Technical Design for a Status & Trends Monitoring Program to Evaluate Extent and Distribution of Aquatic Resources in California

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1. Executive Summary

The 2010 State of the State's Wetlands report(CRNA 2010), issued by the California Natural Resources Agency, found that, while California is a leader in investment in wetland monitoring, protection, and restoration, the state lacks a coordinated statewide system to accurately determine wetland extent and distribution. Consequently, it is not possible to accurately report on the effect of these investments. The principal challenge to accurate assessment and effective monitoring over time is the expense of comprehensive mapping; conservative estimates predict comprehensive mapping of California's wetland resources would cost at least $8.4 million. As a result, the State cannot yet answer the fundamental question "What is the extent of California's wetland, and how is it changing with time?"

Probabilistic mapping can provide a cost-effective alternative for monitoring aquatic resource extent and distribution. Currently, the National Wetland Inventory Status and Trends program (NWI-S&T) and the Minnesota Status and Trends program (MN-S&T) utilize this approach to assess wetland extent and distribution. Elements of these two programs provide a foundation for the development of a status and trends (S&T) program for California. However, given California's geographic and climatic diversity, the design of the national program is not adequate to independently meet the state's information needs.

A probabilistic approach includes three basic elements: i) random placement of sample points across the entire state; ii) wetland map production for small (ideally, 1-16 km^2) plots placed at each point; iii) extrapolation from the random sample plot maps to a statewide estimate of wetland extent. This report provides recommendations for a probabilistic monitoring design for California's aquatic resource S&T. The design was developed to meet the following objectives:

- Report extent (status) and changes in extent (trends) at regular intervals.
- Include estimates for all surface aquatic resources including wetlands, streams, and deepwater habitat.
- Support regional intensification through design flexibility.

Design recommendations for the California S&T program were developed with input from the project's technical advisory committee (TAC) and based on a review of existing S&T programs. A series of design alternatives were identified for various program elements. Each design alternative was tested through repetitive simulation and modeling using the most comprehensive wetland and stream maps currently available. Simulation results allowed statistical comparison of each alternative and resulted in optimized technical design parameters with respect to California's S&T program objectives. Modeling results were discussed with the TAC, who produced the following design recommendations:

1. Use probability-based sample selection and analysis, as opposed to comprehensive mapping or non-probabilistic sample selection methods. Probabilistic sampling is a consistent design parameter in all of the reviewed S&T programs.

2. Select samples by generalized random tessellation stratified (GRTS) sampling without geographic pre-stratification. This will increase precision of estimates and provide a simple mechanism for regional intensification.

3. Use the entire State as a sample frame, rather than relying solely on areas with previously mapped aquatic resources. This will ensure that estimates reflect comprehensive extent and distribution of wetlands and aquatic resources.

4. Map and classify all elements within sample plots, including aquatic resources and upland land use. This will provide information about proximal anthropogenic influences and impacts on wetlands and aquatic resources.

5. Balance plot size with total sample size and the number of aquatic resources and wetlands covered by each sample plot. Small plots are more variable, and therefore require a larger total sample size, but may be more cost-effective at the program level. In contrast, large plots provide more information within each individual plot. We recommend a 4 km^2 plot size as the best balance of these factors for California.

6. Revisit and remap sample plots at regular intervals. This will help identify and track changes in extent and distribution, in addition to distributing mapping costs over multiple years.

7. Maintain static sampling plot locations over time, as opposed to monitoring new locations, or a mix of new and previously observed locations, at each time point. This will increase the accuracy and precision of estimates for both extent (wetland status) and changes in extent (trends).

The recommended program design was validated through a pilot-scale application at 60 plots in the Salinas River Valley and Southern California regions. This validation quantified expected random and systematic errors between map producers and between probabilistic estimates and comprehensive values. These error rates can be used to develop data quality objectives for use during program implementation.

The proposed S&T design will allow the State of California to reliably estimate the extent and distribution of wetlands, streams, and deepwater habitat, and changes over time in a cost effective manner. However, a probabilistic program, such as the proposed S&T program does not result in a “wetland map” for California. The S&T program, combined with other elements such as regional intensive maps, project-based accounting, and analysis of drivers of wetland loss will allow California to meet the needs for a comprehensive strategy to assess wetland gains and losses, and will support condition assessment, and will ultimately facilitate evaluation of the effectiveness of the state’s wetland protection and restoration programs.

2. Introduction and Background

2.1 Need for a California Status and Trends Program

Despite being a national leader in investment in wetland protection, management, and monitoring, California agencies cannot reliably answer essential questions about the extent and distribution of wetlands and streams and how these resources are changing over time. The 2010 State of the State's Wetlands report, produced by the California Natural Resources Agency (CNRA), highlighted the challenges associated with compiling, coordinating, managing, and disseminating information about aquatic resources between a broad range of public and private organizations:

> *A fundamental challenge facing entities entrusted with protecting California's wetlands is the lack of an integrated, comprehensive wetland monitoring and assessment program and the associated data management infrastructure to support it. The actual "state of California's wetlands" will not be fully understood until such a program is in place. An enhanced data management system would not only allow assessment of status and trends, but will facilitate improved coordination among the various entities involved in wetland regulation, management, and protection. Perhaps most importantly, it will improve transparency of wetland programs and information by making it more easily accessible to the public.*

The present project was designed to address one of the information needs highlighted in the State of the State's Wetlands report: accurate monitoring of the extent and distribution of wetlands on a regional and statewide basis (CNRA 2010). All components were developed with the consultation and assistance of a technical advisory committee, comprised of technical experts and managers with expertise in mapping, probability-based program design, and program implementation (Appendix A).

A probabilistic program to monitor the extent (status) and changes in extent (trends) of California's aquatic resources can improve California's ability to accurately monitor aquatic resource extent and answer questions such as, "how much wetland area does California have, where is it located, and how is it changing with time?" Monitoring extent and distribution over time can involve a range of analytical approaches, including: comprehensive mapping; probabilistic monitoring of wetland extent and distribution; project-based accounting and monitoring; investigation of natural and anthropogenic drivers of aquatic resource gain, loss, and conversion; and evaluation of programs and policies. Probabilistic monitoring is the most cost-effective and flexible of these approaches and can greatly enhance the state's capacity to answer statewide and regional extent and distribution programs. A probabilistic status and trends (S&T) program can also quickly and easily expand mapping to areas of the state that have been underserved by prior mapping initiatives.

2.2 Report Organization

This report provides design recommendations for an S&T program optimized for monitoring aquatic resource extent and distribution in California. The report contains six major sections. Section 2 provides introductory and background information for a California S&T program. Sections 3 and 4 present modeling and simulation work that was performed to evaluate and optimize design options for spatial and temporal sampling methodology consistent with the program objectives. Section 5 discusses the evaluation of the recommended design using pilot-scale implementation in two regions within California. Section 6 presents a full set of design recommendations for the California S&T program based on results from the project's modeling and simulation work and pilot-scale implementation.

We intend this report to be used in two major ways. Primarily, the design recommendations can be used by state agencies such as the California Natural Resources Agency (CNRA), Department of Fish and Game (CDFG), or the State Water Resources Control Board (SWRCB) to guide development and implementation of a California S&T program. Secondarily, the methodology used to model, simulate, and optimized the design recommendations could be applied to development of other probabilistic design programs, such as general habitat assessment or aquatic resources condition assessment.

2.3 Context for the California Status and Trends Program

The State of California first attempted to quantify statewide wetland extent and change in the first State of the State's Wetlands report, released in 1998 by California Governor Pete Wilson, the CNRA, and the California Environmental Protection Agency (Cal/EPA) (Wilson et al. 1998). This report was published five years after the first statewide wetlands monitoring program in California was established by Executive Order W-95-93, which proposed "no net loss" as a policy goal for wetland management. The 1998 report measured progress towards this goal by performing an accounting of permitted actions (such as Clean Water Act Section 404 dredge and fill permits) and restoration or conservation activities. The report concluded that wetland area, as measured through permitted actions, had increased in California by 15,129 acres between 1996 and 1997. However, the report touched on issues associated with solely using reported actions to determine wetland gains and losses. These issues included failure to capture illegal or exempt wetland losses, omission of non-regulatory restoration and conservation programs, inability to include natural changes in wetland extent, lack of physical verification to ensure that created wetlands are successful, and absence of information for specific wetland types.

In 2007, the California Water Quality Monitoring Council (CWQMC) was created pursuant to California Senate Bill 1070, which charged the CNRA and Cal/EPA with establishing a council to provide recommendations related to improving the coordination and cost-effectiveness of water quality monitoring, enhancing integration and data sharing across agencies, and increasing public access to data. In addition to the CWQMC, CNRA and Cal/EPA created a number of working groups, including the California Wetland Monitoring Workgroup (CWMW), which was established to address issues specific to wetlands. Subsequently, the CWMW released a strategy document that would embody the principal driving force of wetland monitoring in California.

In 2010, a second attempt at quantifying wetland extent came with the second State of the State's Wetlands report, released by the CNRA (CNRA 2010). In the intervening decade, progress on the National Wetland Inventory (NWI; a comprehensive wetland mapping program operated by the US Fish and Wildlife Service) in California allowed the State to estimate total wetland acreage at 2.9 million. However, the NWI still covered only approximately 80% of the state, and map vintages varied from between the 1970s and the 2000s. In addition, assessment of no net loss was still based on an accounting of regulated or reportable actions. The 2010 report recognized these shortcomings and recommended several solutions, including coordination, standardization, and classification of monitoring, assessment, identification, and mapping across state agencies and partners. These recommendations were echoed and expanded upon by the CWMW strategy document entitled “Tenets of a State Wetland and Riparian Monitoring Program (WRAMP)” (CWMW 2010).

The WRAMP document lays out the objectives and goals of the statewide approach as well as specific recommendations for the different components. The primary stated objective 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. Specific recommendations related to mapping included adoption of a standardized, statewide wetland definition, classification system, and mapping protocol, in addition to investigation of a probabilistic approach to monitoring wetland extent. Prior to beginning the work presented in this report, the only completed component was the standardized definition (Technical Advisory Team 2009a).

2.4 California S&T Program Information Needs and Goals

Mapping is a primary component of the first level in the US Environmental Protection Agency’s (EPA) Level 1-2-3 framework approach to wetland monitoring. Level 1 (L1) refers to landscape level analysis and underpins rapid (L2) and intensive (L3) field-based assessments of condition. Existing elements of California’s L1 strategy include the California Aquatic Resources Inventory (CARI) map, the California Wetland Portal (CWP) and EcoAtlas for tracking of regulated and reportable projects, and coarse indices of landscape stressors such as percent imperviousness or landscape development intensity. Probabilistic assessment of wetland extent, if adopted, would act as a foundational element and an additional component of this broader L1 strategy.

Because L1 activities act as a foundation for all field based monitoring activities, the L1 strategy should ideally address all aquatic resource types in California and provide usable, contemporary information to guide rapid and intensive condition assessments. In addition, the CWMW recommends that L1 strategies provide directly useful information to the public, scientists, and lawmakers, such as:

- Total wetland area in California
- Wetland locations and types
- Changes in wetland extent, composition, or distribution over time

Existing mapping approaches are inadequate to provide contemporary, comprehensive information due to their varying vintages, quality, and incomplete coverage across the state. As an alternative, a properly designed, probability-based sampling program can provide timely and accurate information when implemented as part of a coordinated L1 strategy, freeing up resources to devote to more targeted L1 studies.

2.5 Existing and Emerging Approaches

Current aquatic resource monitoring approaches in California were not designed to accurately measure and report California aquatic resource extent and distribution. For example, California aquatic resource assessment programs, such as the Surface Water Ambient Monitoring Program (SWAMP) operated by the State and Regional Water Boards and the Resource Assessment Program (RAP; http://www.dfg.ca.gov/rap/) operated by the California Department of Fish and Game (CDFG), currently focus on condition assessment and do not provide estimates of resource extent and distribution (SWAMP 2010). Similarly, comprehensive federal mapping programs, such as the US Geological Survey National Hydrography Dataset (NHD; http://nhd.usgs.gov/) and the NWI were produced with variable methodology and do not cover the entire State (USGS 2000, Tiner 2009, USEPA and USGS 2010). In addition, probability-based federal programs, such as the S&T component of the NWI or the Natural Resources Inventory (NRI), both discussed in greater detail below, were not designed to meet state-level reporting and estimation needs (USDA 2007, Dahl 2011). For example, the NWI-S&T contains only 257 observation locations in California, covering only 0.6% of the land area, mostly concentrated along the coast (Figure 2.1).

Figure 2.1. Location of NWI-S&T plots in California. Each plot is 4 mi^2.

For most existing programs, mapping approaches have almost exclusively involved a comprehensive census of aquatic resources. While this type of approach is considered the gold standard, it is also time consuming and expensive. As a result, contemporary information is usually only available for limited geographic areas. Probabilistic S&T programs bring a new approach to monitoring aquatic resource extent and distribution. A probabilistic approach, where a randomly selected portion of the target area is mapped at one time, can be used to fill part of this information gap. Because mapping costs (for both time and money) are reduced, probabilistic sampling can provide population level statistics, for the entire area, more frequently. This summary information could then be used to drive more targeted and intensive mapping, focused on regions or resources of particular interest. If properly designed, the combined approach of probabilistic sampling and mapping, followed by targeted, intensive mapping for specific objectives, can be more cost effective and provide more useful information than comprehensive mapping alone. Figure 2.2 illustrates the difference between a comprehensive and a probabilistic mapping approach.

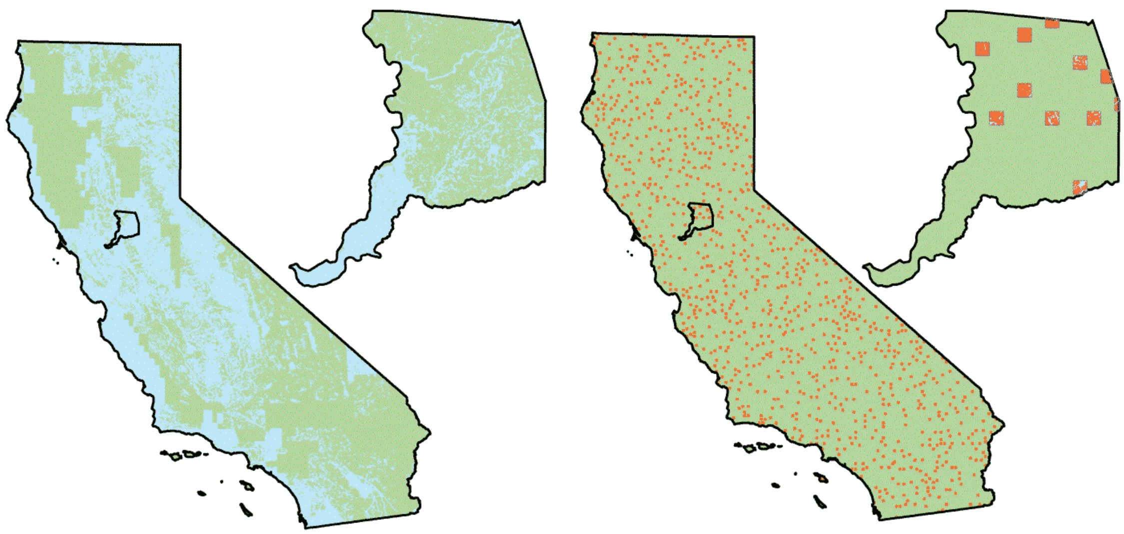


Figure 2.2. Comprehensive (left, blue) and probabilistic mapping (right, orange) for California with a Sacramento county detail. The comprehensive map is the comprehensive NWI map in California (separate from the NWI-S&T shown in Figure 2.1).

2.6 Review of Existing S&T Programs and Statistical Approaches

To guide development of the California S&T program, a review of two federal and one state program was completed. The three relevant programs were:

1. Wetland Status and Trends for the National Wetlands Inventory (NWI-S&T), US Fish and Wildlife Service (USFWS)
2. National Resource Inventory (NRI), US Department of Agriculture (USDA)
3. Status and Trends of Wetlands in Minnesota (MN-S&T), Minnesota Department of Natural Resources (MN-DNR)

A relevant program was defined as one that employs a probabilistic sampling design for estimating total wetland extent in a target geographic area. Programs using probabilistic sampling to select sites for condition assessment were not considered unless the program also included mapping from aerial photography. The two federal programs, NWI-S&T and NRI, are well known and often cited. The Minnesota program, MN-S&T, was the only state program selected after initially reviewing available information on state websites, contacting employees at several state environmental agencies with potentially relevant programs, and requesting suggestions from technical partners across the country. Table 2.1 summarizes the most salient elements of each of the three programs.

Table 2.1. Comparison of probability based sampling programs for wetland S&T

Design Point	NWI-S&T[a]	NRI[b]	MN-S&T[c]
Program objectives	Current, scientifically valid national estimates of wetland extent and changes in extent with time and by wetland class	Regular reports on the status, condition, and extent of land, soil, water and related resources on non-Federal lands and changes in extent with time	Assess changes in wetland quantity, associate changes with causal mechanisms, and provide reports every 3 years on wetland quantity
Stratification of target area	Pre-stratification of the lower-48 states by intersecting state boundaries, 35 physiographic regions, and the coastal fringes	Pre-stratification of all 50 states using a 2 mi by 6 mi grid based on the Public Land Survey (PLS) system; post-stratification of results by state	Post-stratification of results by ecological region
Plot selection method	Plots were randomly placed in each strata based on the expected number and diversity of wetlands; supplemental plots have been added over time to increase the sample size from 3,635 to 4,682	2-stage sampling; first, primary sampling units (PSUs) were randomly selected within each PLS-based grid unit; second, sample points were randomly dropped within each PSU; not all PSUs are observed each time; selection of PSUs for observation is weighted to areas more likely to include water resources; current sample frame includes approximately 300,000 PSUs and 800,000 sample points are included in the NRI frame	Generalized random tessellation stratified (GRTS) sampling placed 4,990 plots across the state; sample size was based on the variability of wetland change acreage in NWI-S&T plots located within MN
Plot area and coverage density	4 mi^2 plots cover 0.6% of the lower-48 states by area; pre-stratification with random placement provides national spatial coverage	160 acre PSUs cover 2-6% of the PLS-based grid by area; 3 points per PSU; PLS-based grid provides national spatial coverage	1 mi^2 plots cover 6% of state by area; GRTS provides statewide spatial coverage
Data source and map production	Predominantly from the National Agricultural Imagery Program; preference is for leaf-off color infrared imagery combined with soil surveys, topographic maps, and previous NWI-S&T maps; wetland and deepwater habitats are classified according to 16 categories derived from the Cowardin classification system; upland habitat is also classified	Images collected specifically for the NRI are used with ancillary soil maps, images from past years, and NWI maps; wetlands are only measured if a sample point falls within the wetland; wetlands are classified into 13 categories derived from the Cowardin classification system	High-resolution, true-color imagery is obtained specifically for the MN-S&T and used with ancillary data including NWI maps and a digital raster graphic with elevation, drainage patterns, roads, and potential wetlands; entire 1 mi^2 area of each plot is delineated and designated as deepwater, wetlands, or upland using 13 categories derived from the Cowardin classification system

Table 2.1. Continued

Design Point	NWI-S&T[a]	NRI[b]	MN-S&T[c]
Mapping Frequency	All existing plots are revisited at variable intervals; frequency has increased with time; most recent report nominally tracked changes between 1998 and 2004 using maps created for images taken between 1996-9 and 2003-5	A supplemented, 5-panel design revisits all included PSUs and sample points either annually or every 5 years; the core sample of 40,000 PSUs is observed annually and 5 panels of 30,000 PSUs are each observed once during a complete cycle	A supplemented, 3-pannel design revisits all plots either annually or every 3 years; the core sample of 250 plots is observed annually and 3 panels of 1,580 plots are each observed once during a complete 3-year cycle
Extent and change estimation method	Wetlands are delineated to their boundaries but care is taken to avoid false changes; changes in extent between subsequent mapping efforts are classified as natural or human induced with 5 land use categories if human induced; estimates of change and extent are based on the fraction of sample area covered by a land cover class and the total area of the strata	Wetland area is categorically measured using 4 bins (<1, 1-5, 5-20, and >20 acres); gains and losses are classified due to apparent cause and land-use; missing historical data is imputed for change measurements; a complex weighting procedure is used to produce the extent and change estimates from point observations	Wetland are delineated to their boundaries but false changes are avoided; changes are classified as direct or indirect; total extent and change are estimated using the fraction of sample plots covered by a land-cover class and the area of the entire state; MN expects to achieve a confidence level of 90% plus or minus 20% if variance estimates based on the NWI-S&T plots are accurate; one cycle has been completed (3 years) and of the 250 core plots, 13 exhibited changes in wetland area

[a] (Shaw and Fredline 1956), (Hammond 1970), (Cowardin et al. 1979), (Frayer et al. 1983), (Dahl and Johnson, C. E. 1991), (Dahl 2000), (2006), (2011), (Dahl and Bergeson 2009), (Tiner 2009)
[b] (Nusser and Goebel 1997), (Goebel 1998), (NRCS 2007), (2009)
[c] (DNR 2006), (2009), (2006), (Kloiber 2010), (Kloiber et al. 2010)

Reviewed programs followed similar basic designs. First, all programs surveyed the entire target area, as opposed to using an existing map of wetlands or aquatic resources as the sample frame. This design choice means that some sampled locations lacked aquatic resources or wetlands. However, these locations were still considered “target” for the respective programs because the lack of aquatic resources was used to help develop an estimate of the mean aquatic resource density in the target region. The alternative to this approach, dropping non-target locations or otherwise constraining the sample to locations with aquatic resources, would have artificially inflated the sample mean. Second, the number or density of aquatic resources or wetlands, including zero values, at the sampled locations was used to produce estimates of aquatic resources or wetlands across the target area. Estimation procedures involved design-based sampling and inference that was dependent on the statistical designs specific to individual programs. Third, static locations were observed over time in order to monitor changes in aquatic resources or wetlands.

Each of the three programs was designed to provide estimates of extent and changes in extent of aquatic resources. The NWI-S&T program was developed to produce national estimates of wetland extent; the NRI program to produce national and state-level estimates of wetland extent and condition on non-federal land; and the MN-S&T to produce state-level estimates of wetland extent. In addition, each program incorporates the use of a probabilistic sample of the entire target area and static locations sampled over time. The NWI-S&T and MN-S&T programs produce wetland maps for square plots, while the NRI program randomly selects individual points within target areas.

2.7 Technical Needs and Challenges in California

California has a number of unique technical challenges that require special consideration through this study. Table 2.2 summarizes the challenges, the major design options available, and some of the pros and cons for each design option. An overarching technical challenge is how to select a design that balances all of the individual technical challenges facing California. For instance, a design optimized to address California's low statewide resource density may reduce program performance with respect to rare aquatic resource types. Appropriate balancing of design priorities will be raised repeatedly throughout this report.

First, California has higher ecologic and geographic heterogeneity than the other state-level program, MN-S&T. California heterogeneity is similar to the national programs, NWI-S&T and NRI, but these programs were developed for national reporting. Therefore, the impact of high ecological heterogeneity on state-level monitoring and reporting will be explored through this study in two ways. One, stratification of the state may increase precision and accuracy by allowing concentration of sample points in areas with higher variability. However, spatial stratification can increase sampling costs and decrease monitoring and reporting flexibility. Two, high ecologic and geographic heterogeneity may mean that fixed sampling locations cannot adequately capture the spatial variability in California wetlands. Unfixed or moving sampling locations allow for observation of a greater fraction of California, and therefore a greater fraction of California's aquatic resources, over time. However, fixed sampling locations can provide increased power for detecting small changes in aquatic resource extent.

