HABITAT VALUE AND TREATMENT EFFECTIVENESS OF FRESHWATER URBAN WETLANDS

Martha Sutula
Jeff Brown
Elizabeth Fetscher
Michelle Mattson
Sharook Madon
Gary Santolo
Earl Byron
Chris Stransky

Southern California Coastal Water Research Project

Technical Report 559 - March 2008

HABITAT VALUE AND TREATMENT EFFECTIVENESS OF FRESHWATER URBAN WETLANDS

Final Report to

Los Angeles Regional Water Quality Control Board
for the Assessment of Effectiveness of Treatment Wetlands for Stormwater BMPs and Compatibility with Wildlife Beneficial Uses,
Agreement No. 04-090-554-0

Martha Sutula, Jeff Brown and Elizabeth Fetscher

Southern California Coastal Water Research Project
Costa Mesa, CA
www.sccwrp.org

Michelle Mattson

Aspen Environmental, Inc.
Carlsbad, CA

Sharook Madon, Gary Santolo and Earl Byron

CH2M Hill, Inc.
Santa Ana, CA

Chris Stransky

Nautilus Environmental, Inc.
San Diego, CA

March 29, 2008

Technical Report 559

EXECUTIVE SUMMARY

With the rising urbanization of the coastal watersheds of southern California, requirements for municipal control of runoff quantity and quality have created a demand for wetlands as effective, low-cost best management practices (BMPs) to improve surface water quality and attenuate storm flows. There is equal pressure to restore, enhance, and create wetlands with multiple objectives (i.e., habitat support, treatment of nonpoint source pollution, flood attenuation, and recreation). The potential risk to wildlife associated with using wetlands for treatment of nonpoint source runoff, and the trade-offs between habitat and water quality objectives are not well quantified. To address this information gap, the Los Angeles Regional Water Quality Control Board and the State Coastal Conservancy funded a study to evaluate habitat value associated with urban wetlands of multiple objectives in southern California. The goal of this project was to provide information on how these urban wetlands can be better managed to increase compatibility with wildlife protection in southern California. A phased approach was used to allow data collected in the first phase drive decisions about objectives and study design in subsequent phases. The four major phases of this project included:

- Develop an inventory of existing urban and treatment wetland projects;
- Conduct a biological survey on a representative sample of urban wetlands to evaluate wildlife beneficial use; and
- Evaluate the exposure and toxicity to wildlife from sediment-borne contaminants
- Conduct analysis on the effectiveness of treatment wetlands using existing monitoring data

Below are the major findings and management recommendations of the study.

Findings with Respect to Habitat Value

Type Conversion typical. The 40 freshwater urban wetlands were highly modified from historical reference, with a large percentage type converted to forms atypical in the landscape.

Appropriate Reference for Urban Wetlands. Reference for basin wetlands in this study represents an underlying shift of baselines towards systems that are historically rare or never present in the southern California landscape. An adequate understanding of reference is hampered by the lack of true reference wetlands, particularly for basin wetlands, which were mostly characterized by perennial ponds. Historical ecology studies show that perennial ponds were less than 1% of wetlands in the San Gabriel River watershed. Type conversion of stream habitat to in-channel basins and the formation of linear channels with extended detention (e.g., the "bioswale") represents habitat for which no reference exists. Thus, the reference sites used in this study reflect a state of "best achievable", but do not necessarily provide an adequate benchmark for the characteristics of a basin wetland in pristine ecological condition. Additional studies of the historical ecology and present day distribution of native flora and fauna associated with freshwater depressional (basin) wetlands is greatly needed to provide a better understanding of reference.

Presence and Magnitude of Risk from Contaminants. Most wetlands posed a risk of elevated sediment contaminants and/or toxicity. Eighteen of the twenty-one urban wetlands were either toxic to the amphipod *Hyalella azteca*, or exceeded a sediment quality guideline, or both. Although limited, reference site sediment chemistry, and toxicity data show a clear distinction from the majority of urban sites. An index of degree of sediment contamination was found to negatively correlate with benthic macroinvertebrate diversity in these wetlands. Macroinvertebrates are a critical link in the food web of wetlands, providing the link between primary producers, detritivores, and higher level consumers such as birds, fish, and amphibians. Adverse effects can also occur on amphibian, birds, and fish can occur via bioaccumulation or direct toxicity.

Pyrethroid concentrations were elevated at all 10 sites that were toxic to *H. azteca*, and the mean pyrethroid quotient was negatively correlated with amphipod survival, suggesting that this class of compounds may have been responsible for much of the toxicity observed in this study. Confirmation studies would need to be conducted in order to determine definitely the source of toxicity, which could be expected to vary at each wetland site. Additional information on seasonal and spatial variation in sediment contaminant concentrations within wetlands is needed to better understand the magnitude of the risk posed to wildlife and to identify management options to mitigate that risk.

Urban Infrastructure Constrains Condition; Management can Mitigate Constraints. Results from the biological survey illustrate urban infrastructure provides basic constraints on "best achievable" wetland condition. Cu, Pb, Zn, PAHs, andcypermethrin were significantly correlated with percent imperviousness of the catchment area, an index of percent urbanization. Sediment toxicity and sediment pyrethroid concentration was not significantly correlated with the degree of urbanization. Site specific factors such as wetland project design criteria, objectives, wetland management and maintenance activities can mitigate to some degree the constraints of the urban landscape. Contaminant source control and pretreatment, good designs for wetland creation, restoration of enhancement as well as active management of stressors (hydromodification, increased sedimentation, contaminant exposure, excessive visitation, predation from urban wildlife, exotic species) may mitigate constraints of urban infrastructure to some degree.

Differences by Project Objective and Design Criteria. Urban wetlands that have been created, restored or enhanced for habitat rather than for water quality may be constrained with respect to potential the type or condition. However, this study did not establish, by weight of evidence across all indicators used, that habitat wetlands had statistically significant, superior condition relative to multipurpose or treatment wetlands. Habitat wetlands posed just as much risk of elevated contaminants as treatment or multipurpose wetlands. Sediment chemistry concentrations and toxicity were not significantly different among habitat, water quality and multifunctional wetlands. These results are similar to those found for several other indicators of habitat quality, including benthic macroinvertebrate and bird diversity. This large variability in condition of habitat wetlands can be attributed to the constraints of urban infrastructure as well as factors such as the lack of maintenance at some habitat sites. Significant differences were found in basin wetlands with respect to dominant habitat type, plant diversity, and physical structure. Habitat wetlands had twice the relative area of riparian habitat and roughly half the amount of open water as treatment wetlands. For treatment wetlands, more emphasis will likely be placed on providing treatment through open water and emergent marsh rather than providing riparian habitat. Multipurpose wetlands had significantly higher plant species richness than habitat or treatment wetlands; establishment of plant cover with native species is a common permit requirement for mitigation wetlands, which made up a significant proportion of the multipurpose sites. Multipurpose and treatment wetlands had significantly lower CRAM physical structure scores than habitat wetlands, which were characterized by oval configurations shorelines, steep slopes and lack of macro- and microtopography. Treatment wetlands must be designed and regularly maintained to optimize treatment capacity, so the physical configuration of these wetlands may be an element required to optimize flow conditions or provide easy access for maintenance and vector control. An increased understanding of the importance of the elements of good physical structure may help improve the quality of habitat provided in restoration and mitigation projects.

Wetland Size and Origin. Wetland size was a major controlling factor on the condition of urban basin wetlands. The majority of urban wetlands in this study were very small, with most basin sites under five acres. The condition of the wetland with respect to indicators were all significantly negatively correlated to size. Wetland size is often constrained by adjacent land uses, especially in an urbanized landscape, so is not a factor that is easily managed, it is an element that can be taken into account when prioritizing sites for restoration. It is also important to recognize the value of small wetlands in a fragmented highly urbanized environment. Particularly in arid or semi-arid, highly urbanized areas, wildlife are attracted to aquatic habitats in great numbers because of natural scarcity of such resources. The condition of the habitat provided by historic wetlands was significantly higher than that provided by wetlands that have been type converted from streams and/or floodplain habitat or created from upland habitat. Basins that were type converted from other historic stream habitat represented 68% of the sample population.

Maintenance Activities Intensity. This study showed that urban wetlands must be maintained frequently to manage the variety of stressors, but not at an intensity or in a manner that may be incompatible with the seasonal cycles of nesting and reproduction.

Findings with Respect to Treatment Effectiveness

Effectiveness of Treatment Wetlands Hampered by Lack of Flow Data Analysis. Data analyses conducted on existing monitoring data are inconclusive because of lack of flow data required to calculate loads. Modeling of wetland BMPs is needed to provide a time-integrated picture of water and contaminant budgets that can lead to better calculations of treatment efficiencies. Standardized monitoring of treatment wetland projects can provide the data needed to develop these models.

Treatment Wetlands Reduce Contaminant Concentrations. Existing monitoring data show that southern California treatment wetlands reduce the concentrations of all constituent of interest [i.e., total and dissolved metals (Cu, Pb, Zn, Se), nutrients (nitrate, ortho phosphate), total suspended solids (TSS), and bacteria (*Enterococcus*, *Escherichis coli*, fecal and total coliforms)] relative to inflow concentrations. For dissolved Cu, Pb, and Zn, southern California treatment wetlands were effective at reducing wet season inflow concentrations to below water quality criteria. Southern California treatment wetlands showed 1-2 order of magnitude reductions in *E. coli*, *Enterococcus,* fecal coliform, total coliform and nutrients. Great variability was found in the effectiveness of removal.

Comparison of Dry Versus Wet Weather Performance. Percent reductions and inflow concentrations can vary greatly by wintertime wet and dry weather and dry season. Percent removal of contaminants typically associated with suspended solids (Zn, Cu and Pb, metals, phosphate, and enteric bacteria) had a higher percent reduction in concentration during wet than dry weather. Differences in concentrations of total metals among outflows by weather and season were generally not significant. Dry season concentrations of nitrate were generally the highest, though only significantly different from wet weather concentrations. No significant differences were observed by weather or season among inflow or outflow concentrations of phosphate. Treatment wetlands may be at times a source of contaminants relative to inflow concentrations, but this typically occurs at low concentrations, indicating that it is not likely to be significant source.

Comparison of Treatment Wetlands in Arid Versus Temperate Climates. Some differences in contaminant concentrations in treatment wetland inflows were found between semi-arid and temperate climates. Total Cu, nitrate and phosphate were higher in the inflows to southern California treatment wetlands relative to temperate sites in the International Stormwater database. The opposite was found for TSS and *E. coli*, and for all other constituents. Removal of dissolved Pb appeared to be more efficient in semi-arid systems, while that of dissolved Zn appeared to be efficient in temperate treatment wetlands. In order to confirm these trends, more would need to be understood about the size, soils, geomorphology and hydrodynamics of the systems from which the data are derived in order to tease out true differences in climate. Engineering “rules of thumb” based on temperate climates should be used with caution when designing sites in Southern California.

Management Recommendations to Improve the Habitat Value and Treatment Effectiveness of Urban Wetlands

The following recommendations are given to improve the habitat value of urban wetlands and improve the monitoring of effectiveness and habitat in all urban wetlands. These recommendations can be organized into four categories: 1) watershed planning, 2) design elements, 3) management and maintenance, and 4) minimum standardized monitoring. Each of the recommendations is explained in full detail in the final section of the report.

Watershed Planning Perspectives

- Consider that a watershed wide plan to reduce urban runoff can be greatly aided by watershed scale conservation and restoration of natural wetlands and riparian areas.
- Reduce potential for onsite exposure and toxicity to contaminants by incorporating low impact development, source control and BMP implementation upstream of wetlands.
- When possible, conduct pretreatment of wetland water source; A variety of treatment strategies, including detention, pre-treatment, treatment, and infiltration, should be incorporated in series in order to maximize removal efficiencies and minimize exposure to wildlife.
- Locate BMPs throughout the watershed.

Creation or Restoration Design Elements

- If habitat is an objective of a project to create wetlands, clearly state what the management endpoints are and how the project design is linked to those endpoints.
- Locate the site in an area that can support wetland hydrology.
- Assess the potential risk from contaminants characterizing major water sources and wetland sediments.
- Design the site to have good physical structure.
- Maximize the diversity of habitats within the wetland and transitional upland areas.
- Design and maintain the wetland and transitional upland buffer to have appropriate width and native vegetation.

Management and Maintenance

- Wetland stewardship is essential in urban areas.
- Manage the hydrology to mimic the natural hydroperiod.
- Manage urban stressors (e.g., contaminants, human visitation, invasive plants, trash, excessive sedimentation, etc).
- Maintain the wetland at a frequency necessary to manage stressors, but not in a manner that is incompatible with the seasonal cycles of nesting and reproduction.

Monitoring

- Conduct Monitoring, with intensity scaled to to the project size, but core standardized elements recommended in report should remain intact regardless of project size.

ACKNOWLEDGEMENTS

The authors of this study would like to thank the owners and managers of each of the wetland sites that were assessed in this study. Without their assistance this study would not have been possible. We would also like to acknowledge the role of the interagency technical advisory committee who provided valuable suggestions and guidance throughout the study. Funding for this study was provided by the Los Angeles Regional Water Quality Control Board (Agreement No. 04-090-554-0), and the State Coastal Conservancy under a contract to the Southern California Wetland Recovery Project.
.

TABLE OF CONTENTS

Executive Summary i
- Findings with Respect to Habitat Value i
- Findings with Respect to Treatment Effectiveness iii
- Management Recommendations to Improve the Habitat Value and Treatment Effectiveness of Urban Wetlands iv
  - Watershed Planning Perspectives iv
  - Creation or Restoration Design Elements iv
  - Management and Maintenance iv
  - Monitoring iv

Acknowledgements v

Table of Contents vi

List of Tables ix

List of Figures xi

Introduction 1
- Background 1
- Goals of the study 1
- Organization of the Report 2

Inventory of Southern California Freshwater Urban wetlands 3
- Abstract 3
- Introduction 4
- Methods 4
  - Definition of Urban Wetlands 4
  - Development of the Wetland Inventory 5
  - Site Selection 6
  - Data Fields and the ACCESS Database Design 6
- Results and Discussion 8
  - Finalized List of Sites 8
  - Database Completeness 10
  - Geographic Distribution of Sites 10
  - Wetland Objective and Origin 12
  - Wetland Type 13
  - Wetland Water Sources 16
  - Wetland Project Type 16
  - Wetland Flow Type 16
  - Catchment Land Uses 17
  - Types of Operation and Maintenance Activities and Frequency 17
  - Types of Monitoring Activities and Frequency 17

Habitat Value of Southern California Urban Wetlands: Results of a Biological Survey 20
- Abstract 20
- Introduction 20
- Methods 21
  - Definition of Target Population 21
  - General Design of Biological Survey 22
  - Data Analysis 24
- Results 25
  - Summary of Urban Wetland Characteristics 25
  - Condition of Wetlands Along a Gradient of Urbanization 26

Impact of Objective on Wetland Condition ... 29
Ecological Condition of Urban Wetlands as a Function of Other Characteristics ... 38
Discussion ... 41
Reference Wetlands in an Urban Landscape ... 41
Landscape Context Provides Basic Constraints on Achievable Condition of Urban Wetlands ... 42
Site-Specific Factors Impacting Condition of the Wetland ... 43
Recommendations ... 45
Study of Sediment Contaminant Chemistry and Toxicity in Urban Wetlands ... 47
Abstract ... 47
Introduction ... 47
Methods ... 48
General Study Approach ... 48
Characteristics of Sites ... 48
Field Methods ... 48
Laboratory Analysis ... 49
Data Analysis ... 49
Method ... 52
Results ... 53
Sediment Chemistry ... 53
Toxicity ... 64
Relationship between Chemistry and Toxicity ... 64
Discussion ... 65
Risk to Benthic Organisms from Sediment Contaminants ... 65
Contaminants Responsible for Toxicity ... 66
Implications of Toxicity to Higher Level Wetland Organisms ... 67
Factors Associated With Sediment Contamination and Toxicity ... 67
Management Recommendations ... 68
Effectiveness of Southern California Wetlands for Treatment of Urban Runoff: An Analysis of Existing Monitoring Data ... 70
Abstract ... 70
Introduction ... 71
Methods ... 74
Results ... 76
Effectiveness of Southern California Treatment Wetlands ... 76
Affect of Wet/Dry Weather and Season ... 76
Semi-arid and Temperate Climates ... 82
Discussion ... 87
Effectiveness of Southern California Treatment Wetlands in Reducing Contaminants ... 87
Differences in Treatment Effectiveness by Dry/Wet Weather or Season ... 88
Difference in Treatment Effectiveness by Climate ... 89
Recommendations ... 90
Summary of Findings and Management Recommendations ... 97
Study Findings ... 97
Inventory of Urban Wetlands ... 97
Biological Survey ... 97
Factors Associated With Sediment Contamination and Toxicity ... 100
Analysis of Existing Monitoring Data to Assess Treatment Wetland Effectiveness ... 101
Management Recommendations to Improve the Habitat Value of Urban Wetlands ... 103
Watershed Planning Perspectives ... 103
Creation or Restoration Design Elements ... 104

Management and Maintenance 105
Monitoring 106
References 107

LIST OF TABLES

Table 1 Final List of Urban and Treatment Wetlands in Southern California. 9

Table 2. Estimated Completeness of the Inventory Database with Respect to Major Categories of Existing Data. 10

Table 3. Indicator and Objective of Sampling Conducted for Biological Survey 23

Table 4. List of CRAM attributes and metrics. The overall site score is calculated as the percentage of the maximum possible site score by summing the attribute scores and dividing the sum by the maximum possible site score. See CRAM Version 4.0 for more details (Collins *et al.* 2006). 24

Table 5. Approximate allocation of urban sites by wetland type and habitat objective............................ 25

Table 6. Summary statistics describing correlation between CRAM index scores and attributes and the percent imperviousness in the surrounding land use and catchment of basin wetland sites. Data for channels are not presented... 27

Table 7. Summary statistics describing correlation between BMI metrics and percent imperviousness of catchment land use. P-valuesα=0.05 and the model R2 for correlation are given. Correlations with surrounding land use (0-100 m) were generally not significant at an α= 0.05. 28

Table 8 Summary statistics describing correlation between plant community metrics and percent imperviousness of surrounding land use (0-100 m). P-valuesα=0.05 for the % imperviousness term and the model R2 are given. 28

Table 9 Summary of ANOVA statistics and post-hoc Tukey's comparisons for CRAM index and physical structure scores by project objective for basins (n=26) and channels (n=14), excluding reference sites. Mean and standard deviation of reference site data shown for comparison. ANOVAs of CRAM biological structure, landscape context, and hydrology were not significant with respect to objective at α=0.05. 32

Table 10 Summary of ANOVA statistics and post-hoc Tukey's comparisons by objective (or plant community metrics by basins and channels, excluding reference sites. Mean and standard deviation of reference site data shown for comparison. Mean values with the same superscript letters are not significantly different (p>0.05) from one another based on pair-wise comparisons. Where no superscripts are shown, no significant differences were found. 34

Table 11. Summary of Kruskal-Wallis, nonparametric test comparisons by project objective for BMI community metrics by basins (N=16) and channels (N=10), excluding reference sites. Mean and standard deviation of reference sites provided for comparison. No significant differences were found. 35

Table 12. Summary of ANOVA statistics and post-hoc Tukey's comparisons for avian survey metrics by basins (N=18) and channels (N=10), excluding reference sites. Mean values with the same superscript letters are not significantly different (p>0.05) from one another based on pair-wise comparisons. Where no subscripts are shown, no significant differences were found. 37

Table 13. Summary of the of the ANOVA statistics and post-hoc Tukey's comparisons (mean and confidence interval) for CRAM index, physical structure scores and native plant species richness by origin for basins (n=29). 39

Table 14. Recommended core "minimum" monitoring for habitat and examples of intensive monitoring that could be required for urban wetlands. 46

Table 15. Reporting levels and analysis methods for chemical constituents measured. 52

Table 16. Freshwater sediment quality guidelines used to evaluate the sediment chemistry concentrations. NG = no guideline. An observed effects threshold of 4 mg/kg was used for Se (USDOI 1998).. 54

Table 17. Summary of constituents exceeding sediment quality guideline thresholds, and an indication of toxicity at each of the study sites. 62

Table 18 Relationship between sediment constituents (contaminant concentrations or amphipod survival) and wetland habitat parameters. 63
Table 19. Characteristics of water quality analyses for southern California wetlands that had available data. 73
Table 20. California Toxics Rule freshwater chronic water quality criteria used for comparison with the effluent data. 74
Table 21. Comparison between natural log transformed inflow and outflow values from southern California wetlands. Paired t-tests were used for the evaluations. 76
Table 22. Comparison between percent reduction between treatment wetlands in semi-arid versus temperate climates. T-tests were used for the evaluations. 82
Table 24. Design overview for monitoring treatment wetlands. 91
Table 25. Annual number of wet and dry weather samples required to be 95% confident what the measured outflow values are in relation to the chronic water quality criteria. NA = not enough samples for analysis. ND = non-detect. 95

LIST OF FIGURES

Figure 1. Diagram of process used to develop urban and treatment wetland inventory. 5
Figure 2. Example of a data entry screen for project contact information. 7
Figure 3. Map showing geographic distribution of inventory sites in southern California. Note inset of map shows an enlargement of site distribution in Orange County, where the largest number of sites was located. Two sites in Imperial County are not shown. 11
Figure 4. Number of inventory sites by county (a) and by county and project objective (b). 12
Figure 5. Distribution of sites by origin (created or natural) and project objective. 13
Figure 7. Examples “basin” class of wetlands 14
Figure 8. Example of “channel” class wetlands. 15
Figure 9. Distribution of sites by wetland type and project objective. 15
Figure 10. Distribution of site by project objective and flow management (passively or actively managed). 16
Figure 11. Distribution of sites by type of maintenance activity and project objective. 17
Figure 12. Distribution of sites by type of wetland monitoring and project objective. 18
Figure 13. Distribution of sites by type of monitoring parameter measured. 19
Figure 14. Yellow triangles designate the location of urban wetlands included in the study. Most sites were located within the coastal watersheds of southern California, located on the pacific coast of the United States. 22
Figure 15. Histogram of size distribution of wetlands by channels (n= 15) and basins (n=29). Wetland boundaries are defined by rules for CRAM assessment area delineation (Collins *et al.* 2006). 26
Figure 16. Scatter plots and linear regressions of native plant species richness versus % imperviousness of surrounding land use (0-100 m, red) and catchment (black), showing a positive relationship between the two. 29
Figure 17. Number of mapped native plant communities as a function of objective and wetland type, showing significantly higher number of plant communities associated with habitat and multipurpose wetlands than with water quality wetlands. Error bars represent 95 % confidence intervals. 30
Figure 18. Distribution of CRAM index scores relative to reference standard sites for basins and channel wetlands. The red bar indicates scores for the reference sites. Black indicates the scores for the urban wetland inventory sites. 31
Figure 19. CRAM physical structure scores and native plant species richness for basin sites as a function of objective. Error bars represent 95 % confidence intervals. 32
Figure 20 Distribution of plant species richness and relative percent cover of invasive plant species in urban sites relative to reference standard sites for basins and channel wetlands. Red bars indicate scores for reference sites. Black bars indicate scores for urban wetland inventory sites. 33
Figure 21. Distribution of BMI taxa richness in urban (study) sites relative to reference standard sites for basins and channel wetlands. 36
Figure 22 Distribution of avian diversity and species richness in urban (study) sites relative to reference standard sites for basins and channel wetlands. Red bars indicate reference sites. Black bars indicate data for urban inventory sites. Correction for size places the reference sites within the top 25% of sites for both indicators. 38
Figure 23. Differences in plant species richness at urban wetland sites as a function of the number of times maintenance is conducted at the site. The category sites with of 1-2X per year has significantly greater plant species richness than the >2 X per year category. 39

Figure 24. Relationships between Buffer Overall Quality and Relative Percent Cover of Invasive Plant Species. 40
Figure 25. Graphic reproduced from Stein *et al.* (2007) on the historic extent and distribution of wetlands in the San Gabriel River watershed. Distribution of historical wetland types in various portions of the San Gabriel River floodplain. Note that approximately 9,300 ha of alkali meadow existed in the tidal fringe area; the plot has been truncated to facilitate presentation. 42
Figure 26. Locations of the wetland sites sampled. The freeways in Ventura, Los Angeles, Orange, and San Diego Counties have been added for reference. 51
Figure 27. Sediment metals concentrations at each site. The dashed line indicates the Probable Effects Concentration, except for Se, where the line is the observed effects threshold from Van Derveer and Canton 1997 55
Figure 28. Sediment concentrations of chlorinated pesticides and PCBs at each site. The dashed line indicates the Probable Effects Concentration. Nondetects were replaced with half the method detection limit. 57
Figure 29. Sediment concentrations of PAHs at each site. The dashed line indicates the Probable Effects Concentration. 58
Figure 30. Sediment concentrations of pyrethroid pesticides at each site. The dashed line is the average sediment LC50 for *H. azteca* from Amweg *et al.* 2005. 61
Figure 31. Relationship between mPECq and BMI richness. The vertical dashed line is the threshold above which toxicity is expected. 63
Figure 32. Survival of *H. azteca* and *C. tentans* exposed to wetland sediments. * = significantly different and <80% of control value. 64
Figure 33 Mean Probable Effects Concentration quotients (mPECq) for sediments from nontoxic and toxic sites. The differences in mPECq between the toxic and nontoxic sites were not significant (p=0.47, ln transformed mPECq). 65
Figure 34. Natural log transformed concentrations of dissolved Cu, Pb, Zn and total Se before and after treatment of the dry and wet weather samples from southern California wetlands. Inflow and outflow mean concentrations (circles) and 95% confidence intervals are shown. The dashed lines indicate the water quality criteria. Significant differences between inflow and outflow samples are indicated by the asterisk (*). Letters indicate statistically similar inflow concentrations, while numbers indicate similar outflow concentrations. 78
Figure 35. Natural log transformed concentrations of nutrients, TSS, and bacteria before and after treatment of the dry and wet weather samples from southern California wetlands. Inflow and outflow mean concentrations (circles) and 95% confidence intervals are shown. Significant differences between inflow and outflow samples are indicated by the asterisk (*). Letters indicate statistically similar inflow concentrations, while numbers indicate similar outflow concentrations. 79
Figure 36. Natural log transformed concentrations of metals vs % reduction for different weather/season categories for samples from southern California wetlands. 80
Figure 37. Natural log transformed concentrations of nutrients, TSS and bacteria vs % reduction for different weather/season categories for samples from southern California wetlands. 81
Figure 38. Natural log transformed concentrations of wet weather total and dissolved Cu, Pb, Zn, total Se before and after treatment by the semi-arid and temperate climate wetlands. Inflow and outflow mean concentrations (circles) and 95% confidence intervals are shown. The dashed lines indicate the California Toxics Rule water quality criteria. Significant differences between inflow and outflow samples are indicated by the asterisk (*). Letters indicate statistically similar inflow concentrations, while numbers indicate similar outflow concentrations. 83

Figure 39. Natural log transformed concentrations of wet weather nutrients, TSS, and *E. coli* before and after treatment by the semi-arid and temperate climate wetlands. Inflow and outflow mean concentrations (circles) and 95% confidence intervals are shown. Significant differences between inflow and outflow samples are indicated by the asterisk (*). Letters indicate statistically similar inflow concentrations, while numbers indicate similar outflow concentrations. 84
Figure 40. Natural log transformed inflow metal concentrations vs % reduction for stormwater samples from temperate (International stormwater database) and semi-arid (southern California) climates. 85
Figure 41. Natural log transformed inflow concentrations of nutrients, TSS and bacteria vs % reduction for stormwater samples from temperate (International stormwater database) and semi-arid (southern California) climates... 86
Figure 42. Sampling effort required in order to determine what the measured outflow copper concentration is in relation to the chronic water quality criterion, for a given level of confidence. 94
Figure 43. Sampling frequency approach for chemicals with reporting levels near California Toxics Rule values. P0 = acceptable probability of exceedance (used for lowest line). P1 = unacceptable probability of exceedance (used for upper line(s)). α1 = probability that increased sampling is mandated when the probability of exceedance is actually below P1. α2 = probability that management action (Pollution Minimization Program, PMP) is mandated when the probability of exceedance is actually below P1. β = probability that reduced sampling is mandated when the probability of exceedance is actually above P0. 96

INTRODUCTION

Background

With the rising urbanization of the coastal watersheds of southern California, runoff from watersheds and discharges of nonpoint source pollution have increased dramatically. Requirements for municipal control of runoff quantity and quality have created a demand for wetlands as effective, low-cost best management practices (BMPs) to improve surface water quality and attenuate storm flows. Concurrent with this need, wetlands and riparian areas have been rapidly disappearing from the landscape, and those that remain are often highly degraded. As a result, there is increasing pressure to restore, enhance, and create natural or artificial wetlands with multiple objectives (i.e., habitat support, treatment of nonpoint source pollution, flood attenuation, and recreation). While the California Nonpoint Source (NPS) Plan calls for protecting and restoring wetlands and riparian areas and using vegetated treatment systems as a means to control NPS pollution, it also specifically states that wetlands and riparian areas should be protected from any adverse effects if they are harnessed to treat NPS pollution.

The extent to which wetlands can provide multiple benefits is not clear. Furthermore, the potential risk to wildlife associated with using wetlands for treatment of nonpoint source runoff, and the trade-offs between habitat and water quality objectives are unquantified. As the number of proposals requesting bond money for treatment wetlands increases, and as municipalities propose to use wetlands to comply with requirements of their NPS and National Pollution Discharge Elimination System (NPDES) permits, there is an increasing need for more information on: 1) effectiveness of treatment wetlands as a BMP for urban runoff, 2) circumstances where wetland BMPs for urban runoff are not compatible with wildlife beneficial uses, 3) modifications to siting, design or management criteria that will mitigate adverse impacts to wildlife, and 4) minimum recommended criteria to monitor habitat value and treatment effectiveness. In light of these stated needs, the State Coastal Conservancy, in consultation with Regional Boards (4, 8, and 9) and other state resource agencies, commissioned a literature review on the habitat value of treatment wetlands. This literature review concluded: 1) the effectiveness of wetlands as a BMP for urban runoff was not well documented, 2) adequate monitoring does not exist documenting whether wetlands used to treat urban runoff become environmental hazards for wildlife, 3) concerns of risk to wildlife are valid based on documented examples of adverse impacts to wildlife in wetlands accidentally receiving urban runoff, but the magnitude of the risk is unknown, and 4) the lack of literature on the effectiveness and impacts of using wetlands to treat urban runoff is most acute for semi-arid and arid climates such as southern California (Sutula and Stein 2003). In addition, the literature review outlined a series of data gaps to be addressed and provided preliminary suggestions for siting, design, and management criteria to that may mitigate adverse impact to wildlife. Further discussion with wetlands managers in southern California identified that the risk to wildlife from contaminant exposure and toxicity were among the highest priority research questions.

Goals of the study

To address this information gap, the Los Angeles Regional Water Quality Control Board and the State Coastal Conservancy funded a study to evaluate habitat value associated with urban wetlands of multiple objectives in southern California. The goal of this project is to provide information on how these urban wetlands can be better managed to increase compatibility with wildlife protection in southern California.

A phased approach was used to allow data collected in the first phase to drive decisions about objectives and study design in subsequent phases. The four major phases of this project included:

1. Develop an inventory of existing urban and treatment wetland projects
2. Conduct a baseline habitat survey on a representative sample of urban wetlands to evaluate wildlife beneficial use

3. Evaluate the exposure and toxicity to wildlife from sediment-borne contaminants
4. Conduct analysis, to the extent possible, on the effectiveness of treatment wetlands using existing monitoring data

This report provides specific recommendations on how the habitat value of urban wetlands can be improved. This information is useful to both state resource and water quality agencies, as well as stormwater agencies and NGOs involved in managing these wetlands.

Organization of the Report

This report has five sections. Section 1 covers the overview of the project with an introduction, goals of study and report organization. Section 2 provides the results of the urban wetland inventory. Section 3 presents the results from studies of sediment contamination and toxicity. Section 4 offers an analysis of effectiveness of treatment wetlands from existing monitoring data, and Section 5 summarizes the study findings and a set of management recommendations.

INVENTORY OF SOUTHERN CALIFORNIA FRESHWATER URBAN WETLANDS

Abstract

This report summarizes the findings of the first stage of the project: the inventory of urban wetlands. The inventory consists of a compilation of existing data on selected wetlands and wetland projects, with the overarching purpose of providing a tool for staff of the Regional Boards, state and federal agencies, as well as proponents of future projects, to access information on location, project objectives, basic site attributes and contact information for existing wetland projects. The information compiled on these sites will also serve as the basis to select sites for subsequent phases of the study. The information is assembled in an ACCESS database (Version 2.0), with documentation on 40 freshwater urban and treatment wetlands located throughout six counties in southern California.

Because no comprehensive wetland inventory yet exists for southern California wetlands, the inventory was developed by first compiling a master list of 85 candidate wetland sites using information obtained from wetland owners, stakeholders, agency staff, and known projects. To screen these sites, a set of selection criteria were developed in consultation with a Technical Advisory Committee (TAC) comprised of staff of state, county, and municipal agencies, water districts, and NGOs. These site selection criteria included, among others, wetland type, geographic location, and water source. These three site selection criteria were used, in addition to logical issues such as permission to compile data and access the project site, to produce a final list of 40 wetland sites. Note because sites were not selected randomly from a comprehensive list of sites, the 40 selected sites may not be representative of the general population of urban wetlands.

Existing data compiled in the inventory includes information on: 1) project contacts, 2) classification, 3) site history, 4) technical design, 5) operations (maintenance and monitoring), and 6) existing reports and site photographs. Overall, approximately 70% of the targeted data types in these categories were compiled for the 40 sites. Information in the database is most complete for contacts, classification, and operations (>90%) and least complete for technical design (ca. 50%).

The wetland inventory contains sites from six counties within the study area; the majority of the wetlands are located in Orange County, Los Angeles County and San Diego County, while the remaining sites are in Santa Barbara, Ventura and Imperial Counties. The inventory of sites was equally distributed among in-channel basins, offline basins, isolated basins and small tributary channels. Few sites were represented by offline diverted channels and swales. Approximately 60% of the sites had habitat as their primary objective, 25% had water quality as the major objective, and the remainder can be classified as multipurpose (habitat and water quality treatment). Approximately 40 %of sites receive flows that are to some degree anthropogenically managed. The types and frequency of maintenance and monitoring activities reported among the inventoried wetlands were highly variable. Vegetation management was most consistently carried out at the sites (90%), while vector control, hydrologic management, and trash removal were commonly practiced for approximately 50% of the sites. Indicators monitored at the 40 wetland sites varied depending upon wetland objective; vegetation monitoring was conducted in most wetlands. The emphasis of monitoring in habitat wetlands was on wildlife, hydrology, and water quality, and, less frequently, sediment monitoring; in treatment wetlands, the emphasis was on water quality and sediment monitoring and to a lesser extent, hydrology and wildlife.

The number of inventoried sites and distribution of characteristics they possess with respect to wetland type, project objective, and flow management were sufficient to undertake subsequent phases of the study (baseline habitat survey and intensive study of habitat value and treatment effectiveness). Additional sites

that could serve as reference standards for both channels and basins may be added to the inventory at a later date.

Introduction

This report summarizes the findings of the first phase of the project: the treatment wetland inventory. The purpose of the treatment wetland inventory is to provide a geographic tool for staff of the Regional Boards, state and federal agencies, as well as proponents of future projects, to access information on location, treatment objectives, basic site attributes and contact information for existing treatment wetland projects. The inventory will also serve as the basis to select sites for further study in Phases 2 - 4 of the project.

This report outlines key procedures used for the development of the treatment wetland inventory, describes the form and content of the inventory database, and summarizes the completeness of the database and the distribution of basic characteristics among 40 wetland project sites selected for the inventory.

Methods

Definition of Urban Wetlands

Two general types of wetlands were of interest.

- Basins – Basins are freshwater depressional wetlands that consist of habitat types known colloquially as ponds, pools, wet meadows, treatment wetlands, or wetland detention basin. They are characterized by a topographic depression that provides for temporary storage of flood flows. They may have a defined inflow or outflow, but neither is required for inclusion in this class.

- Channels – Channels refer to small tributary streams and other freshwater wetlands that are characterized by a linear channel with flowing water of short retention times (<3 h, depending on the size of the site) and a defined inflow and outflow.

For this project, “wetlands” are defined using the Cowardin *et al.* (1979) definition while “riparian” areas are defined using the United States Environmental Protection Agency (USEPA; 2001) definition. Freshwater depressional or basin wetlands and small streams and/or channels were targeted for inclusion in the inventory because they are of primary interest to the agencies and other stakeholders represented by the TAC. These are the types of wetlands that are most likely to be proposed as treatment wetlands or multipurpose projects. While it is recognized that estuarine, lacustrine, and large riverine (channel) systems provide water quality enhancement, as well as habitat, they are not the primary wetlands of interest for this study and were therefore excluded from the inventory of 40 wetland sites. Freshwater depressions/basins and streams/channels can be broken up into six wetland categories or types that help to distinguish them with respect to geomorphic position, hydrology, and physical structure of the wetland. These types are explored in the results section.

Three categories of wetlands and riparian projects are considered as “urban” for this study:

- Habitat Projects - Natural wetlands or riparian areas restored or enhanced for habitat, but with urban runoff as primary water source.
- Multipurpose Projects - Natural wetlands or riparian habitat (restored or enhanced) or wetlands or riparian areas created to serve multiple objectives of habitat and water quality improvement

- Treatment Projects - Wetlands or riparian areas constructed and/or engineered for water quality improvement, with habitat ancillary (as in Kadlec and Knight 1996)

Development of the Wetland Inventory

The first step in the development of the urban and treatment wetland inventory was the assembly of a list of candidate sites (Figure 1). Names and contact information for approximately 85 wetland sites were identified through discussions with staff from various state, federal and regional agencies, nongovernmental organizations (NGOs), water districts and stormwater agencies. To establish contact with wetland site owners and determine the wetlands' eligibility for inclusion in the study, a standard form was developed to collect contact information and basic wetland classification and operational information. This information (Tier I data) was collected either by interviewing the wetland owners or managers on the phone or by e-mailing the form to them. In cases where the data were obtained via phone interviews, the completed forms were sent to the contact for quality control purposes. Of the original 85 candidate sites, Tier I data was compiled on approximately 50 projects.

Inclusion of wetland sites in the final list required: 1) access to existing information on the site be readily available and 2) permission to access the wetland site given by the owner, preferably as a written consent. Sites for which access to additional data was either denied or could not be obtained due to non-responsive contacts were dropped from the final list. Permission to access the wetland site was considered as a necessary for subsequent phases of the project that would involve baseline habitat surveys, and evaluation of treatment effectiveness and compatibility with wildlife beneficial use. Those sites for which permission for access was either denied or otherwise could not be obtained after repeated attempts were made at contacting the owners were screened out of the final list.

Of these projects contained in the Tier 1 database, site selection criteria were used to screen a minimum of 40 sites that had characteristics of interest for this study. For these sites, extensive data would be compiled (Tier 2 data). This effort involved the compilation of more detailed information for all wetlands on the final list of sites including wetland design information from wetland owners, managers, design firms, and consultants. Only information that was readily available and compatible with database format was compiled.

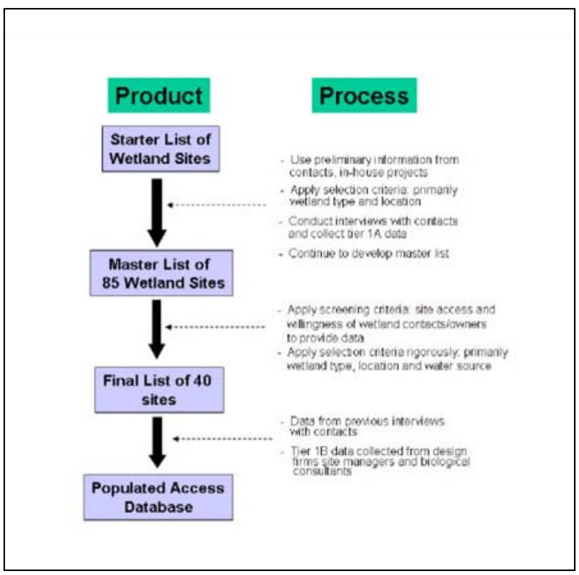


Figure 1. Diagram of process used to develop urban and treatment wetland inventory.

Site Selection

To screen the wetland sites on the candidate site list, the original intention was to use a suite of selection criteria that would be used to balance the composition of projects in the inventory. For this purpose, a set of selection criteria were developed in consultation with a TAC comprised of staff of state, county, and municipal agencies, water districts, and nongonvernmental organizations. These selection criteria considered included wetland type, geographic location of wetlands, water source, catchment land use, contaminants of concern/loading, operations and maintenance, habitat and treatment objectives, water source, treatment design elements, landscape position, site history, and hydrogeomorphic type.

Of these site selection criteria, only wetland type, geographic location, and water source were used to screen the master list and produce a final list of 40 wetland sites for the inventory (Figure 1). Due to the varied response times of contacts, a range in willingness and availability of contacts to provide information, other selection criteria of interest were not used rigorously in the screening process in order to complete the inventory in a timely fashion. Data compiled on these selection criteria will however be useful in subsequent phases of the study to understand the basic differences among sites that might be controlling habitat value or water quality treatment effectiveness.

While the focus of the study is on surface flow wetlands, sub-surface flow (SSF) wetlands, no systems entirely composed of SSF Wetlands are incorporated into this study. This was because the marsh habitat values of SSF wetlands generally do not resemble those of surface flow wetlands.

Data Fields and the ACCESS Database Design

As the preliminary list of candidate sites was under development, the database structure, including specific data fields, was developed to describe the physical, chemical and biological attributes of the wetlands. These data fields form the basic framework of the Access database. The initial Access database was developed at Southern California Coastal Water Research Project (SCCWRP), then beta-tested and modified by CH2M Hill.

The main interface of the database in has several modules for data entry, including contact information for each wetland site, classification information that includes general attributes of the wetland, design information, wetland operational data and site history. Additional modules are also provided to enter information of existing data and reports that have been received from the contacts, Tier 1 data including forms containing data received from contacts on general wetlands attributes and monitoring/maintenance activities, and a module for depositing any historic site photos.

Contact Information. Contact information on wetland owners, wetland operators, and any additional contacts for each wetland site can be found in this module. Figure 2 gives an example of the user interface for data entry, featuring specifically the contact information (Figure 2). Contact information on site owners and site operators is organized into categories that include the name of organization/agency /company, contact name, contact title, full address, phone and fax numbers, email, number of years in ownership and previous owner. The “additional contacts” category includes information on wetland stakeholders (e.g., NGOs, managers of “Friends of….” groups) that in many cases were a valuable source of easily accessible data on wetland sites.

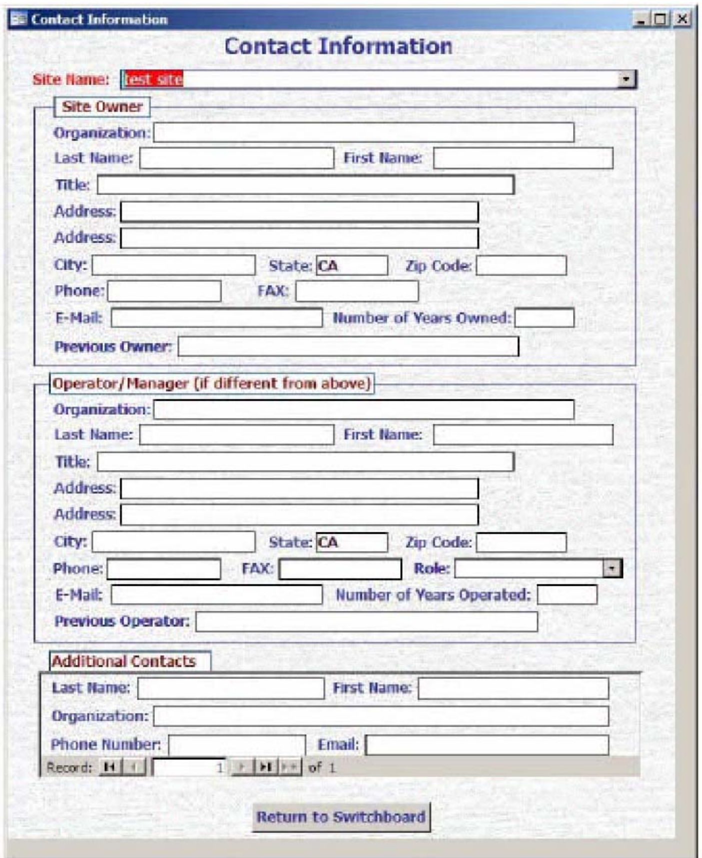


Figure 2. Example of a data entry screen for project contact information.

