanding of wetland systems; and some portions of some module definitions can only be defined in the field, such as seasonally versus perennially tidal estuarine systems. Therefore, the hydrogeomorphology and landscape setting classification structure are not identical to the current list and organization of CRAM modules. Indeed, it is the hope of the TAC that the proposed classification system will drive development and refinement of CRAM modules. This relationship will ensure that mapped and classified wetland can be adequately assessed for condition.

The four proposed modifiers provide additional information for classification of aquatic resources. However, the modifiers may require additional information that cannot be obtained remotely, involve specialized mapping and classification expertise, and may form more of a continuum of overlapping groups than distinct categories. Therefore, the modifiers are excluded from the hierarchy and are optional.

As with any classification system, attributes represent condition at the time of assessment. Wetlands by definition are dynamic systems and thus the "classification" of a site may change over time, boundaries may shift, and the modifiers may be yet more time-variable. For example, the hydrology modifier is particularly sensitive to the dynamic nature of aquatic resources, even when field information is available.

The system as proposed can be modified to satisfy agency and regional needs through addition of new modifiers or finer levels of hierarchical classification. In general, we have erred on the side of broad classes and groups for both the hierarchical classification and modifiers. Care should be taken so that changes do not prevent classifications from being rolled-up into the system outlined here.

Required Hierarchical Classification

The hierarchical classification assigns a hydrogeomorphological "major class" and "class," and a landscape position type, to each aquatic resource. The hierarchy should be applied to wetlands first, and the resulting classification extended to associated non-wetland open waters and aquatic support areas. The exception is for marine systems, which do not form wetlands as defined here. In this case, non-wetland open water marine systems should be classified first and the classification extended to the associated aquatic support area.

Table 1. Hierarchical classification component.

Hydrogeomorphology		Landscape Connection	
Major Class	**Class**	**Type**	**Subtype**
Non-wetland Open Water	Lacustrine	*Same as associated wetland*	
	Riverine		
	Estuarine		
	Marine	Intertidal	Cove Embayment Exposed Shoreline
		Subtidal	Cove Embayment Exposed Shoreline
Wetland	Depression	Floodplain	Defined outlet Undefined outlet
		Non-floodplain	Defined outlet Undefined outlet
	Lacustrine	Structural Basin Topographic Plain	
	Slope	Hillslopes Fan Break in Slope Topographic Plain	
	Riverine	High-gradient	
		Low-gradient	Confined Unconfined
	Estuarine	Canyon Mouth River Valley Mouth Delta	
		Structural Basin	Embayment-Rocky Headland Embayment-Bar Built Lagoon Dune Strand/Dammed
Aquatic Support Areas	*Same as associated Wetland*		

Table 2. Translation of hierarchical classification to function.

		Groundwater		Surface Water		Floodwater	
		Recharge	Discharge	Supply	Replenishment	Peak attenuation	Storage
Depressions	Floodplain	X				X	X
	Isolated with defined outlet						
	Isolated without defined outlet	X				X	X
Lacustrine	Structural basin	X		X		X	X
	Topographic plain	X		X		X	X
Slope	Hillslopes		X				
	Fan		X				
	Break in slope		X				
	Topographic plain		X				
Riverine	High grade				X		
	Low grade, confined		X		X		
	Low grade, unconfined		X		X		
Estuarine	Canyon mouth					X	X
	River mouth					X	X
	Delta					X	X
	Embayment-rocky headland					X	X
	Embayment-bar built					X	X
	Lagoon					X	X
	Dune-strand/dammed					X	X
Marine	Cove, intertidal					X	
	Cove, subtidal					X	
	Embayment, intertidal					X	
	Embayment, subtidal					X	
	Exposed shoreline, intertidal					X	
	Exposed shoreline, subtidal					X	

		Sediment			Cycling		
		Transport	Storage	Supply	Organic matter	Nutrient/carbon	Surface water filtration
Depression	Floodplain		X		X	X	X
	Isolated with defined outlet		X			X	X
	Isolated without defined outlet		X		X	X	X
Lacustrine	Structural basin		X			X	
	Topographic plain		X			X	
Slope	Hillslopes	X		X	X		
	Fan	X		X	X		
	Break in slope	X		X	X		
	Topographic plain			X	X		
Riverine	High grade	X		X	X	X	X
	Low grade, confined	X					
	Low grade, unconfined			X	X	X	X
Estuarine	Canyon mouth		X		X	X	X
	River mouth		X		X	X	X
	Delta		X		X	X	X
	Embayment-rocky headland		X		X	X	X
	Embayment-bar built		X		X	X	X
	Lagoon		X		X	X	X
	Dune-strand/dammed		X		X	X	X
Marine	Cove, intertidal		X				
	Cove, subtidal		X				
	Embayment, intertidal		X				
	Embayment, subtidal		X				
	Exposed shoreline, intertidal		X				
	Exposed shoreline, subtidal		X				