Second, California has lower aquatic resource density than MN-S&T, the other state-level program. This lower density has two potential design implications. One, lower density could lead to use of stratification or a constrained sample frame, so that observation locations are concentrated in areas with expected aquatic resources. However, this design choice relies on assumptions about the distribution of aquatic resources in California, and how those distributions will persist over time. Two, low aquatic resource density may mean that a significant fraction of observed sample plots contain no aquatic resources (are "null" for aquatic resources). This null fraction is related to plot size in that larger plots are more likely to overlap with aquatic resources and smaller plots are more likely to fall between aquatic resources. This relationship also typically leads to higher variability in wetland density for small plots compared with large plots. Given the costs associated with acquiring imagery and verifying the absence of aquatic resources, choice of appropriate plot size for the California S&T program could be significantly impacted by the low density of aquatic resources in California.

Third, California has a number of rare wetland types with special ecological importance, such as vernal pools and fens. Accurate monitoring of rare populations typically requires a modified sampling design. A random sample from across the target area is unlikely to select a significant number of rare individuals, limiting statistical precision. This is particularly the case if a rare population is geographically limited. However, many of the design modifications necessary to increase power for rare populations rely on prior knowledge of the extent and distribution of the population. For example, geographic intensification relies on knowledge of the geographic regions where the rare population is concentrated. As no complete map exists for rare wetland types as of this writing, this type of probabilistic sampling design is not possible at this time.

Table 2.2. Technical Challenges and Design Options for California

Technical Challenge	Design Options	Pros	Cons
Ecologic and geographic heterogeneity	Pre-stratification	May increase precision and accuracy	May increase sampling costs and decrease monitoring and reporting flexibility
	Un-fixed (moving) sampling locations over time	Maximizes the amount of aquatic resources observed over time	May reduce power to detect small changes over time
Low statewide aquatic resource density	Pre-stratification or constrained sampling frame	Concentrates observation in locations with more aquatic resources and reduces the chance that a sample location could be “empty” of aquatic resources	Relies on current (incomplete) knowledge of the extent and distribution of aquatic resources
	Larger sample plot sizes	Larger plots are more likely to include aquatic resources and have lower variability	Larger plots are more expensive to map, reducing the possible sample size
Rare and/or geographically limited aquatic resource types	Modified sampling designs such as pre-stratification	Increases the number of rare or geographically limited aquatic resource types included in the sample	Relies on current (incomplete) knowledge of the extent and distribution of specific aquatic resource types

2.8 Objectives and Basic Design Elements for the California S&T Program

Sampling design options for the California S&T program were evaluated in terms of their ability to meet the following objectives:

- Accurate and precise measures of the extent of aquatic resource types and subtypes
- Monitoring of both current extent and changes in extent over time
- Support regional, management, or hypothesis-driven intensification and implementation

These objectives were developed and defined by the technical advisory committee assembled for this project (Appendix A). Consistent with the California WRAMP document, the design presented here covers all aquatic resource types and subtypes including deepwater, wetland, and stream habitat. The ecological connections between deepwater, wetlands, and streams means that separation of any single element would diminish the scientific basis for the monitoring program. Second, the program design focuses on accurate reporting of both extent (status) and changes in extent (trends) over time. Third, support for regional or question based intensification and implementation can mean many different things but always relies on a flexible design. For example, regional intensification could occur based on a political or ecological region for which more precise or detailed information about aquatic resource extent and distribution is needed. Similarly, question-based intensification could be driven by a desire to monitor certain wetland types or geographic areas in detail or to evaluate certain driving forces for change. Both of these modifications requires an overall program design that is flexible and can be modified as needed.

Program objectives also led to a number of recommended basic design elements, including:

- Use of the California wetland definition and the California aquatic resource classification system (Appendix B)
- Use the entire state as a sample frame and select samples from a regular (square) grid of all of California
- Produce maps of the entire contents of grid cells; all aquatic resources and upland land use for each selected grid cell should be mapped and classified

The California wetland definition and California aquatic resource classification system developed by a statewide technical team in support of the State's Wetland and Riparian Area Protection Policy (WRAPP) are recommended in order to maintain compatibility with other components of California wetland and aquatic resource monitoring and management. These two systems are currently in draft phase and under review by the policy team for the WRAPP.

Using the entire State of California as a probabilistic sample frame has both advantages and limitations; however, overall, the advantages outweigh the potential limitations of this approach. The low density of aquatic resources in California means that a focused monitoring frame, in regions where aquatic resources are already mapped, would focus mapping costs and effort on sample locations most likely to contain an abundance and diversity of aquatic resources. However, this type of focused sample frame relies on use of prior knowledge about the geographic extent of aquatic resources. Because this information is incomplete and of variable quality, we recommend the entire State of California be used as a sample frame for the S&T program. This is the approach used by existing S&T monitoring programs such as the NWI-S&T and the MN-S&T. While it is likely that a fraction of sample plots will be null for aquatic resources, these null plots still provide information about the average density, extent, and distribution of aquatic resources in California. In addition, if a plot is determined to entirely lack aquatic resources and be unlikely to develop aquatic resources, that plot will excluded from intense mapping efforts.

Finally, the entire cell area will be mapped and classified as either aquatic resources or upland. This approach is employed by the NWI-S&T and the MN-S&T and provides a number of benefits. Classification of aquatic resources and upland land use provides information about proximal impacts to aquatic resources and could be used to infer possible causes of aquatic resource extent change. In addition, classification of adjacent land use could potentially be useful for other aquatic or natural resource monitoring and management programs. Therefore, an upland classification system was developed to be used in concert with the California aquatic resources classification system (Appendix B).

2.9 Study Questions

We used simulated sampling of NWI and the NHD to investigate and develop spatial sampling design recommendations. Issues such as sample selection methodology, use of stratification, and plot size were investigated to address the following questions:

- Is probabilistic monitoring of aquatic resource extent feasible in California?
- How can the sampling design balance wetland monitoring with stream monitoring?
- Can the design adequately monitor rare or geographically limited aquatic resource types such as vernal pools or estuaries?
- Can a single plot size balance map production costs with sample variability and precision?

Similarly, the temporal sampling and monitoring design was investigated using simulations performed on modeled changes in aquatic resource extent. Results were used to determine the ideal mixture of fixed and transient sampling locations and to address the following questions:

- What monitoring design balances status (current extent) monitoring with trends (changes in extent) monitoring?
- How does the choice of analysis and estimation methodology impact change detection and estimation?
- Can one design provide accurate and precise status and trends monitoring across all aquatic resource types, temporal trends, and geographic areas?

Finally, a pilot-scale implementation was conducted to validate selection of design options and to examine, in detail, the differences between probabilistic and comprehensive approaches to aquatic resource extent monitoring. The pilot-scale implementation was designed to address the following questions:

- What is the expected variability between mapping professionals?
- How can sampling and mapping procedures be refined?
- Are there differences in estimates of wetland extent between comprehensive aquatic resource maps and maps produced for probabilistically selected sample plots?
- What statistical analysis approach provides the most accurate and precise estimate of aquatic resource extent from the sample information?

3. Spatial Sampling Design Development

3.1 Introduction and Study Questions

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 (Nusser et al. 1998, Gregoire 1999). For large geographic areas, insufficient resources have frequently prevented timely completion and updating of comprehensive aquatic resource inventories (Tiner 2009, Ståhl et al. 2010). 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.

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 (Kloiber 2010, Ståhl et al. 2010, Dahl 2011). 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.

3.2 Background & Review: 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 (Kloiber 2010, Ståhl et al. 2010, Dahl 2011). Therefore, the fraction of the target area covered by the aquatic resource of interest is easily estimated by design-based sampling and inference (Gregoire 1999, Albert et al. 2010).

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.

<u>Sample Selection Method</u>

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 aquatic features 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 (Stevens and Olsen 2003, 2004). Appendix C.2 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, 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).

3.3 Methods: Simulation and Modeling

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 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 3.1). 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 3.1).

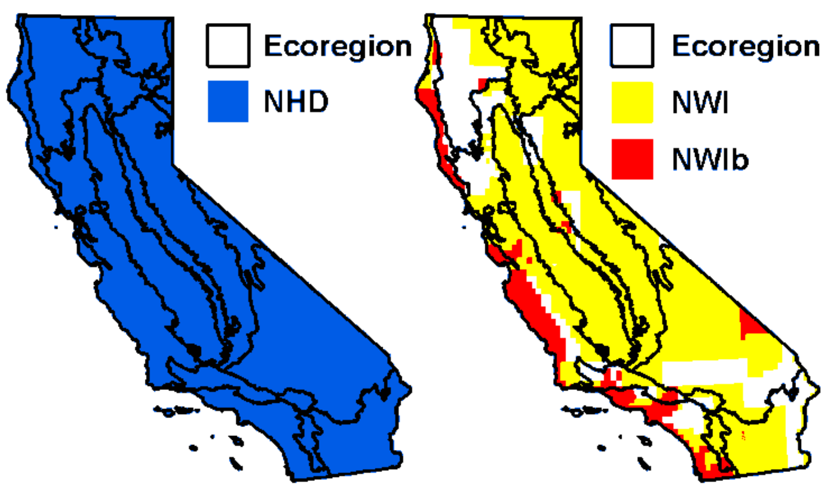


Figure 3.1. Level-III ecoregion boundaries and availability of NHD (A) and NWI (B) digital maps in California. Mapping methodology divides the NWI into maps without (NWI; yellow) and maps with (NWIb; red) buffered streamlines.

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 (Smith et al. 2003, Kozak and Zielinski 2007). 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 3.1), 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):

$$n_i = n\left(\frac{N_i\sigma_i}{\sum_{i=1}^{k} N_i\sigma_i}\right) \qquad \text{for } i \text{ in } 1, 2, \ldots, k \qquad \text{Eq. 3.1}$$

Under optimum allocation, total sample size (n) is allocated based on the population size (Ni) and population standard deviation (σi) 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).

Table 3.1. 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})[a]					
	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})						
	All	Estuarine[b]	Lacustrine	Marine[b]	Palustrine	Riverine	PUSC[d]
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

Table 3.1. Continued

Level-III Ecoregion[b]	NWIb Wetland Density (km^2 km^{-2})[c]						
	All	Estuarine[b]	Lacustrine	Marine[b]	Palustrine	Riverine	PUSC[d]
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] SO refers to Stahler stream order; “NA” is used when NHD streamlines did not have an attributed stream order.
[b] Estuarine and marine subtypes were present in all ecoregions
[c] NWIb extent did not include all ecoregions
[d] Palustrine, unconsolidated shore, seasonally flooded wetlands (Cowardin et al. 1979)

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 (Kloiber 2010, Ståhl et al. 2010, Dahl 2011).

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. 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 3.1) 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 3.1. 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 (Holland and Jain 1981, Duffy and Kahara 2011).

Simulations

We conducted all sampling simulations in R version 2.13.1. Each of the 28 sampling designs was simulated 5,000 times for each dataset, a replication count used by Miller and Ambrose (2000) to give an adequate estimate of variability in the dataset. 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). 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.

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):

$$d_{Cx} = \frac{\bar{x}_x - \mu}{s_x} \qquad \text{Eq. 3.2}$$

This relationship (d_{Cx}) is known as Cohen's d and is an alternative to a t-test for the difference of means. 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. 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):

$$p_x = \frac{s_x}{\bar{x}_x}\sqrt{n_s} \qquad \text{Eq. 3.3}$$

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):

$$\frac{s_{x1}^2}{s_{x2}^2} = \frac{\bar{x}_{x1}^2 / n_{s1}}{\bar{x}_{x2}^2 / n_{s2}} \qquad \text{Eq. 3.4}$$

The equality in equation 3.4 can be re-arranged and, using equation 3.3, 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 ($\bar{x}_f$) (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 $\bar{x}_f$ 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:

$$d_{Cf} = \frac{\bar{x}_{f1} - \bar{x}_{f2}}{s_{f1,2}} \qquad \text{Eq. 3.5}$$

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 (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:

$$E = Z_{1-\frac{\alpha}{2}} \frac{s}{\sqrt{n}} \qquad \text{Eq. 3.6}$$

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):

$$E_P = \frac{Z_{1-\frac{\alpha}{2}}\left(\frac{s_x\sqrt{n_s}}{\sqrt{n_p}}\right)}{\bar{x}_x}100\% \qquad \text{Eq. 3.7}$$

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 (Appendix D). 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 ($\bar{x}_f$):

$$e_m = \frac{n_s\left(cost_{plot}\right)}{n_s\left(1-\bar{x}_f\right)area_{plot}} \qquad \text{Eq. 3.8}$$

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.

3.4 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 3.2). 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).

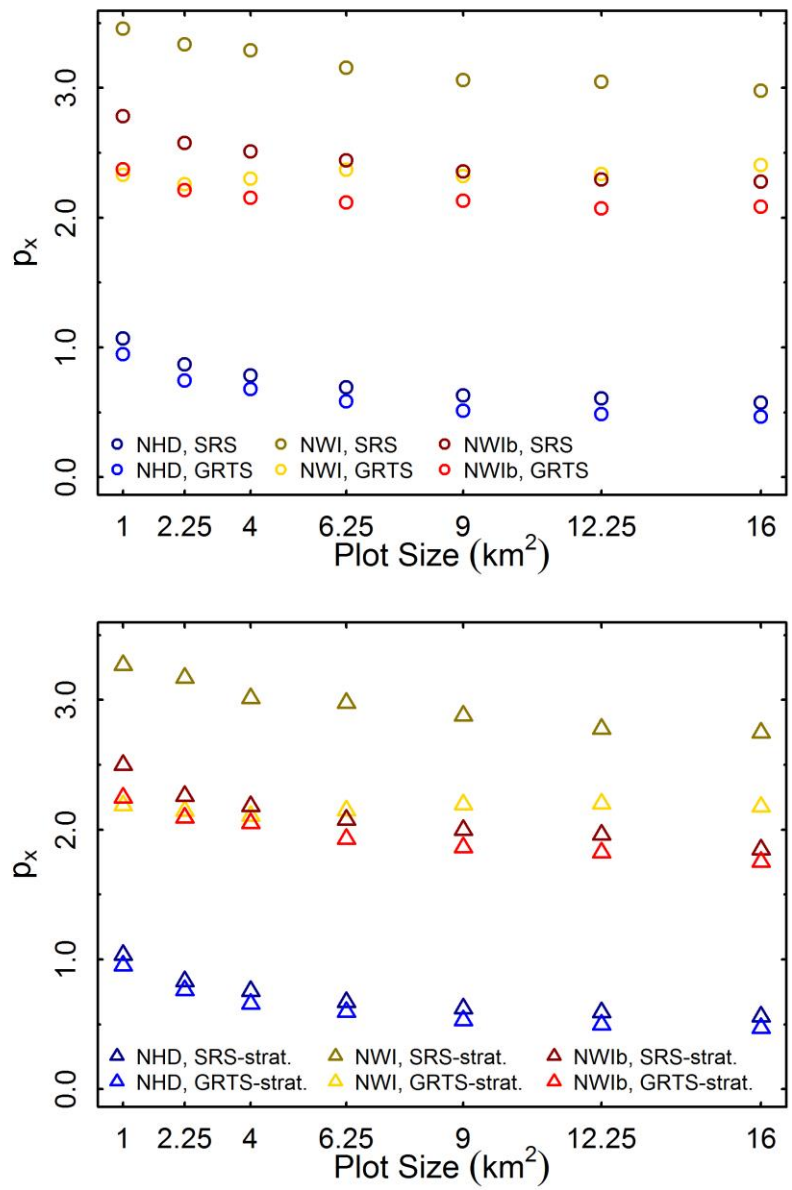


Figure 3.2. px by plot size and sampling method. px values are from the distribution of simulated sample means. Results are shown for SRS (darker shades) and GRTS (lighter shades), with (triangle) and without (circle) stratification by ecoregion, for the NHD (blue), NWI (gold), and NWIb (red).

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 3.3 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 3.3 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 3.3), 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 3.3) 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 3.3), 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 3.3) 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 3.3) 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 3.3) 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 3.3).

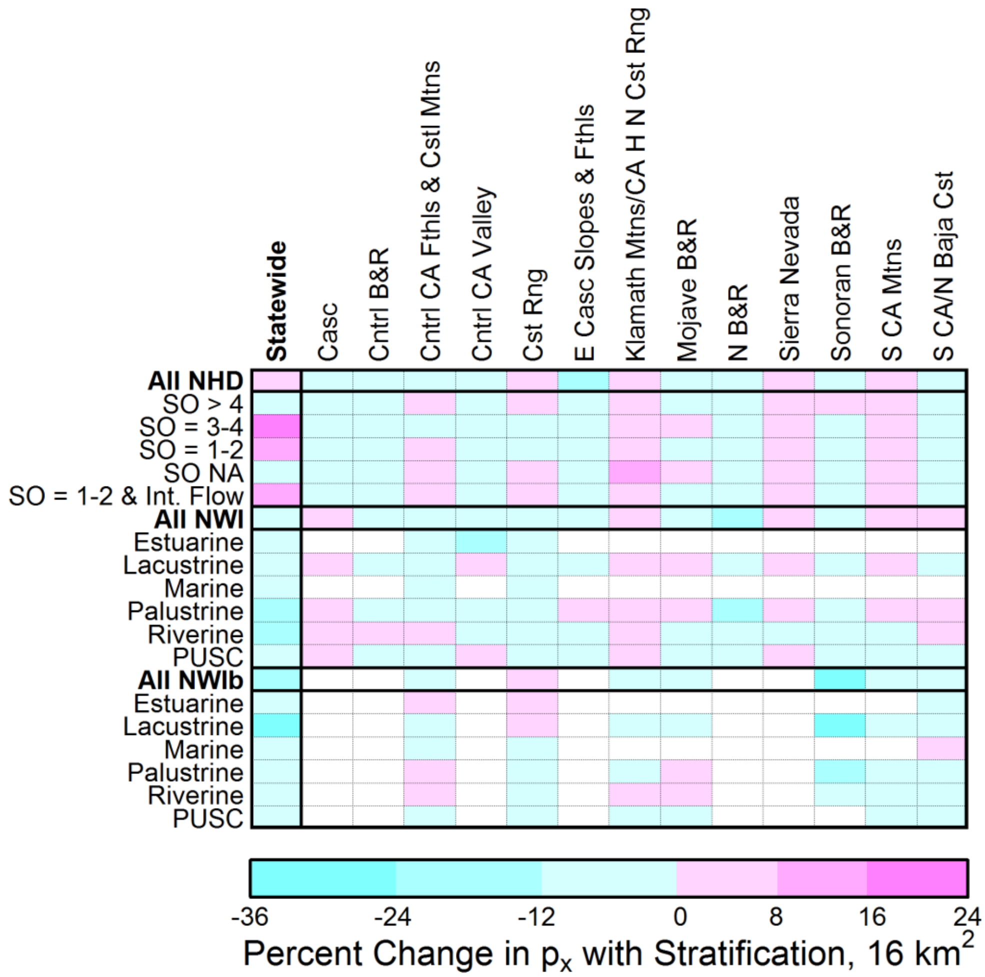


Figure 3.3. Percent change in px between unstratified and stratified sampling stratification. Shown are results for 16 km² plots, sampled using GRTS. Wetland and stream types by row and ecoregions by column. Intensity of cell shading corresponds to degree of increase (magenta) and decrease (teal) in px.

Plot Size

Smaller plot sizes produced more variable estimates of mean wetland and stream density (Figure 3.2) and were more likely to lack aquatic resources, i.e., to have higher null fractions (Figure 3.4). 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.

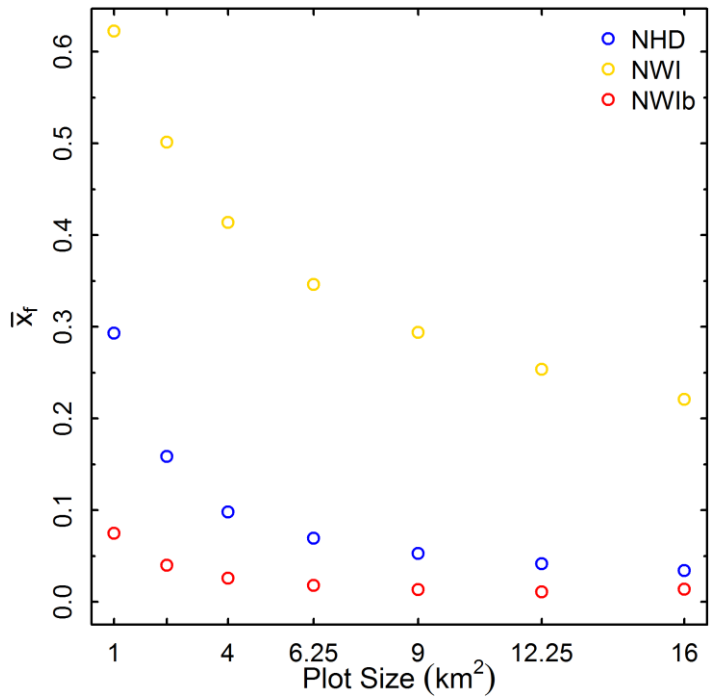


Figure 3.4. Mean fraction of sampled plots with null densities ($\bar{x}f$). Shown are results for GRTS, unstratified sampling. Points indicate the mean fraction of sampled plots lacking aquatic features for the NHD (blue), NWI (gold), and NWIb (red).

We observed two distinct relationships with predicted sample costs (Figure 3.5 and Table 3.2). Only the even plot sizes are shown in Figure 3.5 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 3.5 and Table 3.2). 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 3.2). 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.