Classification Information. This module contains information on wetland origin (natural or created), project type (mitigation or non-mitigation), wetland type (six wetland types documented in this report), wetland objectives (habitat or treatment), and wetland location (longitude, latitude). Additional fields that capture information on landuse in the catchment area of the wetland sites are also included in this module.

Design and Physical Configuration Information. The design and physical configuration information module contains fields that attempt to capture data on influent characterization, contaminant concentration, cell design area and other cell specific data. This includes information such as influent characterization, hydraulic loading rates, average contaminant concentrations, and basic information about the design of wetland projects such as acreage of habitat types, etc.

The design information data proved to be more difficult in compiling than originally anticipated. The reason for this is that there is not a standard method for designing wetlands. Moreover, the data put forward in the design and monitoring reports reflects the values and concerns of these varying design and monitoring firms. For example, only 12 of the 40 sites (30%) have published intended flow rates for the wetland, and 25 of the 40 sites (63%) have published their intended (if designed) emergent plant palettes.

Operational Information. The operational data module is designed to contain information on the monitoring and maintenance activities at each wetland site. Data compiled in this module include:

- Date of project operation
- If monitoring and maintenance is required

- Monitoring information type: This covers what type of data is being monitored at the site, including hydrology, sediment, vegetation, water quality, wildlife, and mapping. Each of these categories, once selected, is linked to pull down menus under two additional fields, parameter type and parameter, where specific information on that data can be selected. For example, if "sediment" is selected under the monitoring type field, then sediment parameters such as bulk parameters, heavy/trace metals, indicator bacteria, nutrients and synthetic organics will be displayed under parameter type.
- Monitoring frequency: The frequency (e.g., monthly, weekly, bi-weekly, etc.) with which the wetland site is monitored can be selected from a pull-down menu in this field.
- Monitoring method: This field describes the method used for the monitoring
- Maintenance information type: A pull down menu containing a list of various types of maintenance activities is available under this field; these activities include vector control, predator control, vegetation management, hydrologic management, sediment removal from wetlands area, structural maintenance (berms and structures associated with the wetland), trash and floatables removal, public access management and security measures.
- Maintenance Frequency: The frequency (e.g., monthly, weekly, bi-weekly, etc.) with which the wetland site is monitored can be selected from a pull-down menu in this field.

Site History Information. Site history information allows text entry about wetland origin, previous wetland type, water source and hydrology management, a checklist of stressors that might be acting on the site, and summary of management activities.

Existing Data and Reports. The existing data and reports module provides a list of all available data on the final 40 wetlands. The format allows for raw data and final reports to be viewed together to create an easy interface between the user and the information. All data collected in this study was input with the data type attached. For example, Additional fields describe the type of data, data format, the file type and the file name of the report.

Site Photographs. This module is designed to serve as a file cabinet, where documented photographs of the sites can be stored and viewed.

Results and Discussion

This section provides the final list of 40 wetland project sites; details the completeness of the database for the inventory data fields; and summarizes the distribution of wetland characteristics within the inventory with respect to wetland location, wetland origin, wetland project type, flow management, wetland objective, wetland type, water sources, catchment land use, wetlands operation and maintenance, and monitoring type and frequency.

Finalized List of Sites

Table 1 gives the final list of urban and treatment wetland projects selected for the inventory. Note that Prado wetlands were on the list, but removed because of catastrophic damage to the wetland system during 2005 flooding the Santa Ana River Basin.

Table 1 Final List of Urban and Treatment Wetlands in Southern California.

Number	Site ID	Site Name	County
1	4	Brawley Wetlands	Imperial
2	20	Imperial Wetlands	Imperial
3	1	Ballona Freshwater Marsh	Los Angeles
4	5	Bridgeport Lake	Los Angeles
5	17	Gardena Willows Wetlands	Los Angeles
6	22	Lower Arroyo Seco Low Flow Channel	Los Angeles
7	23	Madrona Marsh	Los Angeles
8	32	Sepulveda Wildlife Refuge	Los Angeles
9	33	Sims Pond Ecological Preserve	Los Angeles
10	35	Tujunga Ponds	Los Angeles
11	40	Zuniga at the Fritz and Alma Meier Nature Preserve	Los Angeles
12	2	Big Canyon	Orange
13	3	Blackbird Pond	Orange
14	8	Carbon Canyon Basin	Orange
15	9	Carlson Marsh	Orange
16	11	Crown Valley Park - J03P01	Orange
17	13	Duck Ponds - UCI	Orange
18	14	East Alicia	Orange
19	19	Hoag Hospital Mitigation Site	Orange
20	21	La Paz Park On-site Wetlands	Orange
21	24	North Alicia Wetland	Orange
22	29	Prima Deshecha Landfill - Mitigation Site A	Orange
23	30	Prima Deshecha Landfill - Mitigation Site B	Orange
24	31	San Joaquin Marsh - IRWD	Orange
25	37	UCI Marsh	Orange
26	38	Waterfront Marsh	Orange
27	39	West Alicia Wetland	Orange
28	6	Buena Vista Lagoon	San Diego
29	10	Cottonwood Creek Park	San Diego
30	12	Dairy Mart Ponds	San Diego
31	15	Famosa Slough - East Bank Sediment Basin	San Diego
32	16	Famosa Slough - Valeta Street Treatment Wetlands	San Diego
33	28	Penasquitos Canyon Preserve Wetlands - El Cuervo	San Diego
34	27	Old Mission Creek - Bohnett Park	Santa Barbara
35	34	Stork Ranch	Santa Barbara
36	36	Turnpike Bioswale	Santa Barbara
37	7	Camino Real Bioswale	Santa Barbara
38	18	Hill Canyon Mitigation Bank	Ventura
39	25	Oak Canyon Wetlands - Lower	Ventura
40	26	Oak Canyon Wetlands - Upper	Ventura

Database Completeness

The methods component of this section gives an overview of the type of existing data sought for the inventory. The success to which all data fields were complete for each project in the inventory depended on the extent to which the data for a project were readily available and in a format that was compatible with the database.

Table 2 gives the percent completeness of each of the seven major categories of existing data. For some categories, such as contact information and classification information, completeness was 100 % (n = 40). Completeness for operational information and existing data and reports was slightly lower (90%). While data has been received for all sites, an estimate of 90% reflects an acceptable level of certainty that all pertinent reports and relevant operational information about the site were adequately captured in the database.

Table 2. Estimated completeness of the inventory database with respect to major categories of existing data.

Database Module	Estimated % Complete
Contact Information	100%
Classification Information	100%
Design and Physical Configuration Information	40%
Operational Information	90%
Site History Information	40%
Existing Data and Reports	90%
Site Photographs	45%

Other categories, such as design and physical configuration, site history, and site photographs had a much lower estimate of completeness (40 - 45%). In the case of design and physical configuration information, the low percent completeness estimate reflects: 1) low reporting rate on design parameters such as influent characterization, contaminant loading, and cell specific design data, and 2) a lack of standardization in how the data were reported, thus posing quality assurance issues. In the case of site history information (40% complete), site owners are not typically required to keep records of site history; several did so because of interest in the site, but the data for most sites were not readily available.

Geographic Distribution of Sites

The wetland inventory contains sites from six counties within the study area (Figure 3). The predominance of sites in Orange County reflects that sites in this county met the site selection and screening criteria more often than those from other counties. Overall, the majority of the wetlands are located in Orange County (16 sites), Los Angeles County (9 sites) and San Diego County (6 sites), while the remaining sites are in Santa Barbara, Ventura and Imperial Counties (4, 3, and 2 sites, respectively).

Wetland sites with habitat as the primary objective were most abundant in Orange and Los Angeles Counties (11 and 7 sites, respectively), with fewer sites (3 sites each) in San Diego and Ventura counties (Figure 4a). Wetland sites where habitat is rated as the primary objective were not found in Santa Barbara and Imperial Counties. The absence of habitat sites from Santa Barbara and Imperial Counties likely reflects that these counties had fewer wetland sites in the inventory due to fewer information sources. Sites with water quality as the primary objective were present in all counties (1 to 3 sites per county), except Ventura County. Sites where water quality and habitat were rated as equal objectives were only present in Orange County (3 sites), Santa Barbara County (2 sites) and Los Angeles County (1 site; Figure 4b).

Figure 3. Map showing geographic distribution of inventory sites in southern California. Inset of map shows an enlargement of site distribution in Orange County, where the largest number of sites was located.

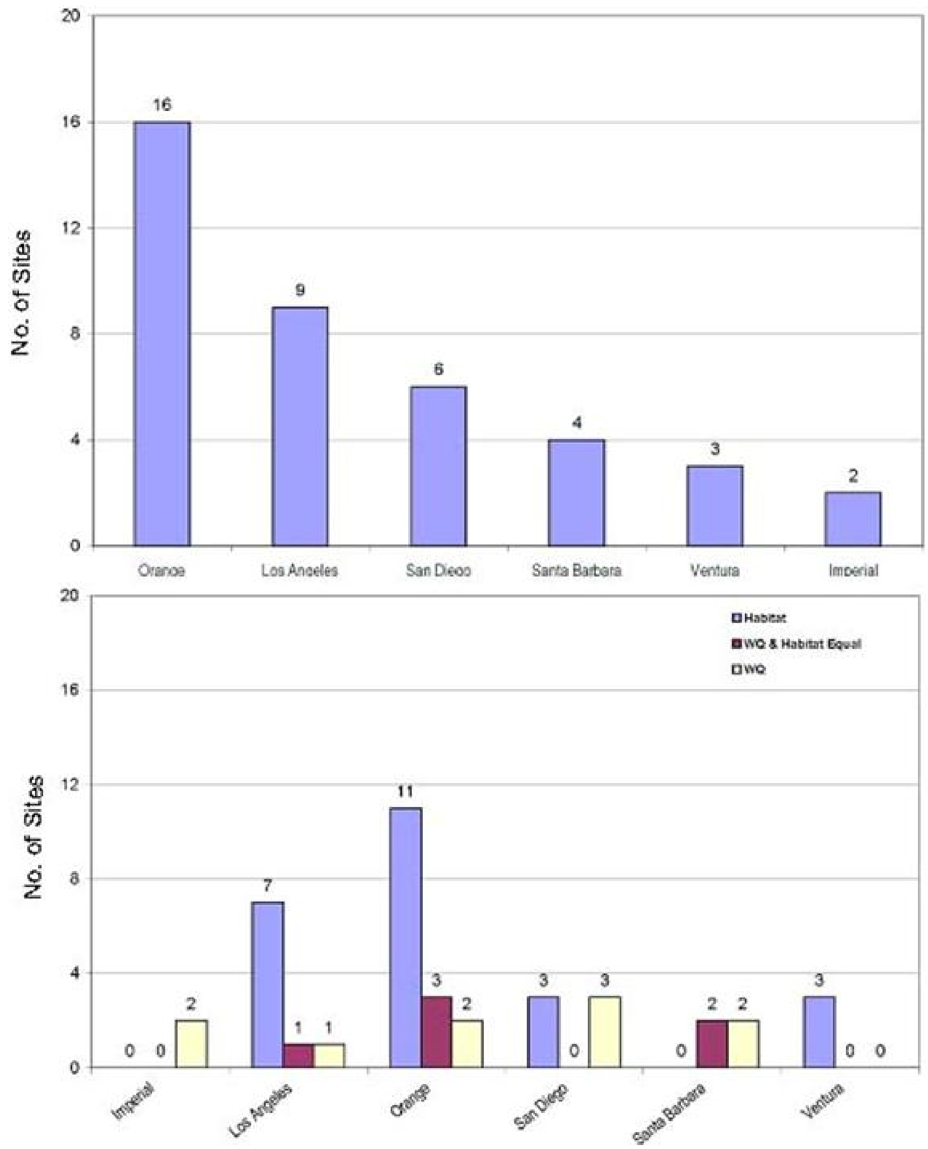


Figure 4. Number of inventory sites by county (top) and by county and project objective (bottom).

Wetland Objective and Origin

The majority (24 out of 40 sites or 60%) of wetlands in the inventory have habitat as their primary objective, followed by 25% of the sites with water quality as their primary objective. Fifteen percent of the sites share habitat and water quality as equally-weighted objectives.

Most (30 out of 40 sites or 75%) of the wetlands are constructed and the remaining are natural in origin. The majority (16 sites) of the created wetlands was built primarily for habitat and ten sites are built for water quality treatment benefits (Figure 5). The remaining sites are constructed to meet both habitat and water quality goals.

Ten out of the sixteen wetlands (63%) built primarily for habitat were created from areas that were previously wetlands (previous “wetland” areas here include riparian edges of streams/rivers and floodplains), while the remaining sites were constructed from areas that were previously upland (Figure 5). For wetlands that were built primarily for water quality treatment benefits, 6 out of 10 sites (60%)

were created from areas that were previously wetlands, while the remaining sites were created from upland areas. Among the wetland sites that share both habitat and water quality as equally-weighted benefits, 50% (2 out of 4 sites) were created from wetland areas and 50% from upland areas (Figure 5). Overall, the majority of the constructed wetlands (18 out of 30 sites or 60%) were created from areas that were previously wetlands and likely reflect the general tendency to locate wetland construction in sites that were historically wetland areas.

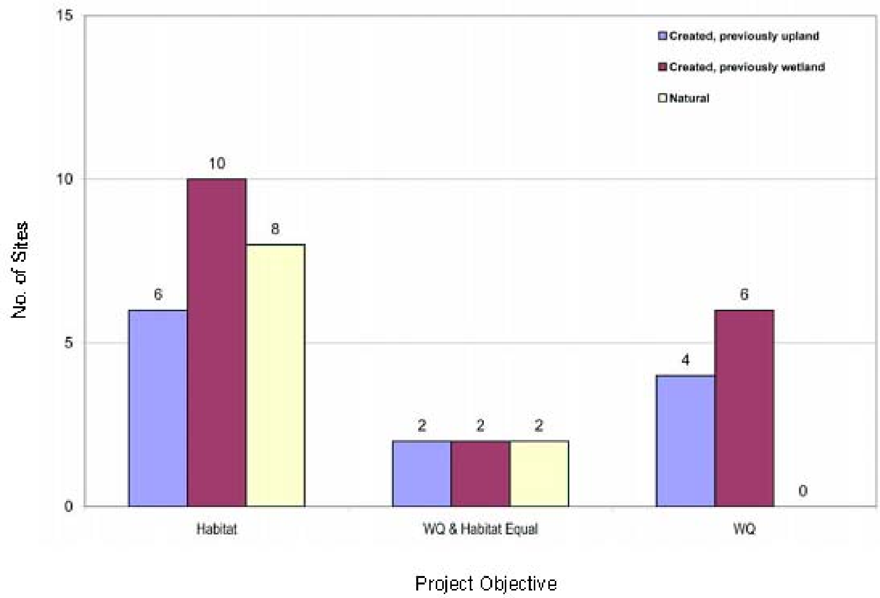
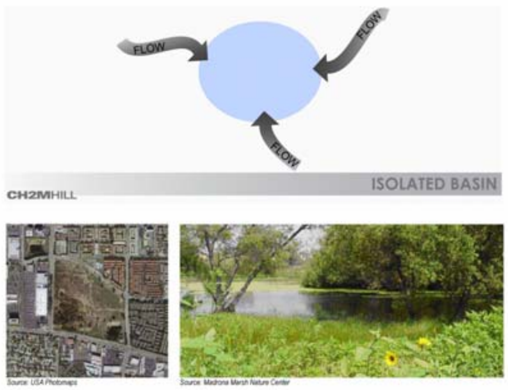


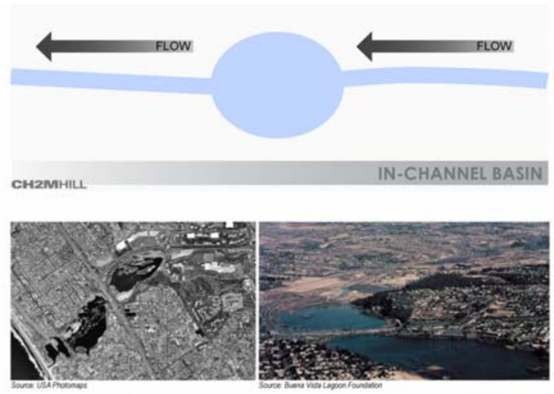
Figure 5. Distribution of sites by origin (created or natural) and project objective.

Wetland Type

Wetland types captured in the survey could be broken down into two major categories: basins and channels. Within these two major categories, a large variety of configurations exist. This section explores the variability found within each category.

Basins. Wetlands in the basin class can generally be grouped into three categories: in-channel basins, isolated basins, and offline basins. In-channel basin are depressional wetlands that are constructed within a stream or river channel, essentially constituting a type conversion from a stream/river to a basin wetland. They range from vegetated wetlands constructed within tributaries, such as low-flow headwaters, to sedimentation basins in stormwater channels designed to capture and retain sediment from runoff. In both extremes, these systems are designed to retain runoff over a period of time in order to promote sedimentation and detain flood flows. Isolated basins occur in topographic depressions, either natural or manmade, in the landscape (Figure 7). Dominant water sources can include runoff from adjacent uplands, point source discharges, precipitation, and groundwater discharge. The direction of flow is normally from the surrounding uplands toward the center of the depression. Elevation contours are closed, thus allowing the accumulation of surface water. Offline basins receive hydrologic inflow from stormwater, creeks or streams that have been built outside of the active channel or floodplain. Water is typically pumped into the wetland. This category may range from wetlands constructed from uplands that were never historically inundated, or wetlands constructed from wetland habitat with no hydrological connection to a stream or river.





Examples of the "basin" class of urban wetlands. *Isolated basins* occur in topographic depressions, either natural or manmade. *In-channel basins* are constructed within historic floodplain habitat. *Offline basins* are constructed in historic floodplains or be constructed from upland; in either case, their water sources are actively pumped to have precise control over the hydrology.

Figure 7. Examples "basin" class of wetlands.

Channels. Three types of channel systems were captured in the inventory: small tributary, offline channels, and vegetated swales. Small tributary channel systems are typically first and second order streams and associated wetland and riparian habitat receiving hydrologic inputs through gravity flow of runoff from drainage networks (Figure 8). Offline channels refer to channelized systems that are created away from the historic channel or flood plan and receive hydrologic inflow from stormwater, creeks or streams by either gravity diversion or by mechanical pumping. The distinction made by this category is the recognition that the wetland may not be configured as a basin, but more like an open flowing channel, creek, or small streams with a distinct inflow and outflow (Figure 8). Vegetated swales are shallow grass-lined channels designed to convey stormwater flows while providing a relatively low level of treatment near the source of the runoff. Because swales may be inundated infrequently, only those swales that show seasonal wetland characteristics by supporting hydrophytic vegetation and/or having hydric soils for a portion of the year will be considered in this inventory (Figure 8).

Figure 9 gives the approximate distribution of these types of wetlands within the basin and channel categories.

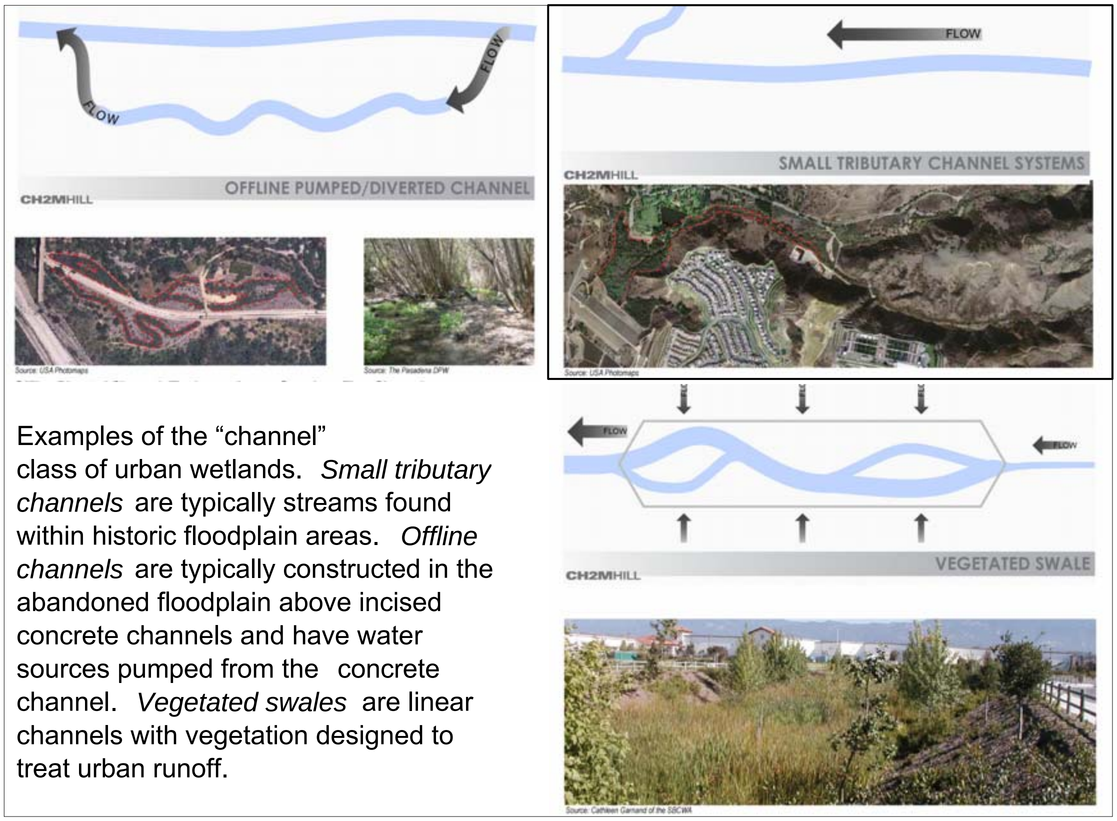


Figure 8. Example of "channel" class wetlands.

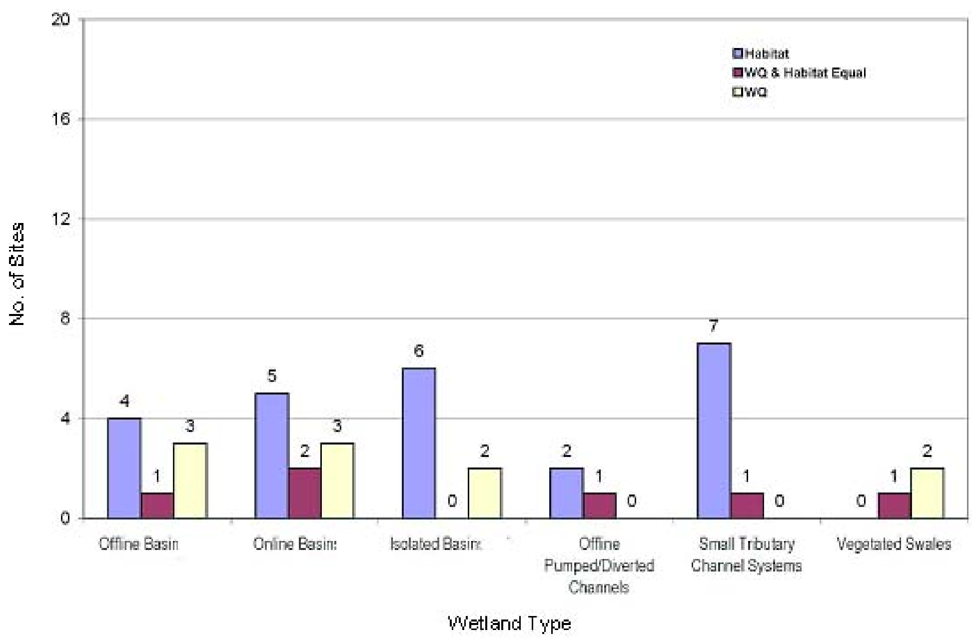


Figure 9. Distribution of sites by wetland type and project objective.

Wetland Water Sources

The majority of the inventory wetlands receive natural or urban runoff as a water source (24 to 27 sites). The next most frequent water sources for these wetlands include streams/rivers (17 sites) or groundwater (14 sites). Agricultural runoff, industrial effluent, and municipal wastewater effluent were relatively uncommon sources of water (2 to 5 sites). Sites identified as receiving industrial effluent and municipal wastewater effluent drew in stream water with upstream municipal and industrial discharge.

Habitat wetlands received water mainly from natural and urban runoff, groundwater, stream or river sources, and to a lesser extent from agricultural, industrial, and municipal sources. Water quality wetlands mainly received water from urban runoff sources, and to a lesser extent from natural runoff, agricultural, stream/river and groundwater. Sites with water quality and habitat as equal objectives received water from natural and urban runoff, groundwater sources, and to a lesser extent streams or rivers.

Wetland Project Type

Fifty-five percent of 40 sites were mitigation projects. Habitat was the primary objective for the majority of both mitigation (64%) and non-mitigation (56%) projects. Water quality improvement was the primary objective of 33% of the non-mitigation wetlands and 18% of the mitigation wetlands.

Wetland Flow Type

Two-thirds of the inventory wetlands (27 out of 40 sites) received flow passively and the remaining third (13 out of 40 sites) received managed flows. Within each flow type, most of the wetlands had habitat as their primary objective, about 25 % had water quality as the primary objective, and approximately 15% shared both habitat and water quality objectives equally (Figure 10).

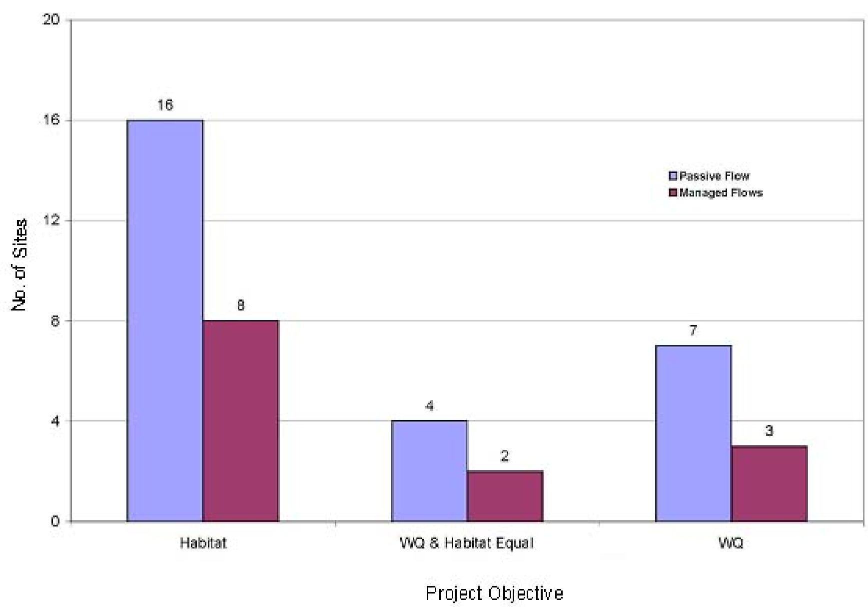


Figure 10. Distribution of site by project objective and flow management (passively or actively managed).

Catchment Land Uses

Catchment land use was reported qualitatively in the Tier 1 data collection (preliminary site information). Catchment size and a breakdown of catchment land use in percentages of year 2000 NOAA C-CAP land cover class were calculated for each wetland using GIS and are reported in the next section.

Types of Operation and Maintenance Activities and Frequency

Of the 40 inventory sites, vegetation management was conducted at all but 4 (Figure 11). Vector control, hydrologic management, and trash/floatable removal were commonly practiced at 22 to 23 sites. Public access maintenance, berm and structural maintenance, sediment removal, and security operations were performed at 9 to 15 sites, while predator control was practiced at 6 wetland sites.

Maintenance activities varied depending upon wetland objective (Figure 11). All wetlands, whether with habitat, water quality, or shared objectives, included some degree of vegetation management, hydrologic management, public access maintenance, vector control, berm and structural maintenance, and trash/floatable removal. Sediment removal, predator control, and security activities were less frequent.

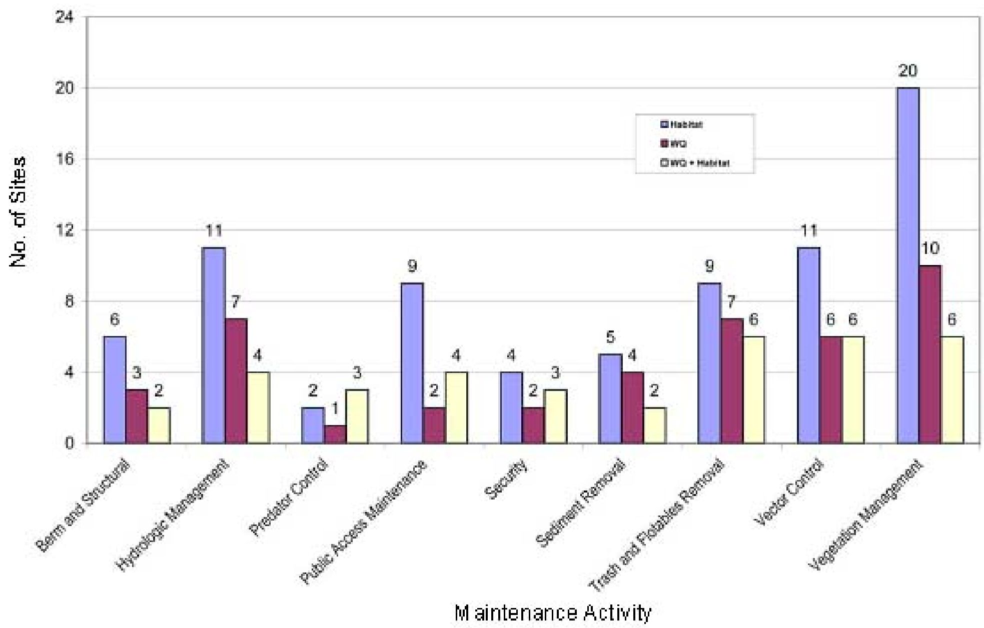


Figure 11. Distribution of sites by type of maintenance activity and project objective.

Wetlands operations and maintenance activities were most often carried out on an as-needed basis ("other"). Operations and maintenance activities were also conducted on a range of schedules spanning from bi-monthly to weekly. None of the wetlands had operation and maintenance activities conducted solely on an annual basis.

Types of Monitoring Activities and Frequency

Monitoring program emphasis varied depending upon wetland objective (Figure 12) but vegetation monitoring was conducted in most wetlands. Habitat wetland activities included wildlife, hydrology, and water quality monitoring; sediment monitoring was conducted less frequently. Water quality wetlands included water quality and sediment monitoring and to a lesser extent, hydrology and wildlife, consistent with the primary treatment goals of these sites. Monitoring at sites with water quality and habitat as equal objectives typically included water quality and vegetation monitoring, and hydrology, sediment, and wildlife less often.

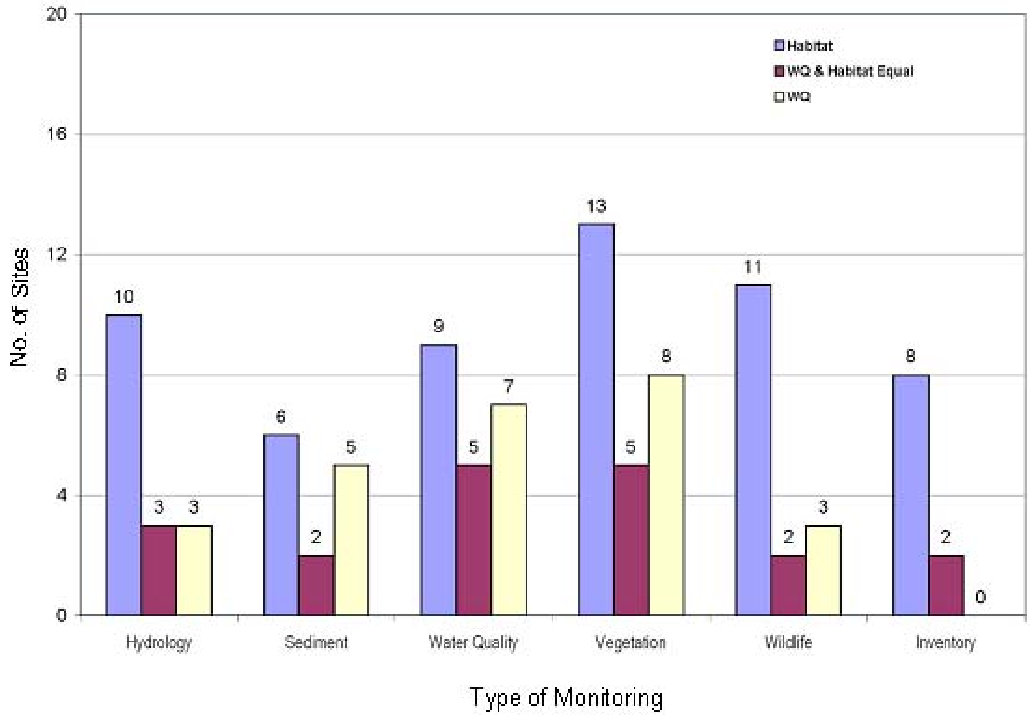


Figure 12. Distribution of sites by type of wetland monitoring and project objective.

The distribution of various monitoring activities conducted at the wetland sites (Figure 13), represents only those sites for which the data were readily available and illustrates the range of wetland parameters monitored at these sites. *Enterococcus* and coliform bacterial indicators, field parameters (i.e., biological oxygen demand (BOD), chemical oxygen demand (COD), conductivity, pH, total organic carbon (TOC), total suspended solids (TSS), water depth, water temperature), nutrients, and vegetation were monitored at 15 sites; wildlife, synthetic organics, and metals were also monitored at 9 to 13 sites). Hydrology, bulk soil parameters (i.e., percent moisture, bulk density, particle size, porosity and TOC), biological inventory (i.e., habitat mapping, vegetation mapping, and other types of mapping) and major inorganics (calcium, carbonate, chloride and magnesium) were monitored at only one to seven sites.

Monitoring activities were most often carried out on a monthly, weekly, and quarterly, as-needed ("other") or bi-annual. Monitoring of habitat wetlands was conducted most frequently on a monthly and bi-annual basis, and less frequently on a weekly, bi-monthly, quarterly, annual and as-needed ("other") basis. None of the monitoring activities in habitat wetlands was conducted on a daily or bi-weekly basis. Water quality wetlands were monitored frequently on a weekly and monthly basis, and less frequently on bi-weekly, bi-monthly, quarterly, annual, and as-needed basis. Wetlands with shared objectives were monitored on a monthly, quarterly, and as-needed basis, and less frequently on a daily and weekly basis. The more frequent (weekly) monitoring of water quality wetlands compared to sites with habitat or shared objectives may likely reflect the level of monitoring required to evaluate treatment effectiveness of these sites.

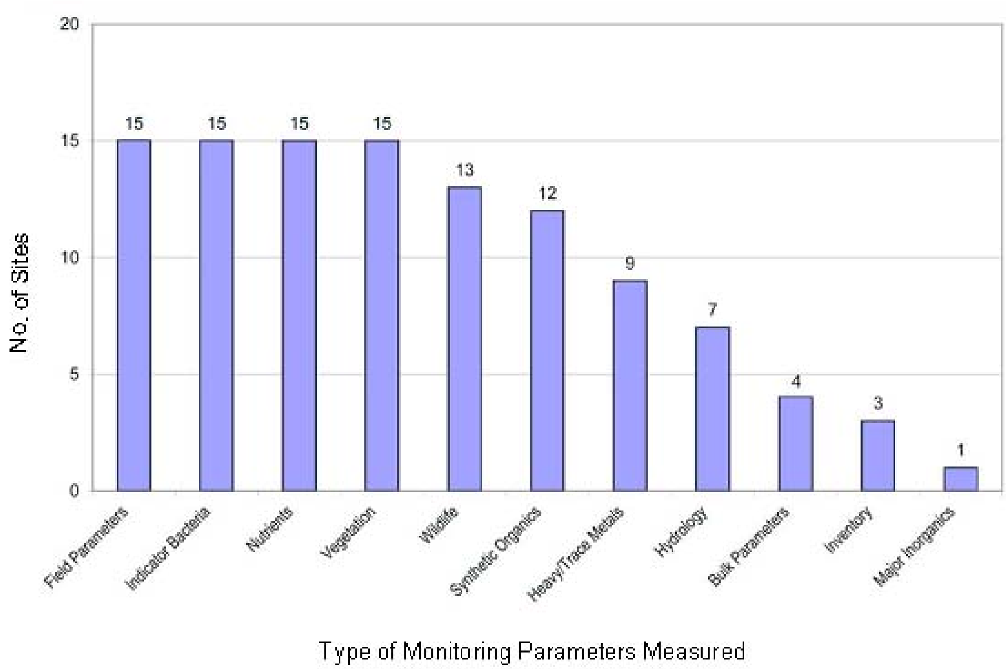


Figure 13. Distribution of sites by type of monitoring parameter measured.

HABITAT VALUE OF SOUTHERN CALIFORNIA URBAN WETLANDS: RESULTS OF A BIOLOGICAL SURVEY

Abstract

With the rising urbanization of the coastal watersheds of California, requirements for control of urban runoff have created demand for treatment wetlands to improve surface water quality and attenuate storm flows. Concurrent with this need, significant wetland and riparian habitat have been lost. Pressure is mounting to restore, enhance, and create wetlands with multiple objectives (i.e., habitat support, treatment of nonpoint source pollution, flood attenuation, and recreation). Information is lacking on the potential risk to wildlife associated with using wetlands for treatment of nonpoint source runoff. The goal of this study is to provide information on how urban wetlands can be better managed to increase compatibility with wildlife protection.

In spring 2006, a baseline biological survey was conducted to evaluate the condition of habitat provided by these sites. The survey focused on both freshwater basins and small streams or channel systems and was conducted in three tiers: 1) wetland and vegetation habitat mapping, 2) rapid assessment of wetland condition using the California Rapid Assessment Method (CRAM) for wetlands at 40 sites, and 3) a more intensive biological survey of 26 sites including vascular plant and benthic macroinvertebrate community composition and bird use.

Results of the survey showed that, while urban infrastructure provide some basic constraints on the achievable condition of wetland basins and channels, site-specific conditions can either mitigate or exacerbate these constraints. The site-specific factors included, but were not limited to: 1) wetland size, 2) project objective and design criteria, 3) intensity of maintenance, and 4) origin of site. The study found that habitat wetlands could achieve what could be considered good condition if: 1) the sites can support wetland hydrology, 2) the hydrology could be managed to mimic natural hydroperiod, 3) the site is designed to have good physical structure, 4) the wetland and buffer had appropriate native vegetation, 5) the wetland is maintained frequently to manage stressors, but not in manner which is incompatible with the seasonal cycles of nesting and reproduction, and 6) issues of bioaccumulation and toxicity are not a factor. This last factor has not been evaluated as a part of the baseline biological survey, but will be addressed in the subsequent phases of the project.

Introduction

Wetlands are known to possess a number of physical, chemical and biological properties that, in addition to providing fundamental support to plant and animal populations, are highly valued by humans. These include: 1) enhancement of surface water quality, 2) water storage and flood attenuation, and 3) aesthetic, commercial, recreational, and educational uses (Mitsch and Gosselink 1993). With the increasing urbanization of coastal areas over the past century, wetlands have been rapidly disappearing from the landscape, and those that remain are often highly degraded (Dahl 1990; Holland *et al.* 1995). In the coastal watersheds of southern California, this problem is particularly acute, with 75% of the approximately 53,000 historic acres already destroyed (California Department of Parks and Recreation 1988). Impacts have been particularly severe for coastal salt marshes (California Coastal Commission 1989, California Department of Fish and Game 1983, Zedler *et al.* 1992), riparian corridors (Faber *et al.* 1989), and vernal pools (Zedler 1987). Concurrent with the loss of wetland habitat, increased runoff from urbanized watersheds and discharges of point or nonpoint source pollution have created a demand for effective solutions to improve surface water quality and attenuate storm flows. As a result, there is increasing interest in restoring, enhancing, and creating natural or constructed wetlands with multiple objectives, e.g., habitat support, treatment of nonpoint source pollution, flood attenuation, and recreation

(Azous and Horner 2000). While it is common for those who manage wetlands for multiple objectives to claim all these benefits, the extent to which these benefits are actually provided and the trade-offs between these objectives is often not clear. In particular, wetlands that maximize water quality improvements do not necessarily provide high quality wildlife habitat, and vice versa (Helfield and Diamond 1997).

This section of the study summarizes the results of the habitat survey of urban freshwater wetlands captured in the inventory (Phase I). The purpose of the habitat survey was to describe baseline biological conditions of urban wetland sites with varying management objectives in comparison to a set of references sites. Specifically, the intent was to answer the following key management questions:

- What is the condition of urban wetlands relative to reference?
- How does ecological condition vary in sites created, restored or enhanced for habitat along a gradient of land use intensity (using percent imperviousness as a proxy)?
- What is the difference in the ecological condition of wetlands whose creation, restoration or enhancement objectives focused on habitat or water quality treatment, or both (multi-purpose)?
- How does the frequency of maintenance impact the condition of urban wetlands?
- How can the habitat value of urban wetlands be improved, whether they are created for natural (habitat value), multipurpose or treatment wetlands?

Methods

Definition of Target Population

Three categories of wetlands and riparian projects were considered "urban wetlands" for this study:

- *Habitat wetlands* - Natural wetlands and associated riparian areas restored or enhanced for habitat, but with urban runoff as the primary water source
- *Multipurpose wetlands* - Natural wetlands or riparian habitat (restored or enhanced) or wetlands or riparian areas created to serve the multiple objectives of habitat creation and water quality improvement
- *Treatment wetlands* - Wetlands or riparian areas constructed and/or engineered for water quality improvement, with habitat as an ancillary objective (as in Kadlec and Knight 1996)

"Wetlands" are defined using the Cowardin *et al.* (1979) definition. Two general types of wetlands were of interest.

Basins – Basins are freshwater depressional wetlands that consist of habitat types known colloquially with such terms as ponds, pools, wet meadows, treatment wetlands, or wetland detention basin. They are characterized by a topographic depression that provides for temporary storage of flood flows. They may have a defined inflow or outflow, but neither is required for inclusion in this class.

Channels – Channels refer to small tributary streams and other freshwater wetlands that are characterized by a linear channel with flowing water of short retention times (<3 h, depending on the size of the site) and a defined inflow and outflow.

These two wetland types were targeted for inclusion in the inventory and subsequent baseline biological survey because they are of primary interest to regional water quality and natural resource managers. All forty urban wetland treatment sites, identified through the first phase of this study, and four reference sites representing "best attainable condition" in undeveloped open space were selected. Sites were distributed throughout San Diego, Orange, and Los Angeles Counties, and parts of Ventura, Riverside, Imperial and San Bernardino Counties in Southern California, USA (Figure 14).