Definitions of Hydrogeomorphology Major Classes

- *Non-wetland open water*: Includes all marine systems and non-marine systems with area greater than 8 ha and average depth greater than 2 m, during the growing season, or greater than the maximum depth from which rooted vascular vegetation grows to the water surface, whichever is deeper. Areas that are temporarily inundated by deep water can be wetlands if such inundation does not persist throughout most of the growing season. Abbreviated NWOW.
- *Wetlands*: Under normal circumstances, a wetland (1) is saturated by groundwater or inundated by shallow surface water for duration sufficient to cause anaerobic conditions within the upper substrate; (2) exhibits hydric substrate conditions indicative of such hydrology; and (3) either lacks vegetation or the vegetation is dominated by hydrophytes. Abbreviated W.
- *Aquatic support areas*: meets one or two, but not all three, of the criterions in the wetland definition and is adjacent to W and/or NWOW. Abbreviated ASA.

Definitions of Hydrogeomorphology Classes

- *Depression (ASA, W)*: closed basin hydrology in topographic lows with no or variable inlets and outlets. System does not include a non-wetland open water portion (greater than 8 ha in area and 2 m in depth). May lack outgoing surface drainage except during flood events or heavy rainfall. Dominant water sources include precipitation and groundwater discharge from shallow saturated zones, nearby streams, or springs. Wetland can fill via surface or subsurface routes. Main loss mechanisms are evapotranspiration and/or infiltration. Many are seasonal and some lack ponding or saturated conditions during dry years. Abbreviated D.
- *Estuarine (ASA, NWOW, W)*: defined by the physical mixing of saltwater and freshwater. Typically has a bidirectional flow (typically tidal) hydroperiod. Often involves wetting and drying during different phases of the hydroperiod. May be saline or hypersaline, with minimal freshwater influence, or saline with a strong freshwater influence. Fully or partially tidal for at least 1 month during most years. Includes sub-tidal and intertidal environments. Tidal channels that do not dewater at low tide or are wider than 30 m are not part of the estuarine wetland. Abbreviated E.
- *Lacustrine (ASA, NWOW, W)*: closed basin hydrology including a non-wetland open water portion (greater than 8 ha in area and 2 m in depth). May or may not be prone to seasonal drying under natural hydrologic regime. Abbreviated L.
- *Marine (ASA, NWOW)*: strongly influenced by bidirectional (typically tidal) hydroperiod. Involves wetting and drying during different phases of the hydroperiod. Saline without strong freshwater influence. Includes sub-tidal and intertidal environments. Abbreviated M.
- *Riverine (ASA, NWOW, W)*: defined by unidirectional flow, but may be tidal (with bi-directional flow) in the lowest geographical reaches in a watershed; not subject to mixing of freshwater and saltwater. Not fully or partially tidal for at least 1 month during most years. Includes channel, active floodplain, and portions of adjacent areas likely to be strongly linked to channel or floodplain through bank stabilization and allochthonous

inputs. Active floodplain refers to periodically flooded area adjacent to and slightly above the active flow zone and can be vegetated or non-vegetated. Abbreviated R.

- *Slope (ASA, W)*: form due to seasonal or perennial emergence of groundwater into root zone. Hydroperiod mainly controlled by unidirectional subsurface flow. Slope wetlands are distinguished from depressional wetlands by having predominantly flow-through hydrology vs. a closed basin. Slope wetlands often exhibit strong dominance by groundwater flow or discharge, although many slope wetlands demonstrate abundant over-surface flow. If surface water moves through a well-defined channel, it is a riverine wetland. Abbreviated S.

Definitions of Landscape Connection Types and Subtypes

- *Break in Slope (S):* abrupt change in gradient such as the edge of a cliff, terrace, or scarp. Slope wetlands typically occur just below the break.
- *Canyon mouth (E)*: estuarine system formed at the mouth of a canyon. Canyons are very common in arid or semiarid regions where down cutting by streams greatly exceeds weathering. Canyons are very narrow, steep-sided (greater than 15%) systems including stream-cut chasms or gorges, the sides of which are often composed of cliffs. As opposed to *river valley mouth* systems, *canyon mouth* systems lack a well-defined channel
- *Confined (R):* width, across which the system can migrate without encountering a hillside, terrace, man-made levee, or urban development, is less than twice channel width or the channel has artificial levees or urban development preventing its migration. Entrenchment is not a consideration.
- *Cove (M)*: a small sheltered recess along a coast, often inside a larger *embayment*, with significantly reduced wave action is due to a naturally narrow inlet. Typically much less than 10 km^2 in area.
- *Defined outlet (D):* system has one or more apparent surface connections to other surface water features such as intermittent streams. Defined outlets function to limit system water level and residence time, particularly during or after precipitation events.
- *Delta (E)*: a typically lobed-shaped or fan-shaped landform formed by sedimentation process at the mouth of a river carrying heavy sediment loads. The Bay Delta is the only estuarine delta in California.
- *Dune Strand/Dammed (E)*: estuarine wetlands that form in the space between dunes. Typically are cut off from a larger estuarine system for significant portions of the year. As a result, the water level may be above or below the adjacent estuary.
- *Embayment (M)*: concave portion of shoreline forming a semi-enclosed indentation, recess, or arm of the ocean into the land or be between two capes or headlands. Larger than a cove, i.e., greater than 10 km^2. An embayment often appears as a crescent shaped coastal configuration of land.
- *Embayment-Bar Built (E)*: a semi-enclosed indentation, recess, or arm of the ocean into the land (i.e., an *embayment*), typically separated from the ocean by a sand-dune or earthen berm.
- *Embayment-Rocky Headland (E)*: a semi-enclosed indentation, recess, or arm of the ocean into the land (i.e., an *embayment*) formed by two rocky capes or headlands.
- *Exposed shoreline (M)*: relatively straight or convex (bending seaward) shorelines that are fully exposed to the waves and currents of the open ocean. Could also include