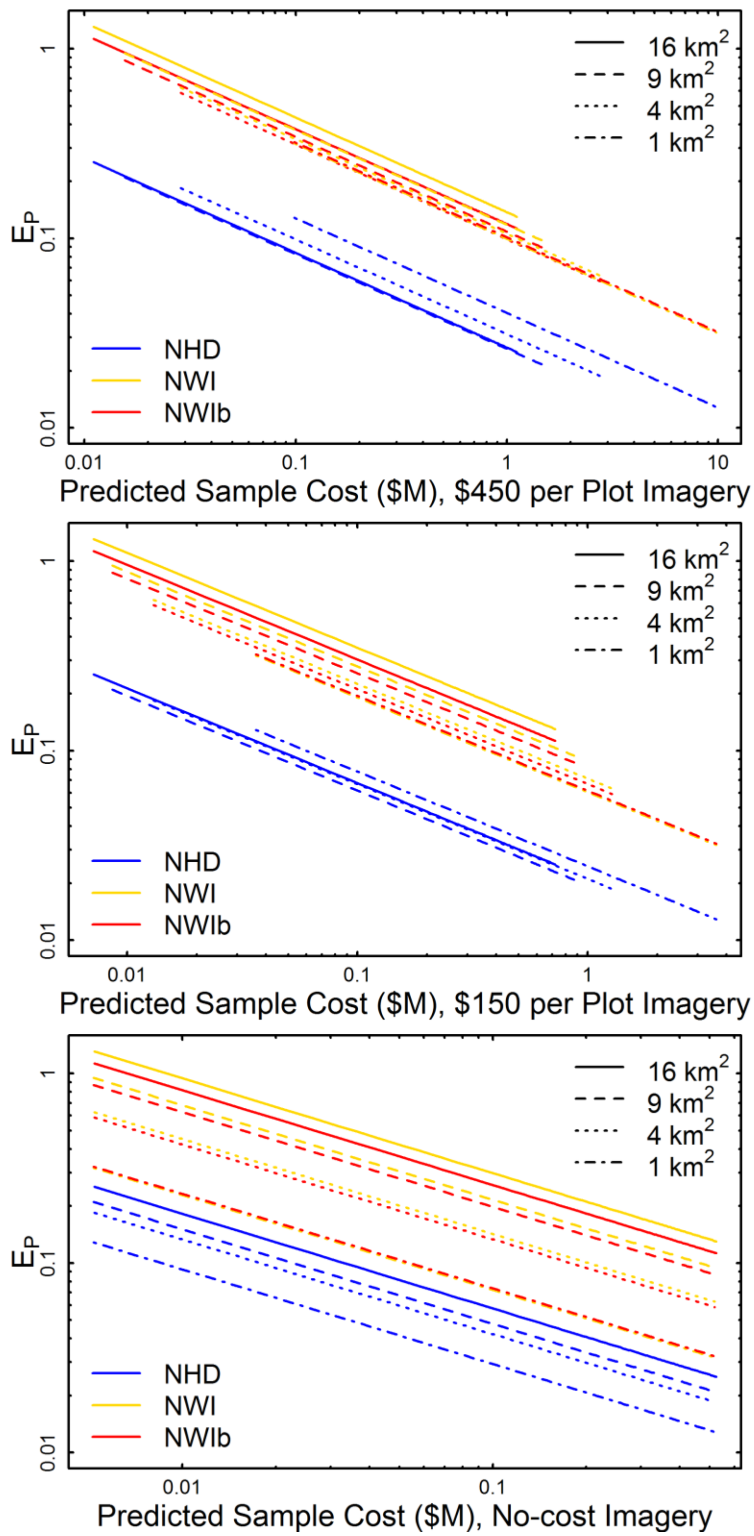


Figure 3.5. Estimated percent error by predicted sample cost. Estimated percent error was calculated as the confidence interval for the mean with α = 0.05. Lines show relationship by data type—NHD (blue), NWI (gold), and NWIb (red)—and plot size—16 km^2 (solid), 9 km^2 (dashed), 4 km^2 (dotted), and 1 km^2 (dashed and dotted). Both axes have log scales.

Table 3.2. Estimated costs for 10% predicted error and cost efficiency for producing maps (em) 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

3.5 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 2003), (2004). 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 (2006). In contrast, the NWI-S&T and the NILS use stratification and systematic sampling, respectively (Ståhl et al. 2010), (Dahl 2011). 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 (Kloiber 2010, Kloiber et al. 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 (Smith et al. 2003, Guisan et al. 2006). 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.

3.6 Conclusions and Recommendations for the California S&T 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 2003, 2004). 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 (Miller and Ambrose 2000, Jongman et al. 2006). 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.

4. Temporal Sampling Design Development

4.1 Introduction and Study Questions

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 (Kloiber 2010, Ståhl et al. 2010, Dahl 2011). 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, Theobald et al. 2007, Ståhl et al. 2010). Spatially balanced designs have been shown to reduce sample variance and increase precision, perhaps by reducing the impact of spatial autocorrelation (Stevens and Olsen 2003, 2004). The effects of spatial balance may also exceed possible increases in precision through stratification (Section 3). Finally, appropriate plot sizes allow program managers to control program costs while still meeting landscape-level information needs (Section 3). 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 (Kloiber 2010, Ståhl et al. 2010, (Dahl 2011). 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 (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, Warren 1994, Ranneby and Rovainen 1995). 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.

4.2 Methods: Simulation and Modeling

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 4.1). 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).

Figure 4.1. Location of study areas within California. Areas were selected based on availability of high-quality, contemporary wetland maps.

[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.

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. Our impact models considered incorporated places, which have legally defined boundaries, and Census-designated places, which are defined for unincorporated areas based on population were used . 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 (http://www.census.gov/geo/www/tiger/tgrshp2010/tgrshp2010.html). 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 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 4.2.

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).

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

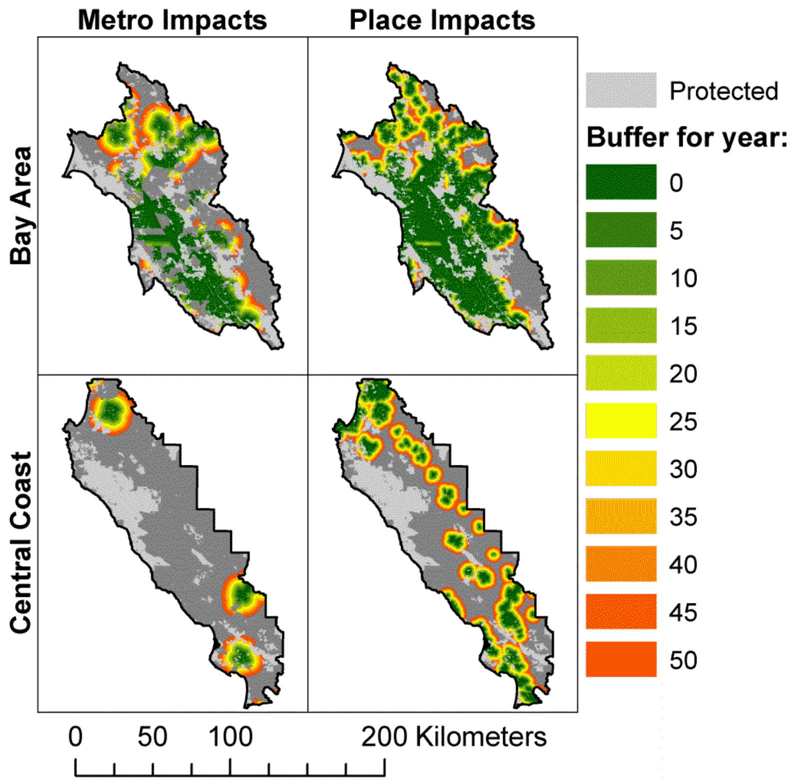


Figure 4.2. Buffers of metropolitan (left) and place (right) impacts for the Bay Area (top) and the Central Coast (bottom) for the duration of the study period.

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. 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 (Section 3). 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.

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 (Section 3). 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:

$$n = N \cdot \rho_{plots}\left(\frac{n_f}{n_f + n_m} + t_{total} \cdot \frac{n_m}{n_f + n_m}\right) \qquad \text{Eq. 4.1}$$

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 Appendix E 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 4.1):

- 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.

Table 4.1. 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 Appendix E.
[b] Change is not calculated for fixed and moving sampling locations.

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 recorded in Table 4.1 include:

- 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 4.1). 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 ($\overline{x}$) and the "true" population value (μ), relative to the distribution standard deviation (s):

$$d_C = \frac{\bar{x} - \mu}{s} \qquad \text{Eq. 4.2}$$

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. 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):

$$f_E = P\left(x < \mu + \frac{\sigma}{\sqrt{n}} \cdot Z_{0.025} \right) + P\left(x > \mu + \frac{\sigma}{\sqrt{n}} \cdot Z_{0.975} \right) \qquad \text{Eq. 4.3}$$

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:

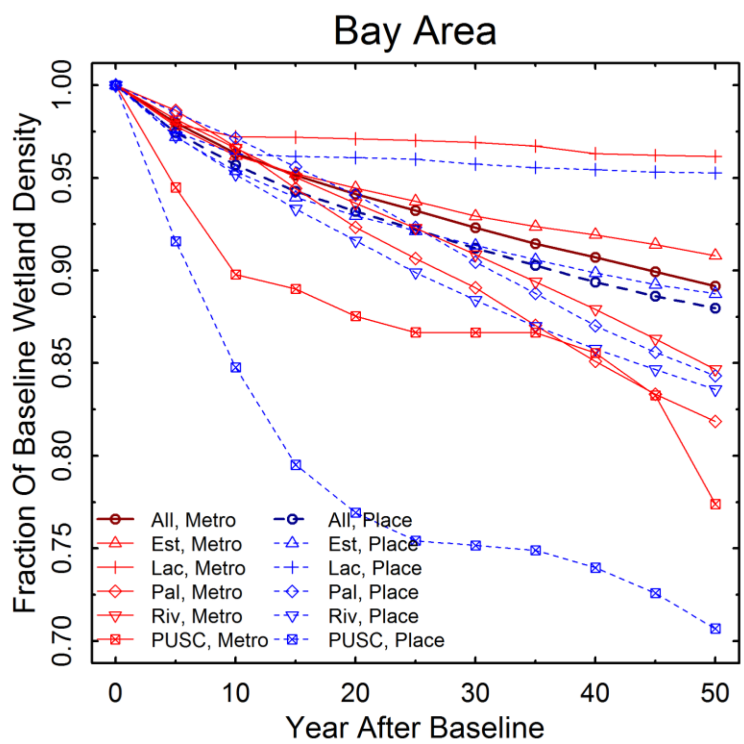
$$E_P = \left(Z_{1-\frac{\alpha}{2}} \left(\frac{s}{\bar{x}} \right) \sqrt{\frac{n}{n_e}} \right) 100\% \qquad \text{Eq. 4.4}$$

In the above, $Z_{1-\alpha/2}$ is the normal distribution value with a cumulative probability of 1 – α/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 (Section 3). 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.

4.3 Results

Model Output

The GIS model produced total wetland density losses between 7.7 and 20% over the modeled time span (Figure 4.3). 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 4.2). Consideration of the empirical variograms (Figure 4.4) 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.



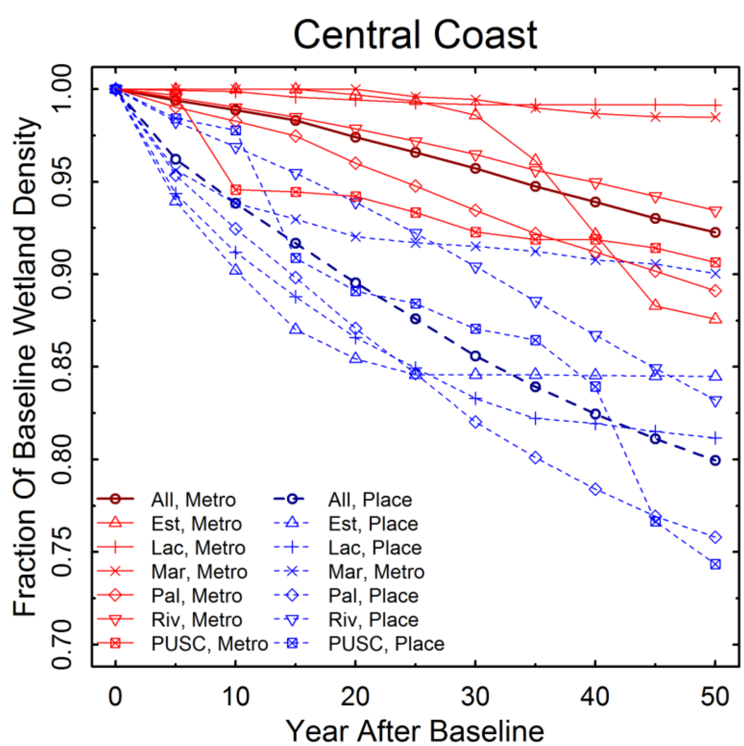


Figure 4.3. Results from modeled impact scenarios. Red solid lines reflect the metro impacts scenario while blue dotted lines reflect the place impacts scenario. Symbols indicate the resource type.

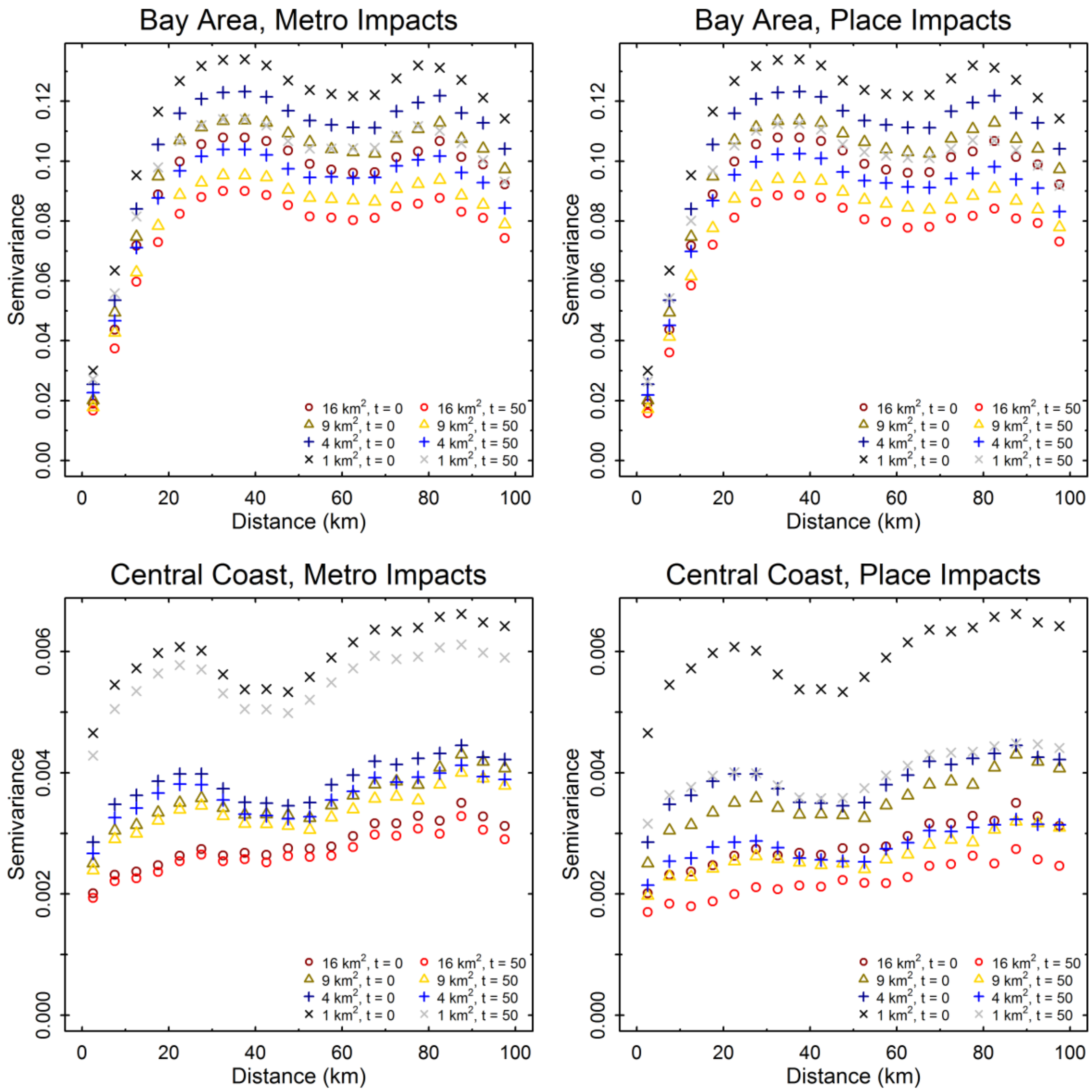


Figure 4.4. Empirical variograms for each study area and impact scenario. Variograms are shown for each plot size at both baseline (t = 0; darker shades) and at the final timepoint (t = 50; lighter shades). Shapes indicate the plot size.

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.

<u>Status</u>

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 4.5 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.

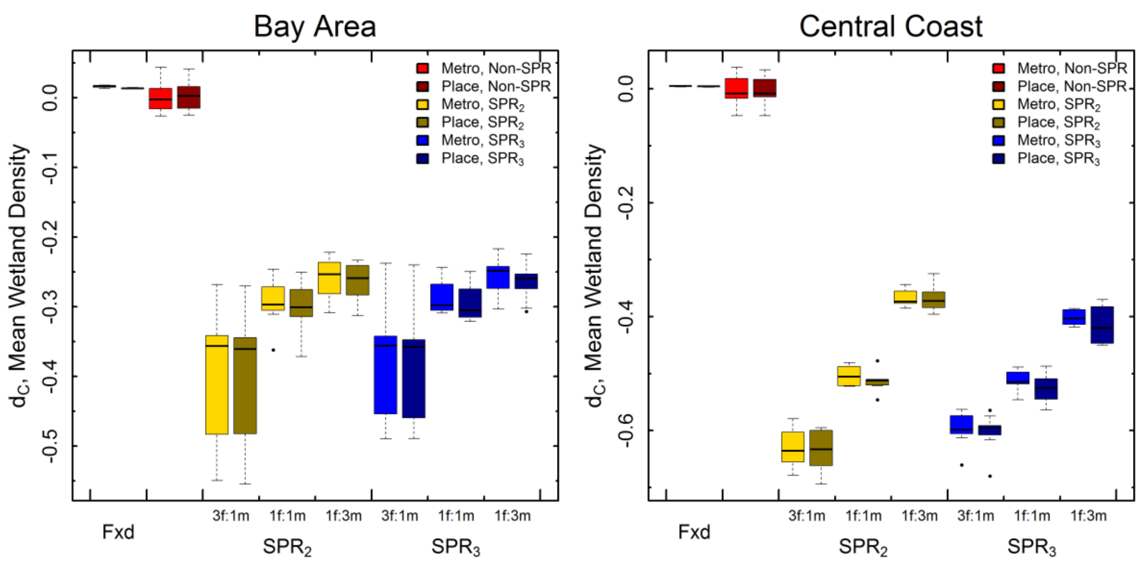


Figure 4.5. dC for mean wetland density, 16 km^2 plot size. Box and whisker plots show the distribution of values for all timepoints. Lighted shades of red, gold, and blue indicate the metro impacts scenario while darker shades indicate the place impacts scenario. Red indicates fixed or moving plot locations; gold indicates SPR based on two timepoints; and blue indicates SPR based on three timepoints.

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 4.6 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 4.3). 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.

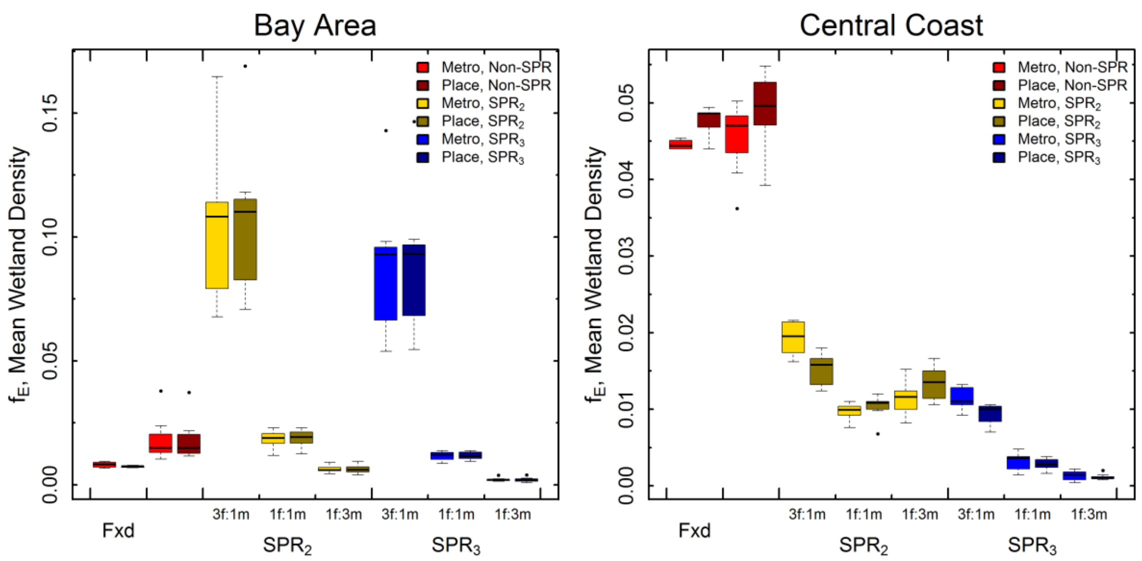


Figure 4.6. fE for mean wetland density, 16 km^2 plot size. Box and whisker plots show the distribution of values for all timepoints. Lighted shades of red, gold, and blue indicate the metro impacts scenario while darker shades indicate the place impacts scenario. Red indicates fixed or moving plot locations; gold indicates SPR based on two timepoints; and blue indicates SPR based on three timepoints.

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 4.7 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.

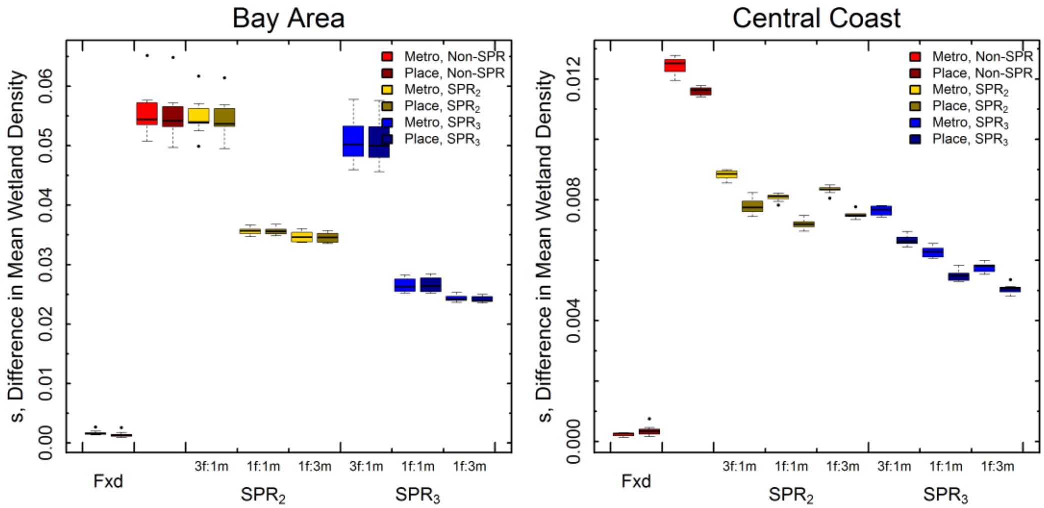


Figure 4.7. s for difference in mean wetland density, 16 km^2 plot size. Box and whisker plots show the distribution of values for all timepoints. Lighted shades of red, gold, and blue indicate the metro impacts scenario while darker shades indicate the place impacts scenario. Red indicates fixed or moving plot locations; gold indicates SPR based on two timepoints; and blue indicates SPR based on three timepoints.

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 4.8 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.