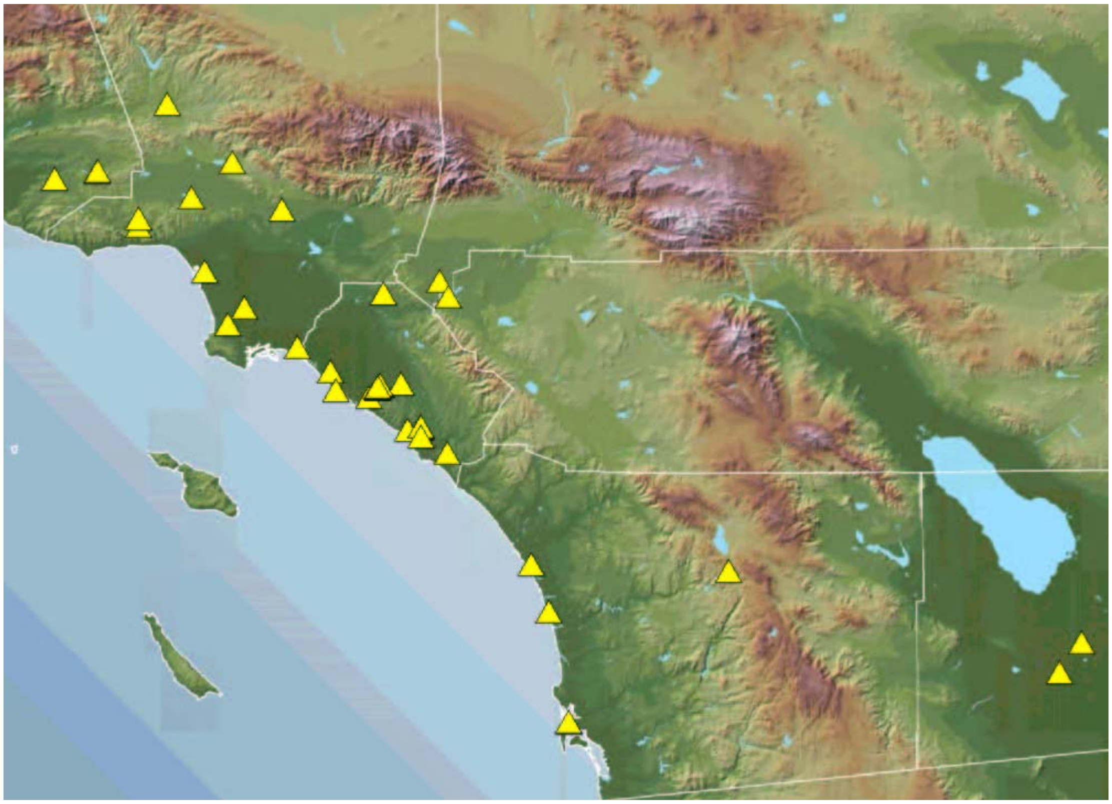


Figure 14. Yellow triangles designate the location of urban wetlands included in the study. Most sites were located within the coastal watersheds of southern California, located on the pacific coast of the United States.

General Design of Biological Survey

The general design of the baseline biological survey consisted of sampling activities to describe baseline ecological condition at three tiers. **Tier 1:** sampling consisted of a landscape assessment of the ecological diversity of wetland and vegetation communities through habitat mapping and characterization of landscape-scale stressors derived from surrounding land use at the catchment scale. Tier 1 activities were conducted on 40 urban wetland sites identified through Phase I activities plus 4 reference sites. **Tier 2:** sampling consisted of a rapid assessment of the general habitat condition using CRAM Version 4.0 (Collins *et al.* 2006, Sutula *et al.* 2006), a field-based method that can be used to rapidly assess the general ecological condition of the site. As with Tier 1 sampling, Tier 2 sampling was conducted on all 44 sites. **Tier 3:** sampling represents more intensive evaluation of habitat condition via a suite of vascular plant, benthic macroinvertebrate, and avian surveys, and was conducted on a subset of the sites (26 sites plus 4 reference sites). Table 3 gives an explanation of the purpose and general approach of each indicator used in the survey. Table 4 gives a list of CRAM attributes and metrics. Sampling activities were conducted from April through June 2006.

Table 3. Indicator and objective of sampling conducted for biological survey.

Sampling Tier	Type	Indicator Name	Objective and Method References
1	Landscape	Wetland and Vegetation Field Mapping	Description of types and extent of wetland habitat types. Wetland and riparian vegetation were mapped using the USFWS (Cowardin *et al.* 1975) and California Department of Fish and Game Vegetation Classification system (Sawyer and Keeler-Wolfe 1995).
1	Landscape	% Imperviousness of Surrounding and Catchment Land Use	Use % imperviousness to assess the intensity of land use surrounding the wetland (i.e., within 100 m of wetland border) and within the wetland's catchment. Source of % Imperviousness data layer was 2001 National Land Cover Database (NLCD). The area defined by the catchment was modeled based on either a 30-m or 10-m digital elevation model using the ARC Hydro component of ARC INFO Version 9.1 (ESRI 2006).
2	Rapid Assessment	General Condition Assessment	General assessment of overall habitat using California Rapid Assessment Method (CRAM Version 4.0, Collins *et al.* 2006 (www.cramwetlands.org, Sutula *et al.* 2006).
3	Plants	Plant Community Composition	Evaluation of plant community composition consisted of the use of line transects (Magurran 1988) to obtain information about the amount of vegetation cover, plant species richness and relative abundances, recruitment, the proportion of invasive species, and the proportion of wetland plant species (Reed 1988).
3	Benthic Macroinverte-brates (BMI)	BMI Community Characterization	The objective of BMI sampling in the wetlands was to characterize the relative abundance and diversity of invertebrates, with approximately equal effort given to the sampling of all major habitat types. Methods used were based on draft California Department of Fish and Game bioassessment procedures for lakes and ponds (CDFG ABL 1996a) and various EPA protocols for BMI sampling in ponded environments (EPA 1990). For flowing waters and channels the methods follow standard California and EPA methodology, as well (ABL 2003, Harrington and Born 2000, EPA 1990).
3	Birds	Bird Use	The objective of bird surveys in the wetlands was to characterize the relative abundance and diversity of birds, with approximately equal effort given to the sampling of all major habitat types. The basic method was a modification of the Area Search method (Ralph *et al.* 1993). The method is quantitative and mimics the method that birders use while searching for birds in an area. This method uses a series of point counts in different plots within the site with the observer moving around in a restricted area to cover the entire area of the wetland.

Table 4. CRAM attributes and metrics.

	Attribute	Metrics
CRAM Index Score	Landscape Context	Landscape Connectivity
		Percent of AA with Buffer
		Average Buffer Width
		Buffer Condition
	Hydrology	Water Source
		Hydroperiod or Channel Stability
		Hydrologic Connectivity
	Physical Structure	Structural Patch Richness
		Topographic Complexity
	Biological Structure	Organic Matter Accumulation
		Interspersion and Zonation
		Number of Plant Layers Present
		Percent of Layers Dominated by Native Species
		Number of Co-dominant Species
		Percent of Co-dominant Species that are Native
		Vertical Biotic Structure

Data Analysis

Data from the urban wetland inventory sites were analyzed to describe patterns in the general characteristics of this population that defined objective, hydrologic management and maintenance intensity and wetland size.

Percent imperviousness was used as a proxy for degree of urbanization in the land use surrounding the wetland site and in its upstream catchment. These variables were regressed as independent variables against CRAM index scores and attributes, for plant community structure metrics, benthic macroinvertebrate metrics, and avian diversity metrics for wetlands in the study, including reference sites.

The impact of project objective (habitat, multi-purpose, and water quality) on the ecological condition of urban wetland sites was evaluated through ANOVAs for mapped habitats, CRAM, plant community structure, benthic macroinvertebrate community and avian survey metrics. Data on reference sites were excluded in the analysis but presented for comparison to the mean value for project objective.

The impact of wetland origin (historic natural, type converted from one natural wetland type to another, or created from upland) on the ecological condition of urban wetland sites (reference sites excluded) was evaluated through an ANOVA for CRAM, plant community structure, benthic macroinvertebrate community and bird use metrics.

The correlations between the intensity of maintenance of the wetland and CRAM index and attribute scores were examined. Maintenance activities could include activities such as vegetation management; water quality or hydrologic maintenance, etc. Intensity of maintenance was defined as a categorical variable with three bins: 1) maintenance activities conducted less than once per year, 2) 1 - 2 times per year, and 3) greater than 2 times per year. Reference wetlands were removed from the data set for this analysis.

Results

Summary of Urban Wetland Characteristics

Table 5 gives an approximate allocation of the 40 urban sites in the study by type and objective. Of the basin wetlands, the number of passively versus actively managed wetlands (i.e., water pumped in order to maintain water levels or hydroperiod) was roughly equal. Of the channels, only one of 15 sites was actively managed via pumping water from an impoundment upstream of the sites. Fifty-one percent of the 29 basin wetlands were on a high intensity maintenance schedule (>2 times per year). There were 2-3 times the numbers of basin wetlands in this category as in the moderate (1 – 2 times per year) or low intensity (<1 time per year) category. In contrast, 80% of channel wetlands were in the moderate or low intensity category for maintenance; no water quality channels had maintenance activities in the high intensity category.

Urban and reference wetland sites (n = 44) were generally small, with the majority of sites falling under 5 acres for both basins and channels (Figure 15). While the results of the inventory of urban wetlands is not a comprehensive inventory of freshwater wetlands in southern California, the distribution of basin sites by origin is notable in that most (63 - 80%) of these systems have been type converted from other historic wetland types (i.e., from estuarine wetlands or rivers and streams to basins). While 20% of treatment wetlands in the inventory have been created from upland areas, most have been created through type conversion from other wetland classes.

Table 5. Approximate allocation of urban sites by wetland type and habitat objective.

Wetland Type	Creation or Restoration Objective	Tier 1 and 2 Sites	Tier 3 Sites
Basins (includes isolated basins and basins with pumped or diverted flows)	Habitat	13	7
	Habitat and Water Quality	8	6
	Water Quality	5	5
Channels (includes Small Tributary Systems and Channels with Pumped or Diverted Flows)	Habitat	4	4
	Habitat and Water Quality	5	4
	Water Quality	5	2
Total		40	28

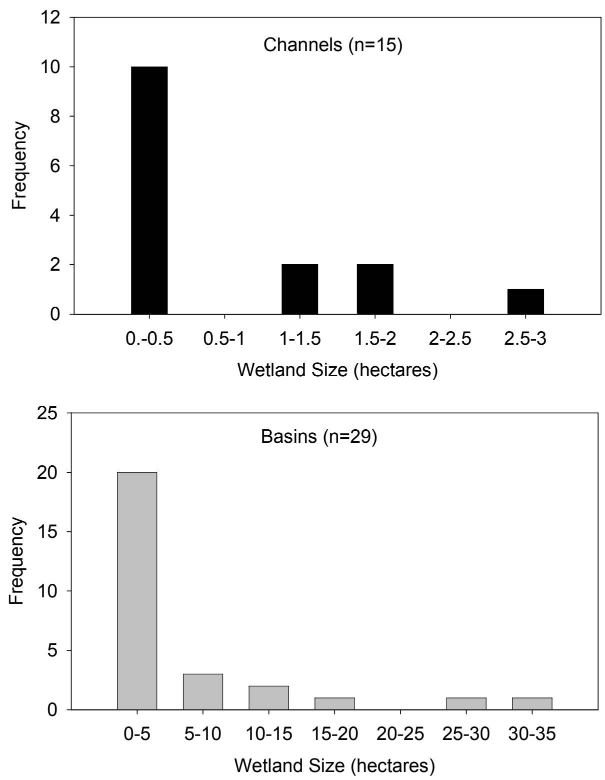


Figure 15. Histogram of size distribution of wetlands by channels (n = 15) and basins (n =29). Wetland boundaries were defined by rules for CRAM assessment area delineation.

Condition of Wetlands Along a Gradient of Urbanization

The condition of the wetland with respect to CRAM index and attribute scores, as well as benthic macro invertebrate community composition, showed a significant negative correlation with increasing % imperviousness (Figure 16; Table 6). Regressions of benthic macroinvertebrate data showed a declining condition in benthic macroinvertebrate community structure with increasing percent imperviousness in the catchment (Table 5). For basins, benthic macroinvertebrate taxa richness, insect richness, percent intolerant species, and percent dominant species, percent Chironomid midges and percent Odonata were all significantly correlated with percent imperviousness at an $\alpha = 0.05$. For channels, only a subset of these was significant (benthic macroinvertebrate taxa richness, insect richness, and percent intolerant species). For both channels and basins, no BMI metrics were significantly correlated with land use in the 100 m surrounding the wetland.

These trends were conserved with respect to plant community composition in channel sites; however, only relative cover of native plants was negatively correlated with an increase in percent imperviousness of surrounding land use for both basins and channels (Table 6, Figure 16). Other metrics such as native species richness and Shannon indices (native species and all species) showed a positive correlation with percent imperviousness in surrounding land use. Regressions of percent imperviousness versus avian community structure indices showed no significant correlations for any of the variables at an $\alpha = 0.05$.

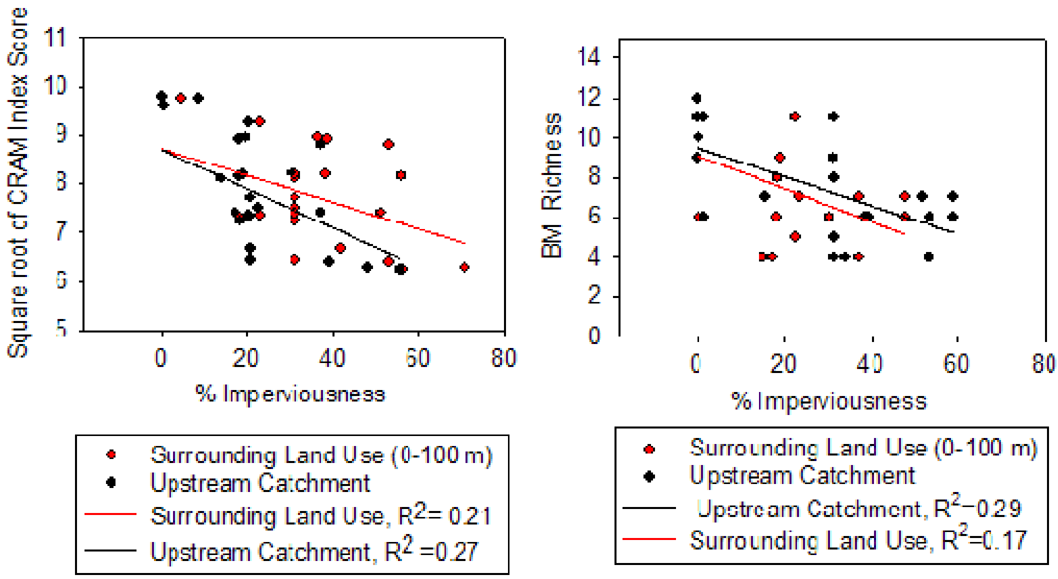


Figure *16.* Linear regression of CRAM index score and benthic macroinvertebrate taxa richness versus percent imperviousness of land use within surrounding 100 m around the wetland and within the catchment for basins. The relationship was significant for basins (p-value = 0.005) but not for channels (p-value = 0.41, R^2 = 0.08).

Table 6. Summary statistics describing correlation between CRAM index scores and attributes and the percent imperviousness in the surrounding land use and catchment of basin wetland sites. Data for channels are not presented.

CRAM Score	% Imperviousness in Surrounding Land Use (0-100 m)		% Imperviousness in Catchment	
	p-value$_{\alpha=0.05}$	R^2	p-value$_{\alpha=0.05}$	R^2
Index Score	0.0005	0.27	0.02	0.21
Landscape Context	0.014	0.24	0.02	0.21
Hydrology	0.02	0.22	0.01	0.26
Physical Structure	0.50	0.02	0.69	0.01
Biological Structure	0.11	0.10	0.19	0.07

Table 7. Summary statistics describing correlation between BMI metrics and percent imperviousness of catchment land use. P-values α = 0.05 and the model R^2 for correlation are given. Correlations with surrounding land use (0 - 100 m) were generally not significant at an α = 0.05.

BMI Metric	Basins, catchment land use		Channels, catchment land use	
	p-value$_{\alpha=0.05}$	R^2	p-value$_{\alpha=0.05}$	R^2
Density (#/m^3)	0.06	0.22	0.85	0.004
Richness (# of taxonomic groups)	0.03	0.26	0.02	0.43
Insect richness (Insect groups only)	0.006	0.38	0.04	0.39
Non-insect richness	0.36	0.05	0.13	0.23
Tolerance value	0.05	0.22	0.75	0.01
% Intolerant species	0.02	0.28	0.04	0.42
% Dominant species	0.03	0.27	0.09	0.28
% Collectors	0.07	0.19	0.69	0.01
% Chironomid midges	0.01	0.36	0.27	0.13
% Odonata	0.02	0.27	0.72	0.01

Table 8. Summary statistics describing correlation between plant community metrics and percent imperviousness of surrounding land use (0 - 100 m). P-values α = 0.05 for the percent imperviousness term and the model R^2 are given.

CRAM Score	% Imperviousness in Surrounding Land Use (0-100 m)	
	% Imperv. p-value$_{\alpha=0.05}$	R^2
Evenness of Layers	0.09	0.29
Relative % Cover Native Plants	0.02	0.27
Relative % Cover of Invasive Plants	0.22	0.12
Native Plant Species Richness	0.007	0.52
Resid. Native Plant Species Richness	0.59	0.09
Shannon Index-All Plant Species	0.09	0.42
Shannon Index-Native Plant Species	0.01	0.53
Plant Species Recruitment	0.39	0.06

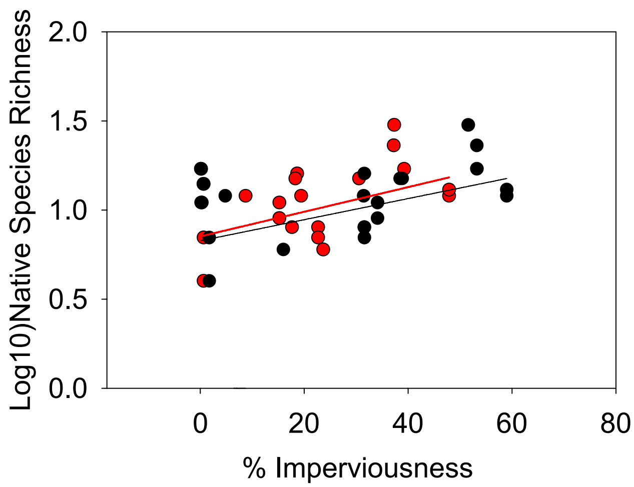


Figure 16. Scatter plots and linear regressions of native plant species richness versus percent imperviousness of surrounding land use (0 - 100 m, red) and catchment (black), showing a positive relationship between the two.

Impact of Objective on Wetland Condition

At a landscape scale, basin wetlands were comprised generally of emergent marsh and represented the greatest percentage of habitat (42%) followed by wetland riparian (35%), with deep open water (>2 m) and shallow open water (<2 m) making up about 20% of the habitat mapped. The relative distribution of habitat in basin wetlands varied by wetland objective; water quality wetlands had double the amount of open water habitat and a quarter to a third the amount of emergent and forested wetland as habitat wetlands. The richness of mapped vegetation communities was also significantly correlated with wetland objective (p-value = 0.01, $R^2 = 0.27$), with wetlands with a habitat or multipurpose objective showing significantly higher richness than water quality wetlands (Figure 17). Vegetation community richness was directly proportional to size for both basins and channels (p-value = 0.001, $R^2 = 0.29$).

Many of the basin wetland class have both wetland riparian habitat and non-wetland riparian habitat in the upland transition zone encircling the basins. In some cases, this riparian habitat was a significant part of the wetland ecosystem. Among basin wetlands, 58% of the non-wetland riparian cover surrounding these systems was non-native and/or invasive, with eucalyptus, ornamental trees, and ruderal shrubs and annual grasses as the most common types. Channels and other types of riverine habitat would be expected to have riparian wetlands in the channel as well as non-wetland riparian habitat in the upland transition zone. Fourteen of fifteen sites had some riparian habitat (either wetland or non-wetland), with native riparian forest dominating 60% of the total non-wetland riparian cover.

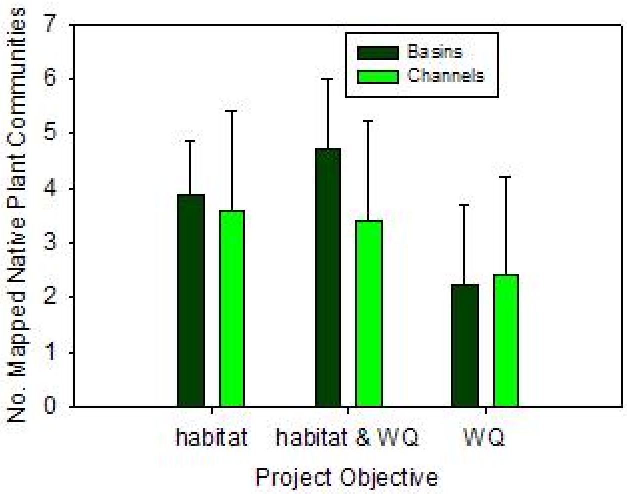


Figure 17. Number of mapped native plant communities as a function of objective and wetland type, showing significantly higher number of plant communities associated with habitat and multipurpose wetlands than with water quality wetlands. Error bars represent 95 % confidence intervals.

Results of rapid assessment of basins and channels show a range of condition relative to reference sites (Figure 18). The mean CRAM index scores for urban sites were 56% (± SD 14) for basins and 45%. (± SD 16) for channels. Site scores were 100% for the single reference channel system and 94% (± SD 2) for the three reference basin wetlands. Urban basin and channel sites scored highest on biological structure (represented mainly by plant community structure); urban basin sites generally scored the worst on hydrology while urban channels scored the worst on landscape context (buffer and linear connectivity of riparian corridor).

Among urban wetland sites, habitat wetlands had the highest mean CRAM index and attribute scores and water quality wetlands had the lowest scores. However, because of high variability among sites, significant relationships were found only for project objective for CRAM physical structure in basins and for both CRAM index score and physical structure for channels (Table 9; Figure 19).

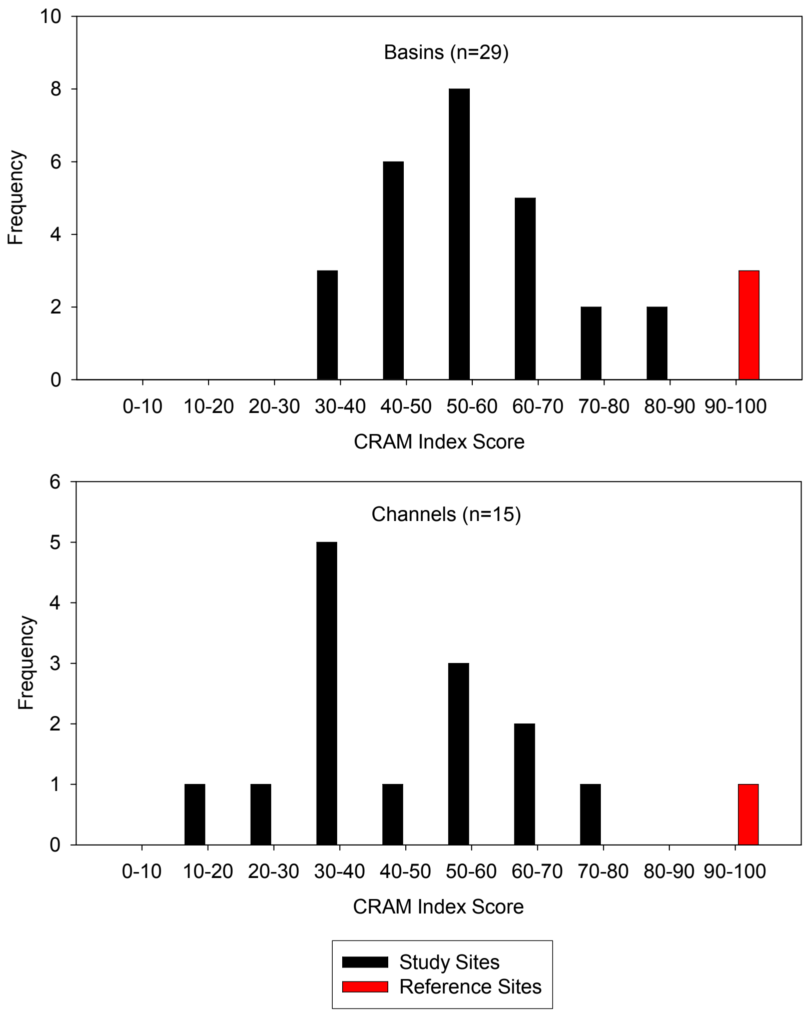
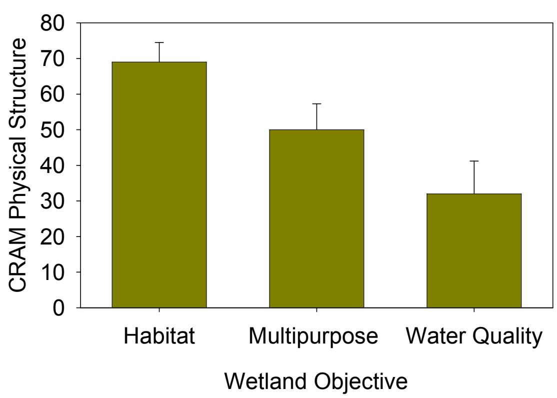


Figure 18. Distribution of CRAM index scores relative to reference standard sites for basins and channel wetlands. The red bar indicates scores for the reference sites. Black indicates the scores for the urban wetland inventory sites.



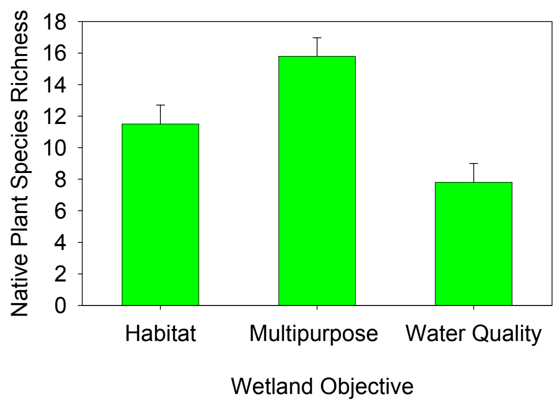


Figure 19. CRAM physical structure scores and native plant species richness for basin sites as a function of objective. Error bars represent 95 % confidence intervals.

Table 9 Summary of ANOVA statistics and post-hoc Tukey's comparisons for CRAM index and physical structure scores by project objective for basins (n = 26) and channels (n = 14), excluding reference sites. ANOVAs of CRAM biological structure, landscape context, and hydrology were not significant with respect to objective at α=0.05.

CRAM Score	Project Objective					
	p-value$_{\alpha=0.05}$	Model R^2	Habitat	Multipurpose	Water Quality	Reference
Basins (n = 26 urban + 3 reference sites)						
CRAM Index Score	0.22	0.18	67(4)	52(6)	49(7)	95(6)
CRAM Physical Structure	0.005	0.34	69(5)	50(8)	32(10)	100(12)
Channels (n=14 urban + 1 reference site)						
CRAM Index Score	0.020	0.50	62(6)	41(5)	35(6)	100
CRAM Physical Structure	0.04	0.42	67(14)	30(13)	13(13)	100

Among the intensive (Tier 3) indicators of wetland condition, there was no clear weight of evidence about the differences among the sites by wetland objective. Comparison of plant community structure metrics for urban sites relative to reference standard sites show variable trends for basins and channels (Table 10). In general, reference standard sites had higher mean species richness and diversity of natives. Urban sites were comparable in magnitude with respect to relative cover of natives and recruitment index. A histogram of native species richness and relative cover of invasive plant species shows that some urban sites can actually score higher than reference sites. This was particularly true for native plant species richness in basins, where the range of native species richness in reference sites varied from 12 to17 versus 4 to 30 for urban sites (Figure 24).

For basins, significant relationships were found by project objective for native plant Shannon evenness and native plant species richness (Table 10; Figure 25). In both these cases, multipurpose wetlands had the highest evenness and richness. For channels, no significant relationships were found for any of the plant metrics. The trends were generally similar to those from basins, except that habitat wetlands tended to have higher values (and in particular, higher evenness, higher native species richness, and Shannon

Index diversity) than multipurpose and water quality wetlands. Water quality wetlands generally had the lowest diversity and richness metrics, but had higher relative cover of natives and lower cover of invasive species than habitat or multipurpose wetlands (Table 10).

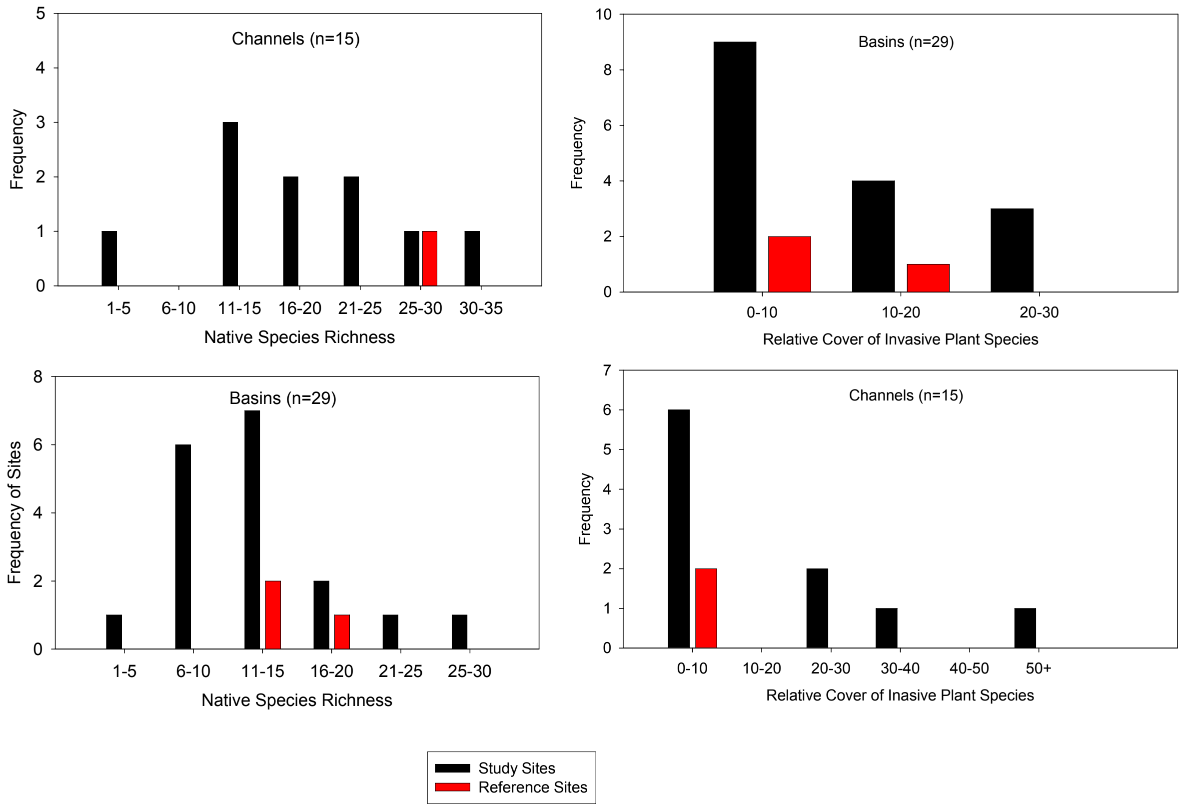


Figure 20. Distribution of plant species richness and relative percent cover of invasive plant species in urban sites relative to reference standard sites for basins and channel wetlands. Red bars indicate scores for reference sites. Black bars indicate scores for urban wetland inventory sites.

Of the benthic macroinvertebrate indicators, mean macroinvertebrate taxa richness, insect taxa richness was higher at reference sites than at urban sites (Figure 21). Most indicators show trends indicating that reference sites are better than urban sites, though the variability in urban sites was high and the sample size too small for meaningful statistical comparison. No significant relationships were found by project objective for BMI metrics in basin wetlands; however, general trends with respect with most metrics consistently pointed to a trend of water quality wetlands having the highest condition scores (Table 11). For channels, the general trends were inconsistent among metrics and no significant differences were observed by wetland objective.

Table 10. Summary of ANOVA statistics and post-hoc Tukey's comparisons by objective (or plant community metrics by basins and channels, excluding reference sites. Mean values with the same superscript letters are not significantly different (p>0.05) from one another based on pair-wise comparisons. Superscripts indicate significant differences.

CRAM Score	Project Objective				
	p-value$_{\alpha=0.05}$	Habitat	Multipurpose	Water Quality	Reference
Basins (n = 18 urban + 2 reference sites)					
Mock Shannon Evenness	**0.008**	^{A,B} 0.5 (0.3)	^{A}0.9 (0.5)	^{B}0.4 (0.3)	0.9 (0.2)
Relative Cover Natives	0.39	87(11)	82(18)	93 (9)	91(7)
Relative Cover of Invasives	0.79	9 (9)	9(8)	6 (8)	8(5)
Native Species Richness	**0.05**	^{A,B} 11.5 (6.2)	^{A} 15.8 (4.7)	^{B} 7.8 (3.9)	14 (3)
Shannon Index All Spp.	0.18	2.0 (0.6)	2.2 (0.5)	1.6 (0.5)	2.0(0.2)
Shannon Index Native Spp.	0.44	1.7 (0.5)	1.9 (0.7)	1.5 (0.4)	2.0 (0.4)
Recruitment	0.06	0.65 (1.7)	13 (10)	0 (0)	3.8(1.8)
Channels (n=10 urban + 1 reference site)					
Mock Shannon Evenness	0.40	1.0 (0.1)	0.8 (0.2)	0.9 (0.1)	1.1
Relative Cover Natives	0.12	83 (13)	63 (26)	95 (1)	96
Relative Cover of Invasives	0.12	8 (2)	20 (4)	2 (3)	2
Native Species Diversity	0.34	23 (4)	16 (11)	14 (1)	14
Shannon Index All Spp.	0.20	2.8 (0.2)	2.3 (0.5)	2.2 (0.1)	2.6
Shannon Index Native Spp	0.20	2.5 (0.4)	2.0 (0.5)	2.0 (0.1)	2.4
Recruitment	0.40	1.3 (1.7)	2.0 (2.3)	0 (0)	12.1

Table 11. Summary of Kruskal-Wallis, nonparametric test comparisons by project objective for benthic macroinvertebrate community metrics by basins (n = 16) and channels (n = 10), excluding reference sites. Mean and standard deviation of reference sites provided for comparison.

CRAM Score	Project Objective				
	p-value$_{\alpha=0.05}$	Habitat Mean (SD)	Multipurpose Mean (SD)	Water Quality Mean (SD)	Reference
Basins (n = 18)					
Richness (# of taxonomic groups)	0.92	6.3 (0.8)	6.7 (1.3)	7.4 (1.0)	10(1)
Insect richness	0.81	3.0(0.7)	3.7(1.1)	4.4 (1.8)	7(1)
Tolerance value	0.93	7.4 (0.4)	6.6(0.7)	6.6 (0.5)	6(1)
% Intolerant species	0.62	0.04(0.01)	0.001(0.06)	0.69(0.06)	4(5)
% Dominant species	0.42	73 (8)	84 (13)	63 (11)	63(12)
% Chironomid midges	0.26	0.4 (5)	1.5 (10)	2.8 (6)	1.7(2.4)
Odonata richness	0.42	0.4(0.2)	0.7 (0.3)	1.2 (0.3)	2(0)
Channels (n = 10)					
Taxa Richness	0.34	12.0 (5)	8.0 (6)	3.5(3)	2
Insect richness	0.61	4.0 (1.4)	4.4 (1.0)	2.5 (1.7)	10-
Tolerance value	0.31	6.6 (0.7)	5.6 (0.5)	6.0 (0.83)	6
% Intolerant species	0.69	0.44 (0.03)	0.01 (0.07)	0.01 (0.08)	5
% Dominant species	0.37	56 (13)	72 (10)	83 (17)	30
% Chironomid midges	0.09	14(8)	23 (6)	22 10)	9
Odonata richness	0.87	0.7(0.3)	0.8 (0.3)	1.0 (0.4)	1

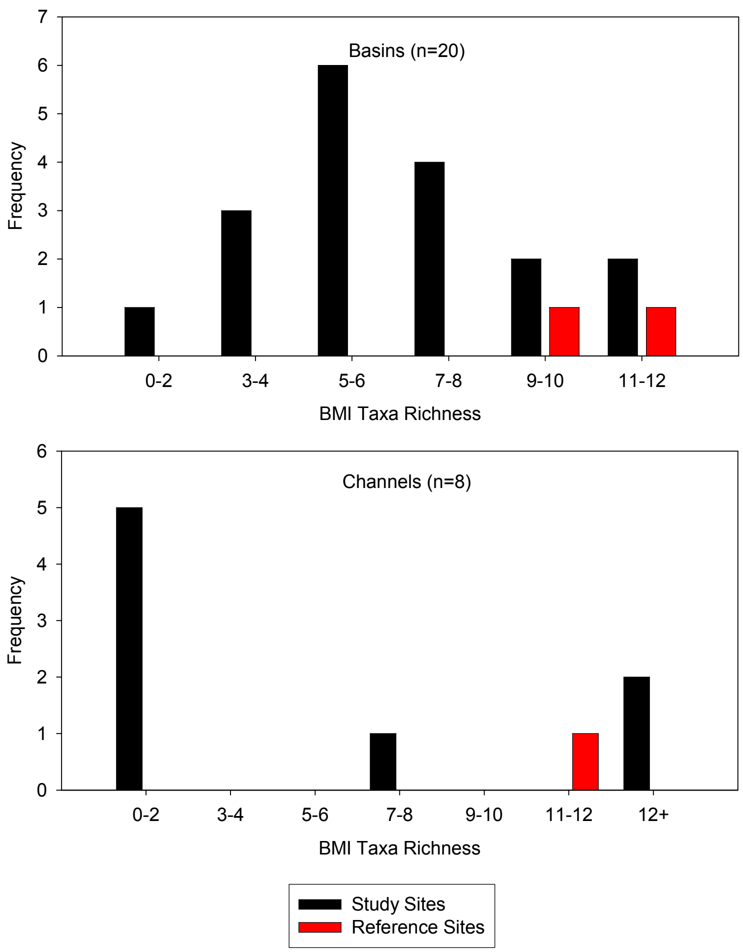


Figure 21. Distribution of benthic macroinvertebrate taxa richness in urban (study) sites relative to reference standard sites for basins and channel wetlands.

Analysis of summary data on avian diversity, richness, abundance, and evenness indices for urban versus reference standard sites showed that some urban wetlands actually scored higher than the reference wetlands (Table 12). Many of the reference sites showed diversity and richness values that were at or just slightly higher than the mean value for the urban sites (Figure 22). One confounding factor with these results is the relationship of these indices with size. All indices were significantly correlated with size with p-value <0.05 and R^2 ranging from 0.30 to 0.36. When size-corrected avian species diversity and richness were compared among urban and reference sites, the basins reference sites were the top-ranked sites for diversity and among the top 25% for richness. With respect to avian species richness, diversity and abundance, no significant relationships were found by project objective for basin wetlands; however, general trends with respect to most metrics consistently pointed to a trend of habitat wetlands having the highest diversity, richness, and abundance (Table 12). For channels, habitat wetlands had significantly higher diversity, richness, and abundance than multipurpose or water quality channels.

Avian diversity and richness were also impacted by the percentage of certain habitat types within basin wetlands. For example, both species richness and diversity declined as the percentage of deep open water increased (p-value = 0.03, R^2 = 0.21), while these indices increased as the percentage of shallow open water increased (p-value = 0.03, $R^2 = 0.21$). Relationships with the percent of wetland riparian habitat and percent emergent marsh were not significant at a p-value = 0.05.

Table 12. Summary of ANOVA statistics and post-hoc Tukey's comparisons for avian survey metrics by basins (n = 18) and channels (n = 10), excluding reference sites. Mean values with the same superscript letters are not significantly different (p >0.05) from one another based on pair-wise comparisons. Where no subscripts are shown, no significant differences were found.

CRAM Score	Project Objective				
	$\text{p-value}_{\alpha=0.05}$	Habitat Mean (SD)	Multipurpose Mean (SD)	Water Quality Mean (SD)	Reference
Basins (n = 18 urban sites + 2 reference sites)					
Avian Species Diversity	0.57	2.4 (0.3)	2.5 (0.3)	2.0 (0.3)	15.0(3.3)
Avian Species Richness	0.63	18(2.3)	17.8(2.7)	15.0 (2.5)	2.0(0.4)
Avian Abundance	0.54	49 (7)	54(9)	^{A} 41 (8)	41.1(1.2)
Channels (n = 10 urban sites + 1 reference site)					
Avian Species Diversity	0.002	^{A} 2.3 (0.4)	^{B} 1.1 (0.4)	^{B} 1.0 (0.6)	15.0
Avian Species Richness	0.003	^{A} 21.3 (1.1)	^{B} 10.0 (0.9)	^{B} 7.0 (1.3)	2.0
Avian Abundance	0.004	^{A} 52 (6)	^{B} 21 (4)	^{B} 21 (3)	40.0

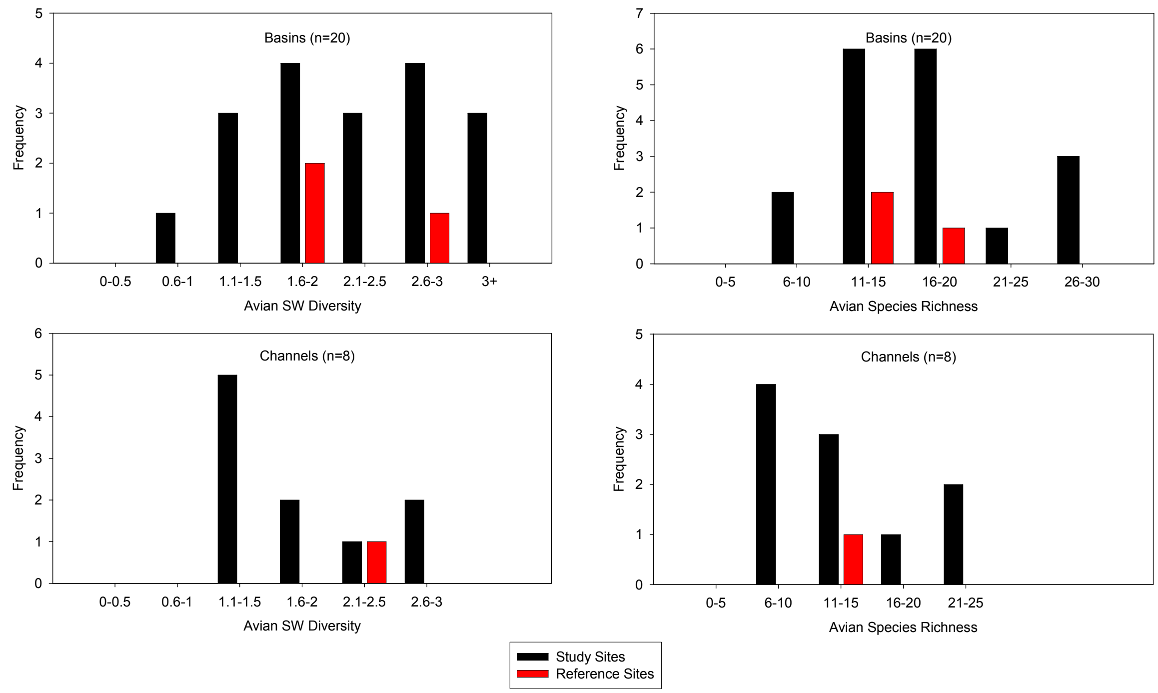


Figure 22. Distribution of avian diversity and species richness in urban (study) sites relative to reference standard sites for basins and channel wetlands. Red bars indicate reference sites. Black bars indicate data for urban inventory sites. Correction for size places the reference sites within the top 25% of sites for both indicators.