relatively straight portions of shoreline with a manmade structure, such as a breakwater or jetty, to artificially decrease wave action or erosion.

- *Floodplain (D)*: a broad, generally flat landform occurring in a landscape shaped by a fluvial or riverine process. For purposes of this classification, limited to the broad plains (wider than 1 km) associated with medium to large river systems subject to periodic flooding. Often have alluvial soils deposited during the flooding events.
- *Fan (S):* a low, outspread, relatively flat to gently sloping mass of sediment material, often shaped like an open fan, deposited by a stream at the place where it issues from a canyon or narrow valley upon a topographic plain or broad valley. Fans also occur where a tributary stream is near or at its junction with the main stream, or wherever a constriction in a valley abruptly ceases or the gradient of the stream suddenly decreases.
- *High-gradient (R):* system has an average gradient above 15%.
- *Hillslopes (S)*: generally steep (greater than 15% slope), high-elevation portion of foothills or mountains.
- *Intertidal (M)*: linear portion of shoreline covered by the great diurnal range (GT) as defined by the National Oceanic and Atmospheric Association (the difference in height between mean higher high water and mean lower low water, the averages for each tidal day observed over the National Tidal Datum Epoch or derived equivalent, created by comparison of simultaneous observations with a control tide station).
- *Lagoon (E)*: a shallow body of water separated from a larger estuarine bay or from the open ocean by a landform such as a sand spit, barrier beach, or reef.
- *Low-gradient (R):* system has an average gradient below 15%.
- *Non-floodplain (D):* surrounding landscape does not meet the definition of *floodplain*.
- *River valley mouth (E)*: tidal areas (brackish and fresh) where a well-defined channel meets an *embayment*, *lagoon*, *cove*, etc. Channel Mouth areas often consist of a *delta* with tidal channels, vegetated marshes, and mud flats. All river valley mouths contain some element of a bar (i.e. are bar-built) unless they have been structurally altered or hardened, in which case the appropriate anthropogenic modifier should be applied.
- *Structural basin (E, L)*: system is located within a pre-existing valley or canyon. Estuarine systems are either exposed by falling sea-levels or invaded by rising sea-levels. Lacustrine systems are filled by a naturally or artificially dammed river, groundwater discharge, and/or surface runoff.
- *Subtidal (M)*: marine system below mean lower low water (see *intertidal*).
- *Topographic Plain (L, S)*: a large level or nearly level (slope less than approximately 3 %) area usually at a low elevation in reference to surrounding terrain. The flat central portion (excluding active *floodplain*) of a large valley (e.g., San Joaquin) would be considered a topographic plain if broader than approximately 3 km on average. Also includes flat, low-elevation areas bordered by marine or estuarine coastline.
- *Unconfined (R):* width across which the system can migrate without encountering a hillside, terrace, man-made levee, or urban development is more than twice the average bankfull width. Unrelated to channel entrenchment.
- *Undefined outlet (D):* system lacks apparent connections to surface stream channels that could limit water levels during or after precipitation events.

Modifiers

Optional but suggested descriptors to provide additional information about the resource. Can be applied to any aquatic support area, non-wetland open water, or wetland. Vegetation and anthropogenic influence can be applied remotely while water regime and substrate should be applied based on field information. Vegetation can be expanded through use of VegCAMP if the necessary information is available and mapping and classification expertise exists.

Vegetation

Broadly classifies the dominant vegetation or lack of vegetation. Multiple modifiers can be used but each modifier should apply to at least 20% of the considered area before it can be included.

- *Non-vegetated*: less than 5% of terrain is vegetated or less than 5% of standing water contains apparent vegetation.
- *Forested*: vegetation has at least 10% canopy cover of woody plant species greater than 3 m in height.
- *Scrub-shrub*: vegetation has at least 10% canopy cover of woody plant species less than 3 m in height, and not more than 10% canopy cover of trees > 3 m in height.
- *Herbaceous*: vegetation has a least 5% cover of non-woody vegetation, and not more than 10% cover of woody vegetation. Should not be used for *emergent* vegetation.
- *Emergent*: vegetation is rooted below water surface and emerges above water level.
- *Floating*: vegetation is rooted below water surface, or is non-rooted, and is evident as a layer on water surface.
- *Submerged*: vegetation is rooted below water surface and does not emerge above water level.
- *Algal*: floating or submerged vegetation lacking true stems, roots, leaves and vascular tissue.