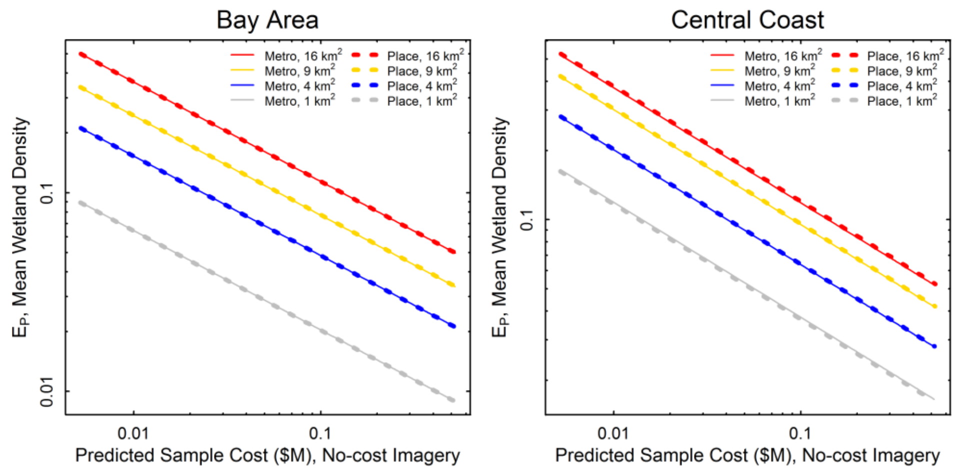


Figure 4.8. Estimated percent error, EP, as a function of predicted sample costs for different plot sizes in the Bay Area and Central Coast. Results are shown for both metro (solid lines) and place (dotted lines) impact scenarios and costs assume use of existing (i.e., no-cost) imagery. Lines indicate the average from all timepoints and colors indicate the plot size.

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 4.9 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.

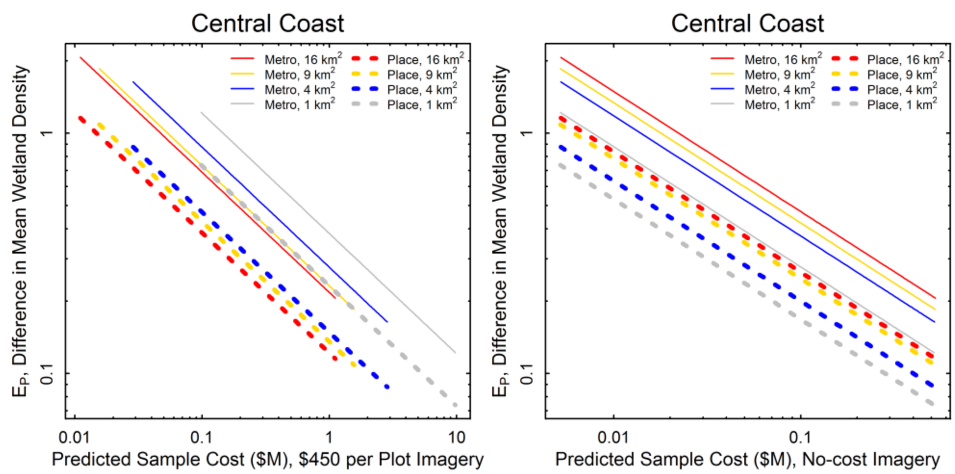


Figure 4.9. Estimated percent error, EP, as a function of predicted sample costs for different plot sizes in the Central Coast. Results are shown for both metro (solid lines) and place (dotted lines) impact scenarios. Cost estimates for use of contract imagery assumed the upper limit for imagery costs ($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.

4.4 Discussion and Implications

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 (Nusser and Goebel 1997, Kloiber 2010, Ståhl et al. 2010, Dahl 2011). 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 (Warren 1994, Scott 1998). 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).

4.5 Conclusions and Recommendation of a Temporal Sampling Design

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.

5. Pilot Testing and Evaluation

5.1 Goals and Questions

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) (Kloiber 2010, Ståhl et al. 2010, Dahl 2011). 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 (Section 3). We evaluated three potential drivers of differences between the sample estimates and the comprehensive values: i) systematic differences in methodology, ii) the likelihood of obtaining a non-representative sample, and iii) issues of inter-mapper variability. We also evaluated different plot sizes, which was a remaining design issue after the previous two studies.

5.2 Methods: Probabilistic Map Production

<u>General Approach</u>

We conducted probabilistic mapping in two regions of California (Figure 5.1). Sample plots were selected using a protocol previously optimized for California aquatic resources (Section 3). Three different groups produced sample plot maps. Inter-group quality checks to ensure consistency and increase accuracy. Once completed, we compared mean aquatic resource density between the sample and comprehensive maps. Inter-mapper variability was evaluated by having all groups produce aquatic resource maps for the same plots. 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. We utilized simulated sampling to estimate the probability of selecting a non-representative sample. Finally, we evaluated different plot sizes by examining nested sample plot sizes and by repeating sampling simulations for a number of plot sizes.

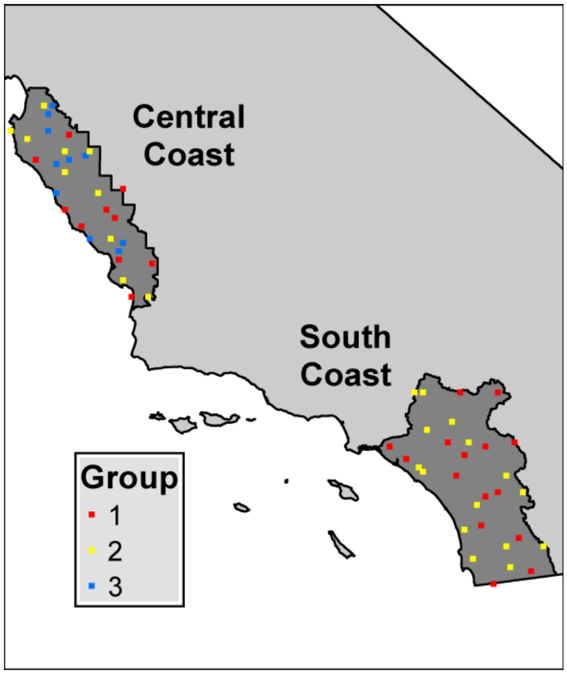


Figure 5.1. Study areas and sample plot locations by mapping group. The third group did not produce sample plot maps in the south coast region.

<u>Study Areas</u>

We selected study areas in the central and south coast of California (Figure 5.1) 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 5.1). 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.

<u>Sample Draw</u>

We selected sample plots using Generalized Random Tessellation Stratified (GRTS) sampling. First, a 4 km square (16 km^2) grid was produced in ArcGIS for each study area. Next, grid cells were converted to points, exported as a shapefile, and the GRTS sample draw performed in R using the *spsurvey* package. We selected thirty plots, without stratification, in each region. Selected plots were divided between the three mapping groups (Figure 5.1; 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 (Stevens and Olsen 2004, Larsen et al. 2008). 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.

<u>Sample Map Production</u>

Mapping groups produced maps consistent with the draft aquatic resource mapping standards for the State of California. Appendix F provides the training materials used for the current study. Appendix G provides a tentative Standard Operating Procedures (SOP) and Quality Assurance Project Plan (QAPP) for the California S&T program. All mapping groups produced maps directly in ArcGIS. Base imagery for all maps was 2005 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 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 (QAQC). 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, QAQC'ed geodatabase for the study plot. Because group 2 was so familiar with the south coast area, they did not perform QAQC on any south coast plots.

Sample Mean Estimates and Comprehensive Mean Values

We produced sample estimates of aquatic resource density by three methods: the GRTS estimator of mean and local variance, the simple mean and variance estimators, and block kriging (Appendix H). The GRTS estimator of the mean utilizes the Horvitz-Thompson estimator, making it equivalent to the simple mean (Diaz-Ramos et al. 1996). 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.

Plot Size

Plot size is an unresolved design issue for the California S&T program. Previous work (Sections 3 and 4) show that while smaller plots are more cost-effective, they are also more likely to miss aquatic resources. We assessed plot size in this study in two ways. First, the sample maps for the 16 km^2 plots were analyzed using nested 9, 4, and 1 km^2 plots. This analysis focused on the probability that each plot size contained different types of aquatic resources. Second, the sampling simulations described below were performed for all four plot sizes. This analysis focused on whether the accuracy of the 95% confidence interval is related to the plot size used.

Inter-Mapper Variability

We examined inter-mapper variability by having all three map production groups map ten additional plots (five in each study area). By mapping the same locations, we could calculate paired differences between each group. Plots for inter-mapper variability were based on a 4 km^2 grid and were selected using the GRTS approach defined above. A 4 km^2 grid was for this application because the plot size analysis (described below) suggested that the 4 km^2 plot size is the best choice for the California S&T program. Maps for the inter-mapper analysis were produced with the same mapping approach described above with two major modifications. First, no external QA/QC was performed as we were isolating the differences between mapping groups. Second, a 2.5 m stream buffer was used.

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 on estimation of wetland extent. These issues should be explored further in subsequent phases of this work.

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 QAQC 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 but modified for California aquatic resources (Brinson 1993). 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 Appendix B.

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 5.1) 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.

Table 5.1. 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

<u>Likelihood of a Non-Representative Sample</u>

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, and whether that relationship was dependent on plot size. To answer these 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 “populations” were 16, 9, 4, and 1 km^2 grids. The 16 km^2 grid was 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 ten different sample sizes. We selected a repetition count of 5,000 based on prior experience with simulated sampling (Miller and Ambrose 2000). The ten sample sizes ranged from 10 to 100 for the 16 km^2 grid, 20 to 200 for the 9 km^2 grid, 30 to 300 for the 4 km^2 grid, and 40 to 400 for the 1 km^2 grid. 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.

This analysis also considered the cost estimates described in earlier sections. Cost estimates are based on two components: map production and image acquisition. Map production costs were assumed to be 25 USD km^{-2} and do not change based on plot size. This estimate is based on the experience of the mapping groups involved in this study. In this analysis, we considered only the use of existing, no-cost imagery.

5.3 Results

The results section contains five 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 evaluate the sample maps for smaller, nested plot sizes. Third, we quantified inter-mapper variability between the three map producers. Fourth, 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. Finally, 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.

<u>Probabilistic Estimates vs. Comprehensive Values</u>

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 5.2). The estimation method used did not influence accuracy but did affect estimated variance. Figure 5.3 provides ordinary kriging interpolations. 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.

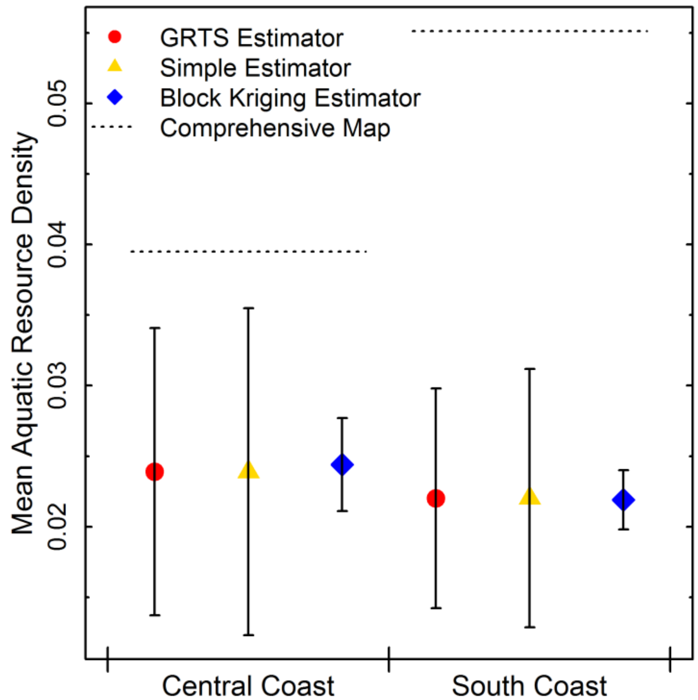


Figure 5.2. Mean aquatic resource density by study area. Red circles indicate the estimated mean of sample maps, plus or minus the 95% confidence interval, based on the GRTS variance estimator. Yellow triangles are based on the simple mean and variance estimator and blue diamonds are based on block kriging. Dotted lines are the mean density from the comprehensive map.

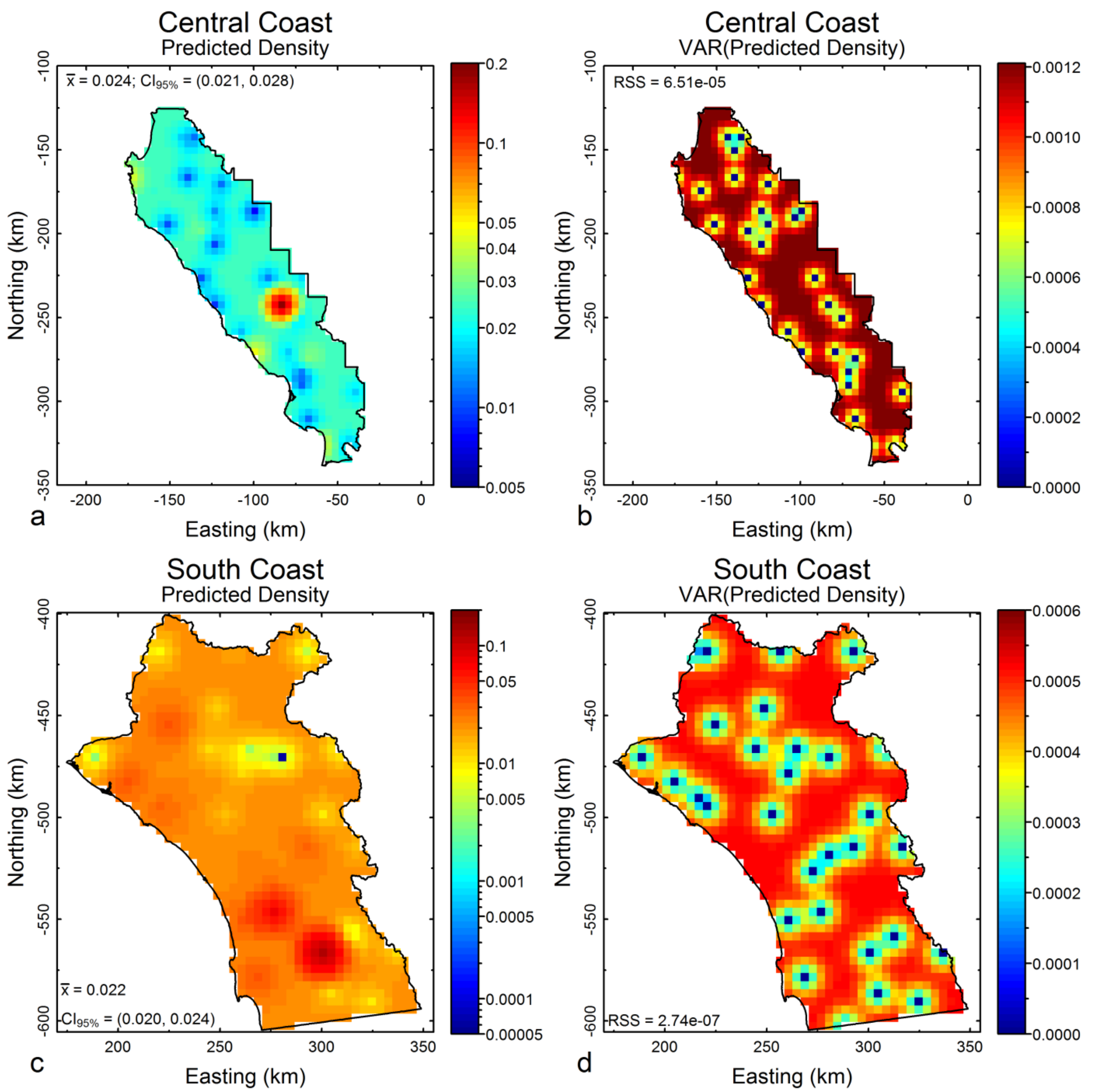


Figure 5.3. Interpolated density (a, c) and variance (b, d) for the central coast (a, b) and south coast (c, d). Ordinary kriging was used with a spherical semivariogram model. Inset means and 95% confidence intervals (a, c) are the block kriging results for each area. Inset residual sums of squares (b, d) are from the ordinary kriging results.

Nested Plot Sizes

In the south coast, all plots contained aquatic resources, regardless of size. In the central coast, all 16 km^2 plots contained aquatic resources but one of the 9 and 4 km^2 plots and two of the 1 km^2 plots lacked aquatic resources (Table 5.2). The number of sample plots declined with subdivision if plots falling on the study area boundary no longer overlapped with the study area. This reduced study plots from 30 to 26 for the central coast and from 30 to 28 for the South Coast. Riverine wetlands were the most common and were present in at least 93% of sample plots for each region, regardless of plot size. For other resource types, substantial decreases in occurrence were seen between the 9 and 4 km^2 or the 4 and 1 km^2 plot sizes. For example, depression and slope wetlands were the second most common resource type, occurring in 67-70% of central coast and south coast 9 and 16 km^2 plots. Occurrence then fell to 52% of 4 km^2 plots and 19% of 1 km^2 plots in the central coast and 60% and 29% in the south coast. Slope resources, the third most common type, followed a similar pattern. In the central coast, 50% of 16 km^2 plots contained slope resources; this fell to 41% of 9 km^2 plots, 30% of 4 km^2 plots, and 15% of 1 km^2 plots. Occurrence rates for the south coast were 37, 30, 20, and 11% of 16, 9, 4, and 1 km^2 plots. Lacustrine aquatic resources were relatively rare. One central coast plot contained a lacustrine resource, but only for the 4 km^2 or larger plots. Three of the 16 and 9 km^2 south coast plots contained a lacustrine resource but this dropped to two of the 4 and 1 km^2 plots. No south coast plots fell on the coastal boundary of the study area so estuarine and marine occurrence could not be assessed for the south coast. In the central coast, two 16 km^2 plots contained estuarine resources; this dropped to one plot for the 9 and 4 km^2 sizes and no plots for the 1 km^2 size. Similarly, four 16 km^2 plots contained marine resources and this dropped to three 9 km^2 plots and no 4 or 1 km^2 plots.

Table 5.2. Number of sample plots containing resource of interest.

Resource Type	**Central Coast**				**South Coast**			
	Plot Area (km^2)				*Plot Area (km^2)*			
	4	*3*	*2*	*1*	*4*	*3*	*2*	*1*
N	30	29	27	26	30	30	30	28
Aquatic Resources	30	29	27	26	30	29	29	26
Depression	20	20	14	5	21	21	18	8
Estuarine	2	1	1	0	0	0	0	0
Lacustrine	1	1	1	0	3	3	2	2
Marine	4	3	0	0	0	0	0	0
Riverine	29	28	27	26	30	29	29	26
Slope	15	12	8	4	11	9	6	3

Inter-Mapper Variability

Differences between mapping groups averaged 29% for total aquatic resource density. Triangle plot analysis and correlation was also moderately good for depressions and riverine resources (Figure 5.4 and Table 5.3). Agreement was not as good for slopes but this may reflect the rarity of slopes in the sampled plots.

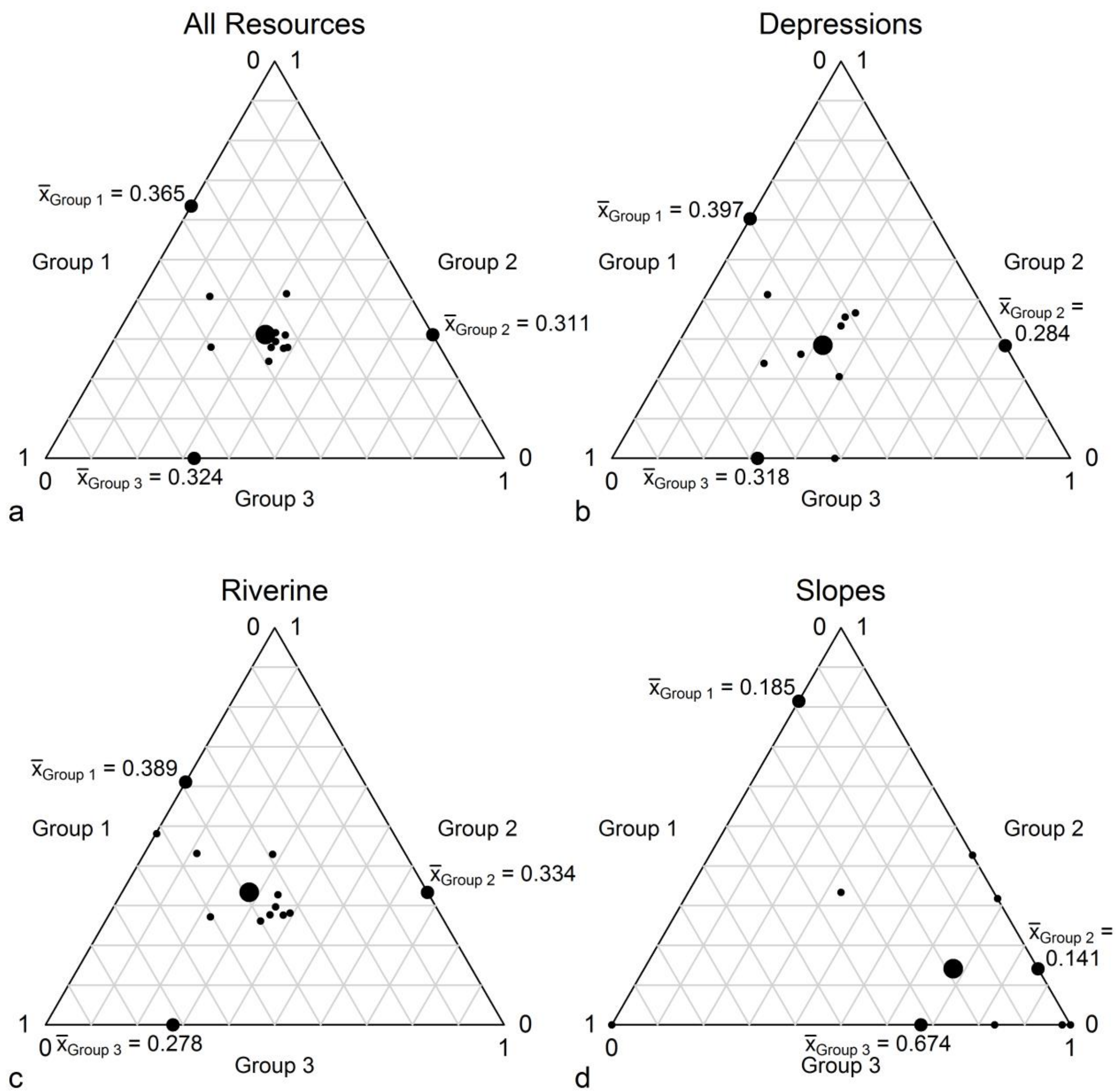


Figure 5.4. Triangle plots comparing mapped resource density between the three mapping groups. The plots converted each density to a value between 0 and 1 and constrained the sum of the three values to 1. Therefore, if all three groups mapped the same density, each group would be assigned the value of 1/3 for that plot. The smallest points represent each individual plot in the inter-mapper exercise. The medium size points on the outside edge of teach plot represent the average value for each group. The largest points in the interior of each plot represent the triangulation of the three averages. Each plot represents either (a) all aquatic resources or individual types: depressions (b), riverine (c), and slopes (d).

Table 5.3. Correlation matrices between the three mapping groups for each resource type.