Ecological Condition of Urban Wetlands as a Function of Other Characteristics

Origin. In basin sites, wetland origin was significantly correlated with the CRAM index and physical structure scores and with plant community metrics such as native plant species richness for basins; Tukey's pairwise comparisons of means show that historic, natural wetlands have significantly higher CRAM scores and native plant species richness than type converted or created wetlands (Table 13). ANOVAs of CRAM biological structure, hydrology and landscape context and plant species diversity and recruitment index were not significant with respect to origin at $\alpha = 0.05$.No correlations were found with respect to origin of channel wetlands for CRAM index or attributes scores (p-value$_{\alpha=0.05}$ >0.05).

Benthic macrovertebrate communities showed significantly higher percent tolerant taxa for wetlands that were maintained at a frequency of >2 times per year (58%) versus those maintained 1 - 2 times per year (19%) or >1 time per year (10%; Kruskal Wallis test, SR p <0.05).

With respect to birds, no significant differences were found among those wetlands that were historic, created, or converted from one wetland type to another (Kruskal-Wallis test; SR p = 0.171, S-W p = 0.115).

Table 13. Summary of the of the ANOVA statistics and post-hoc Tukey's comparisons (mean and confidence interval) for CRAM index, physical structure scores and native plant species richness by origin for basins (n = 29).

Variable	Origin				
	p-value$_{\alpha=0.05}$	Model R^2	Historic	Type Converted	Created
Basins (n = 29)					
CRAM Index Score	0.001	0.28	73(55-95)	56(48-65)	38(24-61)
CRAM Physical Structure	0.001	0.50	78(53-99)	53(43-65)	17(8-32)
Native Plant Species Richness	0.002	0.41	9 (11-26)	10 (7-13)	6(0-10)

Maintenance Intensity. Wetlands that were maintained at an intermediate frequency (1 - 2 times per year) had the highest mean plant species richness, while the lowest richness was associated with the highest frequency of maintenance (Figure 23). A linkage exists between the maintenance of the buffer surrounding the wetlands and the condition of the wetland's plant community. Figure 24 indicates a significant relationship between buffer quality, which is mathematically derived from the three CRAM buffer metrics, and relative percent cover of invasive plant species.

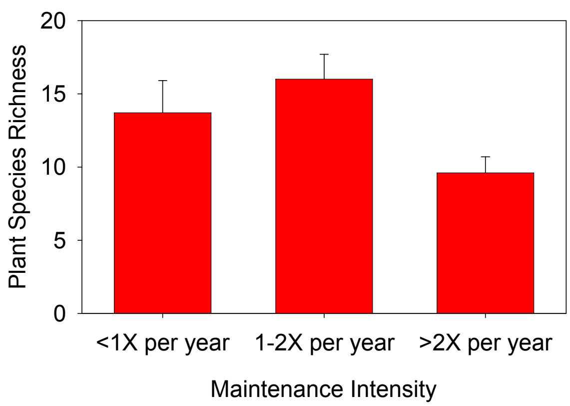


Figure 23. Differences in plant species richness at urban wetland sites as a function of the number of times maintenance is conducted at the site. The category sites with of 1 - 2 times per year has significantly greater plant species richness than the >2 times per year category.

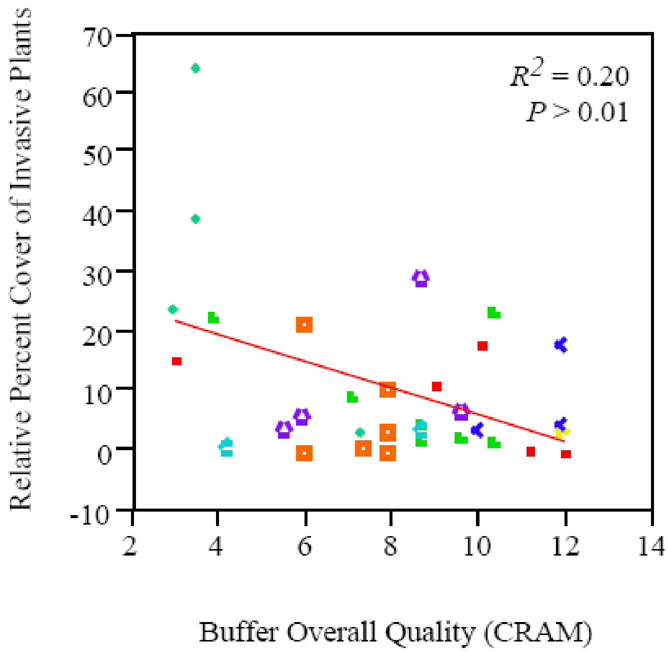


Figure 24. Relationships between buffer overall quality and relative percent cover of invasive plant species.

Discussion

With the increasing urbanization of coastal areas over the past century, the dual problem of declining water quality and loss and degradation of wetland and riparian habitat has popularized the concept of creating or enhancing existing wetlands to promote both habitat benefits and water quality enhancement (Azous and Horner 2000). The extent to which trade-offs exist between habitat and water quality objectives has not been thoroughly investigated (Sutula and Stein 2003). This study attempted to answer four key management questions surrounding the use of wetlands to provide both habitat and water quality benefits:

- How does ecological condition in sites created, restored or enhanced for habitat vary along a gradient of land use intensity (using percent imperviousness as a proxy)?
- What is the difference in the ecological condition of wetlands whose creation, restoration or enhancement objectives focused on habitat or water quality enhancement, or multi-purpose?
- How does the frequency of maintenance impact the condition of urban wetlands?
- Is the origin of the site (historic wetland, wetland created from upland, or type converted wetland) correlated to site condition?

An understanding of what constitutes a wetland habitat in good ecological condition is hampered by the lack of true reference wetlands, particularly for basin wetlands. Thus, baseline biological survey results reflect a state of "best achievable", but do not necessarily provide an adequate benchmark for the characteristics of a wetland in good ecological condition. Results from this baseline biological survey illustrate that, while urban infrastructure provide some basic constraints on "best achievable" wetland condition, site specific factors such as wetland project design criteria, objectives, wetland management and maintenance activities can greatly mitigate the constraints of the urban landscape.

Reference Wetlands in an Urban Landscape

One key assumption in the analysis of the baseline biological survey data is that the reference sites provide benchmarks for the characteristics of a pristine wetland with its biological communities intact. For basin wetlands, this assumption is in question. A recent study of the historical ecology of wetlands in the San Gabriel River Watershed in southern California provides an understanding of the amounts and types of wetlands that once dominated the landscape of the Los Angeles Basin (Figure 15; Stein *et al.* 2007). This study showed that perennial ponds were once less than 0.4% of the approximately 35,000 acres of freshwater wetlands in this watershed; rather, the landscape was dominated by alkali and wet meadows and other types of seasonal wetlands. Two of the three basin reference wetlands were perennial ponds, both of which are relic stock ponds that have been relatively undisturbed since the 1960s. Thus reference for basin wetlands in this study represents an underlying shifting of baselines towards systems that were presumably relatively rare in the southern California landscape. Thus, while these reference wetlands provide a benchmark for "best achievable" for these perennial ponds, additional research is needed to understand the historical ecology of the freshwater wetlands in southern California and the native flora and fauna that once inhabited them. For channels, this issue is present but less of a concern because of the availability of pristine sites in the upper watershed (mostly on national forest land) throughout the region. It is uncertain whether these sites provide an adequate reference for small stream on the coastal terraces, which are now largely disturbed and for which reference sites are no longer available.

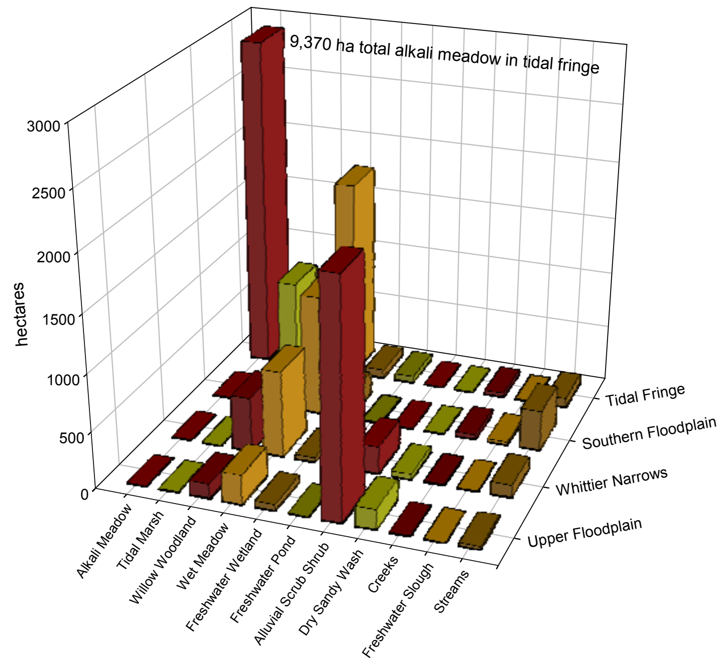


Figure 25. Graphic reproduced from Stein *et al.* (2007) on the historic extent and distribution of wetlands in the San Gabriel River watershed. Distribution of historical wetland types in various portions of the San Gabriel River floodplain. Approximately 9,300 ha of alkali meadow existed in the tidal fringe area; the plot has been truncated to facilitate presentation.

Landscape Context Provides Basic Constraints on Achievable Condition of Urban Wetlands

As the intensity of land use in the area surrounding or just upstream of a wetland increases, stressors increase, thus degrading the condition of the habitat (Mack *et al*. 2006). Stressors can include factors that impact the site at a landscape scale, such as hydro-modification, increased sedimentation, increase in loading of contaminants, or at the site scale, such as excessive human visitation, predation from dogs, cats, and other urban fauna, compaction of soils, etc. Percent imperviousness can serve as just one type of proxy variable for intensity of land use; another example is the Landscape Development Intensity index (LDI; Brown and Vivas 2004).

In this study, we documented the decline in general habitat condition (as measured by CRAM), and benthic macroinvertebrate community condition with increasing imperviousness of either the surrounding, or catchment, land use. However, not all indicators of wetland condition declined with increasing % imperviousness; native plant species richness and plant species diversity actually increased significantly. Several urban wetland sites surrounded by highly impervious land covers had native plant species richness and diversity that exceed that found in reference sites. This indicates that, while degree of urbanization places some basic constraints on what condition a wetland site can be expected to have, site-specific factors can mitigate (or further exacerbate) the effects of the landscape constraints on wetland condition.

Site-Specific Factors Impacting Condition of the Wetland

This study showed that several site-specific factors can influence the ecological condition of urban wetlands. These include:

- Wetland size
- Project objective and design criteria
- Origin of wetland (historic wetland habitat or created from upland)
- Intensity of hydrological management and general maintenance activities

A discussion of how these factors can impact condition is presented below.

Wetland Size. Wetland size was a major controlling factor on the condition of urban basin and channel wetlands. The majority of urban wetlands in this study were very small, with most basin sites under 5 acres. The condition of the wetland with respect to number of mapped habitats, CRAM index scores, plant and avian diversity, and richness metrics were all significantly correlated to size. There are a number of reasons for this relationship. First, larger wetlands naturally have a greater area in which to express macro- and micro-topographic relief and, furthermore, to develop a range of physical patch types such as sediment mounds, point bars, flats, slump blocks, variegated shorelines, etc. This diversity of physical structure provides a range of moisture gradients and habitat niches that are occupied by a wide diversity of wetland flora and fauna (Finkenbine *et al.* 2000)**.**Thus, the larger the wetland, the greater the diversity of the plant and animal species that can be found there. Second, larger wetlands have a smaller ratio of edge to area than small wetlands, and thus provide greater interior spaces that provide better opportunities for foraging, refuge, and reproduction.

Wetland size is often constrained by adjacent land uses, especially in an urbanized landscape; thus, while it is not a factor that is easily managed, it is an element that can be taken into account when prioritizing sites for restoration. It is also important to recognize the value of small wetlands in a fragmented highly urbanized environment. Particularly in arid or semi-arid, highly urbanized areas such as in southern California, wildlife are attracted to aquatic habitats in great numbers because of natural scarcity of such resources (Everts 1997, Lahr 1997).

Project Objective and Design Criteria. This study showed that urban wetlands that are created, restored or enhanced for habitat versus water quality or multipurpose have some basic differences that may constrain the type or condition of habitat that they can provide. At a landscape level, basin habitat wetlands had twice the relative area of riparian habitat and roughly half the amount of open water as treatment wetlands. This difference is understandable from the perspective that treatment in wetlands is achieved through manipulating the hydrology, physical structure and wetland flora to optimize treatment capacity. This is done through 1) manipulating hydraulic retention time to maximize contact time and contaminant settling rates, 2) stabilizing the velocity and distribution of flows and water depths, 3) manipulating the sequence of shallow vegetated areas versus deeper open water areas; 4) manipulating the community structure of wetland plants, and 5) increasing, through design or management, the capacity of the system to act as a continuous sink for the contaminants (Kadlec and Knight 1996). Thus, in a project area with limited size available for treatment, more emphasis will likely be placed on providing treatment through open water and emergent marsh rather than providing riparian habitat, which is an essential feature of both streams and basin wetlands for refuge and buffering from the stressors of an urbanized landscape. Treatment wetlands must be designed and regularly maintained to optimize treatment capacity, so the physical configuration of these wetlands may be an element required to optimize flow conditions or provide easy access for maintenance.

The elements of hydrology, physical structure, and wetland flora that are used to optimize treatment capacity are also manipulated to restore a site to enhance the habitat provided. This study did not establish by weight of evidence across all Tier 1-2-3 indicators that habitat wetlands had statistically significant, superior condition relative to multipurpose or treatment wetlands. Most of the habitat mapping, CRAM, plant community, and avian survey metrics showed habitat sites to be on average in slightly better condition, but these trends were generally not significant and there were many exceptions. The general reason for this large variability in condition of habitat wetlands probably lies in the general unquantified constraints of the urban infrastructure (poor hydrology, lack of buffer, excessive stressors, fragmented connectivity), as well as factors such as the lack of maintenance occurring in some of the urban wetlands. Some multipurpose and water quality wetlands are being maintained better than some habitat wetlands; thus, large variability was found in this dataset.

There were, however, several variables in which significant differences were found by project objective; these differences were instructive because they pointed to how wetlands can be designed or maintained to provide better habitat. First, multipurpose wetlands had significantly higher plant species richness and diversity than habitat or treatment wetlands. This result was confusing until further analysis showed that the majority of these multipurpose wetlands are compensatory mitigation wetlands. For mitigation wetlands, establishment of plant cover with native species is a common permit requirement (Ambrose *et al.* 2006). Comparison of data from these systems with data from reference sites shows that the diversity and richness of plants in these multipurpose sites is being maintained artificially high relative to the community structure of natural wetland plant communities. Second, while native plant diversity and richness may be a component of permit conditions for mitigation sites, it is clear that the physical structure of these sites is not a permit condition. Multipurpose and treatment wetlands had significantly lower CRAM physical structure scores than habitat wetlands. These multipurpose and treatment wetlands were characterized by oval configurations, unvariagated shorelines and steep slopes. Treatment wetlands and many mitigation wetlands normally lack the complex microtopography found in natural wetlands due to the difficulty and associated cost of creating microtopography with construction equipment and the necessity of controlling flood flows with high banks and levees, and requirements for maintenance access. An increased understanding of the importance of the elements of good physical structure in balance with these practical constraints in a wetland may help improve the quality of habitat provided in restoration and mitigation projects. . It may also reduce the need for intensive “biological maintenance”

Origin. Because stormwater and sediment detention basins are among the most adaptable, effective, and widely applied BMPs in urbanizing areas (Schueler 2000), they are often employed as a strategy to meet the multiple objectives of habitat enhancement or mitigation, flood attenuation, and water quality control. Typically, constructed wetlands or stormwater ponds are sited at the lowest elevation of a development site, stream valley, or floodplain. In areas such as southern California, these ponds are often located in headwaters streams. However, these are typically the same areas in which natural wetland or riparian habitat is found and/or stream restoration is targeted. Unfortunately, the physical and biological features that result in optimal stream habitat are very different that the optimal features for treatment of urban runoff. In an effort to achieve both habitat and water quality goals, and because urban development typically generates more runoff than natural landscapes, restored riparian areas and wetlands are often wetter than pre-existing conditions. This results in a “type conversion” of many riverine systems to more palustrine or lacustrine types. The National Research Council (NRC; 2001) has cited this type conversion as one of the greatest source of urban wetland destruction in the last two decades in several regions.

Data from this study showed that 63% of the sites in the inventory have been type converted from other historic wetland types, while only 20% have been created from upland habitat. The condition of the habitat provided by historic wetlands was significantly higher than that provided by wetlands that have been type converted from streams and/or floodplain habitat or created from upland habitat. These findings indicate that, particularly in a region which has experienced high loss of wetlands habitat, watershed management planning should place an emphasis on conserving and restoring historic wetlands to habitat and promoting the creation of water quality wetlands from upland habitat. They also bring into question the appropriateness of converting stream habitat into wetland detention basins to address water quality issues from hydromodification and increased contaminant loading from development (NRC 2001).

Intensity of Maintenance Activities. This study showed that intensity of maintenance activities was significantly related to parameters such as native species richness, where lowest condition was observed in wetlands that were being maintained at frequencies at greater than 2 times per year. Since the data on maintenance frequency collected for the sites was not specific to the type of activity, it is useful to understand what types of activities typically occur through maintenance, and explore impacts on habitat condition from these activities. Maintenance activities for both habitat and treatment wetlands can include: 1) the regulation of influent flow rates and wetland water levels to keep hydraulic and contaminant loading rates or hydroperiod within targeted objectives for the site, 2) removal of sediments that have accumulated in sediment forebays, 3) rotation of discharge sites to allow the wetland to have an extended opportunity to assimilate contaminants and organic matter that create high oxygen demand, and 4) trash removal and vegetation management (Kadlec and Knight 1996). Depending on contaminants of concern and the treatment goals of the constructed wetland, maintenance of vegetation can be minimal, or may involve more intense maintenance including harvesting of plant biomass and eradication of undesirable species through application of herbicide, mowing, and replanting (Kadlec and Knight 1996).

Recommendations

Recommendations for habitat monitoring of future wetland projects should be guided by a core set of consistent indicators that are common to all projects, then a specialized suite of indicators that may answer specific management questions about the wetland. Table 14 gives a core set of indicators.

The suite of project tracking, habitat mapping and rapid assessment core indicators are selected based on extended discussions with the State Water Resources Control Board on the minimum data needed to track the effectiveness of permitted projects within the state of California. “Project” in this capacity is defined as a place in which an action (e.g., restoration, compensatory mitigation, etc.) has resulted in a change of wetland or riparian acreage or condition at a site. Core recommended contaminant monitoring is intended for urban wetlands where issues of toxicity to wildlife would need to be address.

Currently State policy on whether treatment wetlands should be considered wetland projects is unclear. Since treatment wetlands would be considered projects under this definition, until state policy is clarified, these projects should be required to conduct the minimum core monitoring given in Table 14.

Table 14. Recommended core "minimum" monitoring for habitat and examples of intensive monitoring that could be required for urban wetlands.

Type of Monitoring	Management Question	Indicator	Frequency	Spatial Extent of Sampling
Core: Project Tracking	Where are the locations and general information on grant funded projects?	General contact information	At project initiation, with updates as monitoring reports are submitted	N/A
Core: Habitat Mapping	What is the net change in the distribution of wetland and riparian acreage by habitat type resulting from this project? What is the contribution from grant funded projects to the wetland and riparian acreage statewide?	Wetland and riparian habitat mapping at a minimum of 1:500 scale map	Restoration site: Pre-project baseline and "as built" at termination of project Impact site: pre-project baseline	Entire project area
Core: Rapid Assessment	What is the net change in the condition of the targeted wetland as a result of the project?	CRAM	Pre-project baseline and "as built" and, if applicable, pre-project baseline of impact site	Entire project area of both impact and restoration sites
Core: Intensive Monitoring	Are there water column contaminants that are likely to be of concern to wildlife that use the wetland? If so, what are the loads associated with these contaminants?	Pre-construction influent characterization: Water column aquatic toxicity and comprehensive suite of water column contaminant chemistry and flow in wetland inflow during dry and wet weather	EMC and loads from 2 storm events and 24-hr composited concentration and loads from 2 dry weather events	At all major storm drains/creeks that flow into wetland that together comprise 80% or greater of the flow
	Are legacy contaminants likely to be of concern to wildlife at this site? If so, what is the inventory of these contaminants in the sediments?	Pre-construction site characterization: Sediment contaminants and sediment toxicity	Once	Minimum of three sites (near to major water source, mid-wetland, and near outflow)
	Are sediment contaminants reaching levels of concern to wetland fauna?	Sediment contaminants and toxicity	Just after the completion of construction and at project termination ("as built")	Near major water source and near outflow

STUDY OF SEDIMENT CONTAMINANT CHEMISTRY AND TOXICITY IN URBAN WETLANDS

Abstract

Urban wetlands are an important resource with respect to both the habitat they provide as well as the capacity they have to improve water quality of aquatic ecosystems downstream. To better understand the risks, considerations, and trade-offs associated with using wetlands to passively or actively treated urban runoff, sediment toxicity, and chemistry were measured in southern California wetlands that receive urban runoff. Benthic organisms in 18 of the 21 urban wetlands examined were at risk to direct toxicity from sediment contaminants. Most of the sites were either toxic to the amphipod *Hyalella azteca*, or exceeded a sediment quality guideline, or both. Sediment chemistry suggests that pyrethroids may have been responsible for much of the toxicity documented in this study. Concentrations were elevated at all 10 sites that were toxic to *H. azteca*, and the mean pyrethroid quotient was negatively correlated with amphipod survival. Other contaminants were also elevated (including heavy metals, PAHs, and DDE) and could have contributed to the toxicity, although not at as many sites as pyrethroids. An index of degree of sediment contamination (mean Probable Effects Concentration quotient; mPECq) was found to negatively correlate with benthic macroinvertebrate diversity in these wetlands. Sediment toxicity and chemistry concentrations from wetlands used to improve water quality were not significantly different than levels observed in wetlands with habitat as the primary management objective, or multipurpose wetlands (sites that have water quality and habitat as equally important objectives). Sediment toxicity and chemistry values were not related to wetland hydrologic residence time. Several constituents (Cu, Pb, Zn, PAHs, cypermethrin) were significantly correlated with percent imperviousness of the catchment area, an index of percent urbanization.

Introduction

Toxicity to wildlife from contaminants commonly found in urban runoff is among the top concerns of managers of urban wetlands. Elevated levels of contaminants and toxicity have been identified. For example, Katznelson *et al.* (1995) found toxicity in surface waters of a marsh that receives stormwater runoff from an urban area. Additional testing at the site indicated that the toxicity was due to the organophosphorus pesticide diazinon. Brown and Bay (2004) found elevated levels of diazinon in two streams that receive urban runoff. Concentrations of the pesticide were high enough at one of the sites to have been responsible for the observed toxicity. At a wetland that receives urban runoff from a high-density residential area, inflow samples were found to contain elevated levels of dissolved Cd and Ni, and total Al and Se (Brown and Bay 2006). These samples were found to be toxic during two of the four sampling events. Therefore, toxicity to organisms that use urban wetlands for habitat is a legitimate concern.

The purpose of this study was to investigate the sediment toxicity and chemistry of urban wetlands with varying objectives (habitat, multipurpose, water quality treatment). This study had four objectives:

- Quantify the toxicity and sediment chemistry of urban wetland sediments relative to reference wetlands, and how they compare with reference conditions
- Determine the contaminants are likely responsible for that toxicity, as indicated by sediment chemistry
- Determine how sediment contaminant concentrations or toxicity vary as a function of wetland objective, degree of urbanization, and hydrologic retention time
- Provide management recommendations to mitigate the risk of exposure and toxicity

Methods

General Study Approach

Sediment toxicity and chemistry were characterized for urban wetlands in southern California. The wetlands selected for this study were identified in an inventory of southern California urban wetlands by Sutula *et al.* (2005). The wetlands were located in six counties throughout southern California, including Santa Barbara, Los Angeles, Orange, San Bernardino, San Diego, and Imperial Counties. The sites represented different histories, configurations, management objectives, hydrologic retention times, and levels of urbanization in the catchment areas.

Characteristics of Sites

Urban wetlands in southern California are often highly modified systems that reflect the history and management objective at each site. Most of the urban wetlands sampled in this study have been engineered to some degree. Twelve of the twenty-one sites had been modified from existing wetland areas (e.g., widened, deepened, or features such as berms or channels have been added to slow down or re-direct the flow), while five of the sites were created from upland areas that were never historically inundated. The remaining four sites are historic wetlands. Some of the sites that had been engineered to enhance water quality are ponded areas (or a series of ponds) with relatively long retention times that allow particulates and their associated contaminants to drop out of the water column (e.g., IRWD Pond A), while other water quality wetlands have been designed as circuitous channels to maximize the treatment path length for relatively small footprint areas (e.g., WetCAT North). In addition to being less-than-natural due to historical engineering, many wetlands receive water throughout the year, instead of having a more seasonal pattern of runoff influence. Southern California's semi-arid Mediterranean climate is characterized by mild to cool wet winters, and warm to hot dry summers. The average annual rainfall is 38 cm, which accumulates mostly between December and March. The remainder of the year can be quite dry and many wetlands communities are highly dependent on diverted urban runoff during much of the year. The main sources of runoff to many of the urban wetlands in this study were chronic nuisance flows from high-density residential and commercial areas.

Three types of wetland configurations were examined: wetland basins (ponded systems that retain runoff over a period of time to promote sedimentation), wetland channels (flowing channels, creeks or small streams), and type converted wetlands (hybrid wetlands caused by sediment buildup that results in a channelized shallow basin configuration). Most of the urban wetlands had a basin configuration (12 sites), followed by an almost equal number of type converted (5 sites) and channel wetlands (4 sites).

The number of sites in each of the three management objective categories (habitat, water quality, multipurpose) was approximately equal. There were seven wetlands that had habitat as the primary objective, compared to six sites that were managed to enhance water quality, and eight sites that had habitat and water quality as equally important objectives (multipurpose).

Field Methods

Twenty-three wetland sites were sampled between October 22 and November 8, 2007 throughout southern California. This included 21 wetlands that receive urban runoff, and 2 reference sites located in natural areas that do not receive runoff from point or nonpoint contaminant sources (Sespe Creek, Mojave River Marsh). All samples were collected during dry weather. Samples were collected at a location nearest to the primary source of urban runoff in order to characterize the greatest potential effect. As such, the sediment from this location represents a worst-case scenario with respect to potential contamination and toxicity. The top 5 inches of sediment were collected using a shovel, or Ponar grab sampler from an inflatable row boat. Sediment from multiple grabs were composited in the field, and distributed to a precleaned 4-oz glass jar for chemical analysis, or to four 1-L polyethylene jars for toxicity analysis. All containers were held on ice until distributed to the analytical laboratories.

Laboratory Analysis

Chemistry. Samples for trace metals were analyzed by Inductively Coupled Plasma Mass Spectrometry (ICPMS) using EPA Method 6020m (Table 15; Samples for organic constituents (PAHs, PCBs, organochlorine pesticides, organophosphate pesticides, and synthetic pyrethroids) were analyzed by Gas Chromatography/Mass Spectrometry (GC/MS) using EPA Method 8270c. Total organic carbon concentrations were determined by EPA Method 415.1. Chemical analysis of the samples were conducted by CRG Labs, Torrance, CA.

Toxicity. All sediment samples were tested for toxicity using the 10-day *H. azteca* and *Chironomus tentans* survival tests. The tests were conducted in 1-L glass jars containing 2 cm of sediment (approximately 150 ml) and 800 mL of water. Five replicates were used for each sample and the control. Sediments were sieved through a 0.5-mm screen to remove resident organisms and debris. The sediment was then added to the jars, and overlying water (Culligan system treated) added with aeration one day before the animals were added in order to provide a 24-h equilibration period. After equilibration, 10 juvenile *H. azteca* (7 - 10 days old) and 10 *C. tentans* (2^{nd} to 3^{rd} instar) were added to each beaker to start the test. Both species were added to each exposure chamber for the testing, instead of conducting separate exposures. Previous experiments have shown this does not adversely affect the results (Nautilus unpublished data). The exposures were conducted at 20°C. Each test chamber was given 2 g TetraMin® every 48 h during the exposure. At the end of the exposure period, the sediment from the beakers was passed through a sieve to recover the animals, and the number of surviving animals counted. Water quality parameters (temperature, pH, dissolved oxygen, ammonia, and conductivity) were measured on the pore water and overlying water of surrogate water quality beakers at both the beginning and end of the exposure period. A 96-h water only copper reference toxicant test was conducted as a positive control with both species. Toxicity analysis of the samples was conducted by Nautilus Environmental, San Diego, CA.

Data Analysis

The sediment chemistry measurements were assessed by comparing the values to freshwater Probable Effects Concentrations (PECs; MacDonald *et al.* 2000; Table 15). The PEC thresholds are intended to identify contaminant concentrations above which harmful effects on sediment-dwelling organisms are expected to occur frequently. PEC quotients (PECq) were calcutated by dividing each measured concentration by its PEC value. A PECq >1.0 is considered potentially harmful. The mean PECq (mPECq) was calculated following MacDonald *et al.* (2000) as the average PECq for 10 contaminants, including As, Cd, Cr, Cu, Pb, Ni, Zn, total PAHs, PCBs, and sum of DDEs. Samples with an mPECq >0.5 were considered toxic.

Concentrations of Se were assessed by comparing measured values with the observed effects threshold proposed by Van Derveer and Canton (1997; Table 15). There is no PEC value for Se. The authors of the proposed threshold state that the value is preliminary because of the limited data used to derive the threshold (fish and bird data only, no invertebrate studies). However, the threshold has been used by the National Irrigation Water Quality Program (NIWQP 1998) to assess sediment quality. Because invertebrate data were not used to derive the Se threshold, the value was not used to assess the toxicity data in this report, but was be used to help interpret sediment chemistry data.

Pyrethroid pesticides were assessed by comparing measured concentrations with the mean LC50 values (concentrations that cause a 50% reduction in survival; Table 15). There are no PEC thresholds for this class of compounds. The LC50 values were derived from sediment exposure experiments with *H. azteca* (Amweg *et al.* 2005). The LC50 values were used in this study to help interpret the toxicity data, but not to assess the wetland sites. Because these LC50 values were derived from a single study, they do not carry the same weight to assess the sites as the PEC values, which were derived from data collected from many studies, encompassing several sediment types, and included several biological endpoints. A mean pyrethroid quotient was calculated using the same approach used for the mPECq. The concentrations for the individual pyrethroid compounds were divided by their respective LC50, and the average quotient was then calculated for each site.

The toxicity data were assessed with ANOVA and Dunnett's multiple comparison test. The survival data were arcsin transformed prior to analysis. ANOVA on ranks was used when data were non-normally distributed, and an ANOVA with Bonferroni correction was used when there were unequal replicates. Samples were considered toxic if they were significantly different and <80% of the control value.

The relationship between toxicity and habitat parameters was also examined. Spearman correlation was used to compare survival data with CRAM index scores, BMI values, percent imperviousness of the surrounding catchment area, and hydrologic retention time. The habitat parameters were obtained from a previous phase of the study examining the general habitat characteristics of these wetlands.. Increased imperviousness of the catchment area has been shown to contribute to greater contaminant loads in runoff. Greater retention time in wetlands is believed to promote more settling of particles and associated contaminants from the water column (reference). For this study, retention times were categorized as either <1 day, 1 - 7 days, or >7 days.

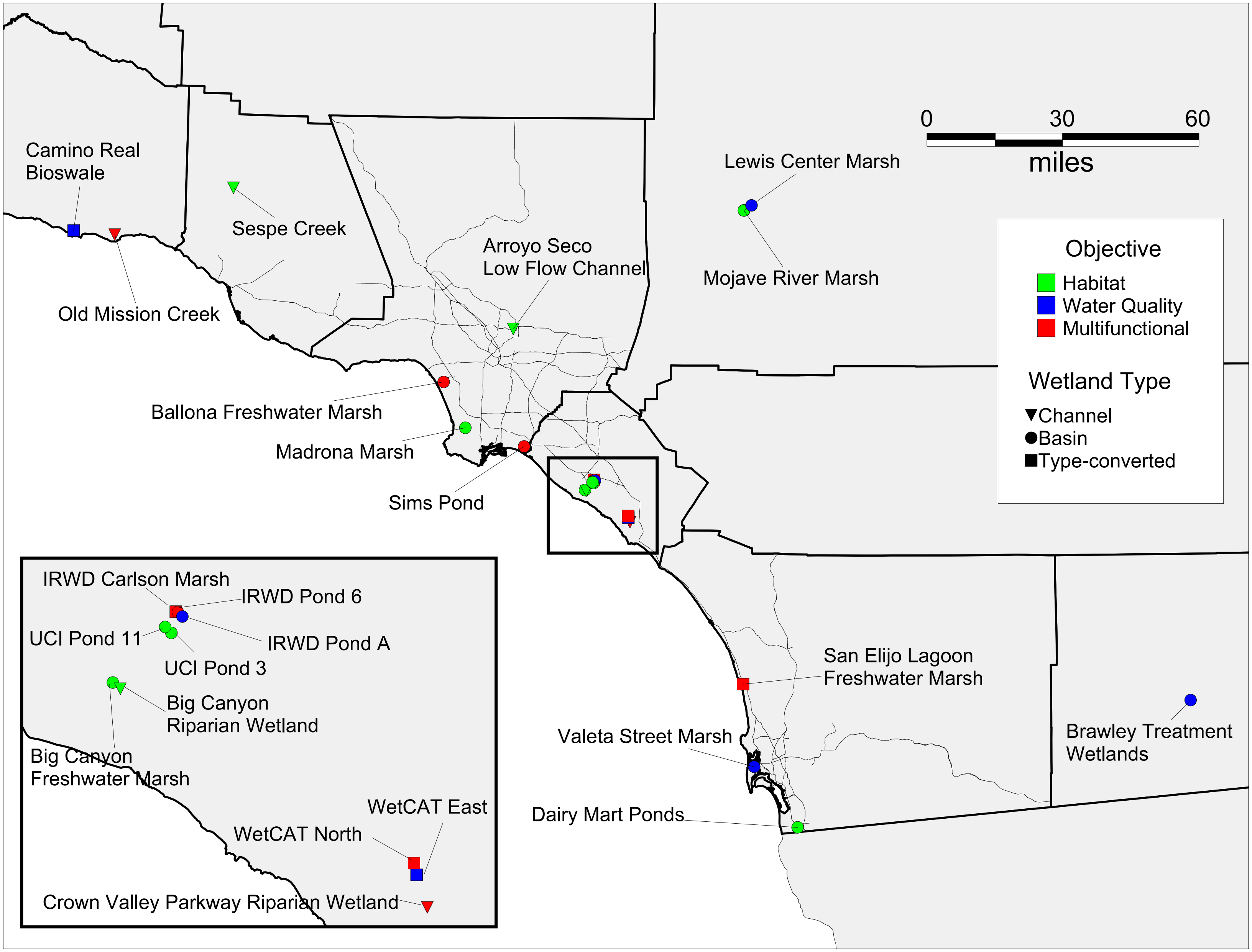


Figure 26. Locations of the wetland sites sampled. The freeways in Ventura, Los Angeles, Orange, and San Diego Counties have been added for reference.

Table 15. Reporting levels and analysis methods for chemical constituents measured.

Analyte	Reporting Level	Method
General		
Total Organic Carbon (mg/kg)		EPA 415.1
Grain size		
Metals (µg/g)		
As	0.05	EPA 6020m
Cd	0.05	EPA 6020m
Cr	0.05	EPA 6020m
Cu	0.05	EPA 6020m
Fe	5	EPA 6020m
Pb	0.05	EPA 6020m
Hg	0.05	EPA 6020m
Ni	0.05	EPA 6020m
Se	0.05	EPA 6020m
Zn	0.05	EPA 6020m
Organic Contaminants (ng/g)		
Organochlorine Pesticides[1]	5, 50	EPA 8270Cm
PCBs[2]	5, 20	EPA 8270Cm
Organophosphate Pesticides[3]	2 - 16	EPA 8270Cm
Synthetic Pyrethroids Pesticides[4]	25	EPA 8270Cm
PAHs[5]	5	EPA 8270Cm

[1] Organochlorine pesticides include: 4,4'-DDD, 2,4'-DDD, 2,4'-DDE, 2,4'-DDT, 4,4'-DDE, 4,4'-DDT, aldrin, BHC-alpha, BHC-beta, BHC-delta, BHC-gamma (Lindane), ahlordane-alpha, ahlordane-gamma, cis-nonachlor, dieldrin, endosulfan sulfate, endosulfan-I, endosulfan-II, endrin, endrin aldehyde, endrin ketone, heptachlor, heptachlor epoxide, methoxychlor, mirex, oxychlordane, toxaphene, and trans-nonachlor.

[2] PCBs include: Aroclor 1016, 1221, 1232, 1242, 1248, 1254, 1260, PCB congener 18, 28, 31, 33, 37, 44, 49, 52, 66, 70, 74, 77, 81, 87, 95, 97, 99, 101, 105, 110, 114, 118, 119, 123, 126, 128+167, 138, 141, 149, 153, 156, 157, 158, 168, 168+132, 169, 170, 177, 180, 183, 187, 189, 194, 200, 201, 206.

[3] OP pesticides include: bolstar (sulprofos), chlorpyrifos, coumaphos, demeton, diazinon, dichlorvos, dimethoate, disulfoton, ethoprop (ethoprofos), fenchlorophos (ronnel), fensulfothion, fenthion, guthion, malathion, merphos, mevinphos (phosdrin), parathion-methyl, phorate, tetrachlorovinphos (stirophos), tokuthion, and trichloronate.

[4] Pyrethroid pesticides include: allethrin, permethrin, bifenthrin, cyfluthrin, cypermethrin, deltamethrin, fenpropathrin, lamda cyhalothrin, prallethrin, and pyrethrins

[5] PAHs include: 1-methylnaphthalene, 1-methylphenanthrene, 2,3,5-trimethylnaphthalene, 2,6-dimethylnaphthalene, 2-methylnaphthalene, acenaphthene, acenaphthylene, anthracene, benz[a]anthracene, benzo[a]pyrene, benzo[b]fluoranthene, benzo[e]pyrene, benzo[g,h,i]perylene, benzo[k]fluoranthene, biphenyl, chrysene, dibenz[a,h]anthracene, dibenzothiophene, fluoranthene, fluorene, indeno[1,2,3-c,d]pyrene, naphthalene, perylene, and phenanthrene, pyrene.

Results

Sediment Chemistry

Most of the urban wetlands had at least one constituent that exceeded a PEC (Table 16; Figure 27). Of the 21 urban wetlands, PEC thresholds were exceeded at 13 sites (62%). For most sites, there were only one or two constituents that exceeded the PEC thresholds, rather than a mixture of contaminants. This was the case for 11 of the 13 sites with PEC exceedances. The most number of PEC exceedences to occur at a site was five. The two reference wetland sites did not have any constituents that exceeded PEC thresholds.

Constituents exceeding the PECs varied among the sites. The most prevalent contaminant was Cd, exceeding the PEC at six sites, followed by Ni and DDE (each elevated at four sites), chlordane (three sites), and Zn (two sites). Fluoranthene, pyrene, Pb, and Cu exceeded their respective PEC at one site each. The incidence of contamination from heavy metals was approximately equal to organics, with PEC exceedances at seven and eight sites, respectively.

Pyrethroid pesticide contamination was prevalent at the urban wetlands (Table 18; Figure 30). Measured values exceeded LC50 values at 13 urban wetlands (62%) by as much as a factor of 137. The highest values were reported at Sims Pond, which receives golf course runoff, and WetCAT North, which receives residential area runoff. Pyrethroid concentrations were below the method detection limit at the two reference wetlands. Organophosphorus pesticide concentrations were below the method detection limit at all sites.

Six locations had sediment concentrations that exceeded the preliminary observed effects threshold for Se (Table 16). All sites that exceeded this threshold were in the Newport Beach/Irvine area. Exceedances ranged from a factor 1.2 at the University of California Irvine Pond 3 to a factor of 14 at Big Canyon Marsh and Big Canyon Riparian wetlands.

Contaminant concentrations were related to the degree of urbanization (percent imperviousness of the catchment area) for some constituents (Table 17). Sediment concentrations of Cu, Pb, Zn, total PAHs and cypermethrin were all significantly correlated with percent impervious area ($r = 0.43$, $r = 0.53$, $r = 0.51$, $r = 0.43$, $r = 0.59$, respectively).

Contaminant concentrations did not appear to be related to the wetland objective or hydrologic residence time (Table 18). Wetlands with habitat as the main objective did not have significantly lower concentrations of contaminants than wetlands used to improve water quality, or those wetlands with habitat and water quality as equally important objectives. In addition, contaminant concentrations were no greater in wetlands where more particulate settlement was expected to occur (predominantly basin wetlands) than in wetlands with shorter residence time (predominantly channel wetlands). The one exception appeared to be for Pb. Concentrations of sediment Pb were significantly different among the three hydrologic retention time categories ($p = 0.04$), with the highest concentrations associated with the longest retention time (>7 days). However, further analysis with multiple comparison testing was unable to verify which categories were significantly different.

One measure of benthic organism community was related to sediment contamination (Figure 31). There was a significant decrease in benthic macroinvertebrate diversity with increasing mPECq values ($r = -0.62$, $p < 0.01$); benthic macroinvertebrate density, on the other hand, did not decrease ($r = 0.14$, $p = 0.55$). The mean pyrethroid quotient was not correlated with benthic community measures ($r = -0.02$ density, $r = -0.09$ richness).

Table 16. Freshwater sediment quality guidelines used to evaluate the sediment chemistry concentrations. NG = no guideline. An observed effects threshold of 4 mg/kg was used for Se (USDOI 1998).