Anthropogenic Influence

Use to describe an observed or apparent anthropogenic influence on the system, likely to impact function and/or condition. Influence can be intentional or unintentional, historical or current, etc. as long as the influence is still apparently impacting the system.

- *Influences on the Whole System:*
  - *Modified*: used when another modifier does not apply but there is obvious evidence of anthropogenic influence.
  - *Remnant*: the current aquatic resource existed prior to establishment of an immediately adjacent anthropogenic disturbance, such as urban development or agriculture. Present resource boundaries could be smaller than historical.
- *Influences on Water Source and Hydroperiod*
  - *Agricultural Runoff:* water source is dominated by an artificially increased input of agricultural runoff — typically escaped or unused irrigation water.
  - *Constrained/Impounded*: modified by a man-made barrier that obstructs the movement of water out of the system to adjacent areas.

- *Diked*: modified by a man-made barrier that obstructs the inflow of water.
- *Ditched/drained*: modified by a man-made structure that functions to drain (usually via subsurface route) the system, thereby altering its natural hydroperiod.
- *Diverted*: anthropogenic modification to otherwise artificially lower the water level.
- *Infiltration*: area receives artificially increased input of treated or untreated water. Water is held for infiltration into a subsurface aquifer.
- *Stormwater Control:* water is held to attenuate flow or until infiltration or evaporation. Also includes systems designed to improve water quality, Typically involving addition of permeable surfaces, filtration, or impoundment.
- *Urban Runoff:* water source is dominated by an artificially increased input of urban runoff.

- *Influences on Substrate and Bank*
  - *Armored*: human actions have artificially consolidated banks and/or bottoms to prevent erosion through placement of concrete, large rocks or boulders, geotextiles, gabions, or other artificial stabilization.
  - *Excavated*: sediment or substrate has been removed to deepen and/or widen the area of inundation.
  - *Filled/graded*: area has had an artificial input of sediment, sand, rock, etc. due to human actions. May be performed to reduce topographic complexity and/or to change slope.
  - *Marine Control Structures*: breakwaters, jetties, groins, seawalls, etc. meant to control erosion, tidal influences, and wave action within an estuary or along a shoreline.
  - *Realigned:* channel has been relocated straightened, or otherwise altered to flow in a different location or pathway and/or through a different type of substrate.
- *Influences Related to Agriculture*
  - *Aquaculture*: standing, flowing, or tidal water used for production of aquatic organisms such as fish, mollusks, algae, etc.
  - *Flooded Agriculture*: cultivation of crops such as rice, wild rice, or cranberries, which require inundated for at least 1 month during the growing season.
  - *Flood Irrigation:* cultivation of crops, often grassy forage crops for hay, by flooding fields to point of saturation or shallow inundation.
  - *Harbors/Marinas/Ports*: open water area where boats are regularly docked, maintained, loaded, or unloaded. Typically have significant modification, armoring, and excavation of the shoreline.
  - *Orchards*: includes vineyards, and other areas planted or maintained for the production of fruits, nuts, berries, or ornamentals.
  - *Ranchland*: area is used for livestock production. Includes hayfields, meadows, managed vegetated areas subject to herbivory by livestock, and non-vegetated areas potentially subject to soil compaction by livestock.
  - *Rangeland:* wildland area used for livestock grazing outside of cultivated ranch and farmland.
  - *Recreation*: area used by humans for activities such as birdwatching, hiking, camping, fishing, biking, recreational vehicles, etc.

- *Row or Sown Agriculture*: soil surface has been mechanically or physically altered for production of crops.
- *Silviculture*: natural or planted forest used for timber production.

Hydrology

This limited set of modifiers applies to duration of tidal, flooded, flowing, or saturated conditions. One system could have up to three modifiers.