All Resources				Depressions			
	Group 1	*Group 2*	*Group 3*		*Group 1*	*Group 2*	*Group 3*
Group 1	1.000	0.758	0.937	*Group 1*	1.000	0.909	0.911
Group 2	0.758	1.000	0.661	*Group 2*	0.909	1.000	0.763
Group 3	0.937	0.661	1.000	*Group 3*	0.911	0.763	1.000
Riverine				**Slopes**			
	Group 1	Group 2	Group 3		*Group 1*	*Group 2*	*Group 3*
Group 1	1.000	0.403	0.941	*Group 1*	1.000	0.209	-0.155
Group 2	0.403	1.000	0.261	*Group 2*	0.209	1.000	-0.004
Group 3	0.941	0.261	1.000	*Group 3*	-0.155	-0.004	1.000

Systematic Methodological Differences

Assumed Stream Width

The assumed stream buffer width had a significant impact on sample estimates of mean aquatic resource density (Figure 5.5a). 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 products. 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 5.5c). However, in the central coast, plot-by-plot differences changed from an average of 14% lower to 71% higher (Figure 5.5b). This suggests that additional methodological differences may exist between the sample and comprehensive maps in the central coast.

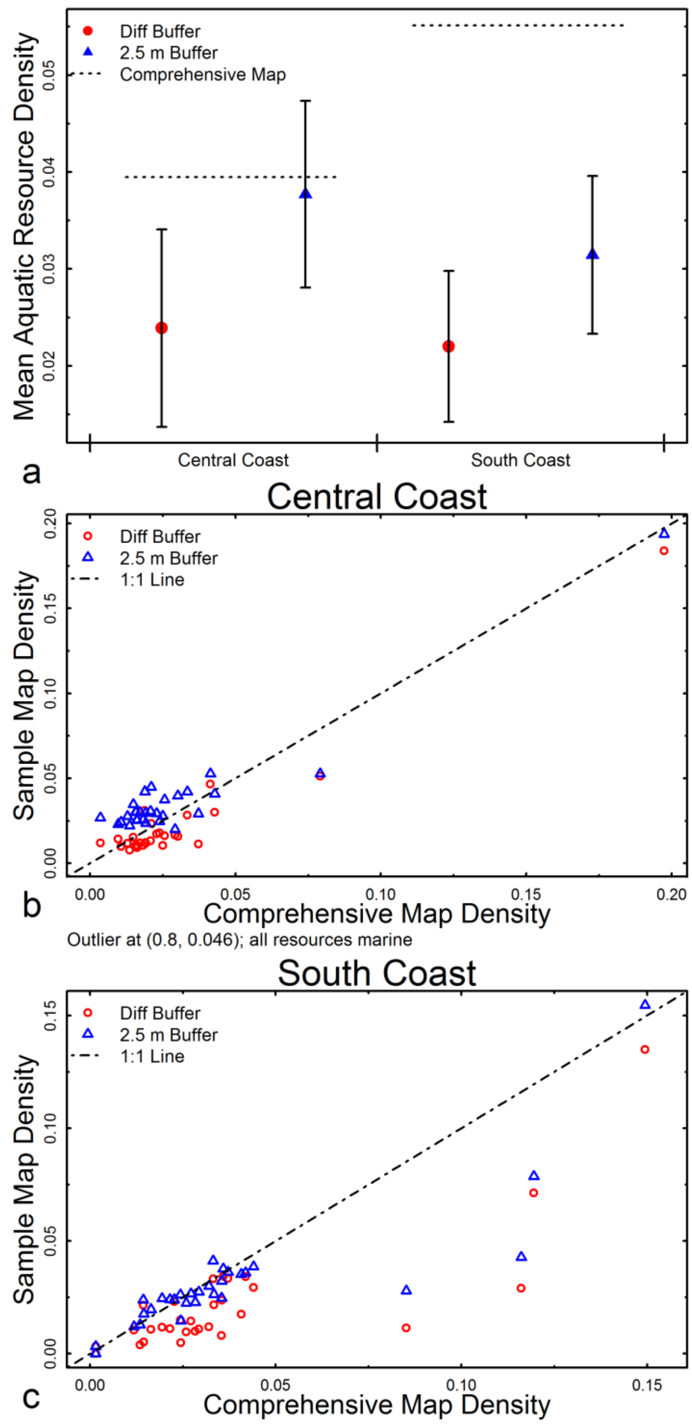


Figure 5.5. Comprehensive versus sample map density by study area. Red circles indicate the estimated mean based on a differential streamline buffer width, plus or minus the 95% confidence interval from the GRTS variance estimator (a). Blue triangles are based on a 2.5 m buffer. Dotted lines are the mean density from the comprehensive map. 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.

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 5.6a). The average plot-by-plot difference, calculated by considering just the maps within the plot boundaries, was 78% in the central coast (Figure 5.6b). 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.

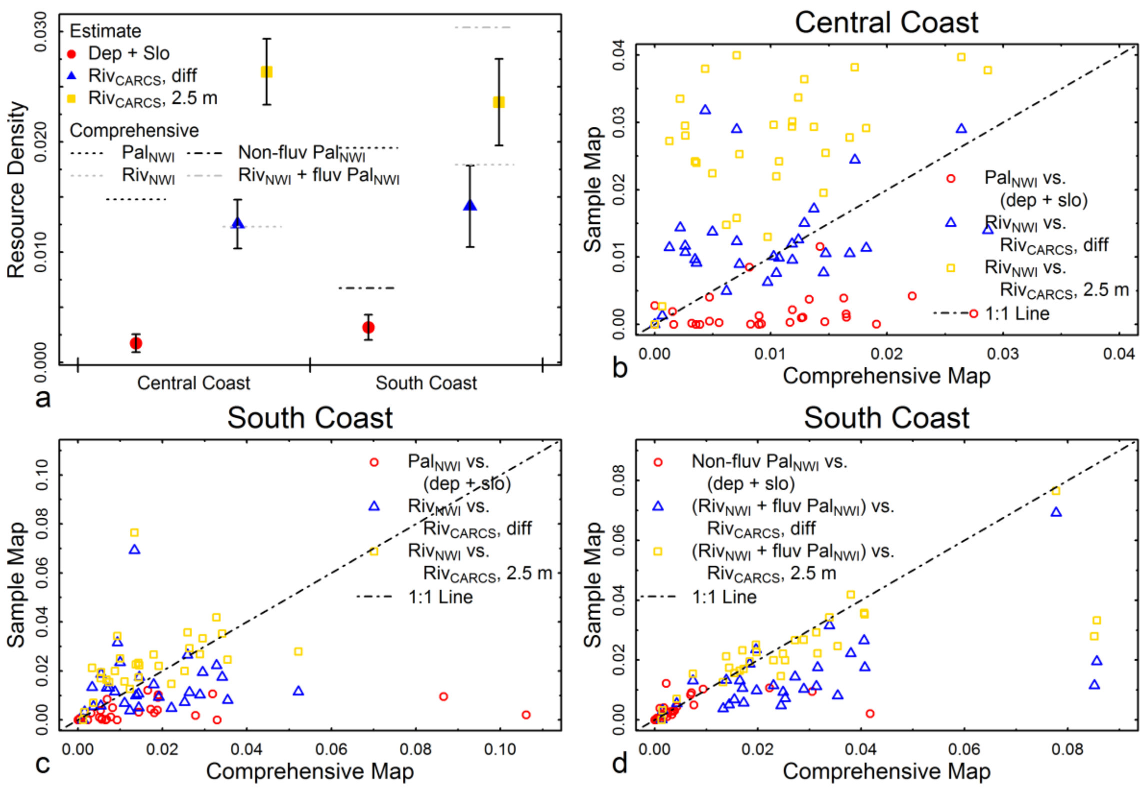


Figure 5.6. Comprehensive versus sample map density by study area and resource type. (a) Red circles indicate the estimated mean density of depressions and slopes. Blue triangles indicate mean riverineCARCS density based on a differential streamline buffer width and yellow squares indicate are based on a 2.5 m buffer. Intervals represent the 95% confidence interval from the GRTS variance estimator. The black dotted line provides the mean palustrineNWI density from the comprehensive map while the black dashed-dotted is the mean non-fluvial palustrineNWI density. The grey dotted line is the mean riverineNWI density from the comprehensive map and the grey dashed-dotted is the mean riverineNWI plus fluvial palustrineNWI 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 palustrineNWI against the sum of depressions and slopes; blue triangles compare riverineNWI against riverineCARCS based on a differential buffer; and yellow squares compare riverineNWI against riverineCARCS based on a 2.5 m buffer. (d) Red circles compare non-fluvial palustrineNWI against the sum of depressions and slopes; blue triangles compare riverineNWI and fluvial palustrineNWI against riverineCARCS based on a differential buffer; and yellow squares compare riverineNWI and fluvial palustrineNWI against riverineCARCS based on a 2.5 m buffer.

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 5.6a). However, plot-by-plot differences decreased from an average of 62% to 0.6% (Figures 5.6c, and 5.6d). 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 5.6a). 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 5.6a). In addition, the average plot-by-plot differences increased from 120% higher to 270% higher (Figure 5.6b). 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 5.6a and 5.6d). 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 5.6c and 5.6d). Agreement was also reduced for comparisons to riverine resources alone in the comprehensive map reduced agreement (Figure 5.6c). 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 5.7a and 5.7b). 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.

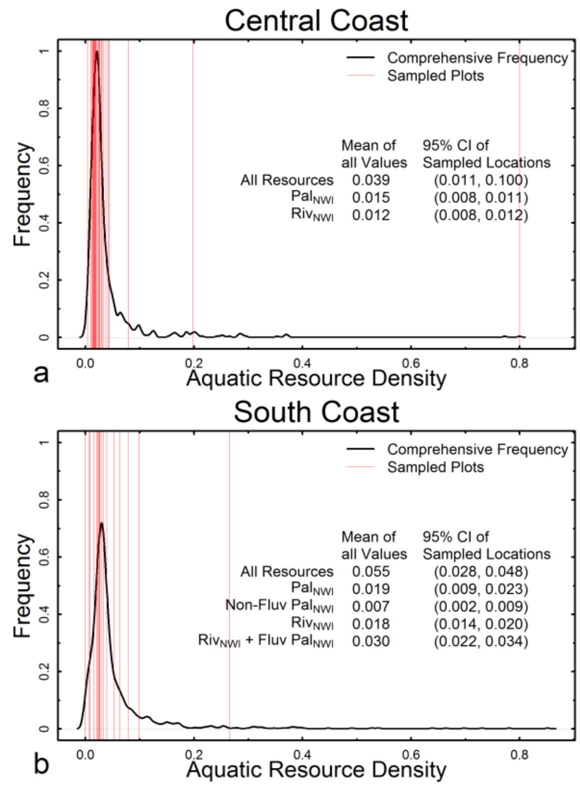


Figure 5.7. Distribution of comprehensive aquatic resource density values and accuracy. (a-b) Solid black line indicates the relative frequency of aquatic resource densities in the comprehensive map for the central coast (a) and the south coast (b). Red lines indicate the plots included in the random 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.

The comparisons in Figure 5.7a and 5.7b 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 5.7b).

Figure 5.8 provides the results of the simulated sampling. Considering a sample size of 30 and a 16 km^2 plot size (the conditions of this study), the sample 95% confidence interval contained the true value 73-80% of the time (i.e., the "empirical" confidence interval is between 73 and 80%). For the largest simulated sample size (100), the interval contained the true value 87-89% of the time. This result suggests that a larger sample size will improve the accuracy of sample estimates. This conclusion 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.

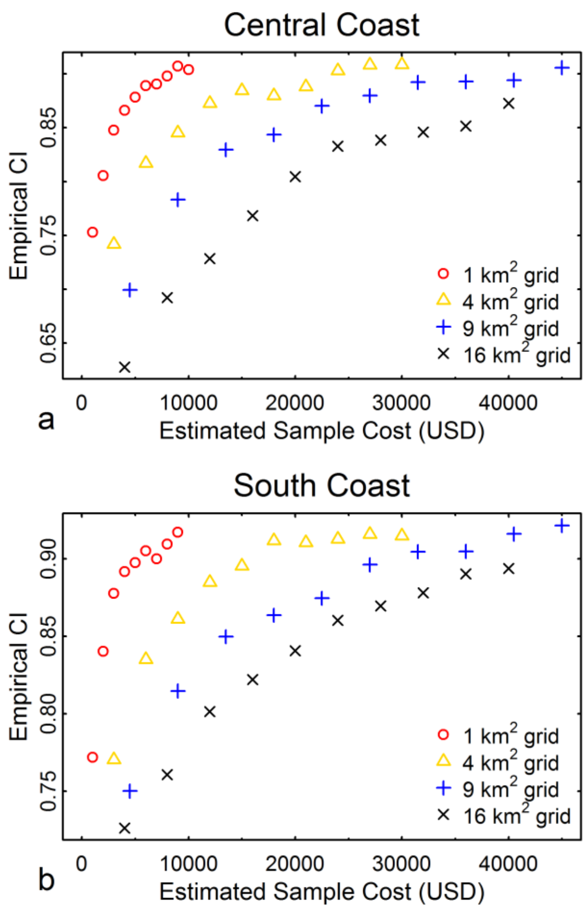


Figure 5.8. Points indicate the results of 5,000 simulations of estimates sample cost (x-axis) for the central coast (a) and the south coast (b). The y-axis is the fraction of the simulations where the calculated 95% confidence interval contained the comprehensive mean. Red circles are for a 1 km^2 grid, yellow triangles for a 4, blue crosses for a 9, and black x's for a 16.

Similar patterns were observed for other plot sizes (Figure 5.8). Comparing performance by plot size also highlighted the cost-effectiveness of smaller plot sizes. For example, a 1 km^2 plot sizes achieved a 90% empirical confidence interval at a predicted sample cost of approximately 8,000 USD in the central coast and 5,000 USD in the south coast. In contrast, a 4 km^2 plot achieved the same empirical confidence interval at a predicted cost of approximately 24,000 USD in the central coast and 15,000 USD in the south coast.

5.4 Discussion and Implications

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. Finally, inter-mapper variability can account for up to 30% difference between plots, thereby increasing the likelihood of differences between sample and comprehensive maps. 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 applying strict training and quality control measures to minimize inter-mapper variability.

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 (Smith et al. 1998, Schmid et al. 2005, Jana et al. 2007). 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 (Appendix B). 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 (Cowardin et al. 1979, Brinson 1993). 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.

<u>Sample Size, Accuracy, and Precision</u>

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).

<u>Sample Size, Accuracy, and Precision</u>

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 (Hirano et al. 2003, Corcoran et al. 2011). Although overall inter-mapper variability was 29%, results from our study for plot-by-plot differences in the south coast suggest that inter-mapper variability map be as low as 5%. 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.

Plot Size

Results in this study for plot size continue to highlight the tradeoffs involved with selecting an appropriate plot size. Considering simply the incidence of different aquatic resource types strongly supports larger plot sizes while consideration of cost-effectiveness supports a smaller plot size. Balancing these competing results depends on the objectives of the particular program. Our opinion is that a 4 km^2 plot size provides the best balance of factors for the California S&T program. For example, the 1 km^2 plot size is more cost effective but provides significantly less diversity of observed resources. In addition, while the 9 and 16 km^2 plot sizes are expected to include a greater number and diversity of aquatic resources, these plot sizes have significant negative implications for overall cost-effectiveness and accuracy. Therefore, we believe that the 4 km^2 plot size is an adequate balance of the factors examined.

6. Design Recommendations for the California Status and Trends Program

6.1 Design Elements

The recommended design provided here was developed to satisfy the following programmatic objectives, developed based on the WRAMP strategy document and with the input from the project TAC (Appendix A):

- Report extent (status) and changes in extent (trends) at regular intervals.
- Include estimates for all surface aquatic resources including wetlands, streams, and deepwater habitat.
- Support regional intensification through design flexibility.

The following design elements are recommended to meet the objectives listed above:

- Utilize probabilistic sampling and analysis methods.
    - Use equal-probability GRTS sampling without stratification (see sections 6.2 and 6.3 for recommendations related to implementing the master sample).
    - Select a square plot size based on consideration and balancing of estimated precision, mapped information, and predicted program costs (see Section 6.4).
- Use the entire state as a sample frame, not just areas with previously mapped aquatic resources.
    - Map and classify the entire contents of sample plots, both aquatic resources and upland land uses, to provide information about spatially proximate upland influences on aquatic resources.
    - Utilize the proposed California wetland definition and aquatic resource classification system (Appendix B) (Technical Advisory Team 2009a), supplemented by the Cowardin et al. (1979) system to provide compatibility with other data sources.
    - Use standard operating procedures (SOP) and a quality assurance project plan (QAPP) (Appendix G) to ensure consistence and comparability between study plot maps and appropriate application of the statistical design. Revise the SOP and QAPP as appropriate, based on advances in mapping technology or program objectives.
- Repeat mapping over time at fixed intervals, ensuring adequate monitoring for changes in extent and distribution and allowing the program costs to be spread across the multi-year cycle.
    - Maintain fixed locations for repeated observations over time and monitoring for changes in aquatic resource extent.
    - Produce estimates of S&T every five years, to ensure results reflect relatively recent conditions (see section 6.3 for recommendations related to map stewardship and section 6.4 for reporting and expected output). Periodic remapping ensures adequate monitoring and distributes program costs over multiple years. Assessing change in resource extent will improve evaluation of ongoing monitoring, permitting, and remediation efforts.

6.2 GRTS Master Sample

One of our principal design recommendations is use of an unstratified, GRTS master sample. This recommendation is strongly supported by the modeling and simulation work comparing GRTS sampling to use of stratification or simple random sampling. In addition, the GRTS sample provides several additional benefits for use as the California S&T program is implemented.

A master sample is a list of sampling locations that provides enough locations for the designed program, plus an "oversample" that can be used to provide additional locations as needed. Additional locations could be required for several reasons, including:

- Expansion of the statewide sample size.
- Regional intensification to meet regional information needs.
- Targeted intensification for hypothesis testing or stochastic event monitoring (e.g., following floods, fires, pest infestation).

Addition of sample plots through a master sample is preferred to performing a new sample draw for a number of reasons. First, the master sample will provide unique locations that have not previously been sampled or observed. Second, under GRTS sampling theory, as long as locations are added in order from the master sample, the entire sample maintains good spatial balance. This applies even if the master sample is filtered to provide only locations in the region of interest or with only the properties of interest. Good spatial balance is considered to reduce the effects of spatial autocorrelation and to increase the precision of the sample. Third, if properly managed, the GRTS master sample can be easily weighted and re-weighted to produce accurate statewide and regional estimates using a common set of sampling locations.

We anticipate that the addition of sample plots to increase the statewide sample size could be conducted for three primary reasons. First, additional program funding could allow the State to add additional sample locations. Second, per plot costs could decrease through advances in imaging technology or map production methods such as fully or partially automated mapping and classification. In addition, experience with the NWI-S&T and MN-S&T indicates that labor costs associated with re-mapping plots during subsequent time intervals may be less than the costs of mapping a new plot. Third, some sample plots may be considered "immune" to changes in aquatic resource extent over time. Examples of this could include a sample plot completely within a wetland or deepwater habitat (such as the San Francisco Bay), or a completely developed sample plot (such as certain industrial areas of Los Angeles County). These plots cannot be dropped from the sample as this would bias sample calculations. However, these plots would not necessarily require imaging and remapping at each timepoint. Instead, plots could be subject to cursory monitoring through no-cost imagery sources, such as Google Earth or NAIP, to verify that they do not require re-imaging and re-mapping. Therefore, the associated cost savings could allow addition of new sample plots.

We view the two other anticipated conditions for adding samples plots, regional intensification or targeted intensification, as supplements to the core objectives of the California S&T program. The additional sample plots can be used to increase the precision of the statewide sample. However, adding plots in certain regions or to meet certain criteria effectively creates statistical strata and prioritizes the regional or programmatic objectives above the objective of producing a statewide estimate to status or trends. Therefore, we recommend that plots should only be added for these reasons if financed primarily by the regions or programs requesting the intensification.

6.3 Master Sample, Map, and Data Management

This section provides recommendations related to management of the GRTS master sample, current and archive maps, and program data. Ideally, all three functions could be managed by a single entity to ensure that all statistical and analytic procedures are followed, consistent with the project SOP and QAPP (Appendix G).

First, a managing entity should be identified and trained in the appropriate selection and maintenance of a GRTS master sample to ensure that all procedures are followed. This entity should be responsible for defining the location of all program sample location, including those used for a regional intensification. Realization of the master sample benefits requires careful management of the master sample list. Appropriate management is necessary to ensure that plots are utilized consistent with GRTS theory and design specifications, and to ensure that plots are weighted appropriately to produce accurate statewide and regional estimates.

Second and closely related, the entity responsible for managing the master sample should provide the technical expertise necessary for producing program estimates from sample plot data. Appropriate analysis of the S&T maps requires experience with GIS and statistical software, GRTS sampling methodology, and access to the GRTS master sample. Therefore, relying on one entity for managing the sample list and producing the estimates would help ensure the statistical reliability of program estimates.

Third, a single entity should be responsible for managing and maintaining current and archive copies of maps produced for the S&T program. Management by a single entity would minimize duplication of data and reduce complications associated with managing different versions of the S&T maps. In addition, the State should consider whether specific plot location information and maps should be made freely available or if they should only be available, by request, to specific parties. Because the California S&T program is designed to monitor the extent and distribution of aquatic resources, there is a concern that free dissemination of the plot locations could lead to unintended modification (e.g., creation, restoration, etc.) of aquatic resources within those plots by third parties. Both the MN-S&T and the NWI-S&T program withhold the geographic location of individual plots except in justified cases. For example, justification for release of locations or maps could include activities such as map production for the statewide program or a regional intensification, change analysis between current and previous timepoints, and use of maps as a sample frame for ambient field condition assessments.

6.4 Plot Size, Precision, Mapped Information, and Program Costs

An appropriate plot size for the California S&T program could not be determined solely from the analysis performed to determine the other program design elements. Instead, we recommend that plot size be treated as a management decision, based on the following considerations and guidelines:

- Utilize a single plot size for use statewide and in all applications of the S&T program design. This will enable compatibility between statewide and regional applications and will allow use of a single master sample.
- Establish a minimum target precision for statewide and regional applications. A target should be established for both status and trends monitoring.
- Consider the expected amount of information contained in the “average” mapped plot.

There is a close and significant relationship between plot size, predicted precision, mapped information, and program costs. Due to the patchy nature of aquatic resources and the specific observation and analysis approach used by S&T programs, small plots tend to have higher variability than large plots. Small plot are also more likely to be completely devoid of aquatic resources (i.e., to be “null” for aquatic resources). The higher variability of small plots means a larger minimum sample size is required to achieve statistically acceptable confidence in status and trends estimates. However, the cost to produce a map for a smaller plot is also expected to be lower than the cost to map a larger plot. Therefore, the larger sample size may or may not increase overall program costs. At the same time, because a higher fraction of small sample plots are null for aquatic resources, a significant fraction of programmatic resources are devoted towards producing maps of empty plots.

Using the best available information from the analysis performed for this report, Table 6.1 provides the minimum sample sizes required to obtain a defined level of precision in mean wetland density (see Section 4). Sample sizes were translated to estimated program costs using three different scenarios for imagery acquisition costs: i) upper-limit for contract imagery, $450 per plot; ii) lower-limit for contract imagery, $150 per plot; and iii) no-cost, existing imagery. Map production costs were assumed to be $25 per km^2. Additional details about the basis for these costs are provided in Appendix D.

The most cost-effective (predicted) sample size depended on the assumptions made about the costs of imagery acquisition. If imagery is assumed to cost $450 per plot, the 4 km^2 plot size was the most cost effective, followed by 1 km^2. If imagery is assumed to cost $150 per plot, or use of no-cost existing imagery is assumed, the 1 km^2 plot size was most cost effective, followed by 4 km^2. Under all assumptions, 16 km^2 was the least cost-effective plot size.