Analyte	Probable Effects Concentration (PEC)	Average LC50
Heavy and Trace Metals		
As	33.0	
Cd	4.98	
Cr	111	
Cu	149	
Pb	128	
Hg	1.06	
Ni	48.6	
Se	4	
Zn	459	
Organochlorine Pesticides		
Chlordane	17.6	
Sum DDD	28.0	
Sum DDE	31.3	
Sum DDT	62.9	
DDT, total	572	
Dieldrin	61.8	
Endrin	207	
Heptachlor epoxide	16.0	
Lindane	4.99	
PCBs, total	676	
PAHs		
Anthracene	845	
Benzo[a]pyrene	1,450	
Benz[a]anthracene	1,050	
Chrysene	1,290	
Fluorene	536	
Fluoranthene	2,230	
Naphthalene	561	
Phenanthrene	1,170	
Pyrene	1,520	
PAHs, total	22,800	
Organophosphate Pesticides	NG	
Pyrethroid Pesticides		
Bifenthrin	NG	4.5
Cyfluthrin	NG	13.7
Deltamethrin	NG	9.9
Esfenvalerate	NG	24
Lambda-cyhalothrin	NG	5.6
Permethrin	NG	90

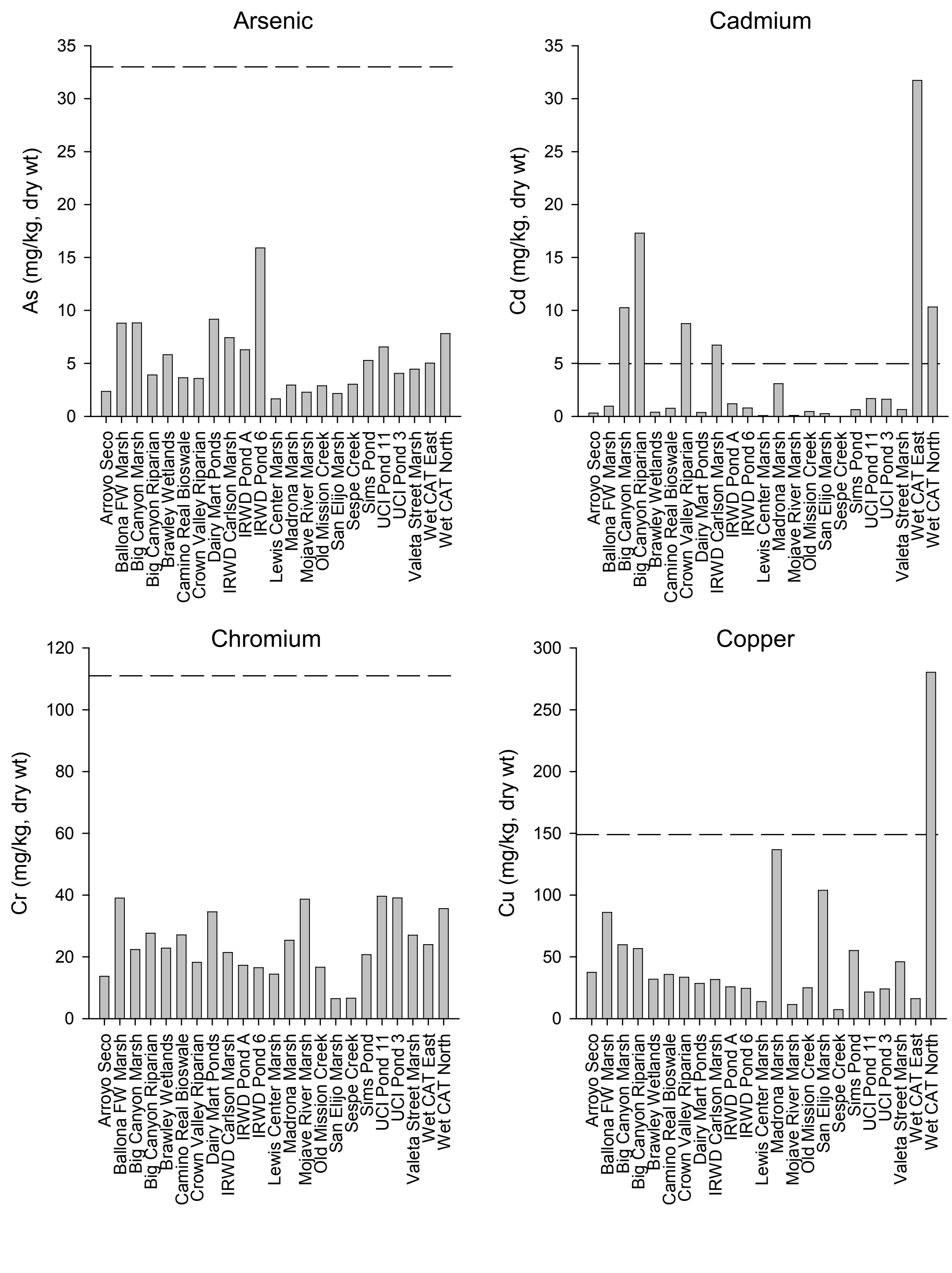


Figure 27. Sediment metals concentrations at each site. The dashed line indicates the Probable Effects Concentration, except for Se, where the line is the observed effects threshold from Van Derveer and Canton (1997).

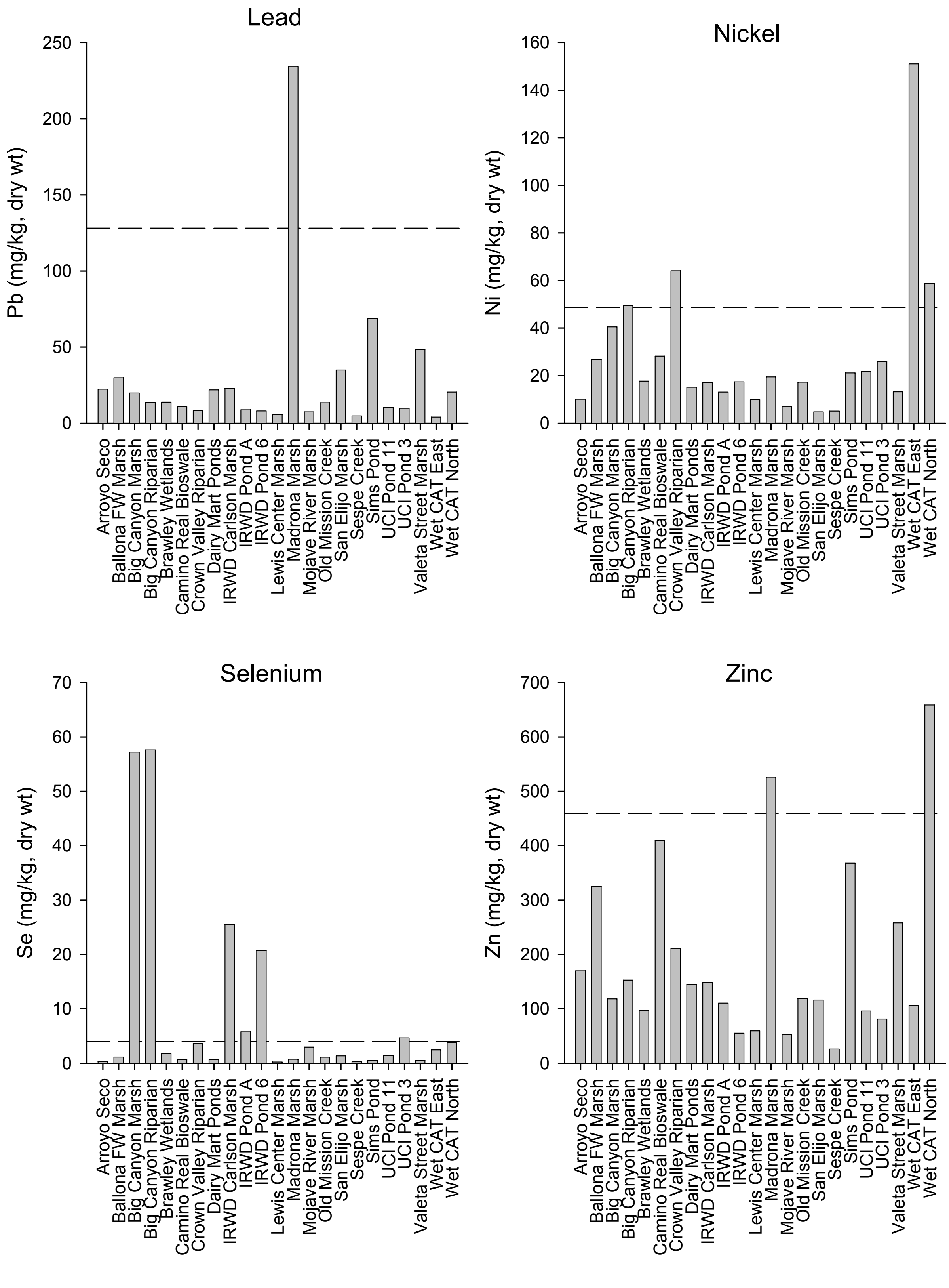


Figure 27. continued.

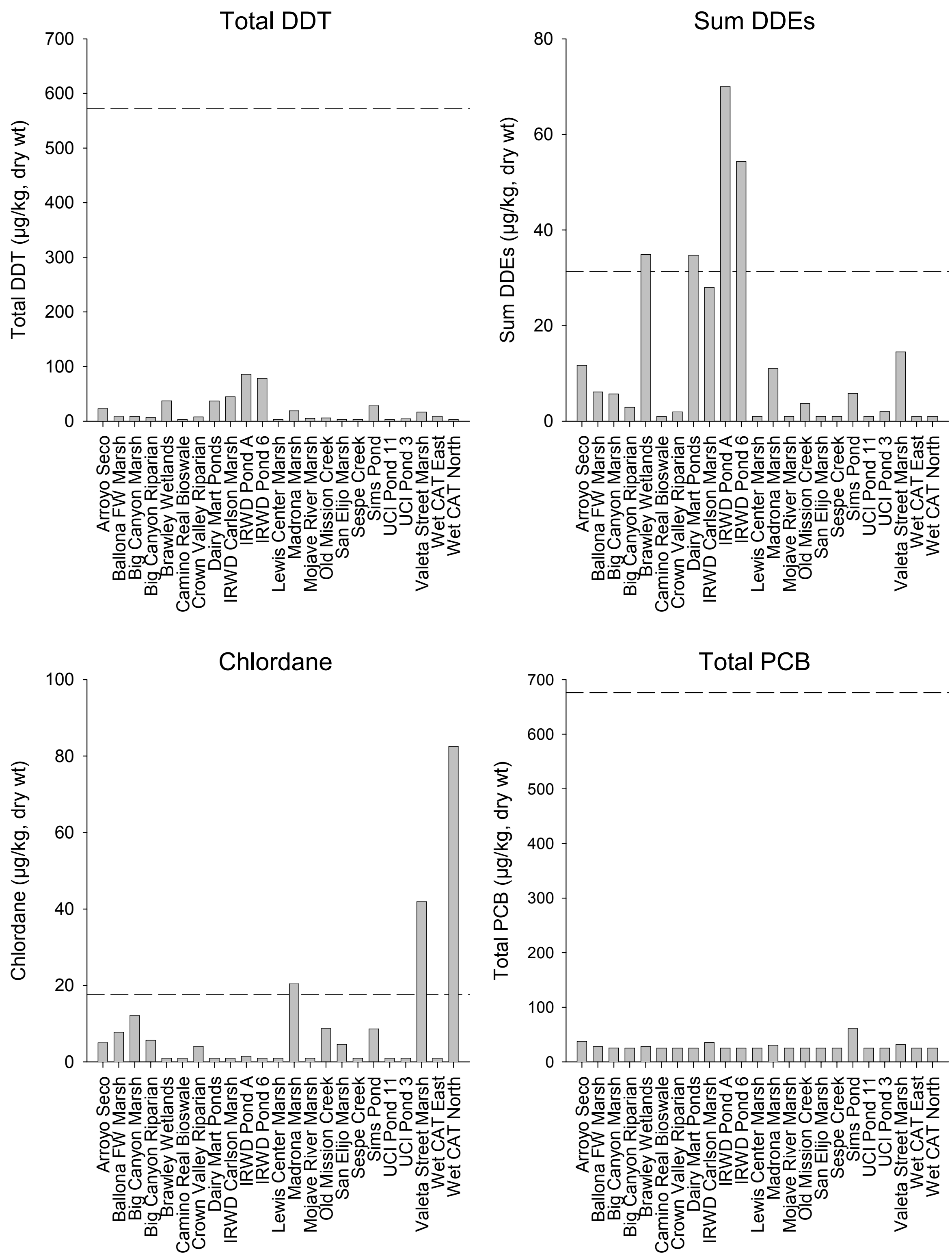


Figure 28. Sediment concentrations of chlorinated pesticides and PCBs at each site. The dashed line indicates the Probable Effects Concentration. Nondetects were replaced with half the method detection limit.

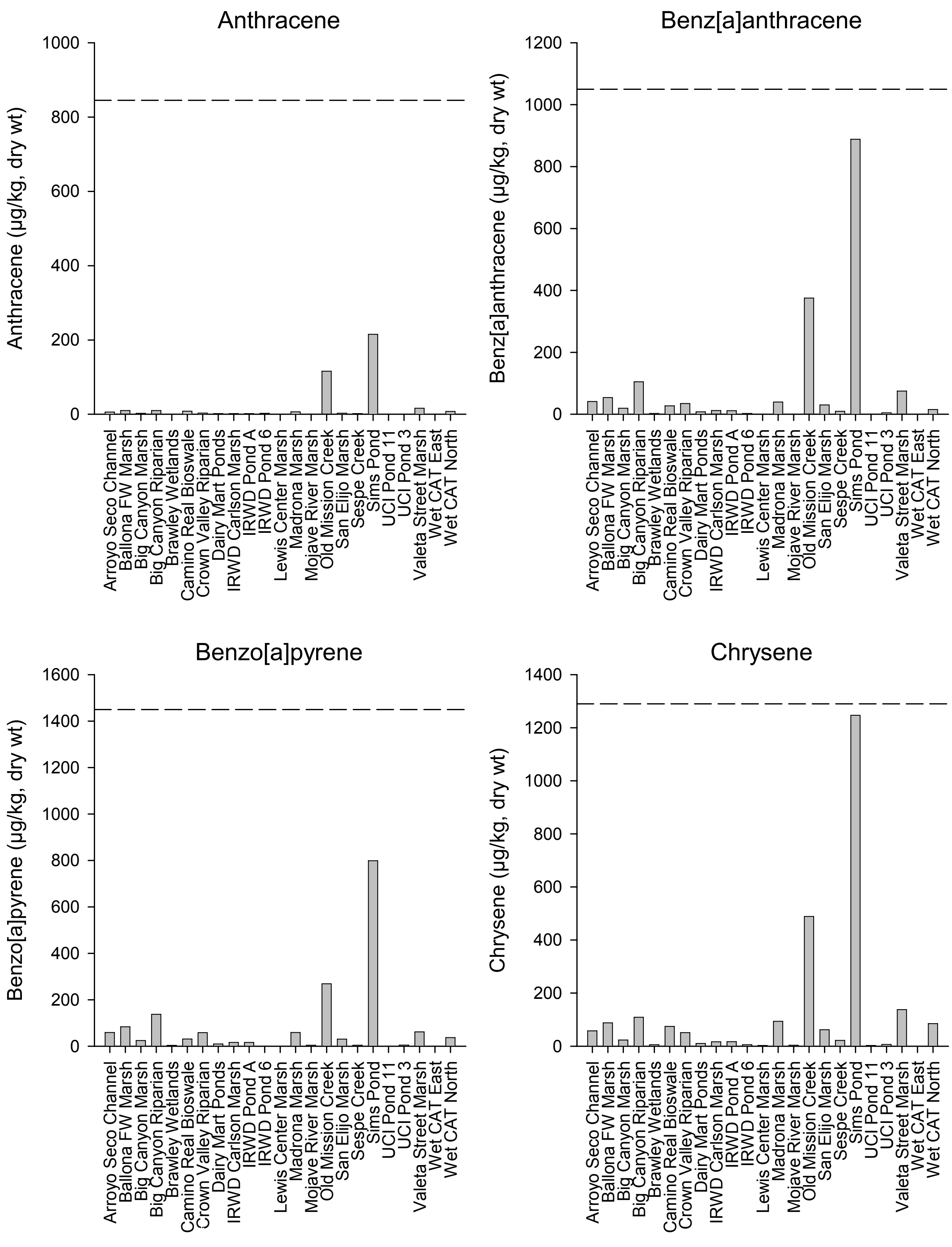


Figure 29. Sediment concentrations of PAHs at each site. The dashed line indicates the Probable Effects Concentration (pec).

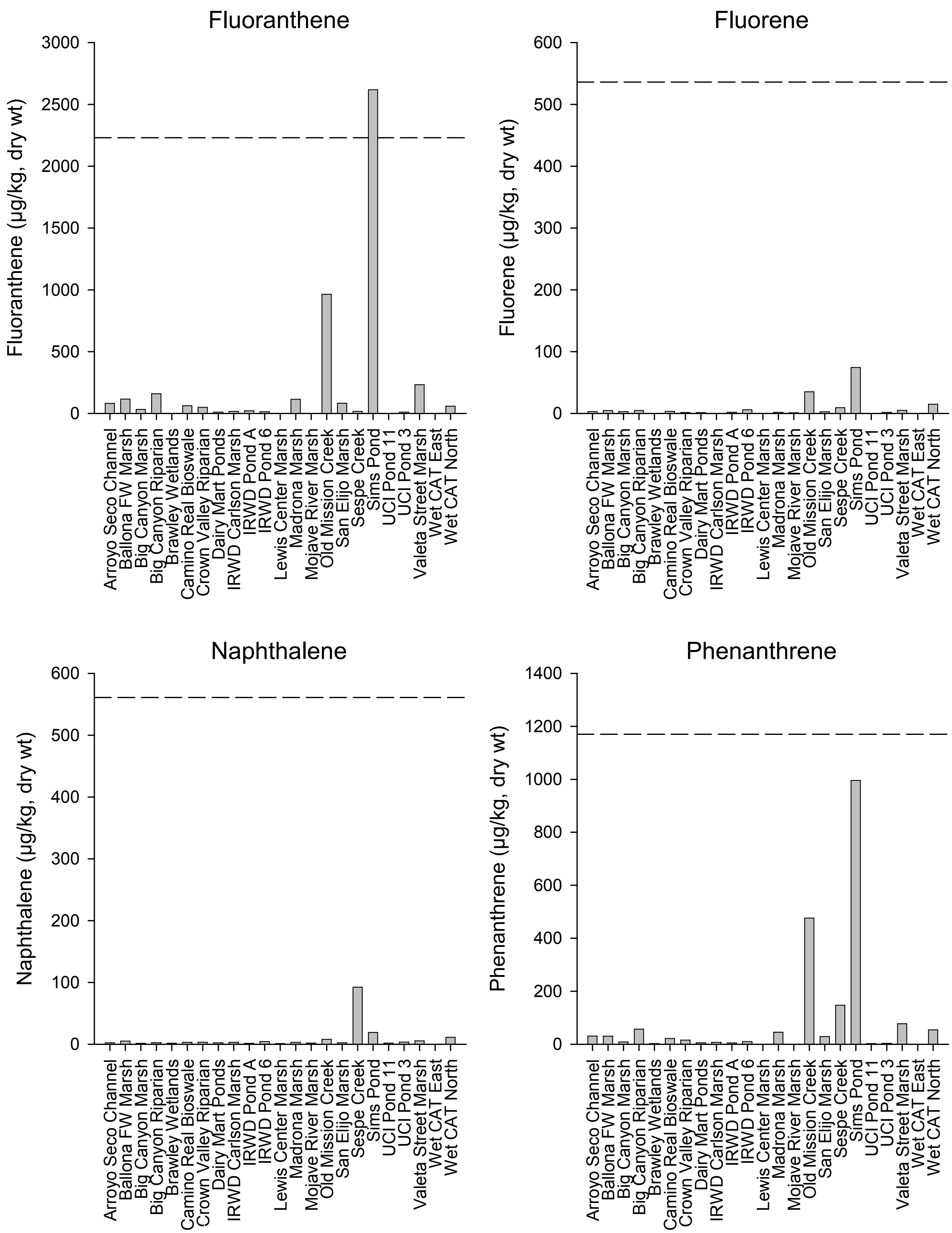
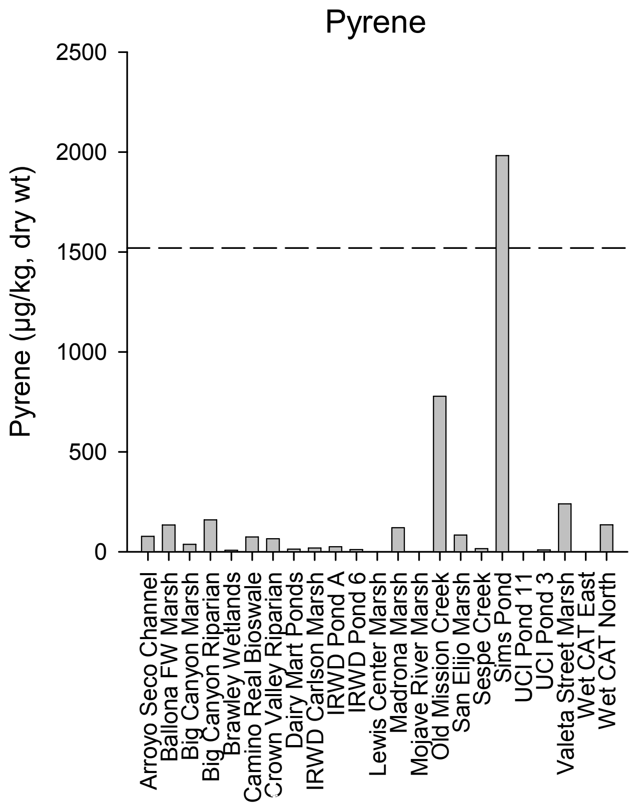


Figure 29. continued.



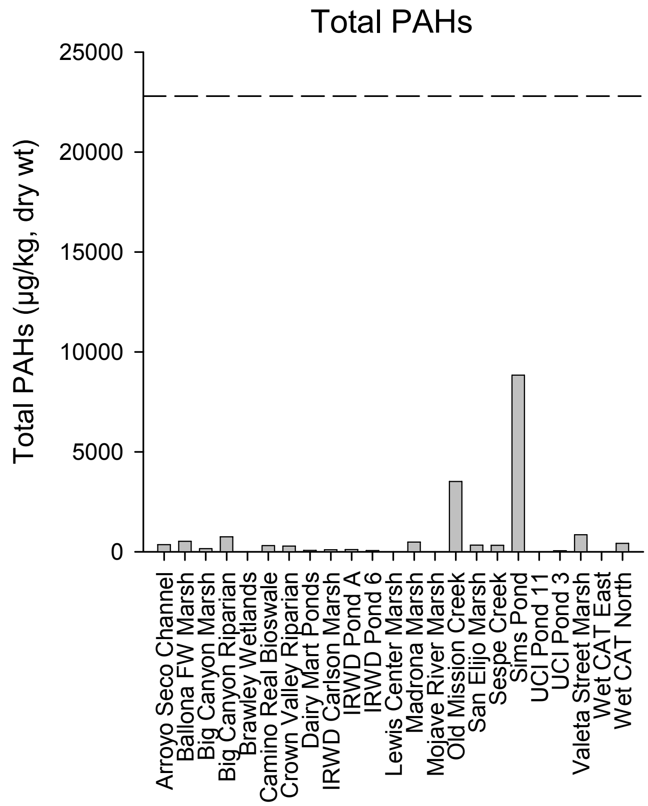


Figure 29 continued.

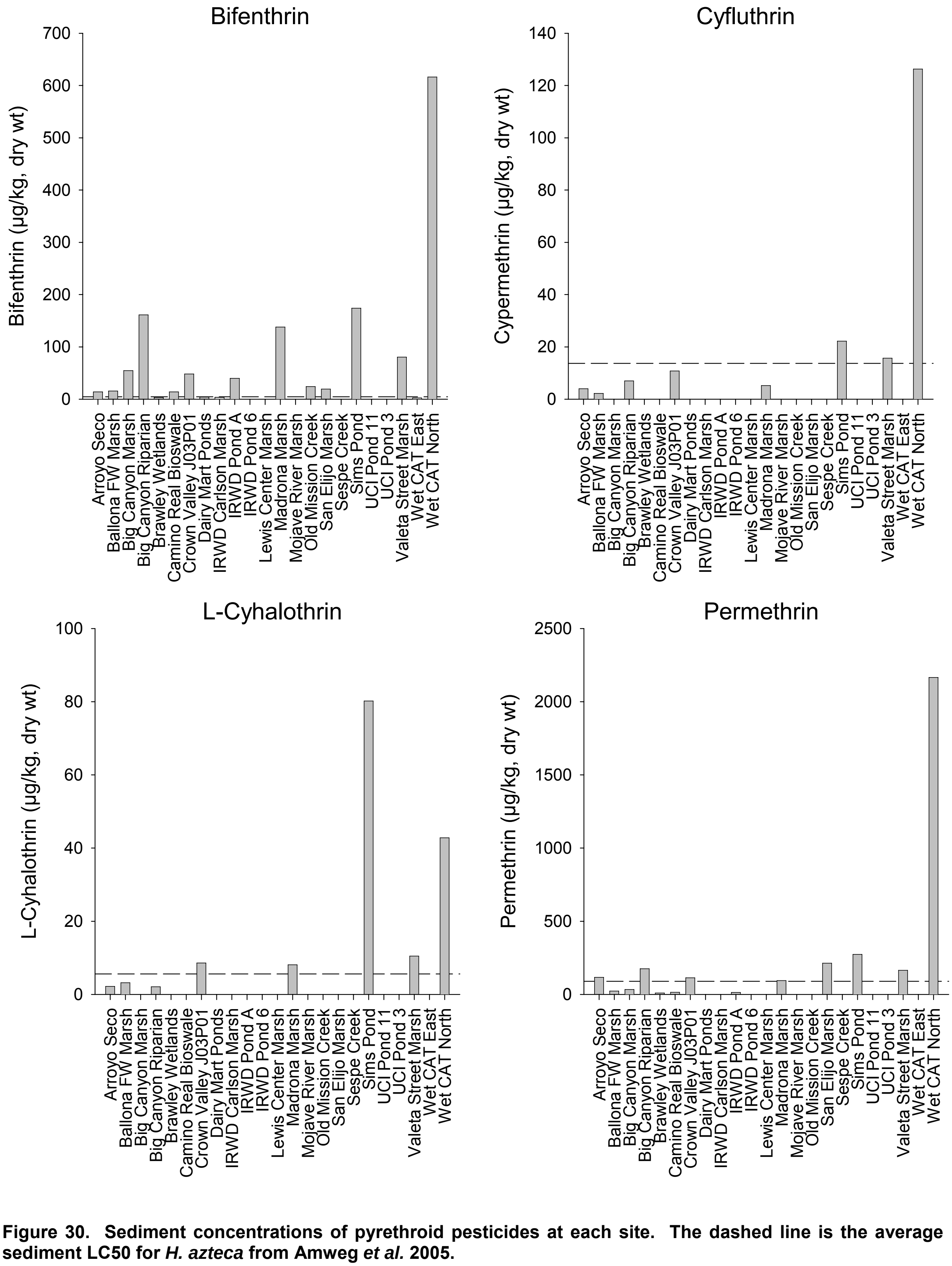


Figure 30. Sediment concentrations of pyrethroid pesticides at each site. The dashed line is the average sediment LC50 for *H. azteca* from Amweg *et al.* 2005.

Table 17. Summary of constituents exceeding sediment quality guideline thresholds and an indication of toxicity at each of the study sites.

Wetland site	Constituents exceeding PEC thresholds	Pyrethroids Exceeding Mean Sediment LC50 for *H. azteca*	Exceed Se Observed Effect Threshold	*H. azteca* Acute Toxicity	*C. tentans* Acute Toxicity
Arroyo Seco Channel	None	Bifenthrin, permethrin	No	Yes	No
Ballona Fresh Water Marsh	None	Bifenthrin	No	No	No
Big Canyon Marsh	Cd	Bifenthrin	Yes	Yes	No
Big Canyon Riparian	Cd, Ni	Bifenthrin, permethrin	Yes	Yes	Yes
Brawley Wetlands	Sum DDE	None	No	No	No
Camino Real Bioswale	None	Bifenthrin	No	Yes	No
Crown Valley Parkway Riparian	Cd, Ni	Bifenthrin, L-cyhalothrin, permethrin	No	Yes	No
Dairy Mart Ponds	Sum DDE	None	No	No	No
IRWD Carlson Marsh	Cd	None	Yes	No	No
IRWD Pond A	Sum DDE	Bifenthrin	Yes	Yes	No
IRWD Pond 6	Sum DDE	None	Yes	No	No
Lewis Center Marsh	None	None	No	No	No
Madrona Marsh	Pb, Zn, chlordane	Bifenthrin, L-cyhalothrin, permethrin	No	No	No
Mojave River Marsh	None	None	No	No	No
Old Mission Creek	None	Bifenthrin	No	No	No
San Elijo Marsh	None	Bifenthrin, permethrin	No	Yes	No
Sespe Creek	None	None	No	No	No
Sims Pond	Fluoranthene, pyrene	Bifenthrin, cyfluthrin, L-cyhalothrin, permethrin	No	Yes	No
UCI Pond 11	None	None	No	No	No
UCI Pond 3	None	None	Yes	No	No
Valeta Street Marsh	Chlordane	Bifenthrin, L-cyhalothrin, cyfluthrin, permethrin	No	Yes	No
Wet CAT East	Cd, Ni	None	No	No	No
Wet CAT North	Cd, Cu, Ni, Zn, chlordane	Bifenthrin, cyfluthrin, L-cyhalothrin, permethrin	No	Yes	No

Table 18 Relationship between sediment constituents (contaminant concentrations or amphipod survival) and wetland habitat parameters.

Constituent	% Imperviousness of Catchment area	Hydrologic Retention Time Category (<1 day, 1-7 days, >7 days)	Objective (habitat, water quality, multifunction)
Cu	0.05	0.69	0.15
Pb	0.01	0.04	0.29
Ni	0.77	0.12	0.99
Zn	0.02	0.64	0.67
PAHs	0.04	0.86	0.09
DDE	0.72	0.59	0.98
Chlordane	0.12	0.63	0.41
mPECq	0.50	0.40	0.64
Mean pyrethroid quotient	0.22	0.94	0.54
Survival	0.52	0.38	0.42

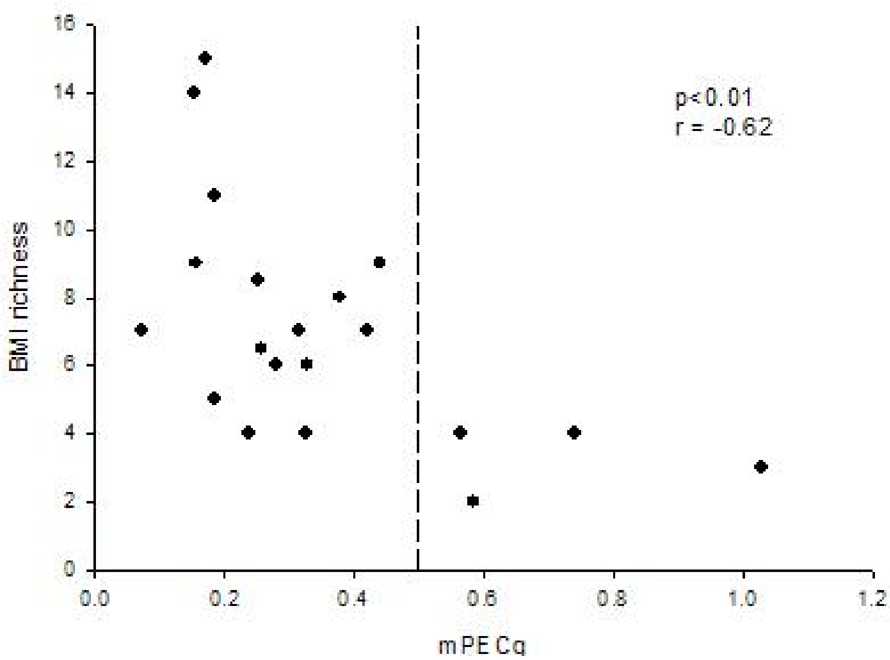


Figure 31. Relationship between mean Probable Effects Concentration quotient (mPECq) and benthic macroinvertebrate (BMI) richness . The vertical dashed line is the threshold above which toxicity is expected

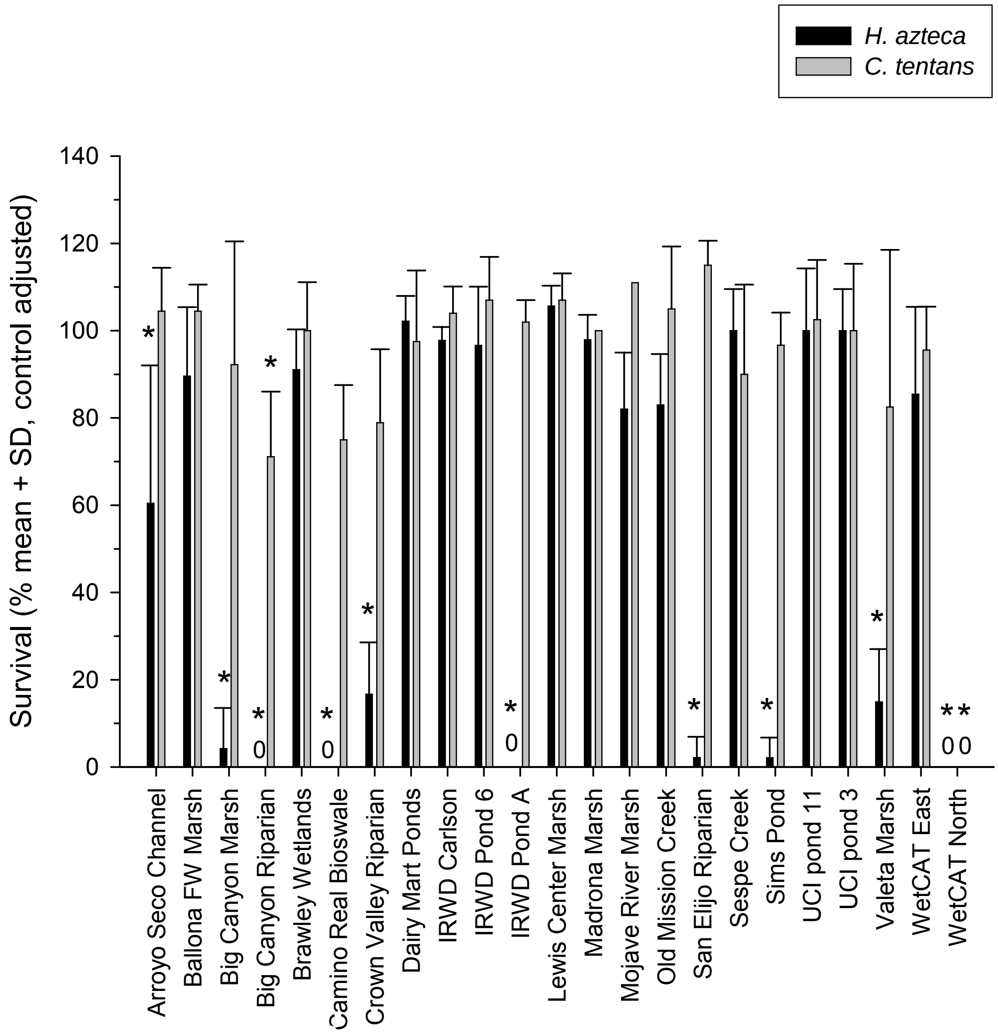


Figure 32. Survival of *H. azteca* and *C. tentans* exposed to wetland sediments. * = significantly different and <80% of control value.

Toxicity

About half of the treatment wetland sites (10 out of 21) were toxic to *H. azteca* survival (Figure 32). Only one site was toxic to *C. tentans* survival (Big Canyon Riparian). Because *H. azteca* survival was the much more sensitive test, results for amphipod survival were used throughout the remainder of the document to identify wetland toxicity.

Toxicity to *H. azteca* was not related to wetland objective. Amphipod survival in sediments from wetlands with water quality as the primary objective was not significantly different than the survival of amphipods exposed to sediments from habitat wetlands, or the multifunctional wetlands ($p = 0.42$). In addition, amphipod toxicity was not related to the amount of urbanization (measured as % imperviousness of the catchment area, $p = 0.52$), hydrologic residence time ($p=0.38$), benthic macrofauna diversity ($p = 0.95$) or CRAM index scores ($p = 0.73$).

Relationship between Chemistry and Toxicity

Pyrethroids were elevated at most sites and could potentially have been responsible for the toxicity at all 10 sites that were toxic to amphipods. There was agreement between pyrethroid exceedance and the toxicity threshold (both exceeded, or neither exceeded) at 18 out of the 21 urban wetlands (86%). Moreover, the mean pyrethroid quotient for the toxic sites was significantly greater than for the nontoxic sites (p <0.01), and amphipod survival was negatively correlated with the mean pyrethroid quotient (p <0.01, r = -0.68).

Constituents with PEC thresholds may have been responsible for the toxicity at 7 of the 10 sites that had amphipod toxicity. Heavy metals exceeded PEC thresholds at four of the toxic sites, with Cd exceedances at all four sites, followed by Ni (three sites), Cu and Zn (one site each). However, almost an equal number of sites had elevated metal concentrations but no associated toxicity (three wetlands). Sediment DDE concentrations exceeded the PEC value at four sites, but only one of those sites was toxic to amphipods. Likewise, the chlordane PEC was exceeded at three sites, with associated toxicity at only one of these sites.

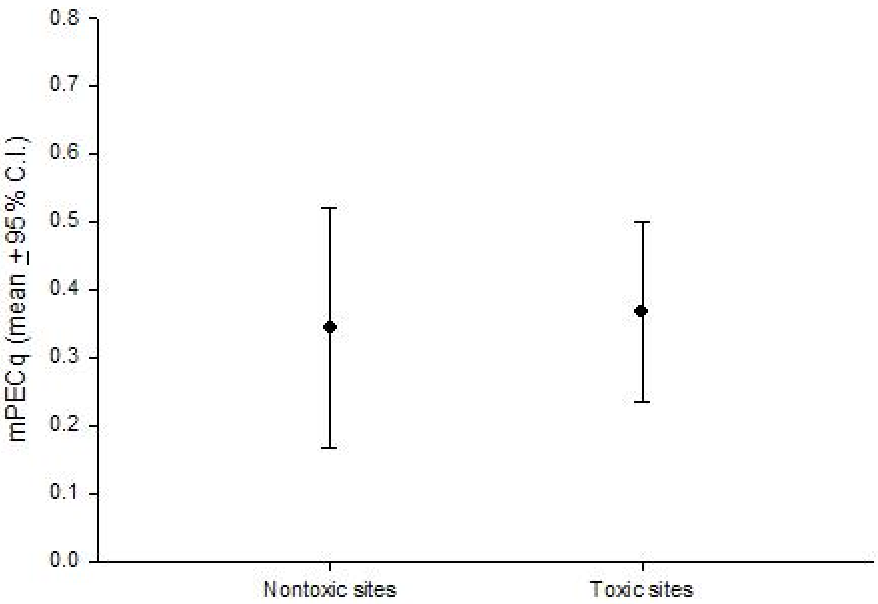


Figure 33. Mean Probable Effects Concentration quotients (mPECq) for sediments from nontoxic and toxic sites. The differences in mPECq between the toxic and nontoxic sites were not significant (p = 0.47, ln transformed mPECq).

Discussion

Risk to Benthic Organisms from Sediment Contaminants

The habitat quality of a wetland is directly related to its water source (Mitsch and Gosselink 1993). Many urban wetlands are directly dependent on wet and dry weather runoff to sustain their wetland hydrology, particularly in urban environments where water is a scarce resource. Conversely, this urban water source provides a source of contaminants that can be harmful to the wildlife that depend upon the wetland for existence. The results of this study indicate that the benthic organisms in 18 of the 21 urban wetlands examined were at risk to direct toxicity from sediment contaminants. Most of these sites were either toxic to *H. azteca*, or exceeded a sediment quality guideline, or both. Sediment contaminant concentration was also found to significantly correlate with decreased benthic macroinvertebrate diversity in these wetlands.

Several previously published studies support the fact that urban wetlands are at risk to contaminant toxicity, with variable results with respect to the magnitude of risk documented. Maltby *et al.* (1995a) found that runoff from highways into a small stream resulted in high concentrations of water and sediment metals and hydrocarbons, and was found to be correlated with changes in benthic community structure. Within these sites, sediment toxicity studies showed a slight reduction of survival of amphipods, with fractionation and toxicity confirmation studies indicating that most of the toxicity was due to hydrocarbons (Boxall and Maltby 1997, Maltby *et al.* 1995b). Yousef *et al.* (1990) and Galli (1988) both found that macroinvertebrate communities in sediments of stormwater ponds were low in diversity with an assemblage characteristic of high pollution stress (chironomid and tubificid worms and dipteran midge larvae). Kaurouna-Reiner and Sparling (1997), in a study of trace metal bioaccumulation in stormwater ponds in Maryland, detected copper, lead, zinc, and occasionally cadmium in tissues of snail, damselflies and other macroinvertebrates, but found limited acute toxicity to amphipods exposed to sediments from these ponds. Most of these studies cited noted the severity of bioaccumulation and toxicity could be expected to be highly variable among wetlands and a function of many site-specific factors, including the mix of contaminants introduced, contaminant loading over time, the hydraulic residence time, sediment bulk characteristics, and age of the wetland (Schueler and Holland 2000).

Sediment contaminant concentrations and toxicity could also be expected to vary both spatially and temporally within a wetland. In this study, sediment samples were taken in a location proximal to the primary source of urban runoff. As contaminant concentration and toxicity could be expected to vary, at minimum, as a function of distance from the source of contaminants, the results of the study are likely to represent, spatially, a conservative estimate of risk for the wetland. In addition, the samples were collected during dry weather, representing chronic nuisance flow conditions. Stormwater is usually a much more complex matrix than dry weather flow, containing higher concentrations and a greater variety of contaminants. It is unclear if sediment quality of wetlands reflects as large of a seasonal change as the overlying water quality. Therefore, additional information about how sediment contaminant concentrations vary seasonally and spatially within wetland is needed. This information would be important to better understand the magnitude of the risk posed to wildlife and to identify management options to mitigate that risk.

Contaminants Responsible for Toxicity

Without fractionation and confirmation studies, the contaminant(s) responsible for the toxicity observed cannot be conclusively identified. However, sediment chemistry suggests that pyrethroids may have been responsible for much of the toxicity documented in this study. Concentrations were elevated at all 10 sites that were toxic to *H. azteca*, and the mean pyrethroid quotient was negatively correlated with amphipod survival. Other contaminants were elevated, and could also have caused toxicity. However, these contaminants were not present at elevated levels to the degree in which pyrethroids were. Pyrethroids were also a better predictor of toxicity, because they had a lower incidence of being elevated in nontoxic sediments than the contaminants with PEC thresholds. This close association may be due to the fact that the pyrethroid thresholds were derived from toxicity tests with *H. azteca* (the species used for toxicity testing in this study), while the PEC thresholds were derived from a variety of biological endpoints (benthic community parameters and toxicity tests with other species). Therefore, a lack of an acute response from *H. azteca* in sediments with elevated PEC contaminants does not necessarily indicate the sediments were not toxic at a chronic level, or that the sediments would not have been acutely toxic to resident benthic organism. While pyrethroids appear to be implicated in the toxicity, a sediment Toxicity Identification Evaluation (TIE) would be useful to help identify the class of compound(s) causing the toxicity.

As a class of contaminants, synthetic pyrethroid pesticides now occupy a large percentage of the residential and commercial pesticide use in the United States (Weston *et al.* 2004). Pyrethroids adsorb to soil particles, so sediments would be expected to be the main repository of these compounds (Gan *et al.* 2005). Pyrethroids are known to be highly toxic to aquatic organisms (Maund *et al.* 2002), and toxicity has been observed in sediments and surface waters in a number of locations in California (Amweg *et al.* 2005, Weston *et al.* 2004). Bifenthrin, found in high concentrations in the sediment in this study, is a restricted use pesticide. Common applications of this pyrethroid pesticide include structural pest control, fire ant control, and vector control (Weston *et al.* 2004).

Implications of Toxicity to Higher Level Wetland Organisms

Adverse effects on higher level wetland organisms (e.g., amphibian, birds, fish, etc.) can occur via bioaccumulation, direct toxicity or via impact from alterations in the food web (Campbell 1995, Cooper 1991, Wren *et al.* 1997, Dunier and Siwicki 1993 *et al.*). In this study, an index of degree of sediment contaminants (mPECq) was found to significantly correlate with benthic macroinvertebrate diversity in these wetlands. Macroinvertebrates are a critical link in the food web of wetlands, providing the link between primary producers, detrital trophic organisms, and higher level consumers such as birds, fish, and amphibians (Ludwa and Richter 2000). The effects of urban runoff on resident macroinvertebrate species diversity and community structure have been well documented. Overall, these results can be summarized as: 1) decreasing overall taxa richness, 2) eliminating or reducing taxa belonging to scraper and shredder functional feeding groups relative to collector functional feeding groups, 3) reducing the taxa richness and relative abundance of orders Ephemeroptera, Plecoptera, Odonata, and Trichoptera (sensitive orders often used as basis for stream biometrics), and 4) reducing or eliminating certain Chironomids taxa, with an increase in the abundance of tolerant Chironomids, oligochaetes, and gastropods (Galli 1988, Ludwa and Richter 2000, Yousef *et al.* 1990). In this study, the magnitude of this alteration is difficult to document because of the limited data on reference sites. Few remaining reference sites for the "basin" class of wetlands exist in southern California (Stein *et al.* 2007). Of these remaining wetlands, most were dry during the sampling period, which occurred during the driest rainfall year on record for southern California. For the two reference sites sampled, sediments were not toxic to test organisms, and did not exceed sediment quality guidelines for any of the contaminants analyzed. Using the reference sites for comparison, urban wetlands tended to have greater concentrations of metals. For example, all 21 of the urban wetlands had greater concentrations of Cu and Zn (ranging by a factor of 1.2 – 24.4 times the reference value for Cu, and 1.1 – 12.6 times the value for Zn), 20 sites had greater Cd and Ni concentrations (3 – 356 times the Cd value, and 1.4 – 21.4 times for Ni), and 19 sites had greater concentrations of Pb (1.1 – 31.3 times greater than the reference site values). Thus, though limited, the sediment chemistry and toxicity data from the reference sites show a clear distinction from the majority of urban sites. Benthic macroinvertebrate taxa diversity was not significantly different between the reference sites and urban sites, suggesting that additional stressors may also be important in controlling invertebrate diversity at these sites.