- *If tidal* — bidirectional flow for at least 1 month of the year or during extreme tidal/wind events.
  - *Irregularly tidal*: Bidirectional flow during extreme tides caused by high water levels or high wind. May stay predominantly non-tidal in some years.
  - *Perennially tidal*: Bidirectional flow, once or twice daily, most days, for at least 11 months of the year.
  - *Seasonally tidal*: Bidirectional flow, once or twice daily, most days, for at least 1 month of the year. May result from seasonal closures of tidal inlets.
- *If flowing* — unidirectional flow in a channel (or bidirectional in tidal rivers) at some time during a normal water year.
  - *Perennial flow*: Flowing water is present for the entire annual cycle; typically occurs in larger geographical areas because of a combination of precipitation and groundwater discharge.
  - *Intermittent (Seasonal) flow*: Flowing water is present for periods of weeks to months following the cessation of precipitation, but not throughout the annual cycle; typically occurs because of a combination of precipitation and groundwater discharge.
  - *Ephemeral flow*: Flowing water is present only during or immediately after precipitation events; typically occurs in small watershed areas as a direct response to precipitation.
- *If flooded* — standing water for at least 1 week of the year.
  - *Perennially flooded*: Standing water throughout year. Only dries completely under extreme drought conditions.
  - *Seasonally flooded*: Standing water for 3-9 months of the year associated with seasonal precipitation patterns.
  - *Temporarily flooded*: Standing water less than 3 months of the year or not associated with seasonal precipitation patterns. May be completely dry in some years.
- *If saturated* — without standing water but water table at or near surface for at least 1 week of the year.
  - *Perennially saturated*: Lacks standing water but water table is at or near surface throughout year. Only dries completely under extreme drought conditions.
  - *Seasonally saturated*: Lacks standing water but water table at or near surface for 3-9 months of the year. Associated with seasonal precipitation patterns.
  - *Temporarily saturated*: Lacks standing water but water table at or near surface less than 3 months of the year; may or may not be associated with seasonal precipitation patterns.

Substrate

Terms are based on the average size of sediments and their expected affect on biota and sediment transport. Different terms are used for flow-through, closed-basin, and tidal systems due to the different responses of these systems to substrate condition.

- *Riverine or flow-through systems:*
    - *Labile*: Greater than 50% of substrate is made up of sand or material less than 2 mm in diameter.
    - *Transitional*: Mixed system with greater than 50% of substrate made up of gravel to cobble sized material (greater than 2 mm and less than 256 mm or 10.1 in).
    - *Consolidated*: Greater than 50% of substrate made up of rock larger than cobbles, bedrock, or another consolidated material such as cemented sandstone. Artificially consolidated channels substrates (e.g., concrete) should receive this modifier and the appropriate anthropogenic modifier (e.g., armored).
- *Lentic or closed-basin systems:*
    - *Unconsolidated bottom*: Substrate of cobbles, gravel, sand, mud, or organic material smaller than 256 mm (10.1 in).
    - *Rock bottom*: Substrate of bedrock, boulders, or stones larger than 256 mm (10.1 in).
- *Marine or estuarine systems:*
    - *Rock*: 50% or greater cover of bedrock or consolidated pavement.
    - *Unconsolidated Substrate*: Less than 50% cover of bedrock or consolidated pavement.
        - *Coarse*: greater than 90% of particles by volume are larger than 2 mm diameter.
        - *Fine*: greater than 90% of particles by volume are smaller than 2 mm diameter.
    - *Faunal Reef*: Extensive structural substrate largely composed of biogenic materials formed by the colonization and growth of mollusks, polychaetes, or fauna other than corals. Reef-building fauna may or may not be present.
    - *Coral Reef*: Substrate or environmental setting largely constructed by the reef-building activities of corals and associated organisms. Live corals may or may not be present.

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APPENDIX I: Classification Systems Reviewed

Seven classification systems were chosen for evaluation:

1. US Fish and Wildlife Service (USFWS), Classification of Wetland and Deepwater Habitat (Cowardin 1979)
2. California Rapid Assessment Method (CRAM) for Wetlands
3. Hydrogeomorphic (HGM) Classification for Wetlands
4. Landscape Position, Landform, Water Flow Path, and Waterbody Type (LLWW)
5. Ramsar
6. California Forest Practice Rules (2010)
7. Canadian Wetland Classification System

Numerous classification systems exist. Systems not created or implemented in the context of large-scale inventory or monitoring programs were excluded. Regional or state-level derivatives of Cowardin or HGM were not specifically evaluated, but considered in light of the review of the overall Cowardin and HGM approaches to classification.

APPENDIX II: Criteria for Assessing Existing Classification Systems

The following criteria were developed to evaluate existing classification systems in light of the goals and intended uses for the California Aquatic Resources Classification System.

1. <u>Represents full range of CA aquatic resource form and function.</u> California has extraordinarily diverse aquatic resources, particularly wetlands and riverine systems, because of its physiographic and climatic variability. The purpose of this criterion is to assure that the classification system(s) captures the extreme forms of wetlands and riverine systems that typify alpine, coastal, desert, and temperate rainforest conditions, and that it captures the major variations in wetland and riverine form along the continuum of conditions between the extremes, to the extent that the variations can be discerned during wetland and steam mapping (see criteria 2 below - this criterion is not about explicitly denoting all the variations in wetland or stream form that might be identified in the field). It may be necessary for separate teams to work concurrently and in a coordinated way on different classification systems for wetlands, riverine systems, and riparian areas. This is because of the complex nature of these different systems and the many experts needed to understand and categorize their variability.
2. <u>Can be applied during mapping.</u> Some classification systems are based entirely on indicators that are evident in aerial images, satellite images, or on maps. Other systems combine such characters with modifiers based on information about management objectives or field conditions that cannot be known without site-specific reports or site visits. The purpose of this criterion is to make sure that the aquatic resources, wetlands, and riverine systems can be classified during wetland mapping without field visits or site reports, other than QA/QC procedures, assuming that the mapping is based on 1-m pixel resolution color imagery viewed at scale 1:5,000 (i.e., based on the draft State wetland mapping protocols).
3. <u>Supports ambient assessment.</u> The classification system should be regarded as part of the comprehensive state wetland and riparian area monitoring program (WRAMP); i.e., mapping, rapid assessment, intensive assessment, and data management. The aquatic resource maps need to serve as the sample frame for rapid and intensive assessment. The