Table 6.1. Required sample size and predicted sample cost for estimating mean wetland density at different confidence levels.

	Plot Size (*km²*)	95% Confidence Interval[a] ± 5%	± 10%	± 15%	± 20%
Required statewide sample size	*1*	8,700	2,200	970	550
	4	7,200	1,800	800	450
	9	7,000	1,800	780	440
	16	6,700	1,700	750	420
Cost *($450/ plot imagery)*	*1*	$ 4,132,500	$ 1, 045,000	$ 460,750	$ 261,250
	4	$ 3,960,000	$ 990,000	$ 440,000	$ 247,500
	9	$ 4,725,000	$ 1,215,000	$ 526,500	$ 297,000
	16	$ 5,695,000	$ 1,445,000	$ 637,500	$ 357,000
Cost *($150/ plot imagery)*	*1*	$ 1,522,500	$ 385,000	$ 169,750	$ 96,250
	4	$ 1,800,000	$ 450,000	$ 200,000	$ 112,500
	9	$ 2,625,000	$ 675,000	$ 292,500	$ 165,000
	16	$ 3,685,000	$ 935,000	$ 412,500	$ 231,000
Cost *(no-cost imagery)*	*1*	$ 217,500	$ 55,000	$ 24,250	$ 13,750
	4	$ 720,000	$ 180,000	$ 80,000	$ 45,000
	9	$ 1,575,000	$ 405,000	$ 175,500	$ 99,000
	16	$ 2,680,000	$ 680,000	$ 300,000	$ 168,000

[a] Total sample sizes and predicted costs under a range of percent errors and confidence intervals can be found in Appendix I.

In addition to considering the costs associated with improving sample precision, Table 6.2 shows the costs associated with producing maps for plots containing aquatic resources. Shown is the expected cost per square kilometer to produce an aquatic resource map, corrected for the expected probability that an individual plot contains aquatic resources. Due to the costs of imagery acquisition and the higher rate of null plots for small plots, the costs per square kilometer increase rapidly as plot size decreases. For comparison, total sample size and predicted costs for a ± 10%, 95% confidence intervals are also provided.

Table 6.2. Cost per square kilometer to produce a plot, corrected for expected null fraction.

Plot Size (*km²*)	**Cost per km²** Corrected for f_{null}			**Total Sample Size and Cost** For ± 10% and 95% CI			
	$450	*$150*	*no-cost*	*N*	*$450*	*$150*	*no-cost*
1	$ 513	$ 189	$ 27	2,200	$ 1, 045,000	$ 385,000	$ 55,000
4	$ 141	$ 64	$ 26	1,800	$ 990,000	$ 450,000	$ 180,000
9	$ 76	$ 42	$ 25	1,800	$ 1,215,000	$ 675,000	$ 405,000
16	$ 54	$ 35	$ 25	1,700	$ 1,445,000	$ 935,000	$ 680,000

The analysis performed for this technical report could not support a definitive recommendation of a single plot size as ideal for all situations. However, based on the competing interests between precision and map production, and after reviewing all analysis and simulation, ***a tentative recommendation is made for a 4 km^2 plot size.***

6.5 Expected Output

Results from S&T program implementation will allow the state to report estimates of statewide or regional average density of aquatic resources and changes in density since the previous reporting timepoint. Reporting could correspond with, or supplement, the State of the State's Wetlands Report and the California intensification of the National Wetland Condition Assessment, or at any interval prescribed by the state agencies. The S&T program will also provide a wealth of information that can be used by a wide range of agencies and programs for additional analysis or as foundation for intensified investigations. Together, the proposed S&T program and the other State monitoring programs can provide a foundation for a coordinated Level 1 monitoring approach for California to provide estimates of wetland extent and change over time. However, achievement of coordinated results will require development of an appropriate Level 1 strategy and ongoing collaboration between the different State, local, federal, and private entities conducting Level 1 monitoring in California.

The California S&T program was designed to produce two types of output. First, the program will produce individual aquatic resource maps for selected sample plots, consistent with statewide and regional applications of the design. The plots mapped as part of this program will, cover a significant fraction of California (3-10% of the total land area). These maps could be used for additional purposes such as training and calibration purposes for other State mapping programs or as a sample frame for ambient condition assessments. However, the S&T program will not produce a contiguous map of aquatic resources and as such is not a substitute for other Level 1 strategy components such as comprehensive mapping. Second, and more importantly, the S&T program will produce estimates of extent, and changes in extent, of aquatic resource types and subtypes. If sampling is appropriately scaled, similar results can also be produced for regional or question-based intensifications.

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APPENDIX A. Technical Advisory Committee for Development of Design recommendations for the California S&T Program

A.1. Membership

Richard Ambrose	*University of California, Los Angeles*
Karen Bane	*California Coastal Conservancy*
Danielle Bram	*California State University, Northridge*
Elaine Blok	*US Fish and Wildlife Service*
Jennifer Cavanaugh	*US Department of Agriculture*
Kristen Cayce	*San Francisco Estuary Institute*
Ross Clark	*Moss Landing Marine Laboratory*
Josh Collins	*San Francisco Estuary Institute*
Tom Dahl	*US Fish and Wildlife Service*
Shawna Dark	*California State University, Northridge*
Tim Duff	*California Coastal Conservancy*
John Eadie	*University of California, Davis*
Charlie Endris	*Moss Landing Marine Laboratory*
Julie Evens	*California Native Plant Society*
Jim Harrington	*California Department of Fish and Game*
Cliff Harvey	*State Water Resources Control Board*
Paul Jones	*US Environmental Protection Agency*
Todd Keeler-Wolfe	*California Department of Fish and Game*
Steve Kloiber	*Minnesota Department of Natural Resources*
Gail Kuenster	*California Department of Water Resources*
Leila Lackey	*Southern California Coastal Water Research Project/ University of California, Los Angeles*
Ben Livsey	*State Water Resources Control Board*
Glenda Marsh	*California Department of Fish and Game*
Kevin O'Connor	*Moss Landing Marine Laboratory*
Bill Orme	*State Water Resources Control Board*
Patricia Pendleton	*California State University, Northridge*
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Pam Rittelmeyer	*California Coastal Conservancy*
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Steve Schoenig	*California Department of Fish and Game*
Chris Solek	*Southern California Coastal Water Research Project*
Eric Stein	*Southern California Coastal Water Research Project*
Martha Sutula	*Southern California Coastal Water Research Project*
Daniel Swenson	*US Army Corps of Engineers*
Denice Wardrop	*Pennsylvania State University*
Dave Weixelman	*US Forest Service*
Adam Wolfe	*California Department of Fish and Game*

A.2. Meeting List

March 11th, 2011 Introduction to goals and context of the S&T program; review and prioritization of technical issues

April 29th, 2011 Evaluation of existing wetland classification systems; drafting recommendations of the California wetland classification system

May 31st, 2011 Address comments and remaining issues for California wetland classification system

November 1st, 2011 Results from simulation of spatial sampling designs; discussion of implications for S&T program design and pilot-scale validation

December 12th, 2011 Review cost-based analysis of plot size; evaluate design of pilot-scale validation

May 2nd, 2012 Results from simulation of temporal sampling designs; discussion of implications for S&T program design

July 19th, 2012 Results from pilot-scale validation and review draft technical report

A.3. Meeting Materials

Agendas, meeting slides, and meeting summary notes available upon request.

APPENDIX B. Draft California Aquatic Resources Classification System

B.1. 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), (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.

B.2. 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.

B.3. 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 (http://www.ramsar.org/cda/en/ramsar-news-latest-classification-system/main/ramsar/1-26-76%5E21235_4000_0__). 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.

B.4. Recommended California Aquatic Resource Classification System

Structure and Approach

The proposed classification system consists of six elements important in aquatic resource classification. The first two elements (hydrogeomorphology and landscape position) are hierarchical (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 modifiers. 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 B.1. Hierarchical classification component.

<table>
<tr><th colspan="2">Hydrogeomorphology</th><th colspan="2">Landscape Connection</th></tr>
<tr><th>Major Class</th><th>Class</th><th>Type</th><th>Subtype</th></tr>
<tr><td rowspan="4">Non-wetland Open Water</td><td>Lacustrine</td><td colspan="2" rowspan="3">Same as associated wetland</td></tr>
<tr><td>Riverine</td></tr>
<tr><td>Estuarine</td></tr>
<tr><td>Marine</td><td>Intertidal</td><td>Cove
Embayment
Exposed Shoreline</td></tr>
</table>

<table>
<tr><td></td><td></td><td>Subtidal</td><td>Cove
Embayment
Exposed Shoreline</td></tr>
<tr><td rowspan="11">Wetland</td><td rowspan="2">Depression</td><td>Floodplain</td><td>Defined outlet
Undefined outlet</td></tr>
<tr><td>Non-floodplain</td><td>Defined outlet
Undefined outlet</td></tr>
<tr><td>Lacustrine</td><td colspan="2">Structural Basin
Topographic Plain</td></tr>
<tr><td>Slope</td><td colspan="2">Hillslopes
Break in Slope
Topographic Plain</td></tr>
<tr><td rowspan="2">Riverine</td><td colspan="2">High-gradient</td></tr>
<tr><td>Low-gradient</td><td>Confined
Unconfined</td></tr>
<tr><td rowspan="2">Estuarine</td><td colspan="2">Canyon Mouth
River Valley Mouth
Delta</td></tr>
<tr><td>Structural Basin</td><td>Embayment-Rocky Headland
Embayment-Bar Built
Lagoon
Dune Strand/Dammed</td></tr>
<tr><td>Aquatic Support Areas</td><td colspan="3">Same as associated Wetland</td></tr>
</table>

Table B.2. Translation of hierarchical classification to function.

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

  - *<u>Diked</u>*: modified by a man-made barrier that obstructs the inflow of water.
  - *<u>Ditched/drained</u>*: modified by a man-made structure that functions to drain (usually via subsurface route) the system, thereby altering its natural hydroperiod.
  - *<u>Diverted</u>*: anthropogenic modification to otherwise artificially lower the water level.
  - *<u>Infiltration</u>*: area receives artificially increased input of treated or untreated water. Water is held for infiltration into a subsurface aquifer.
  - *<u>Stormwater Control:</u>* 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.
  - *<u>Urban Runoff:</u>* water source is dominated by an artificially increased input of urban runoff.
- *Influences on Substrate and Bank*
  - *<u>Armored</u>*: 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.
  - *<u>Excavated</u>*: sediment or substrate has been removed to deepen and/or widen the area of inundation.
  - *<u>Filled/graded</u>*: 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.
  - *<u>Marine Control Structures</u>*: breakwaters, jetties, groins, seawalls, etc. meant to control erosion, tidal influences, and wave action within an estuary or along a shoreline.
  - *<u>Realigned:</u>* 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*
  - *<u>Aquaculture</u>*: standing, flowing, or tidal water used for production of aquatic organisms such as fish, mollusks, algae, etc.
  - *<u>Flooded Agriculture</u>*: cultivation of crops such as rice, wild rice, or cranberries, which require inundated for at least 1 month during the growing season.
  - *<u>Flood Irrigation:</u>* cultivation of crops, often grassy forage crops for hay, by flooding fields to point of saturation or shallow inundation.
  - *<u>Harbors/Marinas/Ports</u>*: open water area where boats are regularly docked, maintained, loaded, or unloaded. Typically have significant modification, armoring, and excavation of the shoreline.
  - *<u>Orchards</u>*: includes vineyards, and other areas planted or maintained for the production of fruits, nuts, berries, or ornamentals.
  - *<u>Ranchland</u>*: 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.
  - *<u>Rangeland:</u>* wildland area used for livestock grazing outside of cultivated ranch and farmland.
  - *<u>Recreation</u>*: 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.

B.5. References

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B.6. 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.

B.7. 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 QAQC 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. Is consistent with nomenclature of CA aquatic resource policies and programs. 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. Can be adequately cross-walked to other systems, especially NWI. 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.

Classification Levels Required Based on Cowardin Habitat Type

	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. Complements the CA Vegetation Manual and mapping effort. 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.

B.8. 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

B.9. 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)		

APPENDIX C. Spatial Sampling Theory

C.1. Simple Random Sampling

In simple random sampling (SRS), each individual, k, and has an equal probability of inclusion probability, π_k,

$$\pi_k = \frac{n}{N}$$

where n is the sample size and N is the population size. Using the Horvitz-Thompson estimator, the sample mean, x_π, is defined by,

$$x_\pi = \frac{1}{N}\sum_{k=1}^{n}\frac{x_k}{\pi_k} = \frac{1}{n}\sum_{k=1}^{n} x_k$$

where x_k is the value of the target variable for the k^{th} observation. The unbiased estimator of the sample variance, s^2, is given by,

$$s^2 = \frac{1}{n-1}\sum_{k=1}^{n}\left(x_k - x_\pi\right)^2$$

and the variance of the sample mean, $V(x_\pi)$, can be defined as,

$$V\left(x_\pi\right) = \frac{s^2}{n}$$

or, with the finite population correction factor, as

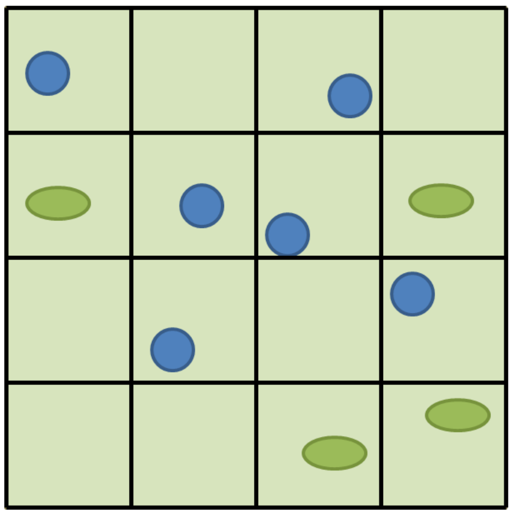
$$V\left(x_\pi\right) = \left(\frac{N-n}{N}\right)\frac{s^2}{n}$$

where the correction factor is used for sample sizes approaching the population size. Selection of an SRS is easily accomplished using a random number generator.

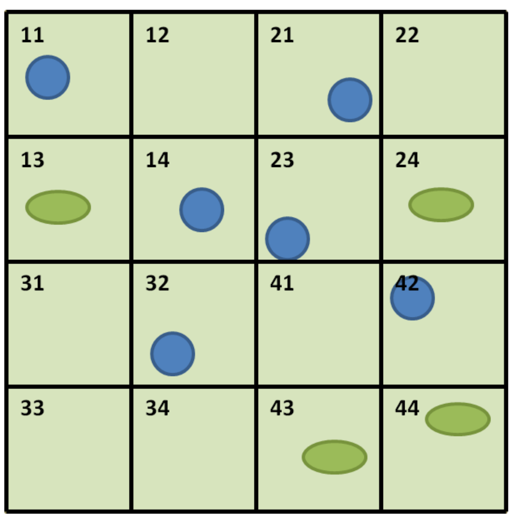
C.2. Generalized Random Tessellation Stratified Sampling (GRTS)

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.

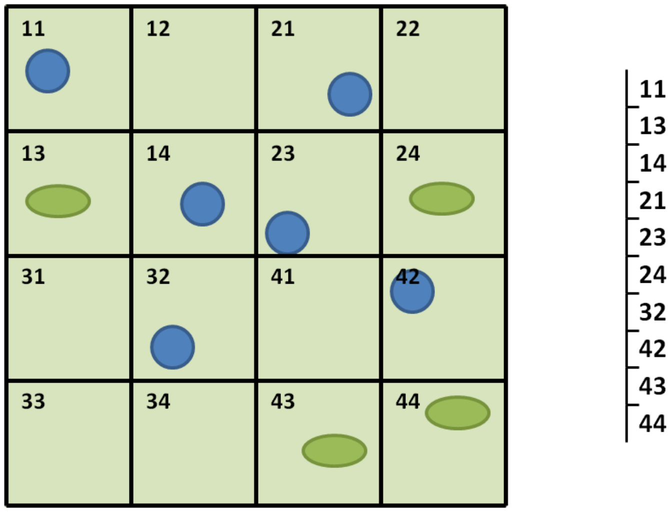
First, a regular grid is placed of the target area and subdivided until there is no more than one individual per grid cell. In our simulations, the grid is subdivided until the target cell size is reached.



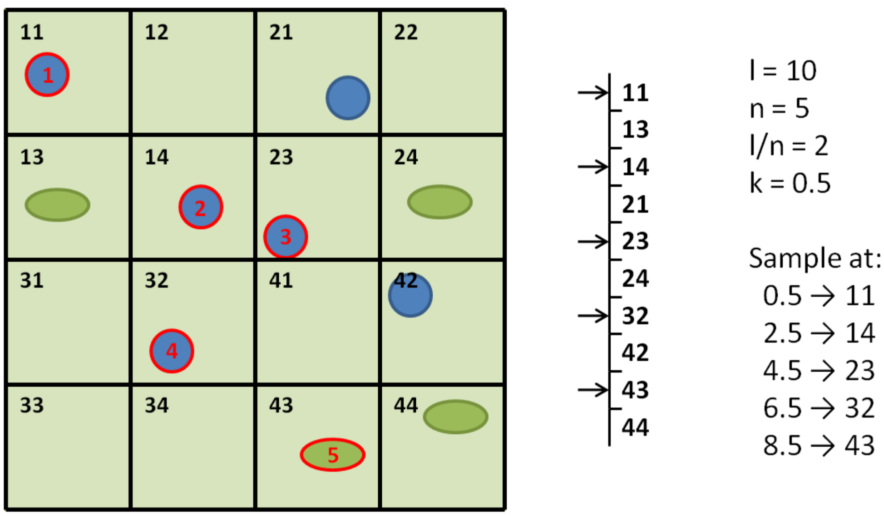
Second, a hierarchical address is assigned to each cell in the grid.



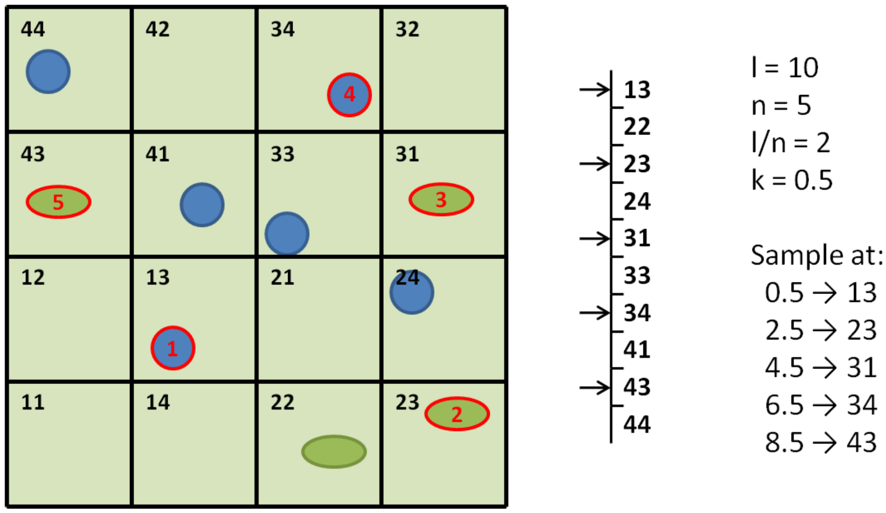
Third, the addresses are placed in order on a line if the cell contains the target. In our simulations, all grid cells overlapping California would be considered to contain the target.



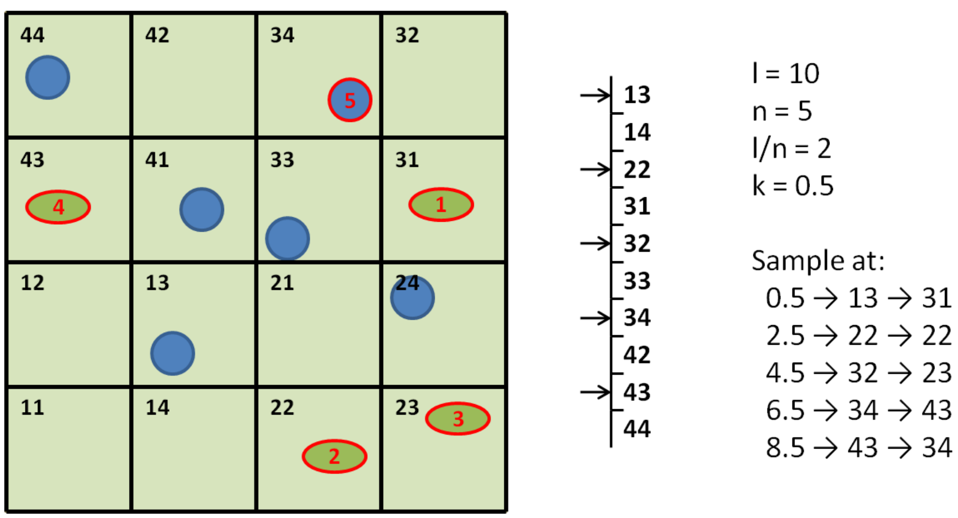
Fourth, a random starting point is placed on the line at a random location *k*, between *0* and *l/n* where *l* is the total length of the line and *n* is the desired sample size. Then, sampling occurs at *k* and all points *k + (i - 1) * (l/n)* where *i* = 2, 3, … , n. When sites are visited in order, 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.



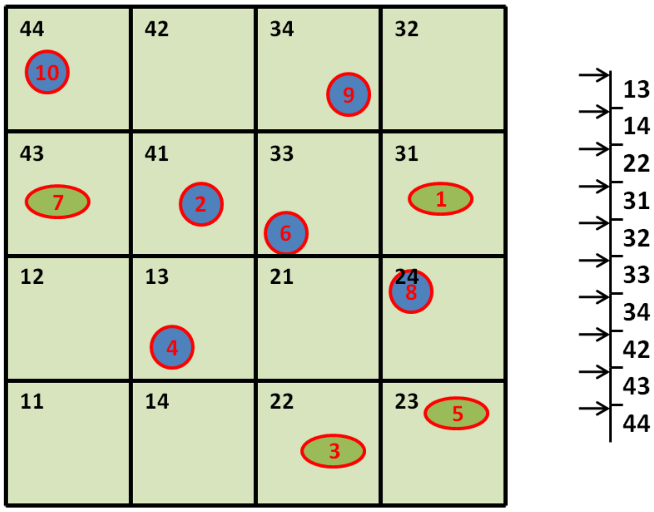
Alternatively, hierarchical addresses could be assigned randomly in the second step. This results in a greater median distance between sequential sample locations while still keeping locations within a somewhat limited area. Substitution of sample locations in the field is no longer efficient but the 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 (1^{st} green)
1.1 → 14 → 41 (1^{st} blue)
2.1 → 22 → 22 (2^{nd} green)
3.1 → 31 → 13 (2^{nd} blue)
4.1 → 32 → 23 (3^{rd} green)
5.1 → 33 → 33 (3^{rd} blue)
6.1 → 34 → 43 (4^{th} green)
7.1 → 42 → 24 (4^{th} blue)
8.1 → 43 → 34 (5^{th} blue)
9.1 → 44 → 44 (6^{th} blue)

C.3. Stratified Sampling with Optimum Allocation

Stratification can be applied to both SRS and GRTS designs by dividing the target population or the target area based on an auxiliary variable. Once the divisions are made, the total sample size can be divided amongst the strata using on a number of strategies. Three of the most commonly used are:

1. Equal allocation divides the sample evenly amongst the strata,
$$n_h = \frac{n}{L}$$
where n_h is the sample size for stratum h, n is the total sample size, and L is the number of strata. This approach is useful when the strata are of similar sizes.
2. Proportional allocation divides the sample amongst the strata in proportion to the size of the strata,
$$n_h = \left(\frac{n}{N}\right)N_h$$
where N_h is the total population size in stratum h and N is the total population size. This approach produces a sample with similar characteristics as the target population for the variables used to define the strata.
3. Optimum allocation divides the sample proportional to the variance in the target variable within the stratum,
$$n_h = n\left(\frac{N_h\sigma_h}{\sum_{i=1}^{L} N_i\sigma_i}\right)$$
where σ_h is the standard deviation within stratum h. Optimum allocation minimizes the overall sample variance.