Factors Associated With Sediment Contamination and Toxicity

Most of the contaminants that were correlated with increasing urbanization (% imperviousness of the catchment area) are associated with automobile use. These include heavy metals, which are associated with brake, tire and engine wear, and PAHs, which are associated with engine oil leakage, and gasoline combustion. While there was a correlation with urbanization, these constituents were rarely present at toxic levels; there were only a few instances where these contaminants exceeded their respective PEC. These less-than-toxic concentrations may have accounted for the lack of correlation between amphipod toxicity and urbanization. However, a more likely explanation is that other contaminants responsible for the toxicity (including pyrethroids) were not correlated with imperviousness. The poor relationship between toxicity and urbanization could also have been due to the fact that the highest percent imperviousness of the catchment area was only 59%. A stronger relationship may have been identified with greater urbanization.

One key management question is concerned with whether treatment wetlands, whose primary management objective is water quality enhancement, present a higher risk to wetland organisms from contaminants than habitat or multipurpose wetlands (wetland sites in which habitat and water quality enhancement functions are equally important). Study results show no significant difference in sediment contaminant concentrations or toxicity among urban wetlands by objective. This result is similar to what was found for several other indicators of habitat quality including benthic macroinvertebrate and bird diversity. The lack of significant differences by objective may be due to the profound equalizing effect that a broad range of stressors present in the urban can have on wetlands. These stressors, including eutrophication, excessive sedimentation, invasive species, habitat fragmentation and disturbance, etc., can confound effects of contaminant toxicity, making trends difficult to identify (Azous and Horner 2001).

Another factor examined in this study that could potentially influence sediment contamination and toxicity is the hydrologic retention time. Retention time can be influenced by the path length of the flow, by the speed of flow through the wetland, or by a combination of both factors. Longer residence times potentially allow a greater amount of particle-associated contaminants to settle out of the water column. In this study, there was no significant difference in sediment contaminant concentrations for most constituents, or toxicity among the different hydrologic retention time categories. The one exception was Pb, which was elevated in the sediments from wetlands with the longest retention times. The lack of relationships among the other constituents could be due to a number of factors. Longer retention times will not necessarily equate with higher sediment contaminant concentrations, if the contaminants are not elevated in the inflow to the wetlands. The sediment contaminant concentrations in this study suggest this was probably not the circumstance for these wetlands. However, if the contaminant load is spread out over a greater area in sites that have longer flow path lengths, then concentrations may not be any different from sites with shorter path lengths. Because the sediments were collected in areas close to the inlet in this study, the concentrations measured may not represent the total contaminant settlement occurring throughout the wetlands. Additional information about how sediment contaminant concentrations vary spatially within the wetlands would be important to better understand the magnitude of the risk posed to wildlife.

Recommendations

Several options exist for ways to avoid the potential impacts from contaminants in urban runoff during the design, construction and maintenance of urban wetlands. These recommendations range from watershed scale planning of conservation and restoration of natural wetlands and riparian areas to onsite design and maintenance features that can prevent the accumulation of sediment contaminants in a wetland.

At the watershed scale, contaminant runoff can be minimized by maximizing the water storage and infiltration opportunities within the developed land uses and outside of existing wetlands (Horner *et al.* 2000). Options exist for improving the water quality of urban runoff before it enters natural wetlands and aquatic habitats. Options should include both source control BMPs and treatment BMPs (including constructed wetlands). In general, source control BMPs (e.g., those which prevent the generation or release of pollutants) are more effective and less expensive than treatment BMPs (Horner *et al.* 2000). Specific recommendations include controlling source of pollutants where possible (e.g., street sweeping, removal of sediment and debris from storm drain inlets, improving infiltration, and other land-based BMPs, etc.). Depending on land use, pretreatment should be provided including oil and grit interceptors in highly industrial sites, highways, etc. (Shutes *et al.* 1997), placement of a primary treatment system such as a sand filter or use sediment collection/settling forebays for treatment of stormwater inflows and for additional treatment of wastewater. Forebays should be designed and located for ease of maintenance, removal of sediment in order to achieve greatest protection of wetland habitat and receiving waters (Schueler 2000). Habitat, multipurpose, or treatment wetlands can be segmented such that the primary treatment is provided in the forebay or initial pond and the remainder of the wetland is used for polishing and/or wildlife enhancement. A variety of treatment strategies, including detention, pre-treatment, treatment, and infiltration, should be incorporated in series in order to maximize removal efficiencies and minimize exposure to wildlife.

Efforts to reduce urban runoff can be greatly aided by watershed scale conservation and restoration of natural wetlands and riparian areas. Given evidence that the presence of riparian buffers greatly reduce contaminant concentrations in runoff before it is discharged to streams and wetlands (Schueler and Holland 2000), an effective strategy for NPS control in a watershed should be coupled with an aggressive program to restore riparian buffers around streams and wetlands and maintain interconnections between wetlands and other natural habitats (Schueler 2000, Horner *et al.* 2000).

Conservation of open space and forest cover, and maintain natural storage reservoirs, drainage corridors including depressions, areas of permeable soils, swales, and intermittent streams should be promoted. Policies to discourage the clearing, filling and channelization of these features (Horner *et al.* 2000) should be developed and implemented. The historic extent of natural wetlands within watershed should be researched and treatment systems should be located where there is no opportunity to restore historic or natural wetlands. Within riparian corridors, this will often involve designing systems that are hydrologically-isolated from surface water flow rather than within the historic floodplain (Schueler 2000). Collectively, these measures can provide a means to reduce the onsite contamination of wetland sediments and can reduce the potential risk to wildlife.

EFFECTIVENESS OF SOUTHERN CALIFORNIA WETLANDS FOR TREATMENT OF URBAN RUNOFF: AN ANALYSIS OF EXISTING MONITORING DATA

Abstract

Examination of the treatment wetland literature shows a general lack of information on the effectiveness of using these wetlands as urban runoff BMPs, particularly with respect to effectiveness of these systems in arid climates and for the treatment of dry weather urban runoff. The importance of looking at treatment effectiveness by during dry weather is increasing as a larger number of NPDES permits are written to cover both winter dry and wet weather as well as summer dry weather loads. To begin to address these data gaps, existing monitoring data for southern California were examined to better understand the contaminant removal efficiency for treatment wetlands, with three principal questions: 1) Do southern California treatment wetlands reduce the concentration of contaminants in dry and wet weather urban runoff? 2) Is there a difference in effectiveness during wintertime wet weather (storm events) versus dry weather (winter or summertime base flow) conditions? and 3) Is there a quantitative difference between the removal efficiencies of temperate versus arid wetlands? Ability to answer these questions was limited because of nearly universal lack of flow data that would allow calculation of percent removal on a basis of loads. Analysis was limited to comparison of concentrations in the influent (entering the wetland) versus effluent (existing the wetland).

Existing monitoring data show that southern California treatment wetlands reduce the concentrations of all constituent of interest [i.e., total and dissolved metals (Cu, Pb, Zn, Se), nutrients (nitrate, ortho phosphate), TSS, and bacteria (*Enterococcus*, *Escherichia coli*, fecal and total coliforms)] relative to inflow concentrations. For dissolved Cu, Pb, and Zn, southern California treatment wetlands were effective at reducing wet season inflow concentrations to below the California Toxic Rule (CTR) water quality criteria. Great variability was found in the effectiveness of removal. Factors influencing this variability include time of sampling as well as other physical, chemical and biological attributes of the wetland. Clearly, modeling of treatment wetland BMPs is needed to provide a time-integrated picture of water and contaminant budgets that can lead to better calculations of treatment efficiencies. Standardized monitoring of treatment wetland projects can provide the data needed to develop these models.

Study data also illustrate that percent reductions and inflow concentrations can vary greatly by wintertime wet and dry weather and dry season. Percent removal of contaminants typically associated with suspended solids such as total and dissolved Zn, Cu and Pb, metals, phosphate and public indicator bacteria contaminants had a higher percent reduction in concentration during wet weather than during dry weather. A few significant differences were found in the inflow and outflow total Cu and Zn during dry weather periods of both the dry and wet season, but these reductions were minor in comparison with those occurring during wet weather. No significant differences were found between inflow and outflow concentrations of total and dissolved Pb during dry weather periods. Differences in concentrations of total metals among outflows by weather and season were generally not significant.

This study indicated that southern California treatment wetlands showed 1 - 2 orders of magnitude reductions in *E. coli*, *Enterococcus,* fecal coliform, and total coliform, regardless of weather or season. Inflow concentrations of indicator bacteria were on average the highest during the dry season, a result consistent with increased biological activity during periods with higher temperature.

Significant reductions in concentrations of nitrate and phosphate were observed from the inflow to the outflow regardless of weather or season, a finding consistent with the widely held understanding that treatment wetlands are a cost-effective means of reducing nutrient concentrations in urban runoff, particularly for nitrate. Dry season concentrations of nitrate were generally the highest, though only

significantly different from wet weather concentrations. No significant differences were observed by weather or season among inflow or outflow concentrations of phosphate.

Data on percent reduction as a function of influent concentration indicate that treatment wetlands may be at times a source of contaminants relative to inflow concentrations. The data indicate that this phenomenon occurred most often during dry weather events, more so during the dry season and during low concentrations, indicating that it is not likely to be significant source during these periods.

Some differences in contaminant concentrations in treatment wetland inflows were found between semi-arid and temperate climates. Total Cu, nitrate and phosphate were higher in southern California treatment wetlands relative to temperate sites in the International Stormwater database. The opposite was found for TSS and *E. coli*, and for all other constituents, no significant difference were observed in the inflow concentrations between treatment wetlands of the two climates. Removal of dissolved Pb appears to be more efficient in semi-arid systems, while that of dissolved Zn appears to be efficient in temperate treatment wetlands. In order to confirm these trends, more would need to be understood about the size, soils, geomorphology and hydrodynamics of the systems from which the data are derived in order to tease out true differences in climate. More careful consistency of sampling and analytical methods would also be required.

Recommendations for future monitoring of treatment wetland BMPs are given. One key recommendation is to require that monitoring be conducted and reported based on load rather than concentration alone. This would necessitate that flows be measured for the period in which the sample was taken, either through direct measurement of flow or via water level recorders with a rating curve to establish flow. Collecting performance data on treatment wetland BMPs in a SWAMP-compatible format will greatly enhance our collective understanding of BMP effectiveness in meeting load reductions.

Introduction

One of the prized functions of wetlands is their ability to enhance water quality (Mitsch and Gosselink 1993). As our watersheds have become increasing urbanized, there is growing interest in treatment wetlands as a BMP to address issues of surface water pollution resulting from point and nonpoint source pollution (Kadlec and Knight 1996). Most of the published research on treatment wetlands has been conducted on wetlands designed to treat municipal or industrial wastewater; a recent review by Strecker (1995) on the subject noted that only a limited number of studies have been published on effectiveness of treatment wetlands as BMP) to treat urban runoff. Furthermore, most of the treatment wetlands for which published monitoring data are available are located in temperate climates. The extent to which these data are applicable to semi-arid and arid regions, such as southern California, is uncertain (Schiff and Sutula 2002, Sutula and Stein 2003), because runoff from semi-arid and arid regions generally have higher concentrations of contaminants than that from mesic or humid areas (Table 18; Caraco 2000). Consequently, stormwater management practices in semi-arid regions such as southern California could require greater either greater pretreatment or more conservative sizing requirements than those in more humid climates (Caraco 2000). With respect to potential land and maintenance costs associated with such practices, better understanding of these differences is needed. Another important data gap is effectiveness of treatment of dry weather urban runoff. Whereas dry season runoff may be somewhat chronic (e.g., steady flow), stormwater inputs are stochastic in nature and subject to short-term, unpredictable pulses of water, sediment, and contaminants that may be several orders of magnitude higher than baseline inputs.

In order to fill these data gaps, analysis of existing treatment effectiveness monitoring data from southern California treatment wetlands is an important next step. Existing monitoring data are difficult to compile and utilize because of time and expense to compile the data, as well as issues of questionable quality control and compatible data collection methodologies. Despite this, existing monitoring data is an underutilized resource. The purpose of this component of the study was to use available existing monitoring data from southern California treatment wetlands to answer three key questions:

- Do treatment wetlands reduce the concentration of contaminants in dry and wet weather urban runoff?
- Is a difference in effectiveness during wet weather (storm events) versus dry weather (baseline flow) conditions?
- Is there a quantitative difference between the removal efficiencies of temperate versus arid wetlands?

Table 19. Characteristics of water quality analyses for southern California wetlands that had available data.

Site	Type of Wetland	Dates Sampled	Sampling Frequency	Type of Sample	Constituents					Data Source
					TSS	Nutrients	Bacteria	Metals	Organics	
Ballona Freshwater Marsh	Basin	4/2003 – 7/2006	Almost Monthly	Grab	X	X	X	X	PAHs, PCBs, Pesticides, VOCs	Read *et al.* 2006.
Big Canyon, Newport	Channel	8/2004 & 10/2004	Twice	Grab	X	X	X	Total	PCBs, Pesticides	WRC 2004
Brawley; New River	Basin	1/2001 – 1/2005	Weekly to Monthly	Grab	X	X	One Event	Dissolved Se only	VOC Two Events	Tetra Tech 2006
Imperial Wetland; Agricultural Drainage	Basin	1/2001 – 4/2005	Weekly to Monthly	Grab	X	X		Dissolved Se only	VOC Two Events	Tetra Tech 2006
La Costa	Basin	Not reported	One Wet Mean EMC, One Dry Mean EMC	Flow-weighted composites for wet; grab for dry	X	X	X	X	VOCs	Caltrans 2004
"	"	2/2000 – 4/2001	11 storms	EMC	X	X	X	X		International Stormwater Database 2003
Prado Wetlands	Basin	7/1998 – 12/2004	Once to Monthly	Not Reported	X	X	X	X	PAHs, PCBs, Pesticides, VOCs	OCWD
San Joaquin Marsh	Basin	1/2002 – 9/2006	Weekly	24-h Flow Composite ; Grab for Bacteria	X	X	X	Total	Pesticides	IRWD
WetCAT East	Channel	2/2001 – 12/2003	Weekly to Monthly	Not Reported	X	X	X	Mn only		CH2M Hill 2004
WetCAT North	Channel	4/2003 – 12/2003	Weekly to Monthly	Not Reported	X	X	X	Mn only		CH2M Hill 2004
WetCAT West	Channel	2/2001 – 12/2003	Weekly to Monthly	Not Reported	X	X	X	Mn only		CH2M Hill 2004
"	"	11/2004 – 3/2005	Monthly	Time-weighted Composite	X			X	Pesticides	Brown and Bay 2003

Methods

An inventory of southern California urban and treatment wetlands conducted in the first phase of this study was used to identify treatment wetlands for which water quality data were available. Nine such treatment wetlands were identified, with one additional site identified from a study by California Department of Transportation (Caltrans; 2004). Most of these wetlands received urban runoff, while one received freeway runoff, and one received agricultural runoff exclusively. Several treatment wetland systems that treat municipal effluent were excluded from this study.

Several types of information related to sampling and analysis were compiled for each site. This included sampling frequency and type of samples collected (e.g., grab, composite), type of constituents analyzed, and constituent concentrations in the wetland influent and effluent (Table 19). The information regarding sampling and analysis for each site was obtained from reports (i.e., Caltrans 2004, CH2M Hill 2004, WRC 2004, GeoSyntec 2006, Read *et al.* 2006, Tetra Tech 2006), or electronic spreadsheets acquired from project managers.

Treatment effectiveness was assessed by comparing effluent concentrations to water quality criteria. The CTR freshwater chronic criteria (criterion continuous concentrations) were used for these comparisons. Effluent concentrations were assessed by comparing the upper 95% confidence limits of measurements combined from each wetland to the appropriate water quality criterion. This was done graphically, by plotting natural log transformed data using Sigma Plot software (version 8.02, SPSS Inc.). While most of the water quality criteria for dissolved metals are expressed as a function of water hardness, most of the wetland datasets did not include measures of hardness. Therefore, the criteria used for comparison in this report were based on an assumed total hardness value of 100 mg/L for all dissolved metals (this corresponds to the chronic criteria values listed in the USEPA 2000; Table 20). Other constituents were evaluated that do not have water quality criteria, including total metals (Cu, Zn, and Pb), nutrients (nitrate and ortho phosphate), bacteria (*Enterococcus*, *E. coli*, fecal coliform, total coliform), and TSS. These constituents were evaluated only for statistical significance between inflow and outflow concentrations. While organic constituents were analyzed at some wetland sites, concentrations were below method detection limits, and were not used in this evaluation.

Table 20. California Toxics Rule (CTR) freshwater chronic water quality criteria used for comparison with the effluent data (USEPA 2000).

Constituent	Water Quality Criterion (µg/L)
Dissolved Cu	9
Dissolved Pb	2
Dissolved Zn	120
Total Se	5

Treatment effectiveness was evaluated two ways: 1) comparison of effluent quality to water quality criteria and 2) calculation of statistical significance between inflow and outflow. Comparison of the effluent quality to water quality criteria is superior to the other method for two reasons. First, percent reduction is often heavily influenced by the inflow concentration. The greater removal rates are associated with high inflow concentrations, which are not necessarily related to the quality of the final effluent. A high removal rate does not guarantee effluent concentrations are meeting water quality standards. Conversely, sites that have relatively low inflow concentrations will have poor removal rates, even if the effluent is meeting water quality standards. Second, water quality criteria are applied for the protection of aquatic life, and are based on laboratory exposures of organisms to contaminants. Aquatic life protection is an important goal for most treatment wetlands.

The data were further evaluated to identify those constituents that were low as a result of treatment by the wetlands, from those that were below water quality criteria prior to treatment, or had inflow concentrations that only marginally exceeded water quality criteria. This was conducted by evaluating the inflow and outflow data for statistical differences through paired t-tests, using log-transformed data with SAS statistical software (version 9.1, SAS Institute Inc.).

While the majority of the data available for this study represented dry weather flow, stormwater samples were collected at most of the sites. In order to assess effectiveness of treatment during dry versus wet weather periods, samples that were collected within five days of a storm event with greater than 0.1 inch precipitation was chosen as representative of wet weather. Rainfall records were obtained from the National Weather Service for the John Wayne Airport, the Carlsbad Airport, the city of Santa Ana, and the town of Thermal near the Salton Sea. Seasonal effect on concentration and reduction was also examined. For this study, dry season samples were considered to be those collected from April through October, while wet season samples were collected from November through March. Therefore, data for three weather/season categories were examined, including: dry weather/dry season, dry weather/wet season, and wet weather (regardless of season).

To investigate the issue of whether differences exist in treatment wetland performance between temperate and arid environments, data from the southern California wetlands were compared with those from other parts of the country, which were obtained from the International Stormwater Database (http://www.bmpdatabase.org). The International Stormwater Database contains inflow and outflow data for metals and TSS that has been collected over the past decade from treatment systems throughout the United States and Canada (Strecker *et al.* 2004). Wetland data from the more temperate climates were obtained from Alabama, Florida, Georgia, Illinois, Maryland, Michigan, Minnesota, North Carolina, Oregon, Texas, Virginia, Washington, and Wisconsin in the United States, and Ontario in Canada.

T-tests were used to compare the differences in treatment effectiveness between wet and dry weather events and arid versus temperate climates. T-tests were applied to log-transformed outflow data, using the Cochran option when variances were unequal. All data from each wetland were pooled for these analyses.

Results

Effectiveness of Southern California Treatment Wetlands

Averaging across wet and dry weather and season, treatment wetlands in southern California significantly reduced the concentrations of each constituent of interest relative to influent concentration [i.e., total and dissolved metals (Cu, Pb, Zn, Se), nutrients (nitrate, ortho phosphate), TSS, and bacteria (*Enterococcus*, *E. coli*, fecal and total coliforms)] ($p < 0.01$ for each constituent, Table 21). Relative to water quality criteria, 95% upper confidence limit of concentrations of dissolved Cu, Pb and Zn in the influent were below water quality criteria. Therefore, the goal of meeting the water quality criteria was met, though not as a result of the wetland treatment processes. For total Se, while concentrations were significantly reduced following treatment, outflow concentrations remained above the water quality criterion.

Table 21. Comparison between natural log transformed inflow and outflow values from southern California wetlands.

	Pairs of inflow/outflow samples	p-value	Inflow Mean and 95% CI	Outflow Mean and 95% CI
Copper (dissolved)	51	<0.01	1.84 ± 0.26	1.30 ± 0.23
Copper (total)	99	<0.01	2.38 ± 0.26	1.69 ± 0.14
Lead (dissolved)	42	<0.01	0.35 ± 0.40	-0.52 ± 0.27
Lead (total)	80	<0.01	1.05 ± 0.52	0.08 ± 0.22
Selenium (dissolved)	285	<0.01	-4.05 ± 0.25	-4.12 ± 0.26
Selenium (total)	73	<0.01	2.36 ± 0.23	2.17 ± 0.21
Zinc (dissolved)	51	<0.01	3.19 ± 0.25	2.65 ± 0.27
Zinc (total)	99	<0.01	3.45 ± 0.27	2.72 ± 0.15
TSS	547	<0.01	3.18 ± 0.07	2.98 ± 0.10
Nitrate	302	<0.01	1.29 ± 0.17	0.84 ± 0.19
Ortho Phosphate	124	<0.01	-0.71 ± 0.15	-1.24 ± 0.20
Enterococcus	191	<0.01	8.43 ± 0.26	4.85 ± 0.27
E. coli	178	<0.01	7.69 ± 0.28	3.44 ± 0.24
Fecal Coliforms	245	<0.01	7.39 ± 0.26	3.82 ± 0.20
Total Coliform	237	<0.01	9.90 ± 0.25	7.29 ± 0.22

Affect of Wet/Dry Weather and Season

Overall, the greatest differences between inflow and outflow concentrations of total Cu, Pb, and Zn in southern California treatment wetlands occurred during storm events (wet weather; Figure 34). A few significant differences were found in the inflow and outflow total Cu and Zn during dry weather periods of both the dry and wet season, but these reductions were minor in comparison with those occurring during wet weather. No significant differences were found between inflow and outflow concentrations of total Pb during dry weather periods. Concentrations of total metals among outflows by weather and season were generally not significant.

The pattern was more variable for dissolved Cu, Pb, and Zn. Significant reductions in the concentrations of these metals occurred during wet weather (Figure 34). For dissolved Zn, significant reductions from the inflow to the outflow also occurred during the dry weather events, while for dissolved Cu significant reductions occurred during the dry season. No significant reductions in dissolved Pb occurred during the dry weather, regardless of season. Dissolved metals in the outflow of the treatment wetlands were all below water quality criteria.

Significant reductions in TSS occurred only during the dry weather events in the wet season (Figure 35). Inflow concentrations of TSS were the significantly higher in the wet weather than in dry weather, but no differences were observed by season or weather event among outflow concentrations.

Patterns in the inflow and outflow concentrations of total and dissolved Se were very different from other metals (Figure 34). No significant decreases in total Se were observed during wet and dry weather events during the wet season, while a slight reduction occurred during the dry season. Likewise for dissolved Se no reductions occurred during the wet season, but slight reductions occurred between inflow and outflow during dry weather events. It is important to note that these data come from a single treatment wetland system.

For *E. coli*, *Enterococcus,* fecal coliform, and total coliform, significant reductions in concentration occurred from inflow to the outflow, regardless of weather or season, generally by 1 - 2 orders of magnitude (Figure 35). The inflow concentrations of these indicator bacteria were generally the highest during dry season; outflow concentrations for these constituents were only consistently significantly different between dry weather events during the wet season and dry season.

Significant reductions in concentrations of nitrate and phosphate were observed from the inflow to the outflow regardless of weather or season (Figure 35). Dry season concentrations of nitrate were generally the highest, though only significantly different from wet weather concentrations. No significant differences were observed by weather or season among inflow or outflow concentrations.

Figures 36 and 37 show the patterns in percent reduction of all constituents as a function of inflow concentration. Total Cu, Pb, Zn, TSS, nitrate, *Enterococcus, E. coli*, fecal and total coliform all show a distinct pattern of increased percent reduction as a function of increased concentration. Notably, these treatment wetlands could be a source of these contaminants, as illustrated by negative percent removals. However, this generally seemed to occur at the lowest concentrations and generally during dry weather, particularly during the dry season. For total and dissolved Se, dissolved Cu, Pb, Zn, and phosphate, there is a general trend towards increased percent reduction with higher inflow concentrations, though the data have much more scatter. Among these constituents, negative percent removals occurred frequently with total and dissolved Se, dissolved Cu, and phosphate.

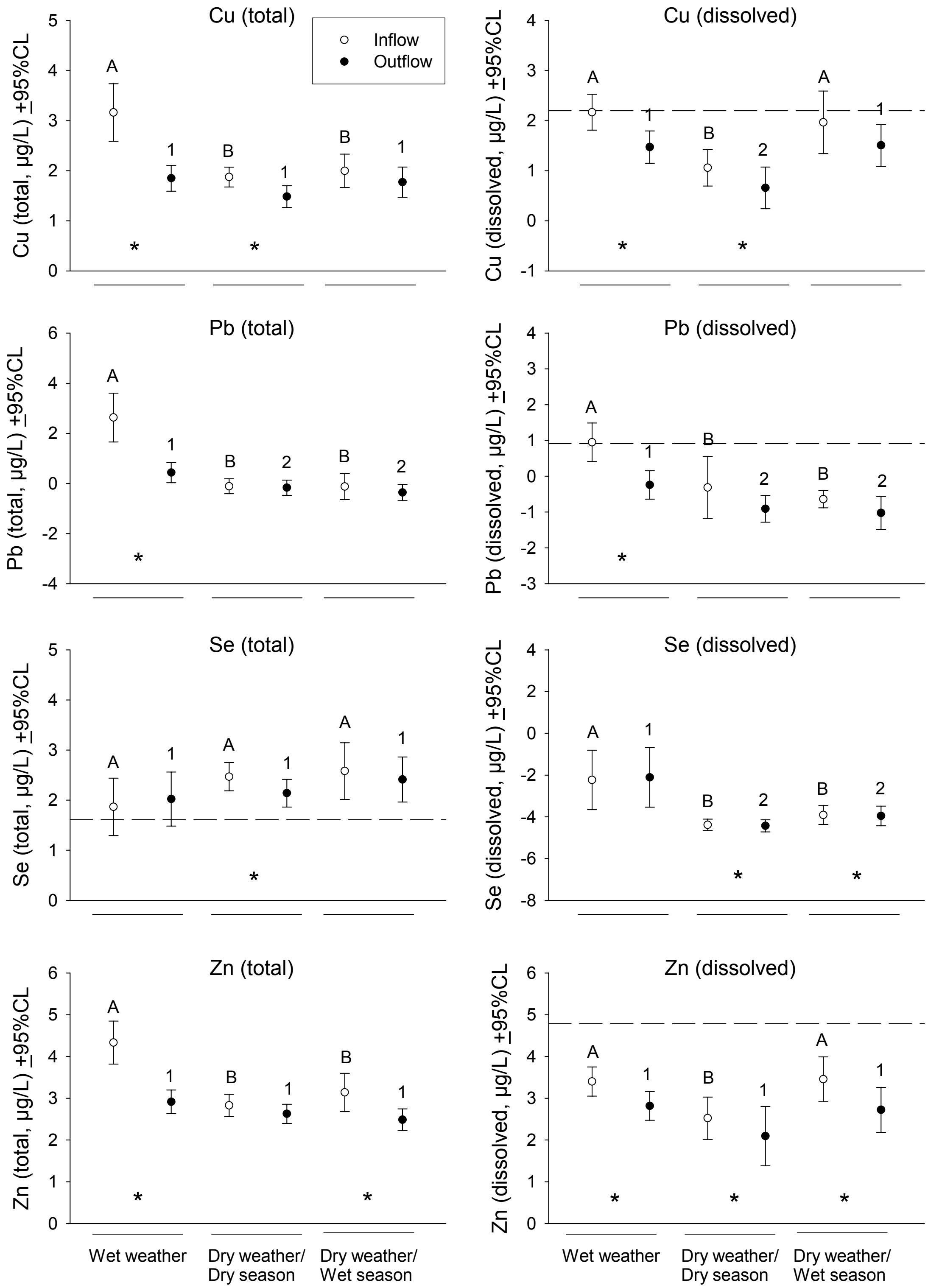


Figure 34. Natural log transformed concentrations of dissolved Cu, Pb, Zn and total Se before and after treatment of the dry and wet weather samples from southern California wetlands. Inflow and outflow mean concentrations (circles) and 95% confidence intervals are shown. The dashed lines indicate the water quality criteria. Significant differences between inflow and outflow samples are indicated by the asterisk (*). Letters indicate statistically similar inflow concentrations, while numbers indicate similar outflow concentrations.

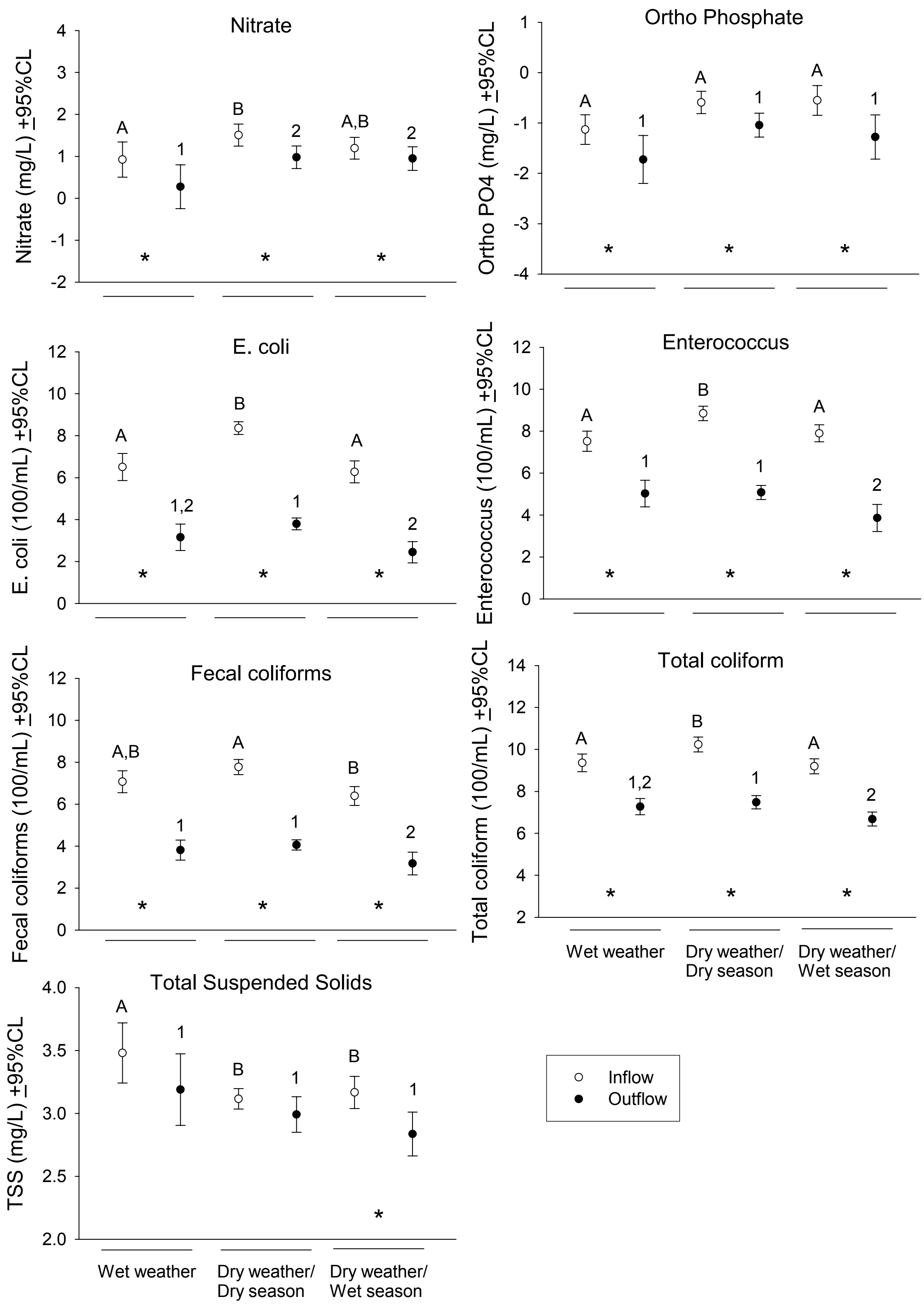


Figure 35. Natural log transformed concentrations of nutrients, total suspended solids (TSS), and bacteria before and after treatment of the dry and wet weather samples from southern California wetlands. Inflow and outflow mean concentrations (circles) and 95% confidence intervals are shown. Significant differences between inflow and outflow samples are indicated by the asterisk (*). Letters indicate statistically similar inflow concentrations, while numbers indicate similar outflow concentrations.

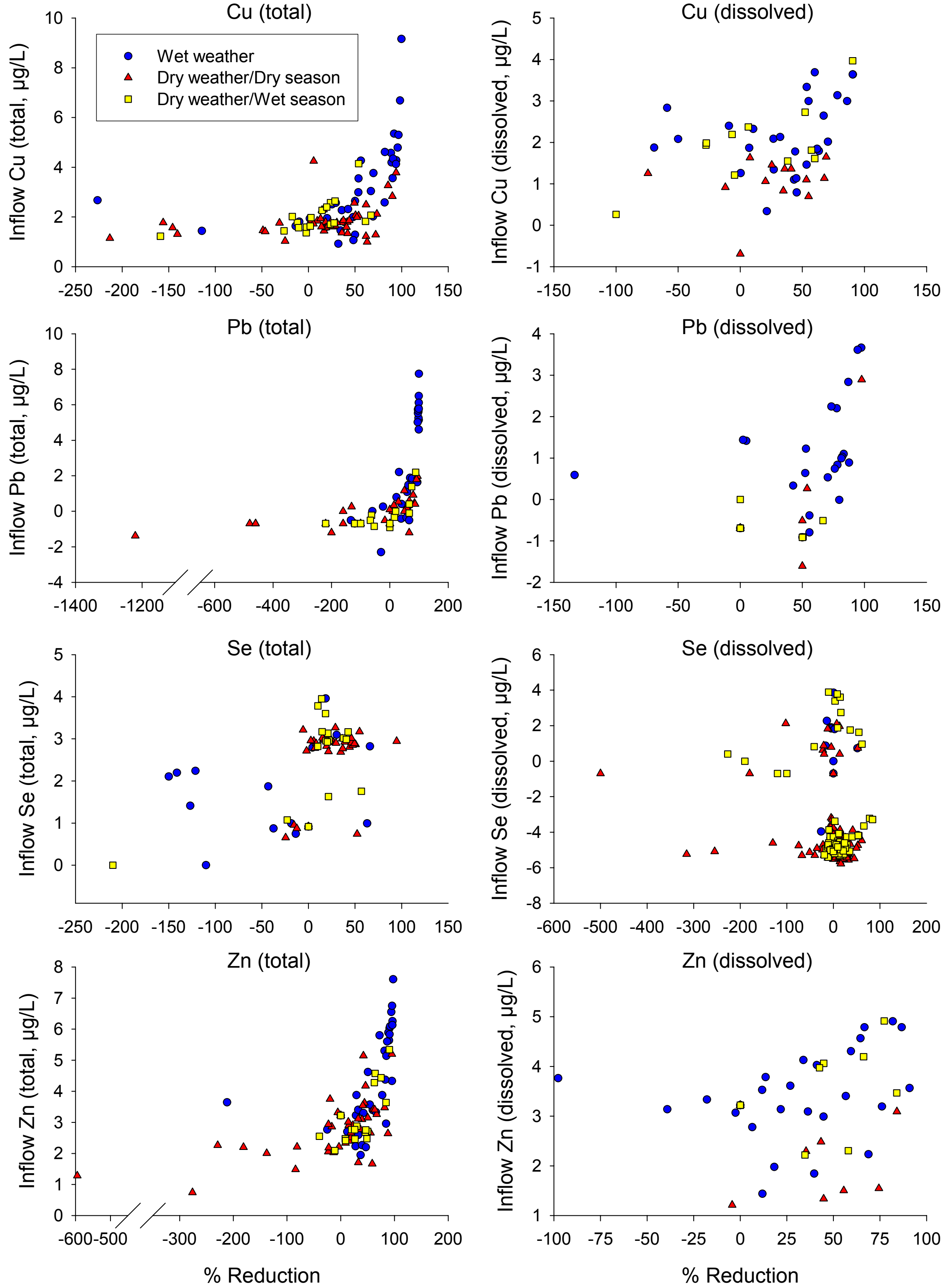


Figure 36. Natural log transformed concentrations of metals vs percent reduction for different weather/season categories for samples from southern California wetlands.

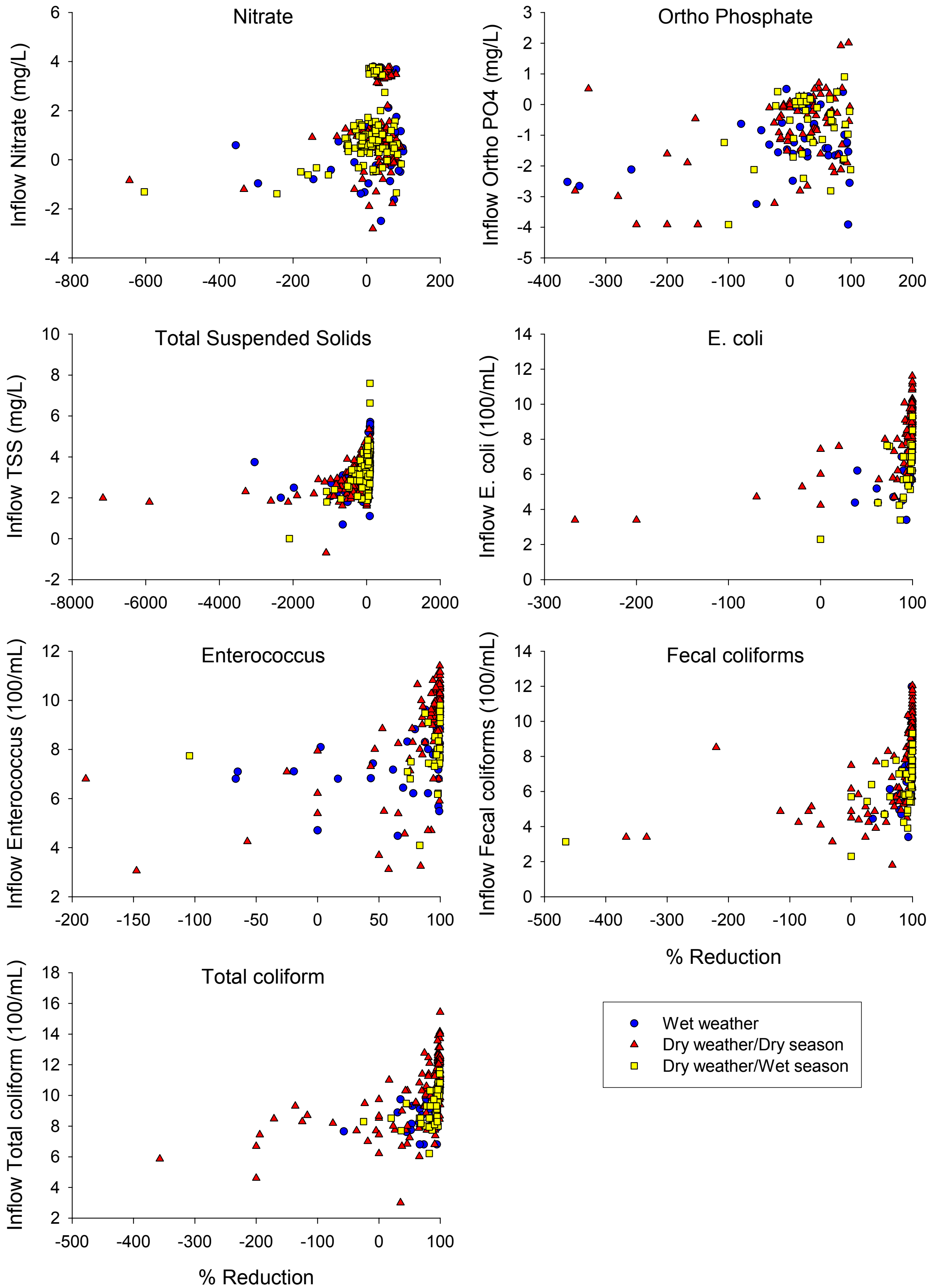


Figure 37. Natural log transformed concentrations of nutrients, TSS and bacteria vs percent reduction for different weather/season categories for samples from southern California wetlands.

Semi-arid and Temperate Climates

As found for semi-arid treatment wetlands in southern California, systems in temperate climates were also effective at significantly reducing concentrations of contaminants from inflow values (Figures 38, 39). For mean total and dissolved Pb, Zn, and dissolved Cu, there were no significant difference between the inflow concentrations between climates (Figure 38). This is also supported by looking at the percent reduction in concentration relative to inflow concentration (Figure 39, 40), which show that for wet weather events, data from the two climates essentially overlap. For total and dissolved Pb, mean outflow concentrations were generally higher for temperate treatment wetlands, while for mean dissolved Zn outflow concentrations were higher for semi-arid wetlands. Inflow and outflow concentrations for total Cu were significantly higher in semi-arid systems (Figures 38, 40). Treatment wetlands sampled in temperate areas seemed to have more a more variable range of % reduction in concentration of dissolved Pb relative to semi-arid systems; dissolved Pb concentrations in temperate wetlands on average fell above CTR water quality criteria; wetlands from semi-arid climates were able to significantly reduce concentrations below this threshold. In contrast, the range of percent reduction of dissolved Zn in semi-arid systems over the same concentration range was lower than in temperate systems (Figure 40).

Mean nitrate and phosphate concentrations were significantly higher for both inflow and outflow of semi-arid treatment systemsrelative to temperate treatment systems (Figure 39). Plots of inflow concentration versus % reduction show the range of nitrate and phosphate inflow concentrations are much higher over a range of % reduction measured (Figure 41). This trend was the opposite of that was found for *E. coli*, where inflow concentrations were much lower in semi-arid wetlands over a range of percent reduction, albeit with limited data in the International Stormwater Database.

TSS was significantly higher in the inflow and lower in the outflow of temperate treatment wetlands relative to semi-arid systems (Figure 39). Inflow concentrations plotted as a function of % reduction in concentration showed a decline in the effectiveness of semi-arid wetlands at a much higher concentration in semi-arid versus temperate treatment wetlands. (Figure 41)

Table 22. Comparison between percent reduction between treatment wetlands in semi-arid versus temperate climates. T-tests were used for the evaluations.