classification system must therefore be consistent with the typology that is dictated by the assessment methods. The State has standard methods of intensive assessment of perennial wadeable streams (the Perennial Stream Assessment Program or PSA). The State is examining how the California Rapid Assessment Method for wetlands and wadeable stream systems (CRAM) might be used in regulatory and other contexts. Although CRAM need not be the basis for the classification system, any proposed system must be able to be related to CRAM.

4. <u>Is consistent with nomenclature of CA aquatic resource policies and programs.</u> A primary goal of WRAMP is to evaluate the performance of the State's policies and programs for protecting and restoring wetlands and riverine ecosystems plus their riparian areas. This means that the classification system needs to recognize the types of aquatic resources, wetlands, and riverine systems that are named in the State's policies and programs. For example, since the State has an Interagency Vernal Pool Stewardship Initiative, it needs a classification system that recognizes the different kinds of vernal pools covered by the initiative.
5. <u>Can be adequately cross-walked to other systems, especially NWI.</u> For the State's effort to map aquatic resources, wetlands, and riverine systems to enjoy federal funding, it must be consistent with, or exempt from, the wetland mapping standards promulgated by the Federal Geographic Data Committee (FGDC). At this time, the FGDC standards require using the Cowardin system of wetland classification based on guidance from the National Wetland Inventory (NWI) of the USFWS. The Cowardin system will be provided for ranking. However, the FGDC standards allow NWI to accept maps that do not strictly use the Cowardin system. NWI knows that many states have their own, unique wetland mapping and classification systems that could benefit NWI through a process of data translation and transference. In the mean time, it should be assumed that California's maps must employ the Cowardin system to comply with Federal standards, based on the following table.

<u>Classification Levels Required Based on Cowardin Habitat Type</u>

	System	**Sub-system**	**Class**	**Subclass†**	**Water Regime**	**Special Modifiers (where applicable)**
Lower 48 States*	Yes	Yes	Yes	Yes	Yes	Yes***
Estuarine & Lacustrine Deepwater**	Yes	Yes	Yes****	Yes*****	Yes	No

† At minimum users should include Subclass for forested, and scrub-shrub classes.
* Includes the lower 48 states.
** Includes the Estuarine and Lacustrine deepwaters of the lower 48 states.
*** Farmed wetlands need only include system and farmed modifier.
**** Classify as unconsolidated bottom unless data indicates otherwise for estuarine and lacustrine deepwater habitats.
***** Users should include Class and Subclass when data are available for estuarine and lacustrine deepwater habitats; for other areas Class will suffice.

6. <u>Complements the CA Vegetation Manual and mapping effort.</u> The State is implementing a statewide initiative to map vegetation (VegCAMP 2007), and has recently expressed interest in integrating vegetation mapping with wetlands and riparian mapping. VegCAMP does not map aquatic resources per se, but does map associations and

alliances of plant species that are indicative of such areas. The wetland maps should help predict plant species composition, and VegCAMP should help identify aquatic resources such as wetlands, riverine systems, and riparian areas. A description of VegCAMP will be provided. But, since it is not a wetland classification system, it will not be ranked.

7. Reflects expected difference in beneficial use or ecological service. The classification system should help managers estimate the kinds of beneficial uses or ecological services that aquatic resources, wetlands, stream ecosystems, and their associated riparian areas are likely to provide. This might be accomplished by annotating maps with information about water source, geomorphic setting, position in drainage network, and land use context. The existing classification systems that address these kinds of factors for wetlands, such as LLWW (Landscape Position, Landform, Water Flow Path, Waterbody Type) of the USFWS, will be ranked. For stream ecosystems, there is a variety of classification systems used to assess fluvial channel physical function (e.g., Montgomery-Buffington), channel behavior (e.g., Rosgen), riverine aquatic life support (e.g., Ca Forest Practice Rules), or salmon support (e.g., Legon-Dietrich). These stream classification systems will be ranked. The classification system would ideally be cross-referenced to the habitat classification system of the California Wildlife Habitat Relationships database. A copy of the CWHR classification system will be provided. But, since it is not a wetland classification system, it will not be ranked.
8. Can be expanded or contracted without requiring new inventories or maps. State policies and programs can shift their focus among aquatic resources, wetlands, and riverine systems. These shifts in focus tend to reveal subtypes that require special attention. For example, the increasing interest in wet meadows is likely to cause more kinds of them to be recognized. The classification system should be adjustable to accommodate such changes in the scope and specific focus of wetland and riverine policies and programs.
9. Is not too elaborate or complicated. Classification can be an expensive aspect of mapping. To minimize the cost, the classification system should be no more complicated or involved than needed to meet the other criteria.