APPENDIX D. Cost Information for Imagery Acquisition and Map Production

D.1. No-Cost Imagery Options

Imagery options for the California S&T program fall into two categories, existing (no-cost) imagery and contracted imagery. The best, and most reliably available, source of existing imagery is the National Agriculture Imagery Program (NAIP) operated by the US Department of Agriculture (USDA).[5] NAIP imagery meets minimum standards for use in the California S&T program. For the past couple of years, NAIP imagery has been available digitally for free through the USDA Geospatial Data Gateway.[6] The California Department of Fish and Game also offers streaming NAIP imagery for use in ArcGIS and other programs.[7]

However, reliance on existing imagery sources, such as NAIP, also imposes limits on the California S&T program. First and foremost, the California S&T program would have no control over the frequency and calendar year of imagery acquisition. NAIP imagery has been acquired for the entirety of California in 2005, 2010, and 2011 and is planned to be repeated in 2012. However, there is guarantee that this schedule of acquisition will continue.

Second, NAIP imagery is acquired during the summer growing season. This timepoint can limit aquatic resource detection and delineation accuracy for two major reasons. One, presence of summer vegetation can obscure boundaries between aquatic resources and upland or obscure aquatic resource entirely. Two, parts of California have a very limited wet season and as such, many temporary or seasonal wetlands and streams may not be visible during the summer acquisition period for NAIP.

Third, while NAIP contractors are supposed to produce imagery with less than 10% cloud cover, this standard is not always be met in all areas and acquisition years. Given the small relative size of S&T plots, cloud cover could be a significant potential issue if this no-cost imagery source is utilized for the California S&T program.

D.2. Contract Imagery Options

Estimates of imagery acquisition costs were obtained from Nick Arentz of Skyview Aerial Photo Inc. SCCWRP has previously contracted with Skyview to obtain imagery for Newport Bay in Orange County. Skyview has several decades of experience in aerial photography in California and the Western US and has substantial experience in the process of imagery acquisition for monitoring programs similar to the proposed California S&T program.

Mr. Arentz was gracious enough to provide recommendations about estimated costs and the bidding process for contracting out imagery acquisition. His comments are reproduced in full below:

[5] http://www.fsa.usda.gov/FSA/apfoapp?area=home&subject=prog&topic=nai
[6] http://datagateway.nrcs.usda.gov/
[7] http://www.dfg.ca.gov/biogeodata/gis/map_services.asp

Leila:

Here are some thoughts on providing the State of California the best value for the collection of aerial imagery by aircraft on a state-wide project with a scattered site distribution:

Goal: Best value for state; Imagery specifications met or exceeded; Project delivered on time.

The 1-meter pixel spec can be achieved by widely-available satellite data (although tasking a satellite for cloud-free imagery (90%) is not possible and other scheduling challenges may exist). It makes more sense project-wide to require a better resolution (in the neighborhood of a 1-foot pixel or better). This will not increase cost because 30cm pixel is achieved in one overhead pass with a film sensor or large format digital sensor (for 4x4 km plot). You could expect a 15cm pixel Ground Sample Distance on a 2x2 km plot.

The NRI/WRP program administered by the USDA has many similarities to the program you described. Following the USDA model for vendor selection as sanctioned by MAPPS, there should be qualification-based selection criteria (QBS). Also, use a Geographic-based selection of vendors as done by the Caltrans Office of Photogrammetry or simply design the cost proposal such that the state is divided into sub-regions and the vendor proposals will be broken down by cost per site per sub-region.

NRI/WRP aerial imagery program administered by the USDA has a 70/30 weighting 70% Qualification Based/30% price proposal

QBS Scoring involves but not limited to the following: Qualifying vendors must submit qualifications showing technical capability and past performance history.

Scoring based on previous related work with the State of CA (and then other similar projects with government agencies). References must be included. Small business and disadvantaged business preferences are applicable. Businesses headquartered in CA receive preference in scoring (due to in-state tax base).

Price proposals:

Based on 1500-2000 sites scattered State-wide and the fact that most qualified vendors are also geographically situated within the State of CA, expect to receive the lowest pricing for the state if the project is awarded to up to 3 qualified flying contractors as opposed to just one vendor. Make it known within RFP that each of the three selected vendors will be awarded a minimum number of sites (say 300) of the total number. This scenario gives incentive for localized vendors to give the best pricing for project sites within sub-regions local to his/her respective base of operation (ie Southern/Central/Northern California) with a sufficient number of sites to employ all available discounts.

Delimiting the project by the dozen or so ecological boundaries would make sense due to terrain issues except for the fact that some of these Ecoregions span a great extent of the state N to S. Spitting the longest (oblong) Ecoregions in half (by latitude) could help make the sites within these regions more competitive from the cost-standpoint.

Skyview Crude Price Estimate:
- $150 - $450 per site
- Scale factors:
- Film-based True Color - CIR - 4-Band Digital Imagery
- 30cm vs 15cm GSD
- Remoteness
- Ground Elevation
- Expected positional accuracy

I would be happy to discuss this further. Feel free to call our office line if you have any questions (especially on a rainy day!).

Thanks,

Nick Arentz, Jr.

Business Development
Skyview Aerial Photo, Inc.
Tel: (951) 445-4434
Fax: (951) 445-4437
www.skyairphoto.com | nickjr@skyairphoto.com

EFFECTIVE IMMEDIATELY: NEW STREET ADDRESS!
37920 Sky Canyon Dr., #1001
Murrieta, CA 92563

D.3. Map Production

Estimates of map production costs were graciously provided from two groups with significant expertise in aquatic resource map production. Both groups considered time and labor costs associated with imagery and auxiliary data management, draft map production, internal quality control, and final geodatabase management. While the groups did not confer in their estimates, both arrived at an approximate rate of $25 per square kilometer.

This figure does not include overhead charges or any additional quality control that the State may need to perform. Depending on overhead rates and the extent of the quality control procedures adopted, this could potentially double the map production costs.

APPENDIX E. Temporal Estimation Methods by Sample Type

E.1. Fixed or 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

E.2. Mixture of Fixed and Moving Locations Using Current and Previous Timepoint (SPR_2)

Mean	$\bar{Y} = \frac{\sum \frac{\bar{Y}_j}{V(\bar{Y}_j)}}{\sum \frac{1}{V(\bar{Y}_j)}}; \quad \mathrm{j} = 2,1$ $\bar{Y}_2 = \bar{Y}_{-2}$ $V(\bar{Y}_2) = \frac{s^2_{Y_{-2}}}{n_{-2}}$ $\bar{Y}_1 = \bar{Y}_{12} + \hat{\beta}_{Y_{12}X_{12}}(\bar{X}_{1\bullet} - \bar{X}_{12})$ $V(\bar{Y}_1) = \left[\frac{\sum(Y_{12_i} - \bar{Y}_{12})^2(1 - r^2_{Y_{12}X_{12}})}{n_{12} - 2}\right]\left[\frac{1}{n_{12}} + \frac{(\bar{X}_{1\bullet} - \bar{X}_{12})^2}{\sum(X_{12_i} - \bar{X}_{12})^2}\right] + \frac{s^2_{Y_{12}} r^2_{Y_{12}X_{12}}}{n_{1\bullet}}$
Change	$\bar{C} = \frac{\sum \frac{\bar{C}_j}{V(\bar{C}_j)}}{\sum \frac{1}{V(\bar{C}_j)}}; \quad \mathrm{j} = \mathrm{S, R}$ $\bar{C}_S = \bar{Y}_{12} - \bar{X}_{12}$ $V(\bar{C}_S) = (s^2_{Y_{12}} + s^2_{X_{12}} - 2COV(Y_{12}, X_{12}))/n_{12}$ $\bar{C}_R = \bar{Y}_{-2} - \bar{X}_{1-}$ $V(\bar{C}_R) = (s^2_{Y_{-2}} + s^2_{X_{1-}})/n_{-2}$
Difference	Difference between current and previous mean
Trend	OLS regression of mean against years since baseline

(Cochran and Carroll 1953), (1962), (1977)

E.3. Mixture of Fixed and Moving Locations Using Current and Two Previous Timepoint (SPR_3)

Mean	$\bar{Z} = \dfrac{\sum \frac{\bar{Z}_j}{V(\bar{Z}_j)}}{\sum \frac{1}{V(\bar{Z}_j)}}; \quad \mathrm{j} = 3,2,1$ $\bar{Z}_3 = \bar{Z}_{--3}$ $V(\bar{Z}_3) = \dfrac{s^2_{Z_{--3}}}{n_{--3}}$ $\bar{Z}_2 = \bar{Z}_{123} + \hat{\beta}_{Z_{123}Y_{123}}(\bar{Y}_{\bullet 2 \bullet} - \bar{Y}_{123})$ $V(\bar{Z}_2) = \left[\dfrac{\sum (Z_{123j} - \bar{Z}_{123})^2 (1 - r^2_{Z_{123}Y_{123}})}{n_{123} - 2}\right]\left[\dfrac{1}{n_{123}} + \dfrac{(\bar{Y}_{\bullet 2 \bullet} - \bar{Y}_{123})^2}{\sum (Y_{123j} - \bar{Y}_{123})^2}\right] + \dfrac{s^2_{Z_{123}} r^2_{Z_{123}Y_{123}}}{n_{\bullet 2 \bullet}}$ $\bar{Z}_1 = \bar{Z}_{123} + \hat{\beta}_{Z_{123}X_{123}}(\bar{X}_{1 \bullet \bullet} - \bar{X}_{123})$ $V(\bar{Z}_1) = \left[\dfrac{\sum (Z_{123j} - \bar{Z}_{123})^2 (1 - r^2_{Z_{123}X_{123}})}{n_{123} - 2}\right]\left[\dfrac{1}{n_{123}} + \dfrac{(\bar{X}_{1 \bullet \bullet} - \bar{X}_{123})^2}{\sum (X_{123j} - \bar{X}_{123})^2}\right] + \dfrac{s^2_{Z_{123}} r^2_{Z_{123}X_{123}}}{n_{1 \bullet \bullet}}$
Change	$\bar{C} = \dfrac{\sum \frac{\bar{C}_j}{V(\bar{C}_j)}}{\sum \frac{1}{V(\bar{C}_j)}}; \quad \mathrm{j} = \mathrm{S, R, U}$ $\bar{C}_S = \bar{Z}_{123} - \bar{Y}_{123}$ $V(\bar{C}_S) = (s^2_{Z_{123}} + s^2_{Y_{123}} - 2Cov(Z_{123}, Y_{123}))/n_{123}$ $\bar{C}_R = \bar{Z}_{--3} - \bar{Y}_{-2-}$ $V(\bar{C}_R) = (s^2_{Z_{--3}} + s^2_{Y_{-2-}})/n_{--3}$ $\bar{C}_U = \bar{Z}_1 - \bar{Y}_1$ $V(\bar{C}_U) = V(\bar{Z}_1) + V(\bar{Y}_1) + 2Cov(\bar{Z}_1, \bar{Y}_1)$ $\bar{Z}_1 = \bar{Z}_{123} + \hat{\beta}_{Z_{123}X_{123}}(\bar{X}_{1 \bullet \bullet} - \bar{X}_{123})$ $V(\bar{Z}_1) = \left[\dfrac{\sum (Z_{123j} - \bar{Z}_{123})^2 (1 - r^2_{Z_{123}X_{123}})}{n_{123} - 2}\right]\left[\dfrac{1}{n_{123}} + \dfrac{(\bar{X}_{1 \bullet \bullet} - \bar{X}_{123})^2}{\sum (X_{123j} - \bar{X}_{123})^2}\right] + \dfrac{s^2_{Z_{123}} r^2_{Z_{123}X_{123}}}{n_{1 \bullet \bullet}}$ $\bar{Y}_1 = \bar{Y}_{123} + \hat{\beta}_{Y_{123}X_{123}}(\bar{X}_{1 \bullet \bullet} - \bar{X}_{123})$ $V(\bar{Y}_1) = \left[\dfrac{\sum (Y_{123j} - \bar{Y}_{123})^2 (1 - r^2_{Y_{123}X_{123}})}{n_{123} - 2}\right]\left[\dfrac{1}{n_{123}} + \dfrac{(\bar{X}_{1 \bullet \bullet} - \bar{X}_{123})^2}{\sum (X_{123j} - \bar{X}_{123})^2}\right] + \dfrac{s^2_{Y_{123}} r^2_{Y_{123}X_{123}}}{n_{1 \bullet \bullet}}$ $Cov(\bar{Z}_1, \bar{Y}_1) = (s^2_{Z_{123}Y_{123}} + s^2_{Z_{123}X_{123}} \hat{\beta}_{Y_{123}X_{123}})/n_{123}$
Difference	Difference between current and previous mean
Trend	OLS regression of mean against years since baseline

(Cunia and Chevrou 1969)

APPENDIX F. Training of Mapping Center Partners & Production of Maps for the Pilot-Scale Implementation and Validation

Three mapping center partners were trained in the production of maps for the proposed California S&T program. This training was performed in the context of map production for the pilot-scale implementation and validation. Standard operation procedures (SOPs) for map production are provided in Appendix G and the design of the pilot is provided in detail in Section 6 of this technical report. Training and map production took place between January and May of 2012.

In January, 2012, participation of the three mapping center partners was confirmed. Partners were the Center for Geographic Studies at California State University, Northridge, Moss Landing Marine Laboratories, and the San Francisco Estuary Institute. SCCWRP established a listserv (sntmapping@sccwrp.org) and file transfer protocol (FTP) site for use in training and the pilot project.

On February 2nd, 2012, a one-day training was held at SCCWRP offices. This meeting was attended by all three mapping partners and involved discussion of mapping production, data management, and quality control standards and procedures. Materials from this meeting are provided as an attachment to this technical report.

Following the February 2nd meeting, work began on map production for the pilot-scale validation. To start, SCCWRP worked with the mapping centers to ensure that all imagery and auxiliary data necessary for map production was available on the project FTP. Then, SCCWRP selected a common training plot, which was then mapped by all three of the mapping partners. The purpose of the training plot was to further train the mapping partners in project SOPs and to ensure a common approach between mapping partners. The results of the training exercise were evaluated on February 24th, 2012, through a group conference call and web-based meeting.

Based on a highly successful training plot, each mapping partner began production of maps for the pilot implementation. Three intermediate conference calls were held to provide updates and discuss questions and potential issues related to the project SOPs. These conference calls were attended by all mapping partners and led by SCCWRP. Calls occurred on March 7th, March 15th, and May 9th, 2012. The calls led to refinement of project SOPs and greater consistency and coordination between the mapping partners.

Maps were produced and fully evaluated by a second professional for accuracy and consistency in boundary delineation and classification. Final maps were delivered in SCCWRP in May of 2012, concluding the training of the mapping center partners and the map production for the pilot project. Results and details on how the maps were evaluated are provided in Section 6 of this technical report. Final maps for all of the plots in the pilot project are also provided as an attachment to this technical report.

APPENDIX G. Standard Operating Procedures and Quality Assurance Project Plan for the California S&T Program

A. Project Management

A.1. Title and approval sheet

Cannot be completed at this time.

A.2. Table of contents

(Omitted)

A.3. Distribution list

Cannot be completed at this time.

A.4. Project/task organization

Cannot be completed at this time.

A.5. Problem definition/background

The procedures outlined in this combined SOP and QAPP are designed to answer the question: what is the area of aquatic resources in California and how is it changing with time? This question has been a requirement of California's aquatic resource monitoring policy since Governor Pete Wilson established California's no-net-loss policy for wetland area in 1993. Information about extent and distribution underpin ambient assessments of aquatic resource condition, scientific study of aquatic resources, and policy and management decisions related to aquatic resources. California has tried twice to answer the question, with the State of State's Wetlands reports released in 1998 and 2010. However, in both instances, the absence of an appropriate method prevented calculation of accurate and precise values.

The procedures outlined here are designed to produce a statistical estimate of the mean density of aquatic resources (as area of aquatic resources per square kilometer) in California at a given point in time, and to estimate the change in aquatic resource density since the previous timepoint. Procedures are also provided for estimating density and changes in density for subregions of California. This information can be used by groups such as managers to guide decision-making and monitoring choices; the technical and scientific community to facilitate monitoring and scientific investigations; and the public to increase understanding of their local and statewide aquatic resources.

A.6. Project/task description

More detailed descriptions of the procedures and tasks are provided and referenced in later sections. Briefly, the project is designed to estimate the aquatic resource density for California, and how that density has changed with time. Therefore, the entire State is used as a sample frame and a probabilistic sample is drawn from a square, 2 km by 2km grid, placed over the entire state. Maps of aquatic resource area, and changes in aquatic resource area, are produced according to the appropriate secondary SOPs and QAPPs.

A.7. Quality objectives and criteria

Numerous quality objectives and criteria will be specified by the secondary SOPs and QAPPs used for image acquisition and map production. Quality objectives and criteria for this specific QAPP should be related to ensuring the reliability of the sampling and analysis process and for assessing consistency between map producers. This will ensure the statistical validity of estimates and reduce systematic errors and biases in estimates and final products.

A.8. Special training/certification

Cannot be completed at this time.

A.9. Documents and records

Cannot be completed at this time.

B. Data generation and acquisition

B.1. Sampling process design

In this project, aquatic resource maps are produced for randomly selected, 4 km^2 sample plots. These maps are used to estimate the mean density of aquatic resources and the change in aquatic resource density over time. Secondary SOP and QAPP documents provide the map production and map quality assurance procedures. This element of this QAPP describes the processes used to select and manage the random sample.

The sampling design is based on a square, 2 km by 2 km grid, placed over the State of California. Grid cells that are entirely outside of the State Boundary should be removed from the sample frame. The sample should be selected using generalized random tessellation stratified (GRTS) sampling. Both ArcGIS and the *spsurvey* package of the statistical language R are required to complete the sampling procedures.

First, use the fishnet tool in ArcGIS to generate a 2 km by 2 km grid. The extent of the grid should be sufficient to cover the entire extent of the State of California (as defined by the 1:24k County Boundaries dataset maintained by the California Department of Fish and Game). The California Teal Albers NAD83 projection (or another equal area projection) is recommended for the fishnet, as an equal area projection is required for GRTS sampling. A random offset, between 0 and 2 km, should be applied to the bottom lefthand corner of the fishnet. Finally, the resulting fishnet should have polygon geometry and label points should be created.

Second, use the California Boundary was used to all intersecting grid cells, and corresponding label points. The resulting set of grid cells and label points will be used to draw the random sample. Add two attributes to the points object, named “X” and “Y,” type = “Double.” Use the calculate geometry tool to calculate the X and Y coordinates, respectively, of the points in meters [m]. Use the same projection as the fishnet to calculate the coordinates.

These coordinates are used by the *grts* function to calculate the distance between points in the sample frame. Therefore, an equal area projection is necessary to ensure the accuracy of this calculation. Export the label points as a shapefile.

Third, use the *spsurvey* package to open the points shapefile and draw a GRTS master sample. Use the *set.seed* function before drawing the sample, using the *grts* function, so that the sample draw can be repeated, exactly at a later time. Refer to documentation for the *spsurvey* package and the *grts* function for specific guidance on function arguments. The design should be a single, unstratified panel with equal probability selection. Use the appropriate arguments to generate an output shapefile containing the sample points. The sample size should be equal to the total number of grid cells. This will ensure that the sample draw never needs to be repeated because additional locations are required. This also provides a mechanism for, theoretically, observing every possible location in the State.

Fourth, use the output shapefile to determine which master sample cells should be used for the S&T effort. This determination should ideally be based on a two-step process and should be completed in ArcGIS. Step one is to determine the subset of the master sample that meets the inclusion criteria for the particular application of the S&T program. One example is a state-wide survey to estimate the mean aquatic resource density for the state of California. The inclusion criterion for this sample is that grid cells overlap with the state. Therefore, the entire master sample will meets the inclusion criteria because only cells that overlapped the state were included in the sample frame. A second example is a regional survey to estimate the mean aquatic resource density for the Central Valley Ecoregion. The inclusion criterion for this sample is that grid cells overlap with the Central Valley Ecoregion. Therefore, an appropriate geospatial dataset should be used to select the portion of the master sample points that fall within the Central Valley Ecoregion.

After the relevant subset of the master sample is determined, the second step is to select sample locations from the subset based on the SiteID number. This number is automatically provided by the *grts* function as an attribute of the master sample shapefile. Using locations in order will increase the spatial balance of the resulting sample. This will increase accuracy and allow use of the GRTS variance estimator, which is more statistically precise than the traditional sample variance estimator. To select locations in order from the relevant subset, simply sort of the SiteID attribute in ascending order. The first n items in this sorted list (where n is the desired sample size based on budgetary constraints and accuracy targets) provide the sample for the particular application of the S&T procedures.

Once sample points are selected, the corresponding grid cell boundaries should be selected and transferred to mapping professionals for production and quality assurance of sample maps. Procedures for map production and change detection are provided by secondary SOPs and QAPPs. Once completed, the sample maps should be transferred to the party responsible for analysis.

The first step in analysis should be to calculate the area-based density of aquatic resources, or changes in density, for each sample plot. This should be calculated by dividing the area of aquatic resource by the sample plot area. If a sample plot falls on the boundary of the target area, only the portion of the sample plot, and the portion of the aquatic resources, that falls within the study area should be considered. The density calculations are best performed in ArcGIS.

After area density is calculated, estimates of mean density, or changes in density, can be produced from the sample data using the *total.est* function in the *spsurvey* package in R. Refer to package documentation for instructions on calculating estimates. Reweighting of sample points will usually be required. The result will be an estimate of mean density with a statistical confidence interval based on the GRTS variance estimator. Numerous additional estimates and analysis options are possible and should be at the discretion of the group performing the analysis.

B.2. Sampling and image acquisition methods

Sampling and image acquisition methods should be described by secondary SOPs and QAPPs related to the production of sample plot maps. Image acquisition may or may not be performed specifically for the S&T program. Existing, no-cost sources of aerial imagery, such as the National Agriculture Imagery Program, meet the quality standards for map production for this program. Regardless of which imagery source is utilized, a consistent image year (ideally the most recent) should be used for all map production and change analysis during a single application of the S&T procedures.

B.3. Sample handling and custody

Does not apply. All imagery and geodatabases will be delivered and processed electronically.

B.4. Analytical methods

Analytical methods relevant to this QAPP include the re-weighting procedure for the sample map. Reweighting is a complicated process that requires knowledge of the GRTS sampling and analysis methodology. Image and geospatial data analysis, interpretation, and processing methods should be described by the appropriate secondary SOPs and QAPPs for imagery acquisition and map production.