Analyte	Semi-arid		Temperate		p-value
	n	% Reduction Mean and 95% CI	n	% Reduction Mean and 95% CI	
Copper (dissolved)	27	35 ± 16	64	29 ± 9	0.49
Copper (total)	37	45 ± 20	264	12 ± 19	0.02
Lead (dissolved)	24	52 ± 20	79	-278 ± 270	0.02
Lead (total)	34	56 ± 21	401	31 ± 9	0.04
Zinc (dissolved)	27	30 ± 16	65	30 ± 41	0.99
Zinc (total)	37	55 ± 18	316	42 ± 10	0.22
TSS	82	-169 ± 112	571	55 ± 7	<0.01
Nitrate	55	15 ± 22	206	21 ± 78	0.38
Ortho Phosphate	46	9 ± 28	65	-107 ± 236	0.37
E. coli	29	26 ± 125	8	83 ± 19	0.38

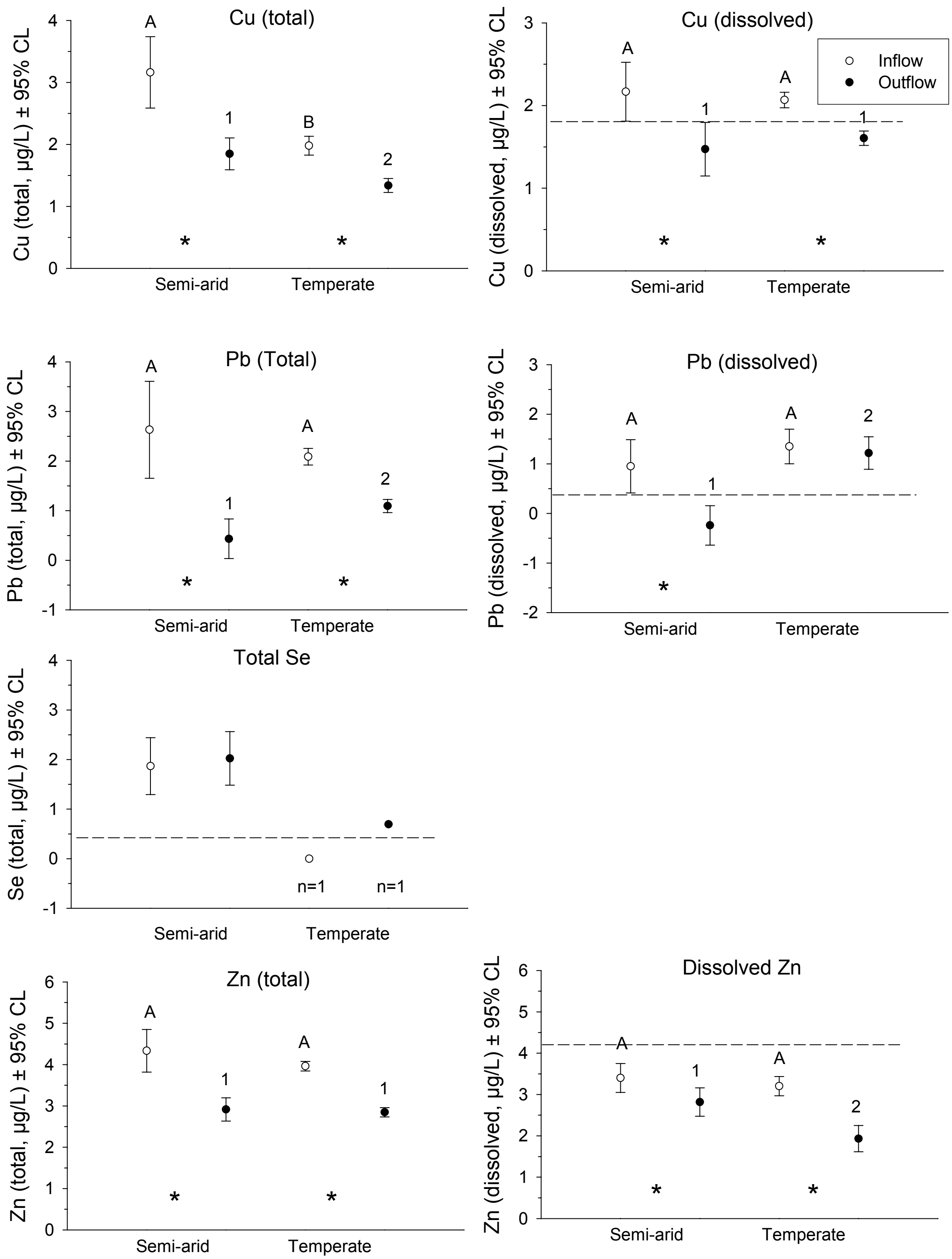


Figure 38. Natural log transformed concentrations of wet weather total and dissolved Cu, Pb, Zn, and total Se before and after treatment by the semi-arid and temperate climate wetlands. Inflow and outflow mean concentrations (circles) and 95% confidence intervals are shown. The dashed lines indicate the California Toxics Rule water quality criteria. Significant differences between inflow and outflow samples are indicated by the asterisk (*). Letters indicate statistically similar inflow concentrations, while numbers indicate similar outflow concentrations.

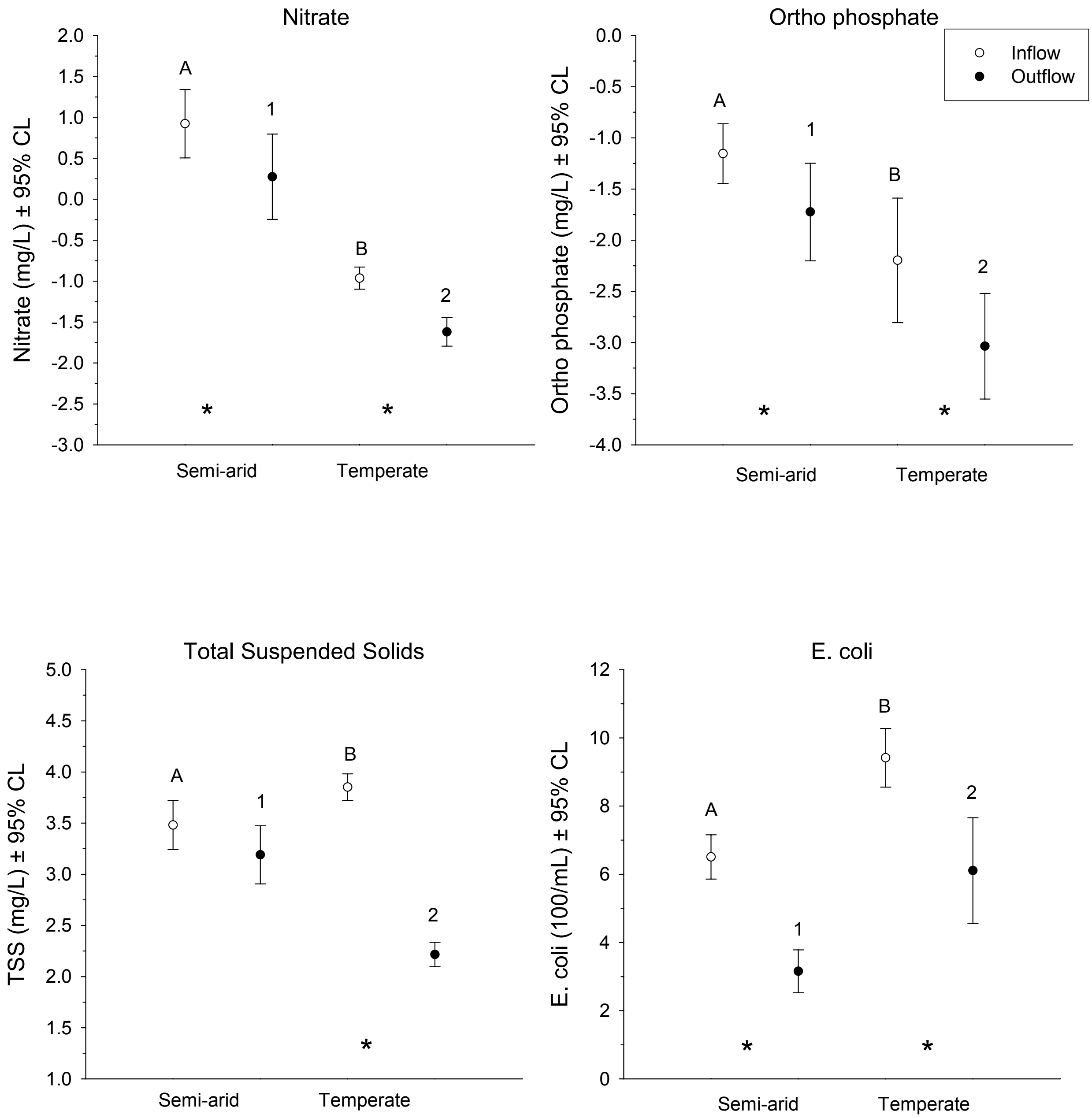


Figure 39. Natural log transformed concentrations of wet weather nutrients, total suspended solids (TSS), and *E. coli* before and after treatment by the semi-arid and temperate climate wetlands. Inflow and outflow mean concentrations (circles) and 95% confidence intervals are shown. Significant differences between inflow and outflow samples are indicated by the asterisk (*). Letters indicate statistically similar inflow concentrations, while numbers indicate similar outflow concentrations.

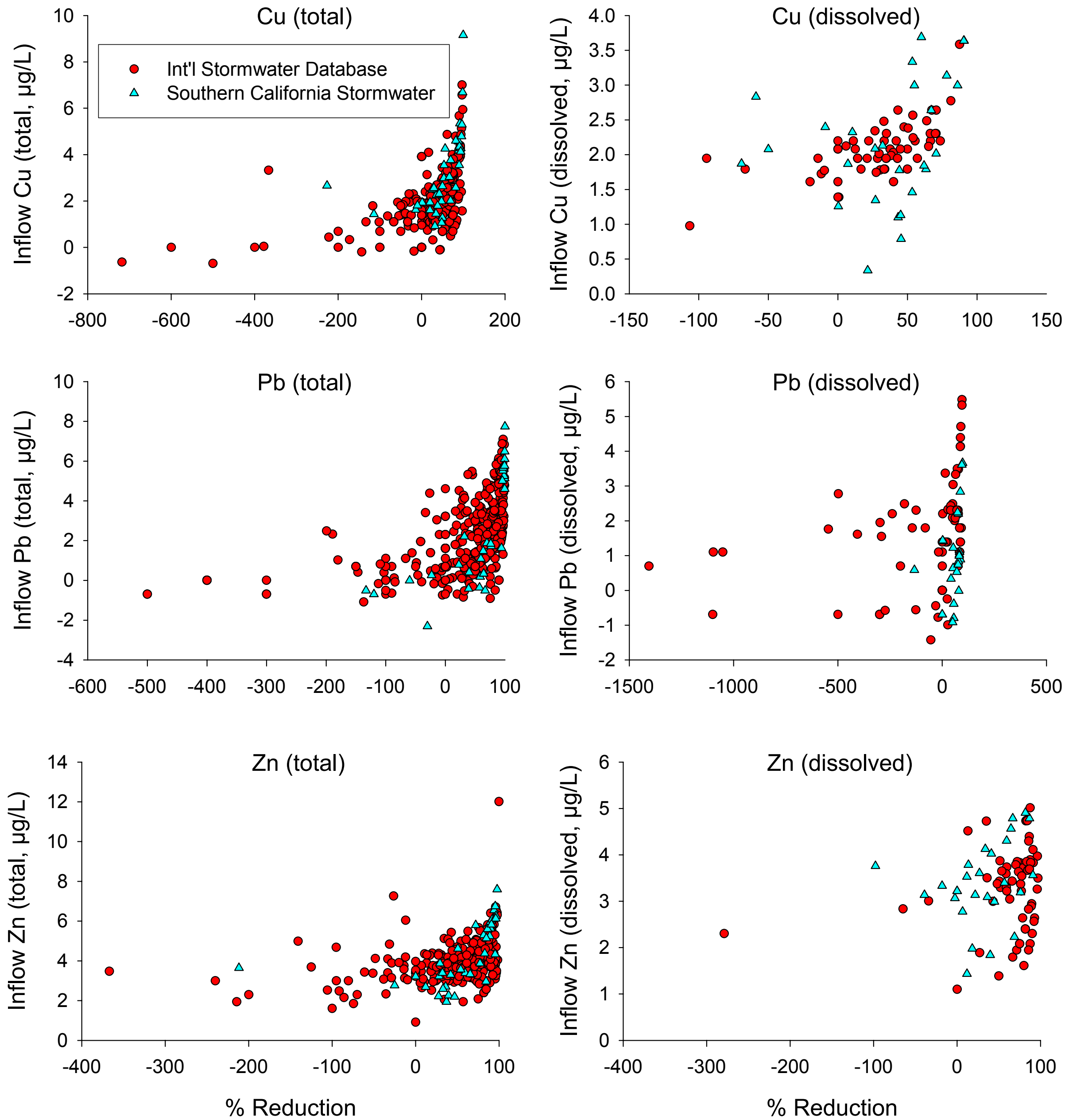


Figure 40. Natural log transformed inflow metal concentrations vs percent reduction for stormwater samples from temperate (International stormwater database) and semi-arid (southern California) climates.

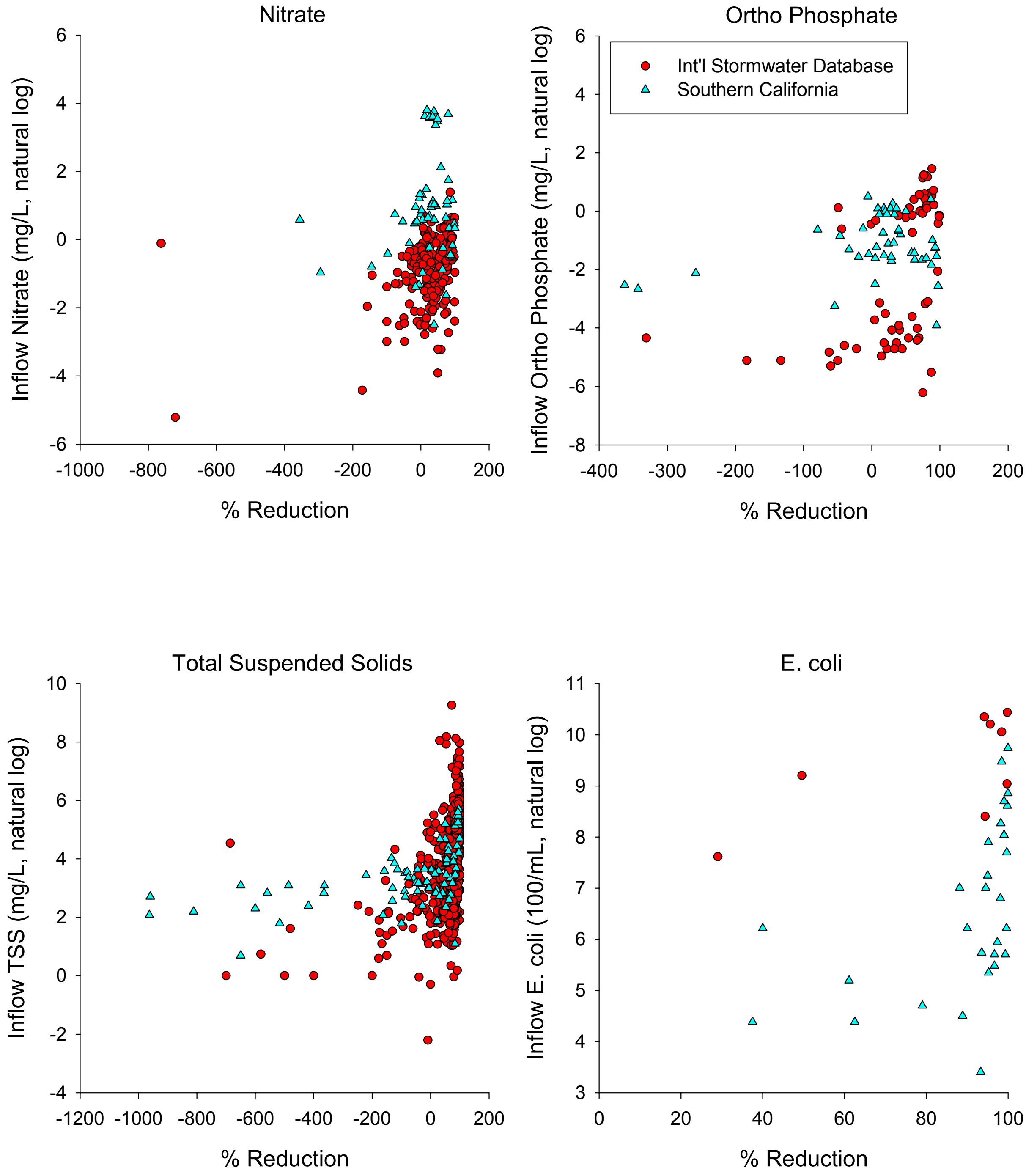


Figure 41. Natural log transformed inflow concentrations of nutrients, total suspended solids (TSS) and bacteria vs percent reduction for stormwater samples from temperate (International stormwater database) and semi-arid (southern California) climates.

Discussion

As our watersheds have become increasing urbanized, there is growing interest in treatment wetlands as a BMP to address issues of surface water pollution resulting from point and nonpoint source pollution (Kadlec and Knight 1996). A review by Strecker (1995) noted that only a limited number of studies have been published on effectiveness of treatment wetlands as BMP to treat urban runoff. Furthermore, most of the treatment wetlands for which published monitoring data are available are located in temperate climates. There is question to what extent these data are applicable to semi-arid and arid regions such as southern California (Schiff and Sutula 2002, Sutula and Stein 2003), where contaminant concentrations could be expected to be higher because wet weather events occur much less frequently. Another important data gap is effectiveness of treatment of dry weather urban runoff. While loads would be expected to be greatest during the wet weather events, "dry weather" in semi-arid areas such as southern California is the prevailing condition. Whereas dry season runoff may be somewhat chronic (e.g., steady flow), stormwater inputs are stochastic in nature, and subject to short-term, unpredictable pulses of water, sediment, and contaminants that may be several orders of magnitude higher than baseline inputs.

To begin to address these data gaps, existing monitoring data from treatment wetlands in southern California and from the International Stormwater Database were examined to better understand the contaminant removal efficiency for treatment wetlands, with respect to three principal questions:

- Is there evidence that treatment wetlands in southern California are reducing the concentration of contaminants in dry and wet weather urban runoff?
- Is there a difference in effectiveness during wet weather (storm events) versus dry weather (baseline flow) conditions and between wet season and dry season?
- Is there a quantitative difference between the removal efficiencies of temperate versus semi-arid wetlands?

Ability to answer these questions was limited by of a lack of flow data that would allow calculation of percent removal on a basis of loads. Analysis was limited to comparison of concentrations in the influent (entering the wetland) versus effluent (exiting the wetland).

Effectiveness of Southern California Treatment Wetlands in Reducing Contaminants

Existing monitoring data for treatment wetlands in southern California showed that these systems are generally effective at reducing the contaminants typically found in runoff, including metals, bacteria, TSS, and nutrients. Southern California treatment wetlands showed significant reductions in *E. coli*, *Enterococcus,* fecal coliform, and total coliform from inflow to the outflow, regardless of weather or season, generally by 1 - 2 orders of magnitude, consistent with published reports in the literature (Vymazal 2005). Significant reductions in concentrations of nitrate and phosphate were observed from the inflow to the outflow regardless of weather or season, a finding consistent with the widely held understanding that treatment wetlands are a cost-effective means of reducing nutrient concentrations in urban runoff, particularly for nitrate (Mitsch and Gosselink 1993, Kadlec and Knight 1996).

Despite these general trends in contaminant removal, a great variability was found in the effectiveness of removal, similar to the findings of other studies (Kadlec and Knight 1996, Strecker 1995). The time period in which the sample was taken is of extreme importance, since many contaminants are known to show a distinct pattern of increased percent reduction as a function of inflow concentration (Kadlec and Knight 1996). This was found to be the case for all total metals, TSS, nitrate, and public health indicator bacteria in this study. Since total metals and indicator bacteria are known to be associated with TSS, this relationship is not surprising, as most basin-type treatment wetlands are effective traps for suspended sediments (Kadlec and Knight 1996). Beyond this, basic treatment wetland size, geomorphology, hydrology, residence times, hydraulic and contaminant loading, biogeochemistry of soils, and other factors have been cited as responsible for additional variability in observed effectiveness (e.g., Kadlec and Knight 1996, Sutula and Stein 2003, Strecker 1995). Clearly, modeling of treatment wetland BMPs is needed to provide a time-integrated picture of water and contaminant budgets that can lead to better calculations of treatment efficiencies. Standardized monitoring of treatment wetland projects can provide the data needed to develop these models.

Data on percent reduction as a function of influent concentration indicate that treatment wetlands may be a source of contaminants relative to inflow concentrations. For wet weather data, this may be an artifact of when the sample was taken, and whether a time lag exists between a gradual decrease in the concentrations of a constituent in the inflow versus outflow. However, this could also be attributed to scouring and resuspension events within the wetland, which could release pore water or particulate-bound contaminants into the surface waters. The data indicate that this phenomenon occurred most often during dry weather events more so during the dry season, indicating that it might be more due to a biologically driven process rather than a physical one (Sutula and Stein 2003). It also occurred for most constituents at low concentrations, indicating that it could be an issue of analytical variability.

For dissolved Cu, Pb, and Zn, southern California treatment wetlands were effective at reducing wet season inflow concentrations to below the CTR water quality criteria. Concentrations in the inflow for these constituents were generally below the CTR water quality standards, so though there was little statistical difference between inflow and outflow concentrations during the dry weather periods, the concentrations are below levels of concern. This was not the case for Se, but it is noteworthy that all data on Se come from a watershed in which Se has a natural source, a Se total maximum daily load (TMDL) is being pursued to address issues of concern, and that other BMPs are being considered to reduce loads.

Differences in Treatment Effectiveness by Dry/Wet Weather or Season

The importance of looking at treatment effectiveness by wet and dry weather and/or by season is increasing as a larger number of NPDES permits are written to cover both winter dry and wet weather as well as summer dry weather loads from the watershed. Good data are needed to describe both conditions, yet most published studies of treatment of urban runoff have focused exclusively on wet weather events (e.g., Strecker 1995).

Percent removal of contaminants would be expected to be lower during wet weather, because constituent concentrations have been shown to be higher and more variable in storm water runoff than in dry weather flow (Kadlec and Knight 1996, Yoon and Stein In press). This was noted in particular for total metal concentrations and phosphate, which are particle reactive and could be expected to greatly increase with increased TSS loads (Gambrell 1993). Treatment wetlands, such as those in the southern California study population, which have a basin morphology, would be expected to serve as a good sediment trap. For most constituents, wet weather inflow concentrations to these wetlands were higher and more variable than dry weather, while outflow concentrations during wet weather were higher but generally not significantly different than dry weather, regardless of season. Another possible explanation for the similarity between dry and wet weather effluent quality is that the treatment process was not by-passed during wet weather. Caltrans (2004) and California Stormwater Quality Association have found that wet ponds with large permanent pools (three times the volume of a 1-year, 24-h storm) tended to have little difference in outflow concentrations between wet and dry weather flow. They suggested that the treatment process is not by-passed during wet weather events, and that the water leaving the wetland is often treated runoff that is being displaced by the more recent wet weather flow. The volume of storm flow in the wetlands in this document may not have been sufficient to have by-passed the treatment process, and the water with longer residence time was captured for the outflow samples.

Inflow concentrations of indicator bacteria were on average the highest during the dry season, a result consistent with increased biological activity during periods with higher temperature (Mitsch and Gosselink 1999). Data showed that during some periods these wetlands could be a source of indicator bacteria, but it typically mostly during the dry season. A number of published studies have suggested that wetlands can act as a source of fecal pollution to surface waters downstream (Grant *et al.* 2001), particularly through the harboring and reproduction of animal sources of fecal pollution in sediments (Sanders *et al.* 2004). Dry season concentrations of nitrate were generally the highest, though only significantly different from wet weather concentrations. No significant differences were observed by weather or season among inflow or outflow concentrations.

Difference in Treatment Effectiveness by Climate

Contaminant concentrations and loads from semi-arid land uses would be expected to be higher than their counterparts in temperate climates. This is because rain events are rare, so pollutant concentrations have longer chance to build up on impervious surfaces prior to wash-off, compared to more humid climate. Drier climates typically have higher sediment and organic carbon loading from open areas because sparser vegetative cover translates to less retention within the watershed. Finally semi-arid climates typically have a larger average rainfall accumulation per event (Caraco 2000). This hypothesis was only supported for total Cu, nitrate and phosphate. The opposite was found for TSS and *E. coli*; for all other constituents, no significant difference were observed in the inflow concentrations between treatment wetlands of the two climates. For TSS and phosphate, a leveling off of percent reduction seems to occur at lower concentrations in treatment wetlands of temperate climates, suggesting removal efficiencies may be higher. Removal of dissolved Pb appears to be more efficient in semi-arid systems, while removal of dissolved Zn appears to be efficient more in temperate treatment wetlands.

In order to confirm these trends, more needs to be understood about the size, soils, geomorphology and hydrodynamics of the systems from which the data are derived in order to tease out true differences in climate. More careful consistency of methods would also be required. Samples at the various wetlands were collected using a variety of strategies, including single grabs, time-weighted composites, and flow-weighted composites. The greatest affect from the different sampling strategies would have been associated with grab samples collected during storm events. This is because contaminant concentrations in storm water can fluctuate considerably over the course of a storm (Yoon and Stein 2007), and results from single grab samples could vary, depending on what part of the storm the samples were collected. However, few of the wet weather samples for this study were collected during the peak storm flow periods. For this study, storm water samples were considered to be those collected within five days of a wet weather event. Therefore, while the wet weather grab samples were most likely influenced by stormwater, the timing of the sample collections was not as important as peak flow.

Recommendations

The data from the existing studies can be used to improve the efficiency of monitoring. The existing designs are inconsistent among wetlands (Table 18), and there does not always appear to be a reason for how often samples were collected or which constituents were analyzed. Most of the data appears to have been collected over relatively short periods, as part of special reconnaissance surveys, in which case a broad range of constituents is justified (e.g., Big Canyon). However, some of the sampling appears to be part of longer-term, routine monitoring projects, which are conducted at greater frequencies and analyzed for a comprehensive suite of constituents (e.g., Ballona Freshwater Marsh, and Prado Wetlands). In addition to improving the efficiency of current monitoring programs, the existing data can be used to create new monitoring designs using the data from the special studies collected during the short-term reconnaissance surveys.

The first step in creating any monitoring program is to identify the purpose for monitoring is needed. A monitoring question should always have some decision value. If there is no answer, or the answer does not trigger a decision, then the need for that information should be critically evaluated. The strategy used to monitor the wetlands will depend on the wetland's primary function, whether for treatment of runoff, or to provide habitat; each wetland type has different management questions to address, each with respective requirements for study design, including sampling frequency, constituents analyzed, and appropriate reference conditions.

The recommended approach can be categorized as part of core monitoring, or a special study. Core monitoring is conducted to evaluate compliance of regulatory requirements, assess current conditions, and to track trends. Core monitoring should be conducted throughout the life of the wetland. Special studies are used to answer specific scientific questions, and are usually conducted over short durations. Special studies are often conducted as adaptive triggers in response to core monitoring, for example to track the source of a contaminant that has recently become problematic. Special studies can also be used for developmental research, for example to investigate ways of optimizing treatment performance, or to model the pollutant concentrations over the course of a storm. Special studies are unique to a particular wetland, whereas the core monitoring design is applicable to all wetlands. Because special studies are site-specific, describing all possible types of specific studies is beyond the scope of this document.

The potential management questions regarding effectiveness of treatment wetlands include:

- *Are concentrations of contaminants in the effluent protective of beneficial uses?*
- *What is the load of selected constituents in the effluent?*
- *How effective is the wetland BMP in reducing loads?*
- *Is the effluent concentration or mass changing over time?*

The necessity for each question and the data required to answer the question are discussed below and summarized in Table 24.

Table 24. Design overview for monitoring treatment wetlands.

Tier	Sampling Conditions	Type of Sample	Suggested Frequency	Indicators	Product
Are water quality criteria achieved?					
Core Monitoring	Wet Weather	Flow-weighted EMC[1]	3 times per year, then per results of sample size analysis	Constituents of interest (constituents that prompted creation of BMP); Dissolved Metals and Total Se	Table or graph of EMCs vs water quality criteria
	Dry Weather	Flow-weighted 24-h Grab Composite	3 times per year, then per results of sample size analysis; QA approach for non-detects	Constituents of interest; Dissolved Metals and Total Se	Table or graph of concentrations vs water quality criteria
Special Studies	Wet Weather	Individual Grab Samples	Depends on goals of special study	Constituents of Interest	Pollutagraph
	Wet Weather	8 Samples per Storm	Depends on goals of special study		Rainfall time series for modeling
	Dry Weather	Grab Composite	Depends on goals of special study	Constituents of Interest	
How effective is the wetland BMP in reducing loads?					
Core Monitoring	Not Applicable	Hydraulic Residence Time	Once	Physical Configuration of Basin	Modeled or measured volume as a function of water level
	Wet Weather	Flow-weighted EMC[1], Flow During Event.	3 times per year, then per results of sample size analysis; QA approach for non-detects	Constituents of Interest; Total Metals	Table or graph of % reduction
	Dry Weather	Flow-weighted 24-h Grab Composite, Flow During Composite	3 times per year, then per results of sample size analysis; QA approach for non-detects	Constituents of Interest; Total Metals	Table or graph of % reduction
Special Studies	Wet Weather	High Resolution Sampling	Depends on goals of special study	Constituents of Interest	Table or graph of changes in mass over the course of a storm event
	Dry Weather		Depends on goals of special study	Constituents of Interest	
What is the load from the BMP?					
Core Monitoring	Not Applicable	Hydraulic Residence Time	Once	Physical Configuration of Basin	Modeled or measured volume as a function of water level
	Wet Weather	Flow-weighted EMC, Flow During Event	3 times per year, then per results of sample size analysis; QA approach for non-detects	Constituents of Interest; Total Metals	Table or graph of effluent load vs TMDL value
	Dry Weather	Flow-weighted 24-h Grab Composite, Flow During Composite	3 times per year, then per results of sample size analysis; QA approach for non-detects	Constituents of Interest; Total Metals	Table or graph of effluent load vs TMDL value
Special Studies	Wet Weather	Individual Grabs	Finite number of events, e.g., 3 times	Constituents of Interest	Pollutagraph
	Dry Weather	Composite Sample and Flow	Increase frequency over core monitoring design	Constituents of Interest	

[1] Minimum rain event to trigger storm sampling is 0.2"; initiate sampling at %120% of baseline flow and terminate at 150% of baseline.

Q1: Are concentrations of contaminants in the effluent low enough to be protective of beneficial uses?

A primary management goal for treatment wetlands is clean up of runoff so it is not a potential risk to the environment. Therefore, the first question addresses treatment effectiveness, by determining if the wetland is able to reduce contaminant concentrations below levels of concern. For regulatory purposes, this is evaluated by comparing constituent concentrations to CTR water quality criteria values. The constituents that should be monitored are those that are known to be present at level of concern, such as those that prompted creation of the BMP, or which have recently become an issue. Some of the analyses conducted at southern California wetlands already reflect this idea. For example, because the WetCAT Wetlands were created or enhanced to provide treatment for bacteria and Mn, the constituents that were monitored in the assessment report by CH2M Hill (2004) are those that are relevant to bacteria (i.e., bacteria, nutrients) and Mn. In contrast, the study by Brown and Bay (2006) at the WetCAT West Wetland was an exploratory investigation examining the ability of BMPs to reduce toxicity, and therefore included analysis of pesticides and a wider variety of metals, but no bacteria.

Sampling frequency will depend on the risk of exceeding a threshold. The goal is to be able to determine what the measured outflow concentration is in relation to the chronic water quality criterion. A greater number of samples required when concentrations are near the threshold, or when variability is high. The optimum number of samples required is be determined with historical data, through a method (termed variance approach) that uses the threshold value, the current average value, the measured variability, the desired confidence level (usually set at $\alpha = 0.05$), and the desired test sensitivity (usually set at $\beta = 0.80$). This method was applied to metals concentrations from those wetlands that had available data. Figure 25 presents the results of the analyses for dissolved copper, which shows that for Ballona Creek Fresh Water Marsh, two samples are necessary for a confidence of 95%, while 28 samples are needed for La Costa, five samples are needed at the Prado Wetlands, and 12 samples are needed for the WetCAT West treatment wetland. The results of the analyses for each of the metals contaminants of interest are in Table 25. Sites that do not have enough data should be monitored three times per year for three years. Then the variance approach can be used to determine if the annual number of samples should be increased or decreased. The sample size analysis should also be revisited when there are changes to the inflow quality, or treatment process.

For those constituents that are below the reporting level, it is not possible to determine the average concentration or variability with any certainty, and therefore a second approach can be used to determine sample frequency. This second approach (termed QA-based approach) does not use proximity to the threshold, or variability. Instead, this approach uses a tiered pass/fail system, in which the sampling effort increases with the number of exceedances. Originally developed for the manufacturing industry, this approach is based on a binomial probability distribution to assess the likelihood of predicting exceedances. If enough samples meet pre-specified quality assurance guidelines (water quality thresholds), then the frequency of QA checks can be reduced and still keep managers confident that the process control is working properly. However, if a QA failure occurs, then the frequency of sampling needs to be increased to the original frequency to restore management confidence in QA. If repetitive QA failures occur, then sampling frequency is further increased until, ultimately, managers are convinced that some management action is necessary to improve performance.

The QA approach is best suited for application to dry weather monitoring in southern California, since the number of storm events is so unpredictable. The variance approach can be used for both wet and dry weather sampling events. During wet weather sampling, event mean concentrations (EMCs) should be collected for the frequency analysis. For dry weather sampling, 24-h grab composites should be collected.

Special studies for the first question could include reducing reporting levels or increasing sampling intensity for modeling purposes, or expanding the list of constituents to identify any new constituents of concern. Toxicity testing could also be used to assess the presence of unmeasured contaminants, with TIEs used to characterize the type of contaminants causing the toxicity. In addition to assessing the presence of unmeasured contaminants, toxicity testing also assesses the additive effect of multiple contaminants that by themselves may be below toxic or standard analytical levels.

Q2. What is the load of selected constituents in the effluent?

The second question is related to mass emissions. As mass-based regulations become more important, such as TMDLs, mass emission monitoring will become critical in evaluating compliance. Although the wetlands themselves probably will not have TMDL regulations, the wetlands may be used to help a sub-watershed meet TMDL goals. Measuring the mass loading in wetland effluent could be useful to managers for comparing among different sources (different wetlands, or different BMP types), and prioritizing resources to improve overall removal.

The constituents to be measured are similar to those for Question 1, only that total metals should be analyzed instead of dissolved metals. This is because for TMDL compliance, the total amount of a metal is important. Constituents targeted for TMDLs will most likely be analyzed as part of a water quality compliance plan for a given wetland. It is critical that flow be measured, either through water level records and a rating curve or through direct flow measurements, in order for estimate percent removal of loads. 12 Because the same data that is collected for concentration compliance will be used for the loads question, the sampling frequency will be the same as for Question 1. The reference condition will be the TMDL as it is applied to the 303(d) listed water body, with the treatment wetland assisting in meeting the mass emissions goals.

Q3. How effective is the wetland BMP in reducing loads?

The third question addresses removal efficiency of the treatment wetland by examining load reduction. Similar to Question 2, the primary role for this question is helping managers to assess a site's role in TMDL compliance. However, whereas Question 2 assesses the relative contribution from each input, this question assesses the ability to diminish the load of the influent, relative to the TMDL. Because this question has similar information needs as Question 2 regarding thresholds, variability, and constituents, the same type data can be used for both questions. Effectiveness can be assessed through graphs of percent load reduction relative to TMDL. Measurement of the physical configuration of the basin provides data to calculate hydraulic residence time as a function of flow. These data can then be used to model integrated load reductions over storms of differing characteristics.

Load reduction is also sometimes expressed as percent removal per acre of BMP. This endpoint of percent removal is only recommended for managers wanting to use this information to assess how several BMPs may be used in tandem to meet a TMDL goal.

Q4. Is the effluent concentration or mass changing over time?

The fourth question is a trends question. Managers will want to know if conditions (concentration or loads) are changing to determine if management actions are necessary (e.g., know when to dredge the wetland, start source identification, or add additional treatment), and also to determine if management actions are having an effect. Sample designs for Questions 1 - 3 can be repeated as necessary to address the timeframe of interest.

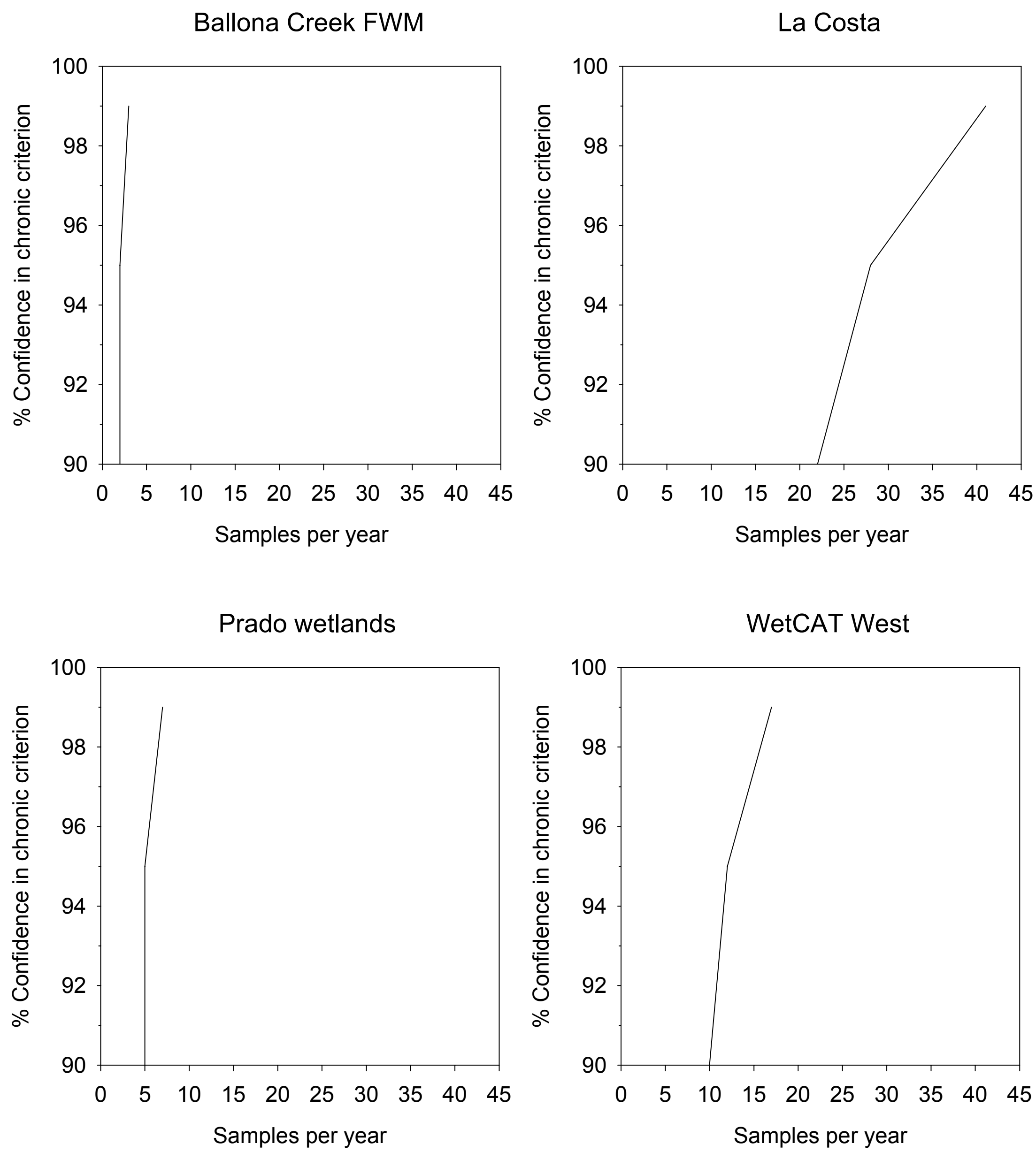
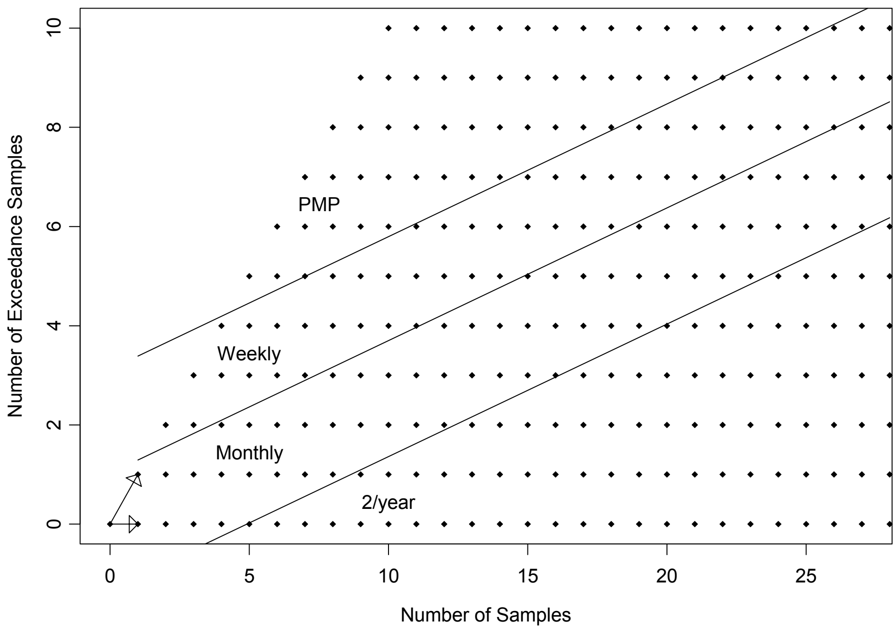


Figure 42. Sampling effort required in order to determine what the measured outflow copper concentration is in relation to the chronic water quality criterion, for a given level of confidence.

Table 25. Annual number of wet and dry weather samples required to be 95% confident what the measured outflow values are in relation to the chronic water quality criteria. NA = not enough samples for analysis. ND = non-detect.

Treatment Wetland	Metal	Dry	Wet	Combined wet & dry data
Ballona Creek Fresh Water Marsh	Dissolved Cu	2	3	2
	Dissolved Pb	2	7	5
	Dissolved Zn	2	2	2
	Total Se	22	17	19
San Joaquin Marsh	Dissolved Cu	NA	NA	NA
	Dissolved Pb	NA	NA	NA
	Dissolved Zn	NA	NA	NA
	Total Se	5	7	5
La Costa	Dissolved Cu	NA	29	28
	Dissolved Pb	NA	16	15
	Dissolved Zn	NA	2	2
	Total Se	NA	NA	NA
Prado	Dissolved Cu	6	NA	5
	Dissolved Pb	ND	NA	ND
	Dissolved Zn	ND	NA	ND
	Total Se	2	NA	2
WetCAT West	Dissolved Cu	8	NA	12
	Dissolved Pb	ND	NA	ND
	Dissolved Zn	2	NA	4
	Total Se	2	NA	2



$P_0 = 0.1$, $P_1 = 0.5$, $\alpha_1 = 0.1$, $\alpha_2 = 0.001$, $\beta = 0.05$

Figure 43. Sampling frequency approach for chemicals with reporting levels near California Toxics Rule (CTR) values. P_0 = acceptable probability of exceedance (used for lowest line). P_1 = unacceptable probability of exceedance (used for upper line(s)). For α_1 = probability that increased sampling is mandated when the probability of exceedance is actually below P_1. For α_2 = probability that management action (Pollution Minimization Program, PMP) is mandated when the probability of exceedance is actually below P_1. For β = probability that reduced sampling is mandated when the probability of exceedance is actually above P_0.

SUMMARY OF FINDINGS AND MANAGEMENT RECOMMENDATIONS

Study Findings

The Los Angeles Regional Water Quality Control Board and the State Coastal Conservancy funded a study to evaluate habitat value associated with urban wetlands of multiple objectives in southern California. The goal of this project is to provide information on how these urban wetlands can be better managed to increase compatibility with wildlife protection in southern California.