APPENDIX III. Upland Classification for the California S&T Program.

- *Natural:* no apparent evidence of constructed surfaces, managed vegetation or agriculture, or increased human visitation
- *Anthropogenic:* area subject to constructed surfaces, managed vegetation or agriculture, or increased human visitation
    - *Industrial/Commercial:* constructed surfaces on more than 80%; contains large constructed surfaces such as shopping centers, warehouses, factories, industrial complexes, above ground storage tanks, etc.
    - *Residential:* constructed surfaces consistent with single or multi-family residences.
    - *Agriculture/Silviculture:* includes rowcrops, orchards, pasture or grazing, hayfields, and silviculture
    - *Recreation:* area of managed or unmanaged vegetation subject to increased human visitation. Includes passive recreation with unmanaged vegetation (activities such as hiking, bird watching, etc.) and active recreation with managed vegetation (ball-fields, golf courses, off-road motorized vehicle use, etc).

- *Transportation:* constructed surfaces for the purpose of human transportation. Includes large constructed areas for the purpose of transportation; principally airports, rail-yards or parking lots.
- *Other:* not described above, barren land, or a transition area

APPENDIX IV: Classification Table with Codes for the California S&T Program

Hydrogeomorphology		Landscape Connection	
Major Class	**Class**	**Type**	**Subtype**
Non-wetland Open Water (O)	Lacustrine (L)	*Same as Associated Wetland*	
	Riverine (R)		
	Estuarine (E)		
	Marine (M)	Intertidal (i)	Cove (c)
			Embayment (e)
			Exposed Shoreline (s)
		Subtidal (s)	Cove (c)
			Embayment (e)
			Exposed Shoreline (s)
Wetlands (W)	Depression (D)	Floodplain (f)	Defined Outlet (d)
			Undefined Outlet (u)
		Non-floodplain (n)	Defined Outlet (d)
			Undefined Outlet (u)
	Lacustrine (L)	Structural Basin (b)	
		Topographic Plain (p)	
	Slope (S)	Hillslopes (o)	
		Fan (a)	
		Break in Slope (k)	
		Topographic Plain (p)	
	Riverine (R)	High-gradient (h)	Confined (f)
			Unconfined (i)
		Low-gradient (l)	Confined (f)
			Unconfined (i)
	Estuarine (E)	Canyon Mouth (c)	
		River Valley Mouth (r)	
		Delta (d)	
		Structural Basin (b)	Embayment-Rocky Headland (r)
			Embayment-Bar Built (b)
			Lagoon (l)
			Dune Strand/Dammed (m)
Upland (U)	Natural (N)	*None*	
	Industrial/Commercial (I)		
	Residential/Housing (H)		
	Agriculture/Silviculture (A)		
	Recreation/Parks (P)		
	Transportation (T)		
	Other (O)		

CHAPTER FIVE: CONCLUSIONS

Recommendations and Implications for Aquatic Resource Extent Monitoring

Recommendations for the California S&T Program

The primary motivation for this study was the development of a probabilistic design for a California S&T program. California currently lacks an appropriate mechanism for producing accurate information about aquatic resource extent and distribution. This information is needed to help fulfill the landscape-level components of the California WRAMP strategy and can also be used to support other State monitoring programs and guide management and policy decisions related to aquatic resources (CWQMC 2010; USEPA 2006). A probabilistic S&T program cannot address all L1 information needs. For example, comprehensive mapping, project based accounting, and other approaches provide additional extent and distribution information that cannot be provided by a probabilistic S&T approach. However, a California S&T program can complement other L1 approaches and is a cost-effective mechanism for providing regional and statewide extent and distribution information.

The objectives for the S&T program design developed here are to provide estimates of extent and distribution, and changes in extent over time, for aquatic resources and subtypes, and to support regional or question-based intensification efforts. Based on these objectives and the simulation work presented in previous chapters, the following recommendations are provided for the California S&T program. First, the S&T program should utilize probabilistic sampling and analysis methods. Specifically, unstratified GRTS sampling provides precision and practicality benefits over SRS sampling. Selection of a GRTS master sample will also provide flexibility for the program to intensify sampling based on regional, programmatic, or scientific information needs. The plot size used should be based on consideration and balancing of estimated sample precision, the information mapped for each plot, and predicted program costs. Our conclusion is that a plot size of approximately 4 km^2 may be the best balance of cost-efficiency and mapped information for monitoring aquatic resource extent in California.