The general concept of a sample weight is the portion of the study area that a sample represents. Thus, if the study grid is 1,000 plots and there are 100 sample locations, each location has a weight of 10. In the sample draw described in section B.1, the sample size is equal to the total number of grid cells. Thus, each sampling location has an automatically calculated weight of exactly 1.

To calculate appropriate weights for basic applications of the S&T procedures, the number of grid cells in the target area should be divided by the sample size used. For example, there are approximately 104,000 grid cells (at a 4 km^2 grid size) in California. The exact number will vary slightly depending on the random offset for the bottom lefthand corner. Thus, if the statewide sample size is set at 2,000, each location will have a weight of 52. Similarly, there are approximately 11,700 cells in the Central Valley Ecoregion (again, the number will vary slightly depending on the offset). If 500 total locations are sampled in this region, each location will have a weight of 23.4. Combination of these two samples (the statewide sample and the Central

Valley intensification) into a single statewide estimate at first appears to be a more complicated weighting problem. However, upon examination, the weights remain identical.

Recalling how sample locations are determined from the master sample, it is unlikely that these two samples are discrete. For example, the first 2,000 locations in the master sample are anticipated to contain approximately 225 locations in the Central Valley (the Central Valley contains 11,700 / 104,000 * 100% = 11.25% of the statewide plots; therefore, a 2,000 plot sample is expected to contain 2,000 * 0.1125 = 225 plots in the Central Valley). As a result, to obtain a total of 500 plots in the Central Valley, we anticipate only adding 275 new plots, bringing the total sample size to 2,275. Weighting for the sample will involve one weight for locations within the Central Valley and one weight for locations outside the Valley. Within the Valley, sample weights are equal to the number of plots in the valley, 11,700, over the number of sampled locations, 500. This produces the same weight as before, 11,700 / 500 = 23.4. Outside the valley, sample weights are calculated as the number of total plots outside the Valley, 104,700 – 11,700 = 92,300, over the number of sampled locations outside the valley, 2,275 – 500 = 1,775, to produce the same weight as before, 92,300 / 1,775 = 52. and one weight for locations outside the Central Valley ((104,700 – 11,700) / (2,275 – 500) ≈ 23).

More complicated cases may arise if cells are excluded or added for reasons other than geographic region. For these and other cases, a specific rational for determining the weights should be outlined through the assistance of a statistician trained in GRTS sampling procedures.

B.5. Quality control

Quality control procedures relevant to this QAPP are threefold. First, are the quality controls related to image acquisition and map generation. These are outlined in the appropriate secondary SOPs and QAPPs. Second, is the overall standard for reliability between map producers. This can be approached in a number of ways but the following provides two examples. Quality control for the S&T program is handicapped because of the lack of a "standard" to judge performance against. Third are standards for analyzing results and producing estimates

First, individual map producers can be trained on a number of well characterized sample plots. These plots would ideally include ground-truthed maps of aquatic resources and would represent a variety of landscapes. Map producers would have to demonstrate accuracy for these maps within a certain threshold in order to be authorized to produce maps for the S&T program. Our experience with inter-mapper variability suggests a threshold between 15 and 30%. Failure to meet the threshold could result in denial of a contract.

Second, sample plots could be randomly selected to be mapped by more than one map producer. Those producers would then have to meet a threshold of agreement — again, a threshold between 15 and 30% is likely appropriate based on our experience with inter-mapper variability. Failure to meet the threshold could result in a requirement to retrain mapping professionals and to revise plots assigned to the producer.

B.6. Instrument/equipment testing, inspection, and maintenance

Does not apply to this SOP & QAPP. However, the SOP and QAPP for imagery acquisition should contain relevant testing, inspection, and maintenance procedures for GPS, camera equipment, etc.

B.7. Instrument/equipment calibration and frequency

Does not apply to this SOP & QAPP. However, the SOP and QAPP for imagery acquisition should contain calibration procedures for GPS, camera equipment, etc.

B.8. Inspection/acceptance requirements for supplies and consumables

Does not apply to this SOP & QAPP. Would only be required as part of the SOP and QAPP for imagery acquisition if hard-copy photography is used.

B.9. Data acquisition (nondirect measurements)

Secondary data will be used in the production of sample plot maps. Therefore, the appropriate secondary SOPs and QAPPs should cover the data acquisition requirements.

B.10. Data management

Data management falls into two categories. First, is the management performed by the mapping professionals. Secondary SOPs and QAPPs should cover this data management.

Second, is data management for the analysis professionals who are producing estimates of status and trends from the maps received from the mapping professionals. This process includes the generation of status and trends estimates. Briefly, the process involves *clipping* plot boundaries to the study area and then *intersecting* the sample maps with the plot boundaries. Then, the density of aquatic resources for each sample plot can be calculated by summing the aquatic resource polygon area (or the area of a change) and dividing by the clipped plot area. This information should be *joined* with the points from the GRTS sample and exported as a shapefile. Then, the results can be analyzed in R using the *spsurvey* package. There are significant opportunities for automating and streamlining this process.

C. Assessment/oversight

Cannot be completed at this time.

D. Data validation and usability

Cannot be completed at this time.

APPENDIX H. Design and Model-based Estimation Theory

H.1. Simple Mean

Simple mean is a design-based estimate and is identical to the equations outlined in Appendix I for simple random sampling. Briefly, the simple mean, x_π, is defined by,

$$x_\pi = \frac{1}{n}\sum_{k=1}^{n} x_k$$

where n is the sample size and x_k is the value of the target variable for the k^{th} observation. The unbiased estimator of the sample variance, s^2, is given by,

$$s^2 = \frac{1}{n-1}\sum_{k=1}^{n}\left(x_k - x_\pi\right)^2$$

and the variance of the sample mean, $V(x_\pi)$, can be defined as,

$$V\left(x_\pi\right) = \frac{s^2}{n}$$

or, with the finite population correction factor, as

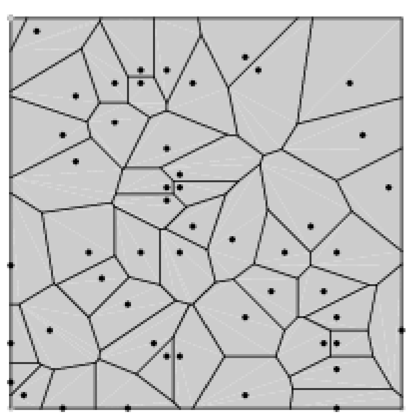
$$V\left(x_\pi\right) = \left(\frac{N-n}{N}\right)\frac{s^2}{n}$$

where N is the total population size.

H.2. Area-weighted Mean

The area-weighted mean uses the Horvitz-Thompson estimator and defines the inclusion probabilities, π_k, of each of the k individuals according to the fraction of the total area covered by their voronoi polygon.

A voronoi polygon for point k delineates all of the points in the target area closer to point k than to any other point:



After construction of the polygons, each point in the sample, $1, 2, 3, \ldots, k, \ldots, n$ has an associated voronoi area, $A_1, A_2, A_3, \ldots, A_k, \ldots, A_n$, and the total area A is equal to the sum of the voronoi areas. Then the mean is defined as a weighted combination:

$$x_{\pi,area} = \sum_{k=1}^{n} \frac{A_k}{A} x_k = \frac{1}{A} \sum_{k=1}^{n} A_k x_k$$

Variance cannot be calculated with this method because there is only one measurement per polygon.

H.3. Inverse Distance Weighting

Inverse distance weighting (IDW), a model-based estimator, produces an estimate for the target parameter at each point in the sampled area. The estimate is a weighted combination of the value of the target variable at each sampled point. The weighting is inversely related to the distance, $d_{e,k}$, between the point being estimated, e, and the sample location, k. Therefore, the value of the target parameter at the estimated location, x_e, is given by:

$$x_e = \frac{\sum_{k=1}^{n} x_k d_{e,k}^{-p}}{\sum_{k=1}^{n} d_{e,k}^{-p}}$$

where x_k is the observed value at sample location k. The weighting power, p, is not objectively set and can be chosen to determine how quickly the weights go to zero. A frequent choice is 2.

IDW does not give an estimate of the uncertainty at the estimated location. To calculate the mean for the target area, the simple mean of all of the points can be taken and the variance calculated for the simple mean.

H.4. Kriging

Kriging, a model-based prediction method, produces predictions of the target variable for every location in the target field, similar to IDW. However, unlike IDW, kriging is able to estimate the error in the predicted value. In addition, while kriging is also a weighted combination of the known values, the weighting functions are less arbitrary than in IDW and the predicted values are selected to minimize the error of prediction. Finally, kriging is an exact interpolator, meaning the observed values are preserved in the predicted surface.

Several different types of kriging have been developed. Three of the most basic are simple kriging, ordinary kriging, and universal kriging. The assumptions of these three methods will be discussed. Then the general process of kriging predictions will be outlined.

Simple kriging assumes: (1) the population mean is known and does not depend on location; and (2) the covariance between every two locations exists and depends only on the distance between the locations, not on the locations themselves.

Covariance measures how multiple variables correlate with one another; a covariance of zero means the variables are not correlated, a positive covariance means the variables are positively correlated, and a negative covariance means the variables are negatively correlated.

The covariance, *C(h)* for the covariance as a function of the distance *h* between two points is mathematically defined as:

$$C(h)=\frac{1}{N(h)}\sum_{N(h)}(x_i-\mu)(x_j-\mu)$$

where *N(h)* is the number of paired points, *i* and *j*, separated by the distance *h*; x_i and x_j are the values of the target variable *x* at the two points; and μ is the (assumed constant) mean of the target variable.

The assumptions of simple kriging are rarely met for real-world populations as the mean is rarely known, or constant, and the covariance may not exist for all points or distances.

In contrast to simple kriging, ordinary kriging assumes: (1) the population mean does not depend on location, but does not need to be known; and (2) the variogram exists and depends only on the distance between the locations, not on the locations themselves. Ordinary kriging does not require that the covariance exists. This could occur if the variance is unbounded (i.e., infinite), such as for Brownian motion.

The variogram represents the average difference in the target parameter for all pairs of points separated by the same distance. The classical estimator of the variogram, *2γ(h)* or the variogram function of the distance *h* between two points, is:

$$2\gamma(h)=\frac{1}{N(h)}\sum_{N(h)}(x_i-x_j)^2$$

The factor of 2 is an artifact of the derivation of the variogram. In practice, the semivariogram, *γ(h)* is typically used and is classically estimated as:

$$\gamma(h)=\frac{1}{2N(h)}\sum_{N(h)}(x_i-x_j)^2$$

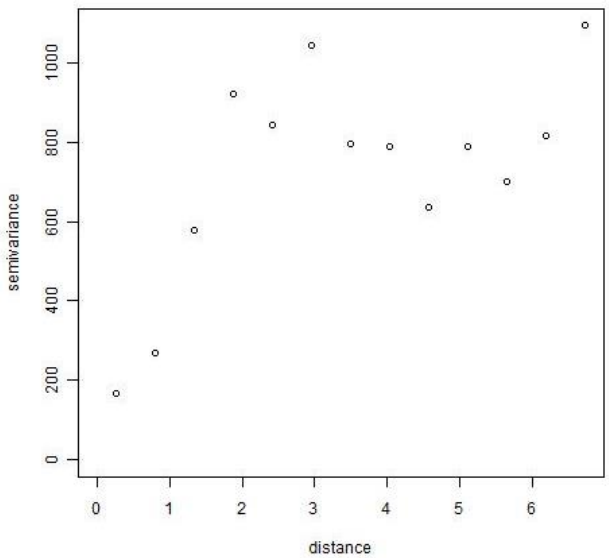
Other variogram estimators exist including the robust estimator:

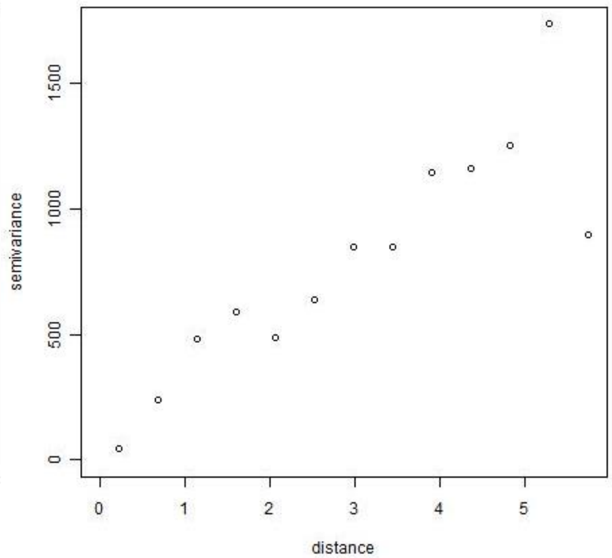
$$2\gamma(h)=\frac{\left(\frac{1}{N(h)}\sum_{N(h)}\left|x_i-x_j\right|^{\frac{1}{2}}\right)^4}{\frac{(0.457+0.494)}{N(h)}} \qquad \text{or} \qquad \gamma(h)=\frac{\left(\frac{1}{2N(h)}\sum_{N(h)}\left|x_i-x_j\right|^{\frac{1}{2}}\right)^4}{\frac{0.457+0.494}{N(h)}}$$

The variogram can exist even when covariance does not because the difference between two points can remain bounded while the product of the difference between the observed and mean value is infinite. However, the variogram will always exist when the covariance exists.

In practice, the assumptions of ordinary kriging are met more frequently than those of simple kriging. However, several real-world populations do not have a constant mean. Violation of this can be identified by plotting the semivariogram with distance, *h*, on the x-axis and the semivariogram estimator, *γ(h)*, on the y-axis. If the variogram levels off after a certain distance *h*, the assumptions of ordinary kriging are met. However, if the variogram is parabolic or continuously increasing, the mean may not be constant.

This is related to a basic principal of spatial statistics, points closer together are more related than points farther apart. This principal is illustrated by the semivariogram on the right (below). In this plot, the semivariogram shows a distinct trend for points close to one another (less than 2); the closer points are more similar and have a smaller semivariogram while the farther points are less similar and have a larger semivariogram. However, points vary from one another (greater than 2) have no relationship as the semivariogram is relatively constant with distance. In contrast, the semivariogram on the left (below) increases continuously with distance, indicating a spatial trend is over-riding the basic principal and the population may not have a constant mean. For example, if a variable in California increases from north to south, the semivariogram for that variable will continually increase as the distance increases.





If the semivariogram suggests a spatial trend is present in the population, a linear model should be built to de-trend the observations. The linear model could include spatially distributed explanatory variables, such as elevation, or even measures of space such as latitude and longitude. After the model is built, a semivariogram should be constructed using the residuals from the model. If this semivariogram no longer has the parabolic shape, ordinary kriging can be performed using the residuals from the linear model.

Finally, universal kriging assumes: (1) the mean is a function of the coordinates and (2) the variogram exists (for the residuals) and is dependent on the distance between points. Universal kriging can be useful when a two-step process using a linear regression model and ordinary kriging would be cumbersome.

The first step in kriging, after the type of kriging is chosen, is to compute the covariogram (for simple kriging) or the semivariogram (for ordinary and universal kriging) and fit a theoretical model to the observations. Several theoretical models have been developed. Because the covariogram, $C(h)$, is related to the variogram, $\gamma(h)$, by:

$$\gamma(h) = C(0) - C(h)$$

only the semivariogram models are presented.

Three of the more common semivariogram models include the linear, spherical, and exponential models. The linear model is given by two components:

$$\gamma(h)_{linear} = \begin{cases} 0, h = 0 \\ c_0 + bh, h \neq 0 \end{cases}$$

where c_0 is intercept and b is the slope. This model does not plateau. In contrast, the spherical model does plateau and is represented by three components:

$$\gamma(h)_{spherical} = \begin{cases} 0, h = 0 \\ c_0 + c_1\left(\frac{3}{2}\left(\frac{h}{\alpha}\right) - \frac{1}{2}\left(\frac{h}{\alpha}\right)^3\right), 0 < h \leq \alpha \\ c_0 + c_1, h \geq \alpha \end{cases}$$

where α is the distance at which the scatter-plot levels off, and $c_0 + c_1$ is the value of the semivariogram after this point. Finally, the exponential model also plateaus but does not remain at a fixed value. This model is given in two components:

$$\gamma(h)_{\text{exponential}} = \begin{cases} 0, h = 0 \\ c_0 + c_1\left(1 - \exp\left(-\frac{h}{\alpha}\right)\right), h \neq 0 \end{cases}$$

Selection of the appropriate semivariogram model is critical as it determines the weighting in later steps of kriging.

After the variogram is modeled, the value at an unobserved location, x_0, is computed. For simple kriging, the predicted value is given by:

$$x_0 = \sum_{i=1}^{n} w_i x_i + \left(1 - \sum_{i=1}^{n} w_i\right)\mu$$

where w_i is the weight given to the observation at location i, and μ is the population mean. The weight is computed using the covariogram model and the distance between the observed and the unobserved location. The weights are computed so they will sum to one. The variance of the predicted value is given by:

$$V\left[x_{0,simple}\right] = C(0) - \sum_{i=1}^{n} w_i C(s_0 - s_i)$$

where $C(0)$ and $C(s_0 - s_i)$ refers to the covariogram model for a distance of zero or for the distance between points s_0 and s_i.

If ordinary kriging is used, the predicted values are calculated by:

$$x_0 = \sum_{i=1}^{n} w_i x_i$$

where the weights are computed using the theoretical semivariogram model. The variance of the predicted value is given by:

$$V[x_0] = 2\sum_{i=1}^{n} w_i \gamma(s_i - s_0) - \sum_{i=1}^{n}\sum_{j=1}^{n} w_i w_j \gamma(s_i - s_j)$$

where $\gamma(s_i - s_0)$ and $\gamma(s_i - s_j)$ are shorthand for the semivariogram model.

Finally, in universal kriging, the predicted value is given by:

$$x_0 = \beta_0 + \beta_1 \sum_{i=1}^{n} w_i y_{1i} + \beta_2 \sum_{i=1}^{n} w_i y_{2i} + \sum_{i=1}^{n} w_i \delta_i$$

when x is predicted by the explanatory variables y_1 and y_2 in the linear model:

$$x = \beta_0 + \beta_1 y_1 + \beta_2 y_2 + \delta$$

where β are the linear coefficients and δ is the error term. The universal kriging predictor can be altered as appropriate for different linear models. The variance of the predicted value is calculated similarly to ordinary kriging.

APPENDIX I. Estimated Sample Sizes and Predicted Total Costs

	Plot Size (km^2)	95% Confidence Interval ± 5%	95% ± 10%	95% ± 15%	95% ± 20%	90% Confidence Interval ± 5%	90% ± 10%	90% ± 15%	90% ± 20%
Required statewide sample size	*1*	8,700	2,200	970	550	6,100	1,600	680	390
	4	7,200	1,800	800	450	5,100	1,300	560	320
	9	7,000	1,800	780	440	5,000	1,300	550	310
	16	6,700	1,700	750	420	4,700	1,200	530	300
Cost *($450/ plot imagery)*	*1*	$ 4,132,500	$ 1, 045,000	$ 460,750	$ 261,250	$ 2,897,500	$ 760,000	$ 323,000	$ 185,250
	4	$ 3,960,000	$ 990,000	$ 440,000	$ 247,500	$ 2,805,000	$ 715,000	$ 308,000	$ 176,000
	9	$ 4,725,000	$ 1,215,000	$ 526,500	$ 297,000	$ 3,375,000	$ 877,500	$ 371,250	$ 209,250
	16	$ 5,695,000	$ 1,445,000	$ 637,500	$ 357,000	$ 3,995,000	$ 1,020,000	$ 450,500	$ 255,000
Cost *($150/ plot imagery)*	*1*	$ 1,522,500	$ 385,000	$ 169,750	$ 96,250	$ 1,067,500	$ 280,000	$ 119,000	$ 68,250
	4	$ 1,800,000	$ 450,000	$ 200,000	$ 112,500	$ 1,275,000	$ 325,000	$ 140,000	$ 80,000
	9	$ 2,625,000	$ 675,000	$ 292,500	$ 165,000	$ 1,875,000	$ 487,500	$ 206,250	$ 116,250
	16	$ 3,685,000	$ 935,000	$ 412,500	$ 231,000	$ 2,585,000	$ 660,000	$ 291,500	$ 165,000
Cost *(no-cost imagery)*	*1*	$ 217,500	$ 55,000	$ 24,250	$ 13,750	$ 152,500	$ 40,000	$ 17,000	$ 9,750
	4	$ 720,000	$ 180,000	$ 80,000	$ 45,000	$ 510,000	$ 130,000	$ 56,000	$ 32,000
	9	$ 1,575,000	$ 405,000	$ 175,500	$ 99,000	$ 1,125,000	$ 292,500	$ 123,750	$ 69,750
	16	$ 2,680,000	$ 680,000	$ 300,000	$ 168,000	$ 1,880,000	$ 480,000	$ 212,000	$ 120,000

	Plot Size (km^2)	85% Confidence Interval ± 5%	85% ± 10%	85% ± 15%	85% ± 20%	80% Confidence Interval ± 5%	80% ± 10%	80% ± 15%	80% ± 20%
Required statewide sample size	*1*	4,700	1,200	520	300	3,700	930	420	240
	4	3,900	970	430	250	3,100	770	340	200
	9	3,800	950	420	240	3,000	750	340	190
	16	3,600	900	400	230	2,900	720	320	180
Cost *($450/ plot imagery)*	*1*	$ 2,232,500	$ 570,000	$ 247,000	$ 142,500	$ 1,757,500	$ 441,750	$ 199,500	$ 114,000
	4	$ 2,145,000	$ 533,500	$ 236,500	$ 137,500	$ 1,705,000	$ 423,500	$ 187,000	$ 110,000
	9	$ 2,565,000	$ 641,250	$ 283,500	$ 162,000	$ 2,025,000	$ 506,250	$ 229,500	$ 128,250
	16	$ 3,060,000	$ 765,000	$ 340,000	$ 195,500	$ 2,465,000	$ 612,000	$ 272,000	$ 153,000
Cost *($150/ plot imagery)*	*1*	$ 822,500	$ 210,000	$ 91,000	$ 52,500	$ 647,500	$ 162,750	$ 73,500	$ 42,000
	4	$ 975,000	$ 242,500	$ 107,500	$ 62,500	$ 775,000	$ 192,500	$ 85,000	$ 50,000
	9	$ 1,425,000	$ 356,250	$ 157,500	$ 90,000	$ 1,125,000	$ 281,250	$ 127,500	$ 71,250
	16	$ 1,980,000	$ 495,000	$ 220,000	$ 126,500	$ 1,595,000	$ 396,000	$ 176,000	$ 99,000
Cost *(no-cost imagery)*	*1*	$ 117,500	$ 30,000	$ 13,000	$ 7,500	$ 92,500	$ 23,250	$ 10,500	$ 6,000
	4	$ 390,000	$ 97,000	$ 43,000	$ 25,000	$ 310,000	$ 77,000	$ 34,000	$ 20,000
	9	$ 855,000	$ 213,750	$ 94,500	$ 54,000	$ 675,000	$ 168,750	$ 76,500	$ 42,750
	16	$ 1,440,000	$ 360,000	$ 160,000	$ 92,000	$ 1,160,000	$ 288,000	$ 128,000	$ 72,000