A phased approach was used to allow data collected in the first phase drive decisions about objectives and study design in subsequent phases. The four major phases of this project included:

- Develop an inventory of existing urban and treatment wetland projects
- Conduct a habitat survey on a representative sample of urban wetlands to evaluate wildlife beneficial use
- Evaluate the exposure and toxicity to wildlife from sediment-borne contaminants
- Conduct analysis, to the extent possible, on the effectiveness of treatment wetlands, using existing monitoring data

The sections below summarize the findings for each component of the study.

Inventory of Urban Wetlands

The inventory of 40 freshwater urban wetlands was used as a means to gather existing data on the basic characteristics of selected wetland sites and provide a means to sample sites for subsequent parts of the study.

General Characteristics of Urban Freshwater Wetlands. The 40 freshwater urban wetlands were generally highly modified from historical reference, with a large percentage of the wetlands were type converted (converted from one wetland class to another) to forms atypical of the southern California landscape. All 40 sites have the urban runoff as their major water source. Approximately 40% of sites have actively managed hydrology. Of these, approximately 60% of the sites have habitat as their primary objective, 25% have water quality as the major objective, and the remainder can be classified as multipurpose (habitat and water quality treatment). About 50% of sites had frequent maintenance activities (>1 time per year), with vegetation management as the most common (90%). Hydrologic management and trash removal was practiced about half of the sites, and vector control at all sites.

Biological Survey

The biological survey was used to gather basic information on the habitat value of the 40 freshwater urban wetlands, plus three reference sites. Habitat value at this stage of the study was defined as the ability to support characteristic flora and fauna. Habitat mapping and assessment of general habitat condition was conducted on all 40 sites; intensive analysis of a fuller suite of indicators including plant community composition, benthic macroinvertebrates, and birds were completed on 29 sites. The biological survey was used to answer the following questions:

- What is the condition of urban wetlands relative to reference?
- Does ecological condition vary as a function of a gradient in land use intensity?
- Is there a difference in the condition of habitat versus water quality or multipurpose wetlands?
- How does the frequency of maintenance impact the condition of urban wetlands?
- How can the habitat value of urban wetlands be improved?
- What is the minimum recommended monitoring for habitat in wetland projects?

Appropriate Reference for Urban Wetlands. Reference for basin wetlands in this study represents an underlying shifting of baselines towards systems that were presumably relatively rare or never present in the southern California landscape. An adequate understanding of reference is hampered by the lack of true reference wetlands, particularly for basin wetlands, which for this study were mostly characterized by perennial ponds.

Recent historical ecology studies in the region showed that perennial ponds were a minor fraction of wetland habitat; the landscape has been dominated by groundwater-supported alkali and wet meadows and other types of seasonal wetlands. The practice of importing water into Southern California, in combination with extensive groundwater withdrawal, has completely changed the water budget of freshwater urban wetlands. Type conversion of stream habitat to in-channel basins and the formation of linear channels with extended detention (e.g., the bioswale) represents habitat for which no reference exists.

The biological survey results reflect a state of "best achievable", but do not necessarily provide an adequate benchmark for the characteristics of a basin wetland in pristine ecological condition. Two of the three basin reference wetlands were relic stock ponds that have been relatively undisturbed since the 1960s, and as such provide "best achievable" reference for basin wetlands. For channels, this issue is present, but less of a concern because of the availability of pristine sites in the upper watershed (mostly on national forest lands) throughout the region. It is uncertain whether these sites provide an adequate reference for small streams on the coastal terraces, which are now largely disturbed and for which reference sites are no longer available.

Additional studies of the historical ecology and present day distribution of native flora and fauna associated with freshwater depressional (basin) wetlands is greatly needed to provide a better understanding of reference.

Site-specific Factors can Mitigate Landscape Restraints. Results from this biological survey illustrate that, while urban infrastructure provides basic constraints on "best achievable" wetland condition, site specific factors such as wetland project design criteria, objectives, wetland management and maintenance activities can mitigate to some degree the constraints of the urban landscape.

This study documented the decline in general habitat condition (CRAM landscape context, physical structure), and benthic macroinvertebrate community condition with increasing imperviousness of either the surrounding, or catchment, land use. However, native plant species richness and plant species diversity actually increased significantly with increasing % imperviousness. Several urban wetland sites surrounded by highly impervious land covers had native plant species richness and diversity that exceed that found in reference sites. This indicates that, while degree of urbanization places some basic constraints on what condition a wetland site can be expected to have, site-specific factors can mitigate (or further exacerbate) the effects of the landscape constraints on wetland condition. Other factors generally ignored in the creation of basin wetlands include the physical structure. Most created urban basin wetlands in the study were characterized by oval shapes with steep slopes. Improved design of physical structure to include a variegated shoreline, an abundance of macro- and micro-topographic elevations within the wetlands will provide a greater number of niches for biodiversity. Thus good designs for wetland creation, restoration of enhancement as well as active management of stressors (hydromodification, increased sedimentation, contaminant exposure, excessive visitation, predation from urban wildlife, etc., exotic species) may to some degree mitigate constraints of urban infrastructure

Wetland Size. Wetland size was a major controlling factor on the condition of urban basin wetlands. The majority of urban wetlands in this study were very small, with most basin sites under 5 acres. The condition of the wetland with respect to number of mapped habitats, CRAM index scores, plant and avian diversity and richness metrics were all significantly negatively correlated to size.

The reason for this relationship is two-fold. Larger wetlands naturally have a greater area in which develop a range of moisture gradients and habitat niches that are occupied by a wide diversity of wetland flora and fauna**.** Larger wetlands have a smaller ratio of edge to area than small wetlands, and thus provide greater interior spaces that provide better opportunities for foraging, refuge, and reproduction.

Wetland size is often constrained by adjacent land uses, especially in an urbanized landscape, so is not a factor that is easily managed, it is an element that can be taken into account when prioritizing sites for restoration. It is also important to recognize the value of small wetlands in a fragmented highly urbanized environment. Particularly in arid or semi-arid, highly urbanized areas such as in southern California, wildlife are attracted to aquatic habitats in great numbers because of natural scarcity of such resources.

Project Objective and Design Criteria. Urban wetlands that are created, restored or enhanced for habitat versus water quality or multipurpose have some basic differences that may constrain the type or condition of habitat that they can provide. However, this study did not establish, by weight of evidence across all indicators used, that habitat wetlands had statistically significant, superior condition relative to multipurpose or treatment wetlands.

Among basin wetlands, habitat wetlands had twice the relative area of riparian habitat and roughly half the amount of open water as treatment wetlands. In a project area of limited size available for treatment, more emphasis will likely be placed on providing treatment through open water and emergent marsh rather than providing riparian habitat, which is an essential feature of both streams and basin wetlands for refuge and buffering from the stressors of an urbanized landscape.

Most of the habitat mapping, CRAM, plant community and avian survey metrics showed habitat sites to be on average in slightly better condition, but these trends were generally not significant. This general reason for this large variability in condition of habitat wetlands probably lies in the general constraints of the urban infrastructure as well as factors such as the lack of maintenance occurring in some of the urban wetlands. Some multipurpose and water quality wetlands are being maintained better than some habitat wetlands.

Multipurpose wetlands had significantly higher plant species richness and diversity than habitat or treatment wetlands. The majority of the multipurpose wetlands were mitigation wetlands, for which the establishment of plant cover with native species is a common permit requirement. Comparison of data from these systems with data from reference sites shows that the diversity and richness of plants in these multipurpose sites is being maintained artificially high.

Multipurpose and treatment wetlands had significantly lower CRAM physical structure scores than habitat wetlands. These multipurpose and treatment wetlands were characterized by oval configurations, unvariagated shorelines and steep slopes. Treatment wetlands and many mitigation wetlands normally lack the complex microtopography found in natural wetlands. Treatment wetlands must be designed and regularly maintained to optimize treatment capacity, so the physical configuration of these wetlands may be an element required to optimize flow conditions or provide easy access for maintenance and vector control. An increased understanding of the importance of the elements of good physical structure in balance with these practical constraints in a wetland may help improve the quality of habitat provided in restoration and mitigation projects.

Origin. The condition of the habitat provided by historic wetlands was significantly higher than that provided by wetlands that have been type converted from streams and/or floodplain habitat or created from upland habitat. Basins that were type converted from other historic stream habitat represented 68% of the sample population. This type conversion is one of the greatest source of urban wetland loss in the last two decades.

Watershed management planning should place an emphasis on conserving and restoring historic wetlands to habitat and promoting the creation of water quality wetlands from upland habitat. One should also question the appropriateness of converting stream habitat into wetland detention basins to address water quality issues from hydromodification and increased contaminant loading from development.

Intensity of Maintenance Activities. This study clearly showed that urban wetlands must be maintained frequently to manage the variety of urban stressors, but not at an intensity or in a manner that may be incompatible with the seasonal cycles of nesting and reproduction. Intensity of maintenance activities was significantly related to parameters such as native species richness, where lowest condition was observed in wetlands that were being maintained at frequencies at greater than 2 times per year or greater. Since the data on maintenance frequency collected for the sites was not specific to the type of activity, it is useful to understand what types of activities typically occur through maintenance, and explore impacts on habitat condition from these activities. Maintenance activities for both habitat and treatment wetlands can include: 1) the regulation of influent flow rates and wetland water levels to keep hydraulic and contaminant loading rates or hydroperiod within targeted objectives for the site, 2) removal of sediments that have accumulated in sediment forebays, 3) rotation of discharge sites to allow the wetland to have an extended opportunity to assimilate contaminants and organic matter that create high oxygen demand, and 4) trash removal and vegetation management.

Factors Associated with Sediment Contamination and Toxicity

Sediment contaminant chemistry and toxicity to benthic macroinvertebrate organisms were assessed at 21 wetland sites at the beginning of the wet season. Specific questions addressed with this component were as follows:

- How does the sediment toxicity and chemistry of urban wetland sediments compare to reference wetlands,
- If toxicity exists, what classes of compounds are the likely causes of that toxicity?
- How do contaminant concentrations or toxicity vary as a function of wetland objective and degree of urbanization?
- What are specific recommendations to mitigate the risk of exposure and toxicity?

Presence and Magnitude of Risk. Most wetlands posed a risk of elevated sediment contaminants and/or toxicity. Most of the sites (18 of the 21 urban wetlands) were either toxic to the amphipod *H. azteca*, or exceeded a sediment quality guideline, or both. An index of degree of sediment contamination (mPECq) was found to negatively correlate with benthic macroinvertebrate diversity in these wetlands.

Sediment contaminant concentrations and toxicity could also be expected to vary both spatially and temporally within a wetland. In this study, sediment samples were taken in a location proximal to the primary source of urban runoff. As contaminant concentration and toxicity could be expected to vary, at minimum, as a function of distance from the source of contaminants, the results of the study are likely to represent, spatially, a conservative estimate of risk for the wetland. In addition, the samples were collected during dry weather, representing chronic nuisance flow conditions. Storm water is usually a much more complex matrix than dry weather flow, containing higher concentrations and a greater variety of contaminants. It is unclear if sediment quality of wetlands reflects as large of a seasonal change as the overlying water quality. Therefore, additional information about how sediment contaminant concentrations vary by seasonally and spatially within wetland is needed. This information would be important to better understand the magnitude of the risk posed to wildlife and to identify management options to mitigate that risk.

Class(es) of Compounds Likely Responsible for Toxicity. Without fractionation and confirmation studies, the contaminant(s) responsible for the toxicity observed cannot be conclusively identified. However, sediment chemistry suggests that pyrethroids may have been responsible for much of the toxicity documented in this study. Pyrethroid concentrations were elevated at all 10 sites that were toxic to *H. azteca*, and the mean pyrethroid quotient was negatively correlated with amphipod survival. Other contaminants were also elevated (including heavy metals, PAHs and DDE) and could have contributed to the toxicity, although not at as many sites as pyrethroids. Confirmation studies would need to be conducted in order to determine definitely the source of toxicity, which could be expected to vary at each wetland site. While pyrethroids appear to be implicated in the toxicity, a sediment TIE would be useful to help identify the class of compound(s) causing the toxicity.

Implications of Toxicity to Higher Level Wetland Organisms. Adverse effects on higher level wetland organisms (e.g., amphibian, birds, fish, etc.) can occur via bioaccumulation, direct toxicity or via impact from alterations in the food web. In this study, an index of degree of sediment contaminants (mPECq) was found to significantly correlate with benthic macroinvertebrate diversity in these wetlands. Macroinvertebrates are a critical link in the food web of wetlands, providing the link between primary producers, detrital trophic organisms, and higher level consumers such as birds, fish, and amphibians.

In this study, the magnitude of this alteration is difficult to document because of the limited data on reference sites. Few remaining reference sites for the “basin” class of wetlands exist in southern California and most were dry during the sampling period, which occurred during the driest rainfall year on record for southern California Although limited, the sediment chemistry and toxicity data from the reference sites show a clear distinction from the majority of urban sites. Benthic macroinvertebrate taxa diversity was not significantly different between the reference sites and urban sites, suggesting that additional stressors may also be important in controlling invertebrate diversity at these sites.

Relationship with Degree of Urbanization. Concentrations of several of the constituents (Cu, Pb, Zn, PAHs, cypermethrin) were significantly correlated with percent imperviousness of the catchment area, an index of percent urbanization. Most of these constituents are associated with automobile use, including engine and brake wear, oil leakage, and gasoline combustion. Sediment toxicity and sediment pyrethroid concentration was not significantly correlated with the degree of urbanization. This may be because agricultural use of pyrethroids would have served as a confounding factor in this analysis.

Project Objective. Habitat wetlands posed just as much risk of elevated contaminants as treatment or multipurpose wetlands. Sediment chemistry concentrations and toxicity were not significantly different among habitat, water quality and multifunctional wetlands. These results are similar to what was found for several other indicators of habitat quality including benthic macroinvertebrate and bird diversity.

The lack of significant differences by objective may be due to the profound equalizing effect that a broad range of stressors present in the urban can have on wetlands. These stressors, including eutrophication, excessive sedimentation, invasive species, habitat fragmentation and disturbance, etc. can confound effects of contaminant toxicity, making trends difficult to identify.

Analysis of Existing Monitoring Data to Assess Treatment Wetland Effectiveness

Existing monitoring data for southern California treatment wetlands were examined to better understand the contaminant removal efficiency for treatment wetlands , with three principal questions:

1. Do treatment wetlands reduce the concentration of contaminants in dry and wet weather urban runoff?
2. Is there a difference in effectiveness during wet weather (storm events) versus dry weather (winter or summer base flow) conditions?

3. Is there a quantitative difference between the removal efficiencies of temperate versus arid wetlands?

Ability to answer these questions was limited by of a general lack of flow data that would allow calculation and comparison of percent removal on a basis of loads. Analysis was limited to comparison of concentrations in the inflow (entering the wetland) versus outflow (existing the wetland).

Treatment Wetlands Reduce Contaminant Concentrations. Existing monitoring data show that southern California treatment wetlands reduce the concentrations of all constituent of interest [i.e., total and dissolved metals (Cu, Pb, Zn, Se), nutrients (nitrate, ortho phosphate), TSS, and bacteria (*Enterococcus*, *E. coli*, fecal and total coliforms)] relative to inflow concentrations. For dissolved Cu, Pb, and Zn, southern California treatment wetlands were effective at reducing wet season inflow concentrations to below the CTR water quality criteria. Southern California treatment wetlands showed 1 - 2 orders of magnitude reductions in *E. coli*, *Enterococcus,* fecal coliform, and total coliform. Significant reductions in concentrations of nitrate and phosphate were observed from the inflow to the outflow regardless of weather or season, a finding consistent with the widely held understanding that treatment wetlands are a cost-effective means of reducing nutrient concentrations in urban runoff, particularly for nitrate.

Great variability was found in the effectiveness of removal. Factors influencing this variability include time of sampling as well as other physical, chemical and biological attributes of the wetland. Clearly, modeling of treatment wetland BMPs is needed to provide a time-integrated picture of water and contaminant budgets that can lead to better calculations of treatment efficiencies. Standardized monitoring of treatment wetland projects can provide the data needed to develop these models.

Comparison of Dry versus Wet Weather Performance. Study data also illustrate that percent reductions and inflow concentrations can vary greatly by wintertime wet and dry weather and dry season. Percent removal of contaminants typically associated with suspended solids such as total and dissolved Zn, Cu and Pb, metals, phosphate and public indicator bacteria contaminants had a higher percent reduction in concentration during wet weather than during dry weather. A few significant differences were found in the inflow and outflow total Cu and Zn during dry weather periods of both the dry and wet season, but these reductions were minor in comparison with those occurring during wet weather. No significant differences were found between inflow and outflow concentrations of total and dissolved Pb during dry weather periods. Differences in concentrations of total metals among outflows by weather and season were generally not significant. Inflow concentrations of indicator bacteria were on average the highest during the dry season, a result consistent with increased biological activity during periods with higher temperature. Dry season concentrations of nitrate were generally the highest, though only significantly different from wet weather concentrations. No significant differences were observed by weather or season among inflow or outflow concentrations of phosphate.

Data on percent reduction as a function of inflow concentration during wet and dry weather periods indicate that treatment wetlands may be at times a source of contaminants relative to inflow concentrations. The data indicate that this phenomenon occurred most often during dry weather events, more so during the dry season and during low concentrations, indicating that it is not likely to be significant source during these periods.

Comparison of Treatment Wetlands in Arid Versus Temperate Climates. Some differences in contaminant concentrations in treatment wetland inflows were found between semi-arid and temperate climates. Total Cu, nitrate and phosphate were higher in southern California treatment wetlands relative to temperate sites in the International Stormwater database. The opposite was found for TSS and *E. coli*, and for all other constituents, no significant difference were observed in the inflow concentrations between treatment wetlands of the two climates. Removal of dissolved Pb appears to be more efficient in semi-arid systems, while that of dissolved Zn appears to be efficient in temperate treatment wetlands. In order to confirm these trends, more would need to be understood about the size, soils, geomorphology and hydrodynamics of the systems from which the data are derived in order to tease out true differences in climate. More careful consistency of sampling and analytical methods would also be required.

Monitoring Recommendations. Recommendations for future monitoring of treatment wetland BMPs are given. One key recommendation is to require that monitoring be conducted and reported based on load rather than concentration alone. This would necessitate that flows be measured for the period in which the sample was taken, either through direct measurement of flow or via water level recorders with a rating curve to establish flow. Collecting performance data on treatment wetland BMPs with consistent sampling methodology and in a SWAMP-compatible format will greatly enhance the collective understanding of BMP effectiveness in meeting load reductions.

Management Recommendations to Improve the Habitat Value of Urban Wetlands

While urban land uses provide some basic constraints on the achievable condition of wetland basins and channels, site-specific conditions can mitigate the constraints of urban infrastructure. Sutula and Stein (2003) developed a series of recommendations, derived from the literature on the siting, design, and management of urban wetland to reduce the risk to wildlife. Results from this study can be used to augment these recommendations by showing that, if implemented, they will aid urban wetlands in attaining “best achievable” ecological condition. These recommendations can be organized into four categories: 1) watershed planning, 2) design elements, 3) management and maintenance, and 4) minimum standardized monitoring.

Watershed Planning Perspectives

Consider that a watershed wide plan to reduce urban runoff can be greatly aided by watershed scale conservation and restoration of natural wetlands and riparian areas. Given evidence that the presence of riparian buffers greatly reduce contaminant concentrations in runoff before it is discharged to streams and wetlands, an effective strategy for NPS control in a watershed should be coupled with an aggressive program to restore riparian buffers around streams and wetlands and maintain interconnections between wetlands and other natural habitats. Promote conservation of open space and forest cover, and maintain natural storage reservoirs, drainage corridors including depressions, areas of permeable soils, swales, and intermittent streams. Develop and implement policies to discourage the clearing, filling and channelization of these features (Horner *et al.* 2000). Limit area of disturbance and mandate tree protection measures during construction (Schueler 2000).Research historic extent of natural wetlands within watershed and locate treatment systems where there is no opportunity to restore historic or natural wetlands. Within riparian corridors, this will often involve designing systems that are hydrologically-isolated from surface water flow rather than within the historic floodplain. Collectively, these measures will provide a means to reduce the onsite contamination of wetland sediments and will reduce the potential risk to wildlife. Prioritize larger sites over smaller sites, sites that have a buffer of undeveloped land use (e.g., open space) surrounding the wetland, and sites in which the hydrologic connectivity of the site to adjacent uplands will not be impeded by high levees or dikes or surrounding development.

Reduce Potential for Onsite Exposure and Toxicity to Contaminants. At the watershed scale, contaminant runoff can be minimized by maximizing the water storage and infiltration opportunities within the developed land uses and outside of existing wetlands. Maximize water storage and infiltration opportunities within the landscape unit and outside of existing wetlands to minimize urban runoff (Horner et al., 2000). Options exist for improving the water quality of urban runoff before it enters natural wetlands and aquatic habitats. Consideration should be given to include both source control BMPs and treatment BMPs (including constructed wetlands). In general, source control BMPs (e.g., those which prevent the generation or release of pollutants) are more effective and less expensive than treatment BMPs. Specific recommendations include controlling source of pollutants where possible (e.g., street sweeping, removal of sediment and debris from storm drain inlets, improving infiltration, and other land-based BMPs, etc.).

When Possible, Conduct Pretreatment of Wetland Water Source. Depending on land use, pretreatment should be provided including oil and grit interceptors in highly industrial sites, highways, etc., placement of a primary treatment system such as a sand filter or use sediment collection/settling forebays for treatment of stormwater inflows and for additional treatment of wastewater. Forebays should be designed and located for ease of maintenance, removal of sediment in order to achieve greatest protection of wetland habitat and receiving waters. Habitat, multipurpose or treatment wetlands can be segmented such that the primary treatment is provided in the forebay or initial pond and the remainder of the wetland is used for polishing and/or wildlife enhancement. A variety of treatment strategies, including detention, pre-treatment, treatment, and infiltration, should be incorporated in series in order to maximize removal efficiencies and minimize exposure to wildlife.

Locate BMPs throughout the watershed. Preferably install several smaller decentralized treatment wetlands throughout the watershed rather than one large treatment wetland (Schueler and Holland 2000). Potential on-site as well as cumulative impacts and benefits should be carefully considered.

Creation or Restoration Design Elements

If habitat is an objective of a project to create wetlands, clearly state what the management endpoints are and how the project design is linked to those endpoints. This will provide clarity about what kind of habitat is being provided and what should be monitored to document whether the project is successful.

Locate the site in an area that can support wetland hydrology. Previous studies have shown the importance of the ability to support wetland hydrology in determining the success of the restoration or mitigation project (Ambrose *et al.* 2006, Mitsch and Wilson 1996, Mitsch and Goselink 1993). Soil maps can aid in determining whether a site previously supported wetland by identifying those soil series (Stein *et al.* 2007) that are characteristic of wetland sediments. Connection to groundwater could ensure that other water sources are available to support wetland hydrology.

Assess the potential risk from contaminants. Consider the objective of the treatment wetland; if habitat benefits are desired, then assess whether the water to be treated or sediments are of adequate quality to support wildlife. Soil/sediment or waterborne concentrations of contaminants should be compared to ecological screening benchmarks such as ambient water quality criteria (USEPA 2002) or sediment quality criteria (e.g., MacDonald *et al.* 2000, Field *et al.* 2002). In addition, the surface water and sediments should be evaluated for contaminants that are known to bioaccumulate in the food web (e.g., mercury, selenium, organochlorines, PCBs, dioxins, etc.). Install BMPs and other pretreatment prior in order to minimize runoff of contaminants upstream of targeted wetland restoration. Assure that water quality criteria are met in the major sources of water to the wetland at the inflow point.

Design the site to have good physical structure. Design the site to have greater opportunities for macro- and micro-topographic complexity, including variegated shorelines, elevational relief and moisture gradients. Create gentle slopes to allow for good plant establishment and diversity. Horner *et al.* (2000) showed that a major urban stress to wetland plants were abrupt and dramatic water level fluctuations caused by storm events in catchments with increased imperviousness. To address this, we recommend designing for moderate water level fluctuations during wet weather storm events (McLean 2000) by creating gentle slopes and seasonally wet areas that only flood during extreme events.

Maximize the diversity of habitats within the wetland and transitional upland areas. Developing a wide variety of habitat types within an urban wetland will enhance wildlife diversity and provide connectivity to other habitats. Habitat diversity can be enhanced by providing for areas of open water, unvegetated flats, emergent marsh, nesting islands for waterfowl, swales and other depressional features, as well as transitional riparian and native upland habitat.

Design and maintain the wetland and transitional upland buffer to have appropriate native vegetation. Because the selection of plant palette is one of the major controls on habitat quality in an urban wetland, whether habitat or treatment wetland, it is important to design and maintain the wetland and transitional upland buffer to have the appropriate native plant communities. This is with respect to not only its species diversity but also its vertical structure and horizontal interspersion of habitat types.

Management and Maintenance

Wetland stewardship is essential in urban areas. This study found that wetlands in urban areas scored better with respect to CRAM when an active wetland stewardship program was in place. Regular maintenance and oversight is needed to control human access to sites and manage the variety of urban stressors that can degrade condition.

Manage the hydrology to mimic the natural hydroperiod. Recent studies suggest that many freshwater wetlands in southern California were seasonal in nature, with natural periods of dry down during the summer months (Stein *et al.* 2007). Even perennial ponds would experience a dry down in this climate. Because wetland animals and plants have evolved to acclimate to certain hydroperiods, alteration of the seasonal nature of that hydroperiod may be causing an alteration in the community structure of plants and animals found there. Additional historic ecology studies need to define what the natural hydroperiod for freshwater wetlands in southern California was and how interannual cycles in rainfall may have modulated that hydroperiod. In wetlands and streams whose hydrology has been disturbed, consider managing stormwater and low flow runoff to match, as close as reasonably possible, the predevelopment hydroperiod and hydrodynamic (Horner *et al.* 2000).

Manage urban stressors. Contaminants, human visitation, invasive plants, trash, excessive sedimentation were among the most commonly found stressors in the urban wetlands in this study. Regular maintenance and oversight is needed to mitigate the impact of these stressors on the wetlands. Stressor identification should be conducted at each site and a program of maintenance and directed activities developed to address these stressors.

Maintain the wetland at a frequency necessary to manage stressors, but not in a manner that is incompatible with the seasonal cycles of nesting and reproduction. In urban environments, maintenance of a wetland and its surrounding buffer is important to mitigate the impacts of stressors. This can include removal of non-native vegetation, trash, actions to control access to the site by humans and other predators (cats, dogs, and other urban wildlife), management of hydrology, dredging of sediments, etc. The good condition of several sites found in highly urbanized areas was due to either a requirement for periodic maintenance (i.e. via a permit condition) or a highly dedicated group of volunteers who provided a mechanism to keep the impacts of urban stressors at bay. Many of the stressors challenging the lowest scoring urban habitat wetlands in this study were disturbances that could be mitigated through appropriate maintenance and improved stewardship (e.g., invasive plants, poor buffer quality, etc.).

These maintenance activities, though beneficial, can also result in short-term disturbance to the native wetland flora and fauna. Results from this study suggested that maintenance activities that occur at higher frequencies may become a disturbance. More in-depth research is needed to understand with greater precision where these tradeoffs lie. However, managers of those sites in which urban stressors appear to be successfully mitigated are guided by the general rule that maintenance activities must be scheduled around critical periods of nesting and reproduction.

Monitoring

Conduct monitoring. Monitoring program components should be a function of project objective (habitat, treatment, multipurpose). Intensity of monitoring should be scaled to the project size, but core standardized elements should remain intact regardless of project size. This will give state agencies the ability to report on the impact of projects on wetland quality and quality, as well as BMP effectiveness (where applicable).

Core monitoring for habitat should include filling and updating the project tracking form (www.wetlandtracker.org), documentation of change in wetland and riparian acreage by major habitat types, pre- and post-project general habitat condition using CRAM or an equivalent rapid method. Core monitoring should also include monitoring for contamination and toxicity (see table 14 for detailed recommendations)

Core monitoring for treatment effectiveness should include monitoring of both flow (i.e. via water level) and concentration in wetland influent and effluent at specified periods of interest (wet and/or dry weather). Number of events monitored will depend on the type of question and precision of the answer required (see Table 24 for more detailed recommendations).

REFERENCES

Aquatic Bioassessment Laboratory (ABL). 2003. California Stream Bioassessment Procedure (Protocol Brief for Biological and Physical/Habitat Assessment in Wadeable Streams). California Department of Fish and Game. Aquatic Bioassessment Laboratory, Water Pollution Control Laboratory.

Ambrose, R.J. Callaway and S. Lee. 2006. An evaluation of compensatory mitigation projects permitted under the CWA Section 401 by the California State Water Quality Control Board 1992-2001. www.waterboards.ca.gov/cwa401/docs/wetlandmitstudy_rpt.pdf

Amweg, E.L., D.P. Weston and N.M. Ureda. 2005. Use and toxicity of pyrethroid pesticides in the Central Valley, California, USA. *Environmental Toxicology and Chemistry* 24:966-972.

Azous, A. and R. Horner. 2000. Wetlands and Urbanization. Lewis Publishers. Boca Raton, Florida.

Boxall, A.B.A. and L. Maltby. 1997. The effects of motorway runoff on freshwater ecosystems. *Archives of Environmental Contamination and Toxicology* 33:9-16.

Brown, M. and B. Vivas. 2005. Landscape Development Intensity Index. Envrionmental Monitoring and Assessment. 101:289-309.

Brown, J. and S. Bay. 2004. Organophosphorus pesticides in the Malibu Creek watershed. pp. 94-102 *in*: S.B. Weisberg and D. Elmore (eds.), Southern California Coastal Water Research Project 2003-2004 Biennial Report. Southern California Coastal Water Research Project. Westminster, CA.

Brown, J. and S. Bay. 2006. Assessment of Best Management Practice (BMP) effectiveness for reducing toxicity in urban runoff. pp. 207-226 *in*: S.B. Weisberg and D. Elmore (eds.), Southern California Coastal Water Research Project 2005-2006 Biennial Report. Southern California Coastal Water Research Project. Westminster, CA.

California Department of Transportation (Caltrans). 2004. BMP retrofit pilot program. Final report. Report CTSW-RT-01-050. Caltrans, Division of Environmental Analysis. Sacramento, CA.

California Department of Transportation (Caltrans). 2006. Treatment BMP technology report. CTSW-RT-06-167.02.02. Caltrans, Division of Environmental Analysis. Sacramento, CA.

California Coastal Commission (CCC). 1989. California Coastal Act of 1976 as Amended January 1989. California Coastal Commission. San Francisco, California.

California Department of Fish and Game (CDFG). 1983. A plan for protecting, enhancing and increasing California's wetlands for waterfowl. California Department of Fish and Game. Sacramento, California

California Department of Parks and Recreation (CDPR). 1988. California wetlands, an element of the California Outdoor Recreation Planning Program. California Department of Parks and Recreation. Sacramento, California

Campbell, K. R. 1995. Concentrations of Heavy Metals Associated with Urban Runoff in Fish Living in Stormwater Ponds. Archives of Environmental Contamination and Toxicology 27.

Caraco, D. S. 2000. Stormwater Strategies for Arid and Semi-arid Climates. In H. K. Holland [ed.], The Practice of Watershed Protection. Center for Watershed Protection.

CH2M Hill. 2004. Wetland Capture and Treatment (WetCAT) Network Agreement No. 01-122-259-0. CH2M Hill, Santa Ana, and City of Laguna Niguel Public Works and Engineering. Laguna Niguel, CA.

Collins, J.N., E.D. Stein, M. Sutula, R. Clark, A.E. Fetscher, L. Grenier, C. Grosso, and A. Wiskind. 2007. California Rapid Assessment Method (CRAM) for Wetlands. Version 4.0. 151 pp.

Cooper K. Effects of pesticides on wildlife. In: Handbook of Pesticide Toxicology. Volume 1. Hays W.J. and Law E.R. (eds). Academic Press, New York, New York.

Cowardin, L., V. Carter, F. Golet and E. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. Office of Biological Services, US Fish and Wildlife Service. Washington D.C.

Dahl, T.E. 1990. Wetland losses in the United States 1780's to 1980's. US Fish and Wildlife Service. Washington D.C.

Dunier, M., and A. K. Siwicki. 1993. Effects of Pesticides and other Organic Pollutants in the Aquatic Environment on Immunity of Fish: a Review. Fish and Shellfish Immunology 3: 423-438.

Everts, J.W. 1997. Ecotoxicology for risk assessment in arid zones: some key Issues. *Archives of Environmental Contamination and Toxicology* 32:1-10.

Faber, P.A., E. Keller, A. Sands and B.M. Massey. 1989. The ecology of riparian habitats of the Southern California Coastal Region: A Community Profile. US Fish and Wildlife Service. Carlsbad, California

Finkenbine, J.K., J.W. Atwater and D.S. Mavinic. 2000. Stream health after urbanization. *Journal of the American Water Resources Association* 36:1149-1160.

Galli, F.J. 1988. A Limnological study of an urban stream stormwater management pond and stream ecosystem. George Mason University. Fairfax, VA.

Gambrell, R. 1994. Trace and Toxic Metals in Wetlands: a Review. Journal of Environmental Quality **23:** 883-891.

Gan J., Lee S.L., Liu W.P., Haver D.L., and Kabashima J. N. 2005. Distribution and persistence of pyrethroids in runoff sediments. Journal of Environmental Quality. 34:836–841.

GeoSyntec Consultants and Wright Water Engineers, Inc. 2006. Analysis of treatment system performance. Download from website (http://www.bmpdatabase.org).

Grant, S. B., Sanders B.F., Boehm A.B., Chu A.K., Gouldin M., McGee C.D., Gardiner N.A., Jones B.H., Svejkovsky J., Leipzeig G.V. and Brown A. 2001. Generation of Enterococci Bacteria in a Coastal Saltwater Marsh and Its Impact on Surf Zone Water Quality. Environmental Science and Technology 35: 2407-2416.

Harrington, J. and M. Born. 2000. Measuring the Health of California Streams and Rivers. A Methods Manual for: Water Resource Professionals, Citizen Monitors, and Natural Resources Students. 2nd ed. Sustainable Land Stewardship International Institute. Sacramento, CA.

Helfield, J.M. and M.L. Diamond. 1997. Use of constructed wetlands for urban stream restoration: a critical analysis. *Environmental Management* 21:329-341.

Holland, C.C., J.E. Honea, S.E. Gwin and M.E. Kentula. 1995. Wetland degradation and loss in the rapidly urbanizing area of Portland, Oregon. *Wetlands* 15:336-345.

Horner, R., A. Azous, K. Richter, S.A. Cooke, L. Reinelt and K. Ewing. 2000. Wetlands and stormwater management guidelines. pp. 299-324 *in*: R. Horner [ed.], Wetlands and Urbanization. Lewis Publishers. Boca Raton, Florida.

Kadlec, R.H. and R.L. Knight. 1996. Treatment Wetlands. CRC Press. New York, New York.

Katznelson, R., W.T. Jewell and S.L. Anderson. 1995. Spatial and temporal variations in toxicity in an urban-runoff treatment marsh. *Environmental Toxicology and Chemistry* 14:471-482.

Lahr, J. 1997. Ecotoxicology of organisms adapted to like in temporary freshwater ponds in arid and semi-arid regions. *Archives of Environmental Contamination and Toxicology* 32:1-10.

Ludwa, K.A. and K.O. Richter. 2000. Emergent macroinvertebrate communities in relation to watershed development. pp. 47-67 *in*: R. Horner [ed.], Wetlands and Urbanization. Lewis Publishers. Boca Raton, Florida.

MacDonald, D.D., C.G. Ingersoll and T.A. Berger. 2000. Development and evaluation of consensus-based sediment quality guidelines for freshwater systems. *Archives of Environmental Contamination and Toxicology* 39:20-31.

Mack, J. 2006. Landscape as a predictor of wetland condition: an evaluation of the Landscape Development Index (LDI) with a large reference wetland dataset from Ohio. *Environmental*. 120:221-241.

Magurran, A.E. 1988. Ecological diversity and its measurement. Princeton University Press. Princeton, NJ.

Maltby, L., A.B.A. Boxall, D.M. Forrow, P. Calow and C.I. Betton. 1995a. The effects of motorway runoff on freshwater ecosystems. *Environmental Toxicology and Chemistry* 14:1079-1092.

Maltby, L., A.B.A. Boxall, D.M. Forrow, P. Calow and C.I. Betton. 1995b. The effects of motorway runoff on freshwater ecosystems. *Environmental Toxicology and Chemistry* 14:1093-1101.

Mitsch, W.J. and R. Wilson. 1996. Improving the success of wetland creation and restoration with know-how, time, and self-design. Ecological Applications 6:77-83.

Maund SJ, Hamer MJ, Lane MCG, Farrelly E, Rapley JH, Goggin UM, Gentle WE. 2002. artitioning, bioavailability, and toxicity of the pyrethroid insecticide cypermethrin in sediments. Environmental Toxicology and Chemistry 21:9–15.

Mitsch, W.J. and J.G. Gosselink. 1993. Wetlands. Second Edition ed. Van Nostrand.

National Irrigation Water Quality Program (NIWQP). 1998. Guidelines for interpretation of the biological effects of the selected constituents in biota, water, and sediment. National Irrigation Water Quality Program information report No. 3. U.S. Department of the Interior. Washington D.C.

National Research Council (NRC). 2001. Compensating wetland loss under the Clean Water Act. National Academy Press. Washington D.C.

Ralph, C.J, G.R. Geupel, P. Pyle, T.E. Martin and D.F. DeSante. 1993. Handbook of field methods for monitoring landbirds. Gen. Tech. Rep. PSW-GTR-144-www. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. Albany, CA.

Read, E., E. Strecker and J. Howell. 2006. Ballona freshwater marsh: Annual report of monitoring, operation and maintenance year 3: October 1, 2004 – September 30, 2005. Geosyntec Consultants, Portland Oregon.

Reed, P.B. 1988. National list of plant species that occur in wetlands: California (Region 0). U.S. Fish and Wildlife Service. Biological Report 88(26.10).

Sanders B., F. Arega, and M. Sutula. 2006. Modeling the dry-weather tidal cycling of fecal indicator bacteria in surface waters of an intertidal wetland. Water Research Volume 39 (14): 3394-3408

Sawyer, J.O. and T. Keeler-Wolf. 1995. A manual of California vegetation. California Native Plant Society. Sacramento, CA.

Schiff K. and Sutula M. 2004. Organophosphorus pesticides in stormwater runoff from southern California. Environmental Toxicology and Chemistry. 23(8):1815-1821.

Schueler, T.R. 2000. Environmental impact of stormwater ponds. *in*: H.K. Holland [ed.], The Practice of Watershed Protection. Center for Watershed Protection. Philadelphia, Pennsylvania.

Schueler, T.R. 1995. Stormwater pond and wetland options for stormwater quality control. National conference on urban runoff: enhancing urban watershed management at the local, county, and state level. EPA/625/R-95/003. Washington D.C.

Schueler, T.R. and H.K. Holland. 2000. The practice of watershed protection. Center for Watershed Protection. Phiadelphia, Pennsylvania.

Shutes, R.B.E., D.M. Revitt, A.S. Mungur and L.N.L. Scholes. 1997. The design of wetland systems for the treatment of urban runoff. *Water Science and Technology* 35:19-25.

Southern California Coastal Water Research Project (SCCWRP). 1992. Surface runoff to the Southern California Bight. pp. 19-28 *in*: J.N. Cross and C. Francisco (eds.), Southern California Coastal Water Research Project Annual Report 1990-1991 and 1991-1992. Westminster, CA.

Stein, E.D., S. Dark, T. Longcore, N. Hall, M. Beland, R. Grossinger, J. Casanova and M. Sutula. 2007. Historical ecology and landscape change of the San Gabriel River and floodplain. Southern California Coastal Water Research Project. SCCWRP Technical Report #499. Costa Mesa, CA.

Strecker, E., M. Quigley, B. Urbonas and J. Jones. 2004. State-of-the-art in comprehensive approaches to stormwater. *The Water Report* 6:1-10.

Strecker, E.W. 1995. The use of wetlands for stormwater pollution control. National conference on urban runoff: enhancing urban watershed management at the local, county, and state level. EPA/625/R-95/003Washington D.C.

Sutula, M., E. Stein, S. Madon and S. Cole. 2005. Inventory of urban and treatment wetlands in southern California: a summary of findings. Draft report to the Los Angeles Regional Water Quality Control Board for the Assessment of Effectiveness of Treatment Wetlands for Stormwater BMPs and Compatibility with Wildlife Beneficial Uses Agreement. No. 04-090-554-0. Costa Mesa California

Sutula, M. and E. Stein. 2003. Habitat value of natural and constructed wetlands used to treat urban runoff – a literature review. Technical report 388. Southern California Coastal Water Research Project. Westminster, CA.

Tetra Tech. 2006. Performance evaluation of the New River demonstration wetlands. Tetra Tech. Pasadena, CA.

United States Environmental Protection Agency (USEPA). 1990. Macroinvertebrate field and laboratory methods for evaluating the biological integrity of surface waters. EPA/600/4-90/030. U.S. Environmental Protection Agency, Office of Research and Development. Washingon D.C.

United States Environmental Protection Agency (USEPA). 2000. Methods for measuring the toxicity and bioaccumulation of sediment-associated contaminants with freshwater invertebrates (Second Edition). EPA/600/R-99/064. Washington D.C.

USDOI (1998). Guidelines for Interpretation of the Biological Effects of Selected Constituents on Biota, Water and Sediment.

Van Derveer, W.D. and S.P. Canton. 1997. Selenium sediment toxicity thresholds and derivation of water quality criteria for freshwater biota of western streams. *Environmental Toxicology and Chemistry* 16:1260-1268.

Weston DP, You J, Lydy MJ. 2004. Distribution and toxicity of sediment-associated pesticides in the agriculture-dominated waterbodies of California's Central Valley. Environ Science and Technology 38:2752–2759.

WRC Consultant Services. 2004. Big Canyon water quality results August 2004. Report to Community Conservancy International. Newport Beach, California.

Wren, C. D., C. A. Bishop, D. L. Stewart, and G. C. Barrett. 1997. Wildlife and Contaminants in Constructed Wetlands and Stormwater Ponds: Current State of Knowledge and Protocols for Monitoring Contaminant Levels and Effects in Wildlife. Canadian Wildlife Service.

Yoon, V.K. and E. Stein. 2007. Contribution of natural catchments to levels of metals, nutrients, and solids in storm water. *Water, Air, and Soil Pollution*. Volume 190: 193-195.

Yousef, Y.M., M. Wanielista, J. Dietz, L. Yin and M. Brabham. 1990. Final report-efficiency optimization of wet detention ponds for urban stormwater management. Florida Department of Environmental Regulation. Talahassee, Florida.

Zedler, J., C.S. Nordby and B.E. Kus. 1992. The ecology of Tijuana Estuary, California: a national estuarine research reserve. National Oceanic and Atmospheric Administration Office of Coastal Resource Management, Sanctuaries and Reserves Division. Silver Spring, Maryland

Zedler, P.H. 1987. The ecology of Southern California Vernal Pools: A Community Profile. Biological Services, U.S. Fish and Wildlife Service. Carlsbad, California.

Yousef, Y.M., M. Wanielista, J. Dietz, L. Yin and M. Brabham. 1990. Final report-efficiency optimization of wet detention ponds for urban stormwater management. Florida Department of Environmental Regulation. Talahassee, Florida.

Zedler, J., C.S. Nordby and B.E. Kus. 1992. The ecology of Tijuana Estuary, California: a national estuarine research reserve. National Oceanic and Atmospheric Administration Office of Coastal Resource Management, Sanctuaries and Reserves Division. Silver Spring, Maryland

Zedler, P.H. 1987. The ecology of Southern California Vernal Pools: A Community Profile. Biological Services, U.S. Fish and Wildlife Service. Carlsbad, California.