Second, the entire state should be used as a sample frame, not just areas with previously mapped aquatic resources. This will ensure the sample is not biased towards previously mapped areas. In addition, mapping and classification approaches should be carefully coordinated and calibrated between mapping professionals to ensure that results are compatible and can be combined to produce regional and statewide estimates. Mapping and classification should include the dominant land use of upland areas, in order to provide additional information about possible impacts on aquatic resources. If the draft or final CARCS system is used, but program managers want to ensure that information is compatible with national mapping efforts such as the NWI, the Cowardin classification system should be employed in addition to CARCS (Cowardin et al. 1979). Next steps in S&T program development include a standard mapping and classification procedures, together with an appropriate, error-checked classification crosswalk between the CARCS system and others.

Third, mapping should be repeated at regular intervals to produce estimates of change in aquatic resource extent and distribution. Fixed locations are expected to provide the best precision and least bias for estimating trends. A next step in S&T program development is a change assessment and classification methodology, based on fixed plots and remapping, for accurately quantifying gains, losses, and type conversions for aquatic resources. Sample locations could be added over time if additional resources become available; a regional, programmatic, or scientific intensification is conduced; or mapping costs decrease. Sample plots could also be added to “replace” plots considered impervious to changes in aquatic resource extent. Such imperviousness could occur if plots are completely covered by aquatic resources or by development. The impervious plots cannot be dropped from the sample, as this would bias estimation of extent and distribution; however, the plots do not require regular imaging and mapping and thus the resources available from not re-mapping those plots could be devoted to increasing the total sample size.

Implications for Aquatic Resource Monitoring

The California S&T program design, and the work conducted to develop the design, have implications for the development of other California aquatic resource monitoring programs. The implications could also potentially apply to other environmental populations with a spatial component, and populations outside of California. First, the design recommendation for the S&T program may be useful design options for other aquatic resource monitoring programs. For example, recommendations for an unstratified GRTS design may be a preferred starting point when designing a monitoring program. Besides the statistical advantages we observed relative to SRS, GRTS sampling also provides certain practical advantages, such as the master sample. For example, in addition to ensuring spatial balance, the master sample simplifies sample draw management if multiple parties are involved and prevents the need to perform a supplementary draw if additional sample points are required. In addition, the likely usefulness of fixed, moving, and SPR locations could be considered for the particular population. While the California S&T program design recommendation is for fixed sampling locations, other populations may be better served by moving or SPR locations. For example, if the monitoring approach has the potential to impact the population, or the monitoring emphasis is on current conditions instead of temporal trends, moving or SPR locations could be appropriate.

Second, simulated sampling approaches, develop to help select the probabilistic design, can be used to help design other monitoring programs and develop additional design parameters. However, this is only possible in instances where a suitable population, such as the NWI and NHD used here, is available for simulation. Absent a suitable existing population, an artificial population must be produced. The artificial population should, ideally, represent the best available information about the variability and spatial distribution of the true population, so that simulations represent, as well as possible, the true population. A number of artificial populations, with different spatial distributions and levels of variance, could also be used to account for possible bias introduced by using a single population. This approach, use of a variety of artificial population, was used in our study of temporal sampling designs.

Finally, a pilot scale implementation is strongly recommended for all large-scale monitoring programs. Our experience shows that this process can identify areas of the design and monitoring procedures that may benefit from modification prior to full-scale implementation. If funds do not exist for a full pilot, or the program is small enough, initial rounds of the program could be used to identify modifications and improvements for use in future rounds.

Implications for Aquatic Resource Management in California

This research was motivated by a need for aquatic resource extent and distribution information in California. In addition to needing extent information for management and scientific purposes, monitoring of extent and changes in extent has been a requirement of California’s aquatic resource policy since Governor Wilson established California’s no-net-loss policy in 1993 (CNRA 2010; Wilson et al. 1998). Because the primary existing approach, comprehensive mapping, is prohibitively expensive over an area the size of California, the State still did not have extent information, or a method for producing this information. Therefore, the results of this study will be instrumental for determining the best approach for producing aquatic resource extent information in California.

Beyond meeting the policy requirement of monitoring aquatic resource extent, results from the S&T program can support additional components of California's aquatic resource monitoring and management programs. For example, individual sample plots can be used to provide a sample frame for field-based assessments of aquatic resource condition. Several areas of California have not been mapped for aquatic resources or mapping occurred more than a decade prior. Therefore, S&T sample plots can provide an opportunity to extend probabilistic field studies to under-mapped areas.

In addition, the California S&T program can provide a key component of California's L1 strategy. L1 monitoring includes several approaches to assessing landscape level properties of aquatic resources. This includes comprehensive aquatic resource mapping, project based accounting of aquatic resource impacts, etc. Probabilistic estimates of aquatic resource extent and distribution cannot replace all components of the L1 strategy but a probabilistic approach is a cost-effective mechanism for providing extent and distribution information and can support implementation of other elements of the L1 strategy. Together, the landscape level information as a whole, such as that provided by the S&T program, can enhance California's aquatic resource management by providing a foundation for management decisions and a basis for assessing protection and conservation policies.